Examining the health of subtidal reef environments in South Australia

Part 4: Assessment of community reef monitoring and status of selected South Australian reefs based on the results of the 2007 surveys

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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

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Executive summary

Concern over the degradation of Adelaide's metropolitan reefs has led to several Reef Health scientific surveys since 1996 and the ongoing community Reef Watch monitoring program. The last survey was conducted in 2005, and the subsequent report provided a health ranking for a number of reefs adjacent to Adelaide. The present report extends the 2005 survey report by:

1. Providing an up-to-date assessment of the condition of Adelaide' s reefs;
2. Comparing the condition of Adelaide's reefs in 2007 with the 2005 survey to determine whether there has been any change in reef health rankings; and
3. Comparing the 2007 scientific and community data for monitoring reef health to assess the efficacy of community monitoring.

The reefs of Adelaide and Fleurieu Peninsula showed the same broad pattern in 2007 as when they were previously surveyed two years earlier as part of the same project. Based on the line-intercept transect data, there were two major groups of sites, representing the northern and the southern reefs and, quite separate to these two groups, the two apparently impacted reefs -- Broken Bottom and Semaphore. The inclusion this year of a reef to the north of these (and yet away from the influence of metropolitan Adelaide) allowed us to demonstrate that the poor condition of these reefs did not simply represent the northern extent of a natural north-south geographic trend. Rather, some other influence, probably associated with urban Adelaide, was evident.

Long-term trends since 1996 seem to indicate a general improvement in the status of reefs along this coast. This may be a biotic reflection of the cessation of some dredging operations or of a decrease in the nutrient loading from wastewater treatment plants, and provides circumstantial evidence that such an improvement in practices has the potential to allow recovery of impacted reefs. Nevertheless, the poor condition of the reefs closest to metropolitan Adelaide indicates that further improvements are required.

A comparison between the dataset of the Reef Health program and that collected by Reef Watch, a community-based monitoring initiative, showed a very similar picture when employing multivariate analysis to study the line-intercept transect data. To a large extent, where discrepancies arose, it is likely to be the result of medium-scale spatial variation (i.e. sampling different areas) rather than a real difference between the data

collectors. Having said this, there were some minor taxonomic issues, which, once addressed, will improve the Reef Watch monitoring.

Assessment of the reefs based on the Reef Health Index (Turner et al. 2007) demonstrated greater disparity between comparable datasets than was the case for the LIT analysis – i.e. the 2005 v 2007 datasets, and the Reef Health v Reef Watch datasets. This is probably due to inadequate methodology for the assessment of mobile fauna (in particular, greater temporal replication is required), and the fact that the mathematical model used to calculate the index is in its early stages and will develop greater utility and accuracy with continuing use and development.

The following recommendations are made:

- Reef monitoring should continue;
- In combination with some professional guidance, community-based monitoring programs (in particular Reef Watch) offer an excellent vehicle for this work which should be encouraged and resourced appropriately;
- A broader range of reefs should be surveyed, possibly at the expense of the frequency of re-survey;
- The protocols utilised by the Reef Health program should be continued with the following modifications:
    - Transects should be marked with permanent endpoints;
    - Photographic transects should be adopted where possible;
    - Alternative methods of assessing mobile fauna for the reef health index are required; and
    - Attention needs to be paid to calculation of individual indices, particularly the appropriateness of a "null" score.
- Improvement to water quality since the mid-1990s should be lauded and continued improvement should be encouraged if we are to see recovery of the most impacted reefs.

1 Introduction

The 2007 Reef Health program represents a continuation of a series of surveys that have focussed on an assessment of the status of South Australian reefs, particularly those off the coast of Adelaide. Not only does it re-examine sites, which have been the subject of previous study in order to assess change, but it also evaluates the efficacy of surveys performed by community groups (specifically Reef Watch).

Survey programs designed to monitor the status of the reefs of Adelaide and Fleurieu Peninsula began with a series of surveys in 1996 (Cheshire et al. 1998) that examined reefs off Semaphore, Broken Bottom, Glenelg, Hallett Cove, Noarlunga and Aldinga. This series of surveys showed that there was a clear biotic gradient from south to north whereby the robust brown macroalgal cover demonstrated in the southern sites was replaced with foliaceous red algae further north. That study identified important physicochemical gradients along this geographic range that may have resulted in this change, the most important of which were a natural decrease in wave energy, and the anthropogenic impacts of eutrophication and sedimentation. However, the actual causes remained speculative.

In 1999, the above reefs were re-surveyed, and the scope was broadened with the addition of Southport, Moana and Horseshoe Reefs (Cheshire and Westphalen 2000). The conclusion on this occasion was that the southern and central reefs appeared not only healthy, but a number were improving in terms of robust brown canopy cover. The northern reefs (Broken Bottom and Semaphore), whilst degraded, appeared relatively stable. Of concern was the high cover of the mussel *Brachidontes rostratus* at Horseshoe Reef and its increasing cover at Noarlunga. Mussels can act to trap sediment and form a dense monospecific mat, excluding all other species (Lubchenko and Menge 1978, Petraitis 1995).

The Reef Health program was subsequently significantly expanded with the support of a wide variety of agencies (see Acknowledgements), and in 2005, a major set of surveys examined the reefs, not only of Adelaide and Fleurieu Peninsula, but also those of Yorke Peninsula. This provided a wider geographic context within which to gain an understanding of the status of our reefs. The subsequent report (Turner et al. 2007) represents the best broad-scale assessment of the reefs of the region to date. Utilising a wide variety of biotic and physical parameters (including macroalgal cover, fish communities, invasive taxa, mussel cover, turfing algae and sediment levels), the Turner

et al. report marked the first time an index had been distilled from multiple data sources to provide a simple measure of the health of these systems. It also maintained a multivariate ordination approach such as had been employed in previous years.

As was the case in the previous assessments, a south-to-north gradient of decreasing cover of large Phaeophycean (brown) macroalgae and increasing cover of smaller (foliaceous and turfing) Rhodophycean (red) macroalgae was observed. Whilst a number of sites demonstrated greater canopy cover, some reefs raised concern because of trends toward greater bare substrate since 1999, sometimes at the expense of robust brown macroalgal canopy. To a large degree, the conclusions drawn from the calculated reef health index mirrored those based on the ordination of communities. There was a gradient of good to poor health from south to north, with Horseshoe Reef standing out as being in poor condition, which was not predicted based on geographic location. Again, the anthropogenic factors of eutrophication, sedimentation and turbidity were raised as possibilities for the poor condition of reefs. However, due to a lack of both historical data and suitable reefs for comparison north of Adelaide, the conclusion that this trend was due to human impacts was still debatable as it was confounded by the natural south - north gradient. The first of these issues has been dealt with through a historical reconstruction of data for the Port Noarlunga/Horseshoe Reef area (Connell et al. 2008). The second issue was partially addressed in the current study by surveying an additional reef north of Adelaide in NE Gulf St Vincent.

Importantly, Turner et al. (2007) also aired the potential for community-based monitoring. This was specifically addressed in Turner et al. (2006). Reef Watch is a community environmental monitoring program, managed by the Conservation Council of South Australia. Training is provided to recreational SCUBA divers in order to allow them to collect data describing the composition of reefs so that a useful understanding of reef status can be obtained. This training is provided by experts in a variety of marine biological fields in South Australia. The overlap of reefs surveyed in the same year (2007) by Reef Watch and scientists associated with the Reef Health program, and the broadly similar methods employed, provided an invaluable opportunity to compare the two types of data.

The present report has the following objectives:

1. Provide an up-to-date assessment of the condition of selected reefs along Adelaide's metropolitan coast;
2. Provide a comparison of the 2007 surveys to those conducted earlier to determine whether there has been a shift in the structure of the biological communities and reef health rankings;
3. Provide a comparison of data collected by community divers from the Reef Watch program and scientific divers from the Reef Health program collected during 2007; and
4. Appraise the methods and indices used in this program to assess the "health" of reefal ecosystems.

2 Methods

2.1 Survey sites

Many of the reefs surveyed in 2005 were re-surveyed in 2007 (Figure 1), but using a slightly modified protocol to decrease survey times and to enable comparison with community data. The indices used in the 2007 surveys are listed in Table 1.

From March to June 2007, surveys were conducted at 15 sites over 12 reefs across eastern GSV, including the Adelaide metropolitan region (Figure 1; Table 2). An additional site (Northern Reef) to the 2005 study was surveyed in NE GSV between Port Gawler and Port Wakefield to help disentangle the natural and anthropogenic gradients. The coordinates of the Northern Reef were supplied by commercial fishermen and are confidential.

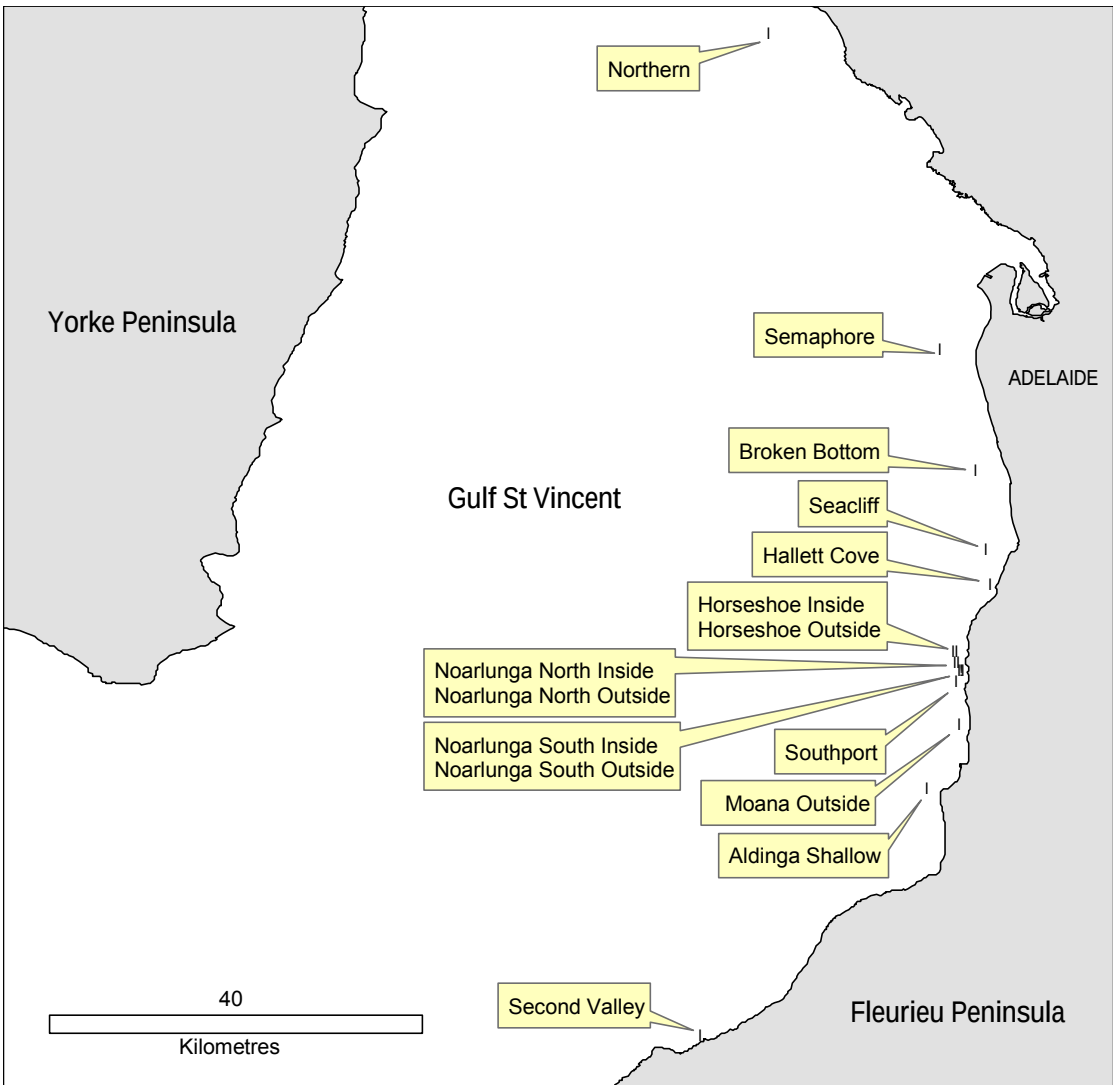

Figure 1. Location of the 15 reef sites surveyed in eastern Gulf St Vincent during 2007. Note that the northern reef is shown only in the general area of its actual location.

Table 1. Eleven indices considered from the 2007 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

Table 2. Locations of the eastern Gulf St Vincent reefs surveyed in 2007. Coordinates are based on the WGS84 datum. The final column indicates whether the sites were also surveyed by Reef Watch (RW), a community-based organization.

Site name	Abbreviation	Depth (m)	Latitude (°S)	Longitude (°E)	Reef Watch Survey?
Northern	NRT	4	Confidential	Confidential	
Semaphore	SEM	8	34 50.826	138 26.757	
Broken Bottom	BRB	9	34 57.801	138 28.817	
Seacliff	SCF	12	35 02.398	138 29.491	RW
Hallett Cove	HAL	5	35 04.418	138 29.661	RW
Horseshoe Inside	HSI	5	35 08.276	138 27.775	
Horseshoe Outside	HSO	5	35 08.365	138 27.483	
Noarlunga North Inside	NNI	5	35 08.930	138 27.695	RW
Noarlunga North Outside	NNO	5	35 08.849	138 27.782	RW
Noarlunga South Inside	NSI	5	35 09.420	138 27.979	RW
Noarlunga South Outside	NSO	5	35 09.415	138 27.925	RW
Southport	SOU	4	35 10.065	138 27.736	
Moana Outside	MSO	5	35 12.551	138 27.863	
Aldinga Shallow	ASH	5	35 16.254	138 25.971	
Second Valley	SVA	5	35 30.583	138 12.889	RW

2.2 Scientific Reef Health Survey

Scientific surveys were conducted at all 15 reef sites. Four 50 metre transect lines were systematically located from points haphazardly chosen at each site. The four transects were actually arranged as pairs with two transects extending in opposite directions from a common start point for each pair. On each transect, SCUBA divers conducted a standard set of sampling protocols (see Appendix B in Turner et al. 2007), consisting of:

- Habitat descriptions including reef topography, environmental conditions such as the level of sedimentation, and dominant life forms in the area of the transect;
- A pelagic fish survey, whereby species, abundance and length were recorded for all fish within a 5 m belt of the 50 m transect (2.5 m either side of tape);
- Cryptic fish and large mobile invertebrate survey along a 1 m belt beside the transect;
- A line-intercept transect (LIT) recording benthic assemblages over the first 20 m of the transect;
- An invasive species survey specifically searching for the presence of a range of known invasive taxa across the area.

Complete descriptions of the sampling regime are presented in Appendix A. Updated site descriptions are presented in Appendix B.

2.3 Community Reef Watch Survey

Reef Watch surveys were conducted at seven of the 15 reef sites (Table 2). The Reef Watch divers undertook a modified version of the full scientific survey, which included the pelagic fish survey, the belt (invertebrate) transect, and the LIT. However, as expertise amongst the group varied, and dive sites were selected based on personal preference rather than a systematic sampling regime, differing numbers of each type of survey were completed at each site:

Second Valley: 2 fish surveys, 2 invertebrate transects, 2 LITs

Noarlunga South Outside: 5 fish surveys, 3 invertebrate transects, 2 LITs

Noarlunga South Inside: 1 fish survey, 1 invertebrate transect, 2 LITs

Noarlunga North Outside: 4 fish surveys, 1 invertebrate transect, 5 LITs

Noarlunga North Inside: 2 fish surveys, 2 invertebrate transects, 2 LITs

Hallet Cove: 2 fish surveys, 3 invertebrate transects, 4 LITs

Seacliff: 2 fish surveys, 2 invertebrate transects, 2 LITs

Note that when fish and invertebrate surveys were done, a complete 50 m transect was always used. However, the LITs were not always of the same length. It is also worth noting that whilst abundance data were calibrated taking into account the length, width and number of transects, there is no simple way to estimate the effect of differing sampling efforts on species richness. Thus the assumption has to be made that sampling effort is adequate in all cases to estimate total species richness.

For the purposes of comparison with Reef Health surveys, Reef Watch surveys were conducted in autumn 2007, with the exception of Second Valley, which was surveyed three months earlier, in summer 2007. Approximate locations were used for all sites except the Noarlunga Outside locations, where actual SARDI GPS coordinates were used. As an example of the spatial variation in survey locations, coordinates of the Reef Watch survey at Seacliff were recorded and indicated that the survey was conducted 343 m from the Reef Health survey of the same reef.

2.4 Data manipulation - index calculation

Data from the 2007 field surveys were manipulated according to the protocols outlined in Turner et al. (2007) to produce a raw value for each indicator. These values were then compared against threshold levels and scaled to produce a final index score.

The following (sections 2.4.1 – 2.4.6) is reproduced from Turner et al. (2007) a previous report in this series. Note that some of the original indices were calculated relative to an overall mean or median. In order to avoid the problem of a changing baseline (or criteria), the means and medians remained a function only of the 2005 dataset for the current analysis rather than altering to include data from 2007.

2.4.1 Indices of areal cover

Four areal cover indices were considered; 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 to some extent 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

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

2.4.2 Indices of abundance

The fish species considered as being site-attached are listed in Appendix C. 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

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. 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. Although incorporating size as well as number, this index will be referred to as "abundance" hereafter. Calculation followed the same procedure as for fish abundance.

2.4.3 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.

2.4.4 Indices of species richness

Macroalgal species richness was based on data obtained from the line intercept transects. Mobile invertebrate species richness was calculated from the invertebrate transects. 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.

2.4.5 Scaling of indices

Upper and lower threshold values were determined for each index based on available information and expert advice (see Turner et al. 2007 for justifications of each). 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, 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, and therefore a "null" score was recorded.

For each indicator, raw figures matching or falling outside of the threshold range were given values as defined in Table 3. 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 (see Figure 2).

Table 3. Critical thresholds and index parameters used for scaling the indices of reef health. These values were maintained from the 2005 surveys and where medians had to be calculated, this was done on the basis of 2005 data to avoid moving baselines on the basis of new data.

	Threshold		Index value			
Index name	**Lower**	**Upper**	**<Lower**	**Lower**	**Upper**	**>Upper**
Areal cover indices						
Areal cover of canopy macroalgae	20	60	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

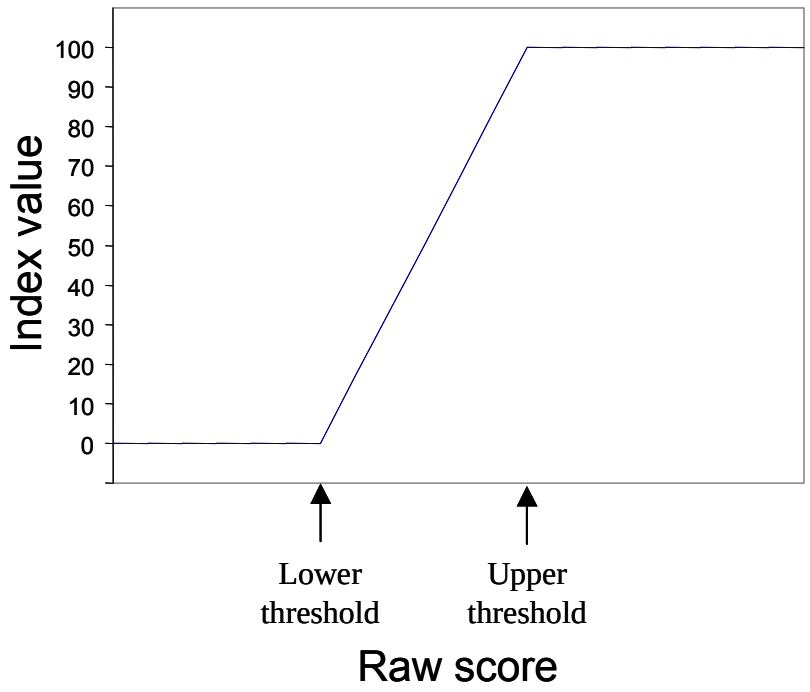

Figure 2. Example scaling of a positive index (a negative index would be the mirror of this plot). Raw figures less than the lower threshold received the minimum value (0), between the two thresholds the index was scaled linearly between the minimum and maximum (100), and for raw scores greater than the upper threshold the maximum index value was recorded.

2.4.6 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 should not necessarily be allocated to the Poor Condition category, but are of greater concern than those described as being in Good Condition. 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.

2.5 Data analyses

Three major issues were examined:

1. Spatial patterns of reef assemblages in eastern GSV from the 2007 survey;
2. Changes to Adelaide and Fleurieu Peninsula metropolitan reefs, 2005 – 2007; and
3. Comparison of scientific and community-based assessments of reef status in 2007.

In each of these instances, two forms of analysis were employed - multivariate analysis (in particular, nonmetric multidimensional scaling (nMDS)) of the LIT data and calculation of health indices as detailed in the previous section. In order to maintain consistency with the previous (2005) surveys (Turner et al. 2007), the same forms of analysis were employed and the indices calculated in the same fashion. Suggestions as to how this method might be improved are nonetheless included in the discussion.

2.5.1 Biotic patterns of reef assemblages and reef health in 2007

Non-metric Multidimensional Scaling (nMDS) was applied to the LIT datasets to provide an ordination which reflected the similarity of the sites to each other in a two dimensional format. Analysis was performed using PC-Ord 4.0 (McCune and Mefford 1999). LIT data was represented in terms of the cover of 7 functional groups to avoid issues created by the variable level of taxonomic ability amongst the divers and to reduce the noise associated with small scale changes in microhabitat that are likely to be unrelated to the larger scale gradients that we were interested in. These groups were canopy brown algae; understorey brown algae; understorey green algae; understorey red algae; turfing algae; animals; and bare substratum.

Note that at each site, the sum of these covers was 100%. The Bray-Curtis distance measure was utilised with a maximum of 100 iterations and a step length of 0.20. Stress is reported on each ordination, but was always < 15%.

The ordinations are presented with an overlay displaying axes relating to each of the functional groups. The length of the functional group axis indicates how well the axis is correlated with position on the ordination. These axes point in the direction of a more significant contribution from that particular functional group.

This analysis was applied to the average composition of the sites (as described by the 4 transects), and to the individual transects to obtain an estimate of the relative within-reef variability.

Stacked column graphs, describing the average percentage composition of each of the reefs, were also constructed from the LIT data to provide an alternative view of the relationships between sites.

2.5.2 Changes in reef health 2005 - 2007

The change in health of the reefs between 2005 and 2007 was assessed through ordination of the LIT data for all sites in common to both points in time. The method is described fully above. In addition to an ordination of the 2005 and 2007 data, we were able to obtain data from the 1996 and 1999 surveys detailed in Cheshire et al. (1998 and 2000). This allowed the creation of an ordination with reefs represented at up to 4 points in time and provided the basis for a longer-term analysis.

Multivariate analysis of change is not simple. There are a variety of methods available and each has its own strengths and weaknesses. Whilst the vectors between points representing the same site at successive points in time give an indication of the scale of change and to some degree the direction, Collings (1998) identified that the vectors were not directly comparable and instead proposed calculating the percentage change between times and plotting that. Thus the points plotted are a representation of the change. Similarity of change is thus represented by proximity of two points on the ordination. This form of analysis was also applied to the data.

Change analysis was done by calculating the absolute difference in percentage cover of each functional group and then categorizing these scores into a 9-point scale. An nMDS ordination was then performed, based on the Bray-Curtis distance measure. It was not relativised / standardised. The categories for the 9-point scale are detailed in Table 4.

Table 4. The magnitude and direction of change represented by each of the categories in a 9-point scale. These categories provided the dataset for ordination of the 2005 – 07 change.

Category	Change in raw % cover
1	>25% decrease
2	20%-25% decrease
3	15%-20% decrease
4	5-15% decrease
5	5% increase to 5 % decrease
6	5% - 15% increase
7	15%- 20% increase
8	20% - 25% increase
9	>25% increase

These values represent the absolute change in percentage cover, rather than as a proportion of the original value. For example, if a raw score for a functional group at a site was 10% cover in 2005 and 28% in 2007, this would have represented a change of 18% and been a category 7 change, as would a change from 80% to 98%.

The changes from 2005 – 07 for each functional group were also represented graphically by means of column graphs in order to provide an alternative visual assessment of the results of the ordination.

Reef health indices and status were calculated in the manner described previously for both 2005 and 2007, and the differences detailed for the individual indices, the overall index and the three point "traffic light" status.

2.5.3 Comparison of community-based and professional reef monitoring data

A comparison of the picture obtained from the community-based organization Reef Watch and the professional Reef Health program was conducted on the basis of both the ordination of LIT data and the "reef health index".

An ordination of all points representing the averages of the sites according to each set of LIT data was conducted as has been described previously. In addition to an ordination of all sites common to both Reef Health and Reef Watch with all points displayed together, a version was produced which provided separate pictures of the sites as surveyed by the two different methods (professional and community-based). Note that

this was based on the original analysis of all points together rather than representing two additional separate analyses. It simply presents the points from each method in separate pictures. This was done because, whilst the original depiction of all sites allows comparison of each site as surveyed by the two methods, the latter presentation better demonstrates the overall patterns identified by a single method and allows comparison of those patterns.

In addition to ordination of site averages, an ordination of the Reef Health transects is presented which only includes transects from those sites also assessed by the Reef Watch program. This is performed to demonstrate the spatial variability of the transects which reflects on the reliability of the estimated average of each of those sites. Distance measures and step lengths were as previously described.

Again, stacked column graphs demonstrating the composition of each site as assessed by each program (Reef Health and Reef Watch) are presented, as is the comparison of the reef health index.

3 Results

3.1 Patterns of reef assemblages and reef health in 2007

In 2007, a reasonably clear pattern exists in the biota of the sites. Two groups of relatively similar sites exist, with an additional two sites that are quite dissimilar to all others (Figure 3). It is evident that a group of sites, including Hallett Cove, Southport, Noarlunga North Outside, Moana, Aldinga and Second Valley, are characterised by a high cover of Phaeophycean canopy algae (>60%; see also Figure 4). Of these, Second Valley has the highest canopy cover, and Hallett Cove the least. At the latter site, brown understorey algae are quite prevalent. The second group of sites is more diverse (as indicated by the greater spread of points in Figure 3). This group consists of Seacliff, Horseshoe Reef Inside and Outside, all the Noarlunga sites with the exception of Noarlunga North Outside (which fell in the previous group) and the northern reef. These sites all demonstrated a moderate (25-40%) cover of canopy algae and moderate amounts of turfing algae, red understorey and sessile animals. Northern Reef is interesting in that it demonstrates a reasonable degree of canopy (26%), but still supports a substantial turf community (46%). This may explain why it is found on the fringe of this second group of sites. The final two sites, Broken Bottom and Semaphore, were both characterised by an almost total lack of canopy, but differed greatly in terms of the major contributor to cover. More than 50% of the substrate at Broken Bottom was covered by turf, while Semaphore had an abundance of understorey species (>85% cover).

Some idea of the biotic variability of a site can be gained from ordinating each of the four separate LITs used to describe each site and observing the degree of similarity between transects within a site. The same general pattern is evident as when the averages were ordinated, and the same relationships with the functional group axes were demonstrated. However, whilst some sites demonstrated a high degree of similarity between transects, others were far more variable (Figure 5). Sites demonstrating a high degree of similarity (i.e. transects are found close together on the ordination) were generally those dominated by a particular functional group (i.e. those found toward the periphery of the ordination of averages in Figure 3). Thus, the transects of the canopy dominated sites of Second Valley, Aldinga, Noarlunga North Outside, Moana and Southport were relatively tightly clustered (within a site) in terms of their biotic composition, as were Broken Bottom and Semaphore. Far greater variation is found between the transects describing sites with a more even spread of cover amongst the

functional groups. Thus the Noarlunga sites (except Noarlunga North Outside), the two Horseshoe Reef sites, Northern Reef and Seacliff all demonstrated a moderate to high degree of variability between transects within a site. Because of the degree of variability between transects within sites, it is possible to define only Broken Bottom and Semaphore as individual sites and two broad groups consisting of multiple sites.

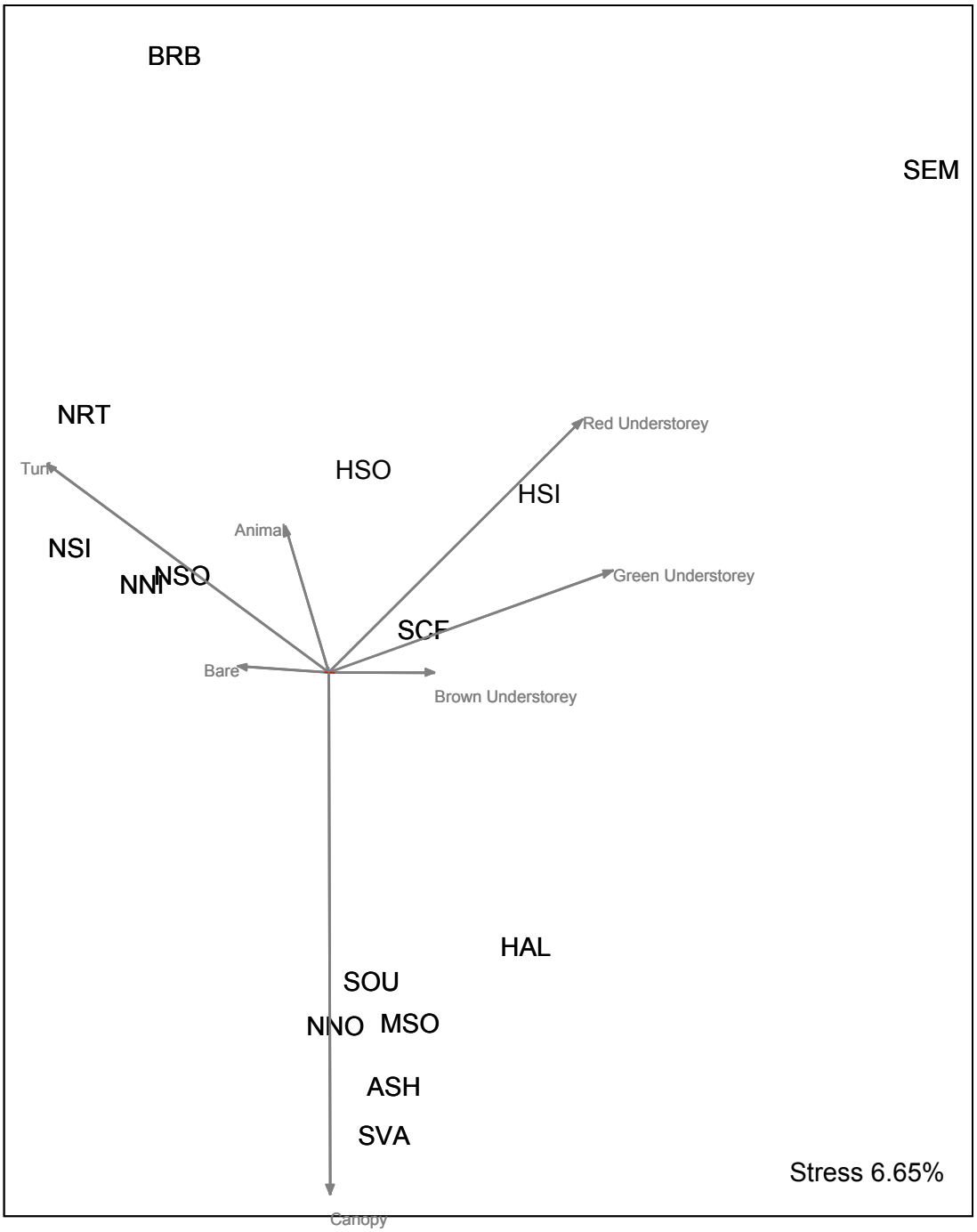

Figure 3. nMDS ordination of sites of the 2007 Reef Health surveys on the basis of functional groups described by LIT data. Overlaid upon this ordination are vectors representing the associations between sites and the functional groups describing them. Length of vector is a representation of the strength of the relationship. Abbreviations are described in Table 2.

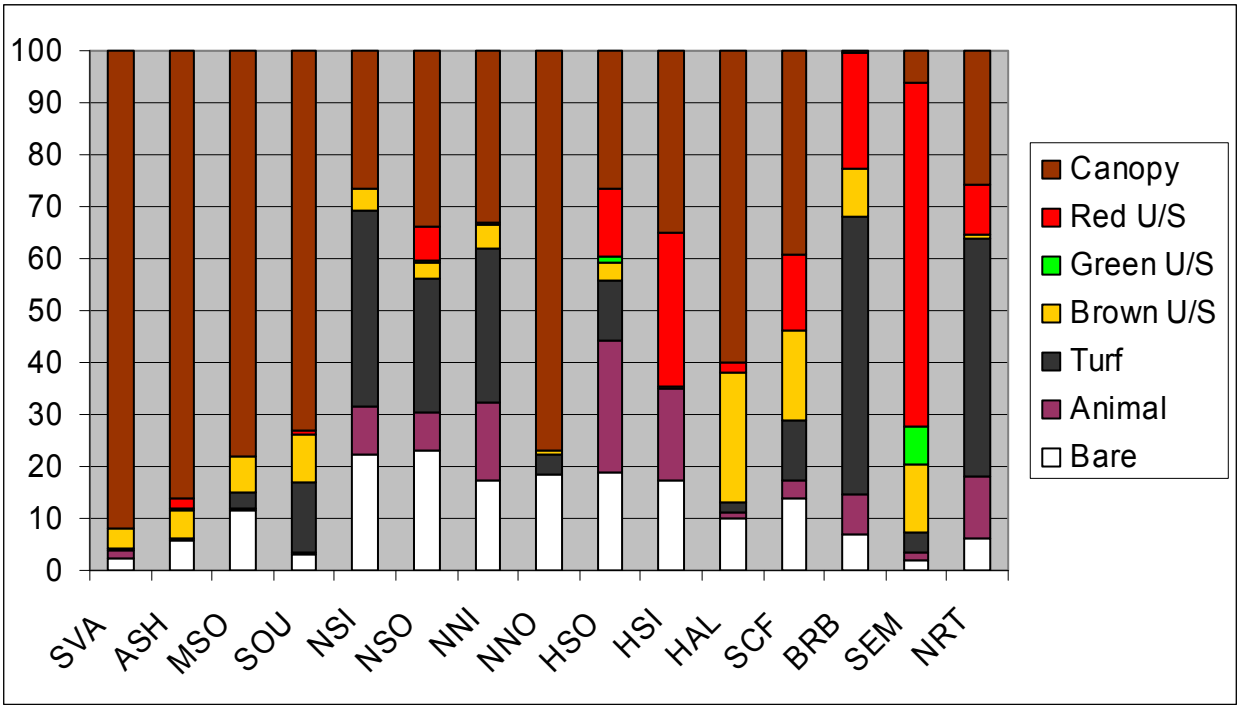

Figure 4. Average cover (%) of each functional group at each site of the 2007 surveys as demonstrated by LIT data.

Calculation of the Reef Health Index, resulted in three sites being classified as being in "poor" condition, four sites in "good" condition, and 8 in the intermediate, "caution" category (Table 5). The scores used as cut-offs for each of these categories were <35 or >65, for consistency with the original (Turner et al. 2007) report. The scores ranged from a low of 18 at Broken Bottom to 92 at Second Valley.

The sites classified as being in poor health were Northern Reef, Broken Bottom and Semaphore. The sites classified as "good" were Second Valley, Moana, Southport and Hallett Cove. All other sites demonstrated an intermediate health index and were classified as requiring "caution".

Variability across sites within each of the individual measures of health was often substantial (see Table 5):

- Areal cover of canopy macroalgae varied to the maximum extent possible, registering 0 at Semaphore and Broken Bottom to 100 at several of the southern sites;
- Turfing macroalgae generally made little difference to the overall score, usually registering a null value Two sites (Northern Reef and Broken Bottom) exceeded 40%

cover and therefore scored zero, whilst Noarlunga South Inside scored 2 (out of 100). All other sites had null values as they had <25% turfing cover;

- Mussel cover, bare substrate cover and presence of invasive species had no impact on the overall score as they were too low at all sites to warrant an index score;
- Abundance of site-attached fish varied almost across the entire range of possible scores, from 5 at Northern Reef to 100 at many of the more southerly sites;
- Abundance of mobile predators did not vary in a manner associated with geographic location. Most (9 of 15) sites scored the maximum possible 100. Aldinga scored lowest with 14 of a possible 100 points;
- Abundance of blue-throated wrasse appeared related to the north-south gradient, but was highly variable. It ranged from 100 at many of the southern sites to zero at 4 sites (Northern Reef, Semaphore, Noarlunga North Outside and Aldinga);
- Evidence of high sedimentation, which is a subjective assessment, scored zero at 6 sites, 3 of which were at Noarlunga. The remaining 9 sites were allocated a "null" score;
- Richness of mobile invertebrates was not as variable as many of the other indices, scoring between 26 (Northern Reef and Moana) and 63 (Horseshoe Outside);
- Richness of macroalgae varied in a manner associated with the north-south gradient between 17 (Northern Reef) and 100 (all sites south of Noarlunga).

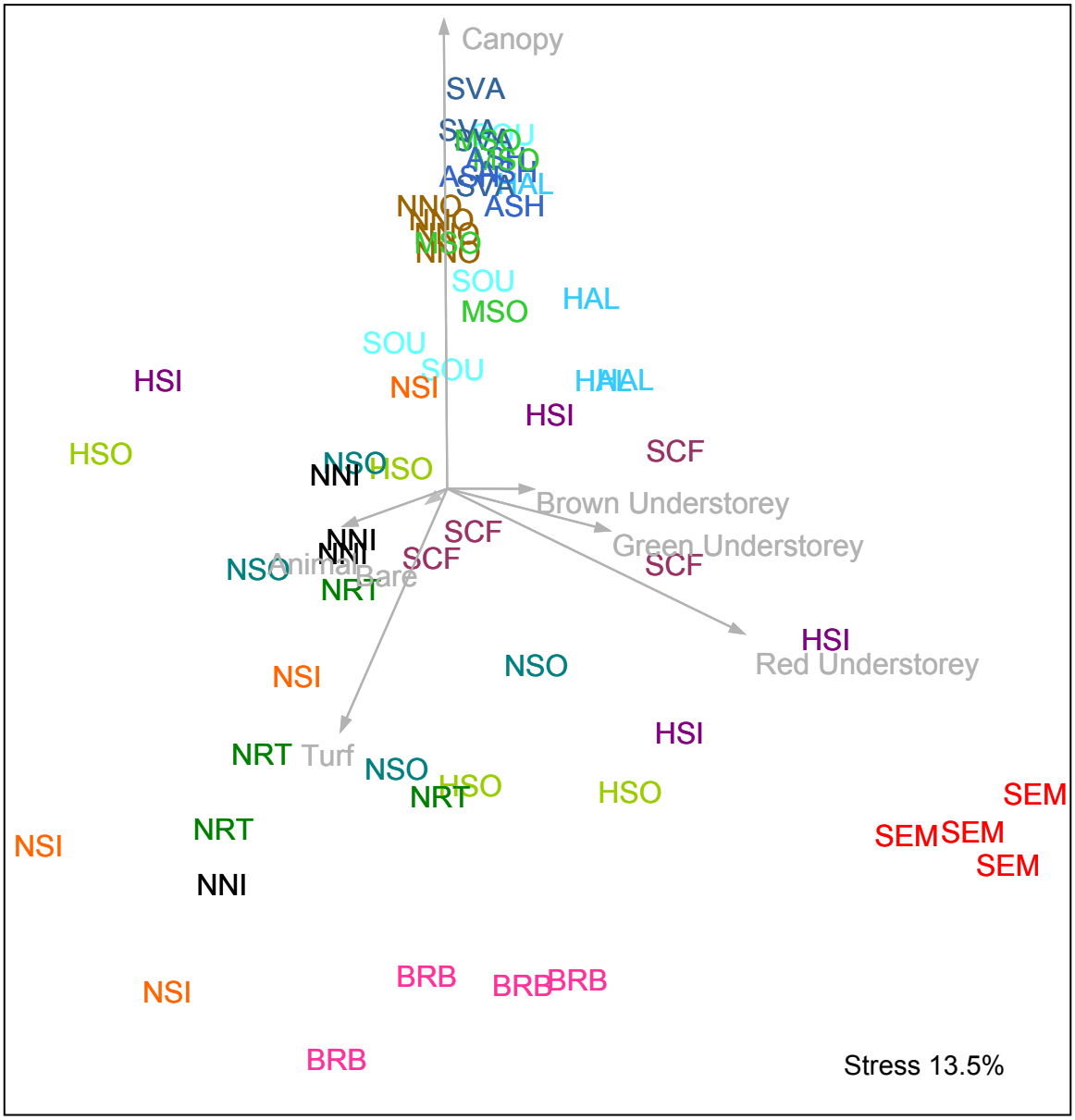

Figure 5. nMDS ordination of all transects surveyed during 2007 by the Reef Health program. Transects from the same site are coded with the same site code and colour. Axes demonstrate the relationship between the sites and the % cover of each functional group. The length of the axes demonstrates the relative strength of this relationship. Abbreviations for sites follow Table 2.

Table 5. Summary of indicator and overall scores (from 0-100 in all cases except for the presence/null indices) for all reefs of the 2007 surveys. 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
Northern Reef	7	0			5	100	0		0	26	17	**19**	**Poor**
Semaphore	0				7	71	0		0	31	41	**21**	**Poor**
Broken Bottom	0	0			15	35	7			47	21	**18**	**Poor**
Seacliff Reef	48				100	71	39			36	41	**56**	**Caution**
Hallett Cove	99				100	100	100			42	36	**79**	**Good**
Horseshoe Inside	42				32	100	24		0	36	22	**37**	**Caution**
Horseshoe Outside	16				24	100	24			63	34	**44**	**Caution**
Noarlunga North Inside	33				100	100	100		0	42	22	**57**	**Caution**
Noarlunga North Outside	100				100	100	0		0	42	100	**63**	**Caution**
Noarlunga South Inside	10	2			93	100	95		0	57	32	**49**	**Caution**
Noarlunga South Outside	33				100	100	49			52	31	**61**	**Caution**
Southport	100				42	100	100			52	100	**82**	**Good**
Moana Outside	100				100	35	100			26	100	**77**	**Good**
Aldinga Shallow	100				100	14	0			21	100	**56**	**Caution**
Second Valley	100				100	92	100			57	100	**92**	**Good**

3.2 Changes in reef health 2005 – 2007

Ordination of Line Intercept Transects

There is a broad similarity across nearly all sites in the change indicated by the ordination of line intercept transects (Figure 6). For all sites except Broken Bottom there is a shift over time from the bottom of the ordination to the top, which according to the overlaid functional group axes could be associated with lower levels of bare substrate, sessile animals and turf and higher levels of canopy. It is important to remember that the ordination is a simplification that seeks to represent many features in two dimensions. Collings (1998) identified the fact that simple comparison of the vectors representing change can be misleading, particularly for sites occupying different regions of the ordination. Thus, having noted the broad similarities in temporal change across sites, it is necessary to investigate this with alternative approaches.

An nMDS ordination was created of the *change* between 2005 and 2007 in each of the functional groups used to describe the LITs (Figure 7). Whilst there were no obvious discrete groups in this ordination, it is apparent that there are a group of sites around the periphery of the ordination which demonstrate substantial change along one or more functional group axes. These are HSI and HSO which are associated with large loss of turf and gain in canopy; NSI which is associated with a gain in turf, but a loss of sessile animal cover; BRB which also experienced a turf gain at the expense of bare substrate; and Semaphore which appears associated with a loss of bare substrate and a gain in red understorey. The other sites are clustered toward the centre of the ordination, indicating that similar, less substantial changes have been evident at these sites.

It is, however, important to remember that position on the ordination is determined by all functional groups. Thus what appears to be a large increase along one axis could represent a large decrease along a directly opposing axis, or a response along several axes at some angle to the original. It is therefore essential to examine the composition of each of the sites to confirm the picture provided by the ordination. It is evident from Figures 8 and 9 that across the period 2005-2007, it is canopy cover and bare substrate which have changed most consistently between sites, with most sites (10 of 14) having increased their level of canopy cover (and of the 4 which lost canopy, none lost more than 5.2% cover), and 13 of 14 sites demonstrating decreased levels of bare substratum, and the 14th demonstrating an increase of only 0.18%.

Taking a longer-term view (11 years), it was possible to ordinate all sites surveyed in the 1996, 1999, 2005 and 2007 surveys, many of which were common to more than a single survey period (Figure 10). This gives, at least for some sites, an indication of the similarity of the sites at different points in time, within the context of the overall spatial variation exhibited by the entire range of sites.

It is evident from this ordination that the relationships between sites at any point in time are generally repeated at other points in time. Semaphore and Broken Bottom have been surveyed since 1996, and have remained relatively consistent, and different from all other sites. Surveyed since 1999, Noarlunga North and South Inside, Noarlunga South Outside, Horseshoe Reef (Inside and Out) are generally relatively consistent and, interestingly, similar to Northern Reef in 2007 (the only time it was sampled). It does appear that Horseshoe Reef Outside and (particularly) Inside are unusual in 2007, containing a greater proportion of red understorey and canopy, and less turf and bare substrate, than seen in previous surveys. Hallett Cove has demonstrated a relatively consistent biota since 1999, and Seacliff has also remained relatively similar over the period it has been surveyed — 2005 and 2007.

Having said that the pattern between sites is relatively consistent, it is important to note that across the years, there has been a general trend for most sites to converge toward the canopy dominated community typified by Second Valley and Moana. Noarlunga North Outside, Aldinga and Southport all showed movement toward this type of community in 2007. Whilst Southport and Noarlunga North Outside were similar by 1999 (the earliest point in time at which they could be compared), Aldinga demonstrated quite a different sort of community in both 1996 and 1999. Whilst other sites may not have reached the same state / composition, there has been a general trend toward this state across these years for most reefs. This can be interpreted as a trend away from bare substrate, animals and turfing algae, toward one dominated by canopy.

Much of the dynamics evident in the ordinations involving all sites at all time periods can be summarised by the ratio of canopy cover to the summed total of bare, turf and animal cover. This ratio was constructed on the basis of the assumptions made consistently throughout the Reef Health program that, within the photic zone, high canopy cover is a sign of a reef in good condition, whilst high cover of animals, bare substrate or turfing algae may be a sign of a degraded reef. The understorey categories were not included in the ratio as their presence does not necessarily indicate either good or poor ecosystem

function. This ratio provides a very useful index in itself of reef health. In every case, the ratio was highest in the most recent (2007) survey, and in all cases where four readings were available (i.e. since 1996), each survey showed an increase in this ratio over the previous one (Figure 11).

Reef Health Indices

Comparison of the results obtained in the surveys of 2007 with those obtained two years previously indicates that the overall status of the reefs has remained relatively consistent, although the status of some reefs has changed. Four reefs have improved (3 from poor to caution and 1 from caution to good), and one (Noarlunga North Outside) has decreased from "Good" in 2005 to "Caution" in 2007 (Figure 12; Table 6). With the exception of the apparent degradation of Noarlunga North Outside, this is consistent with our findings from the ordination of the LIT data.

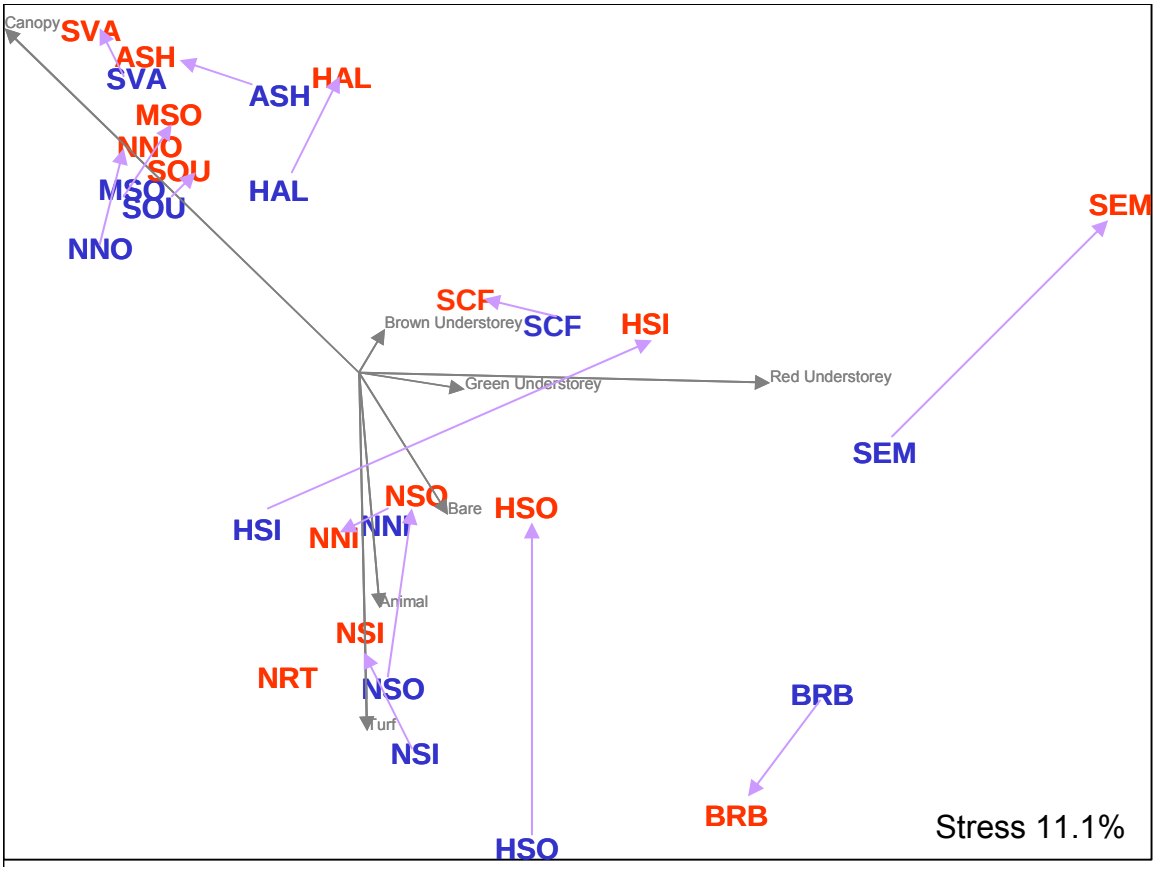

Figure 6. nMDS Ordination showing biotic relationship (i.e. similarity) between sites of the Reef Health program as surveyed in 2005 (in blue) and 2007 (in red). The temporal change is indicated by the purple vector arrow. Abbreviations for sites follow Table 2

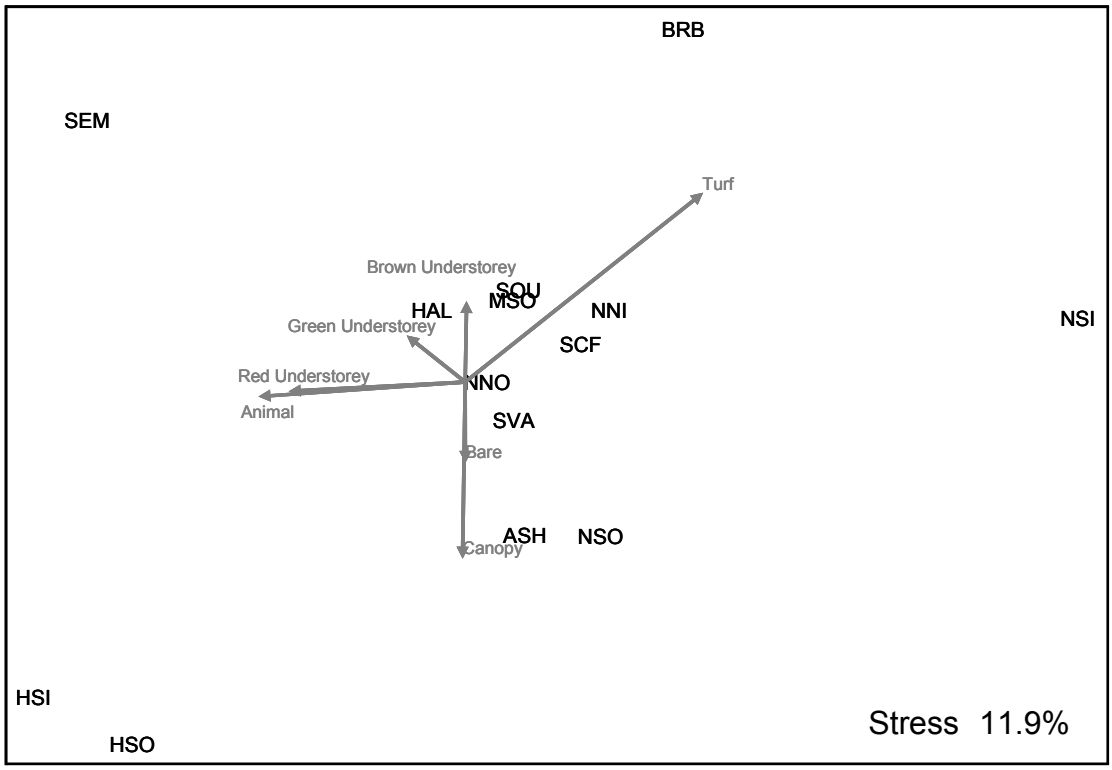

Figure 7. nMDS ordination demonstrating the similarity of change from 2005 to 2007 in percentage cover of reefs of the Reef Health program. Sites which are close on the ordination demonstrate similar biotic change. Abbreviations for sites follow Table 2.

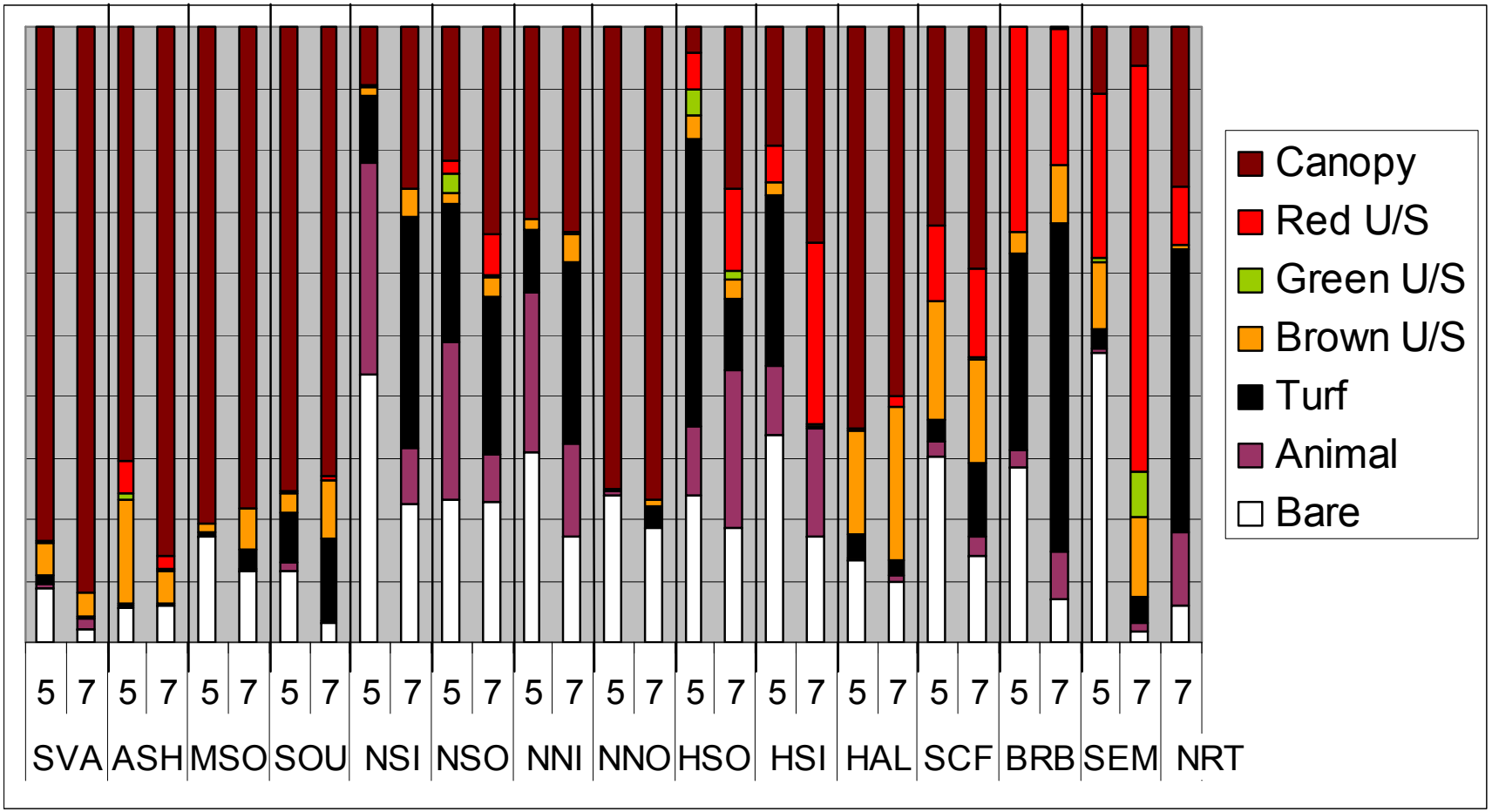

Figure 8. Composition of reefs studied as part of the 2005 and 2007 Reef Health surveys described by functional group cover of LIT. Abbreviations follow Table 2.

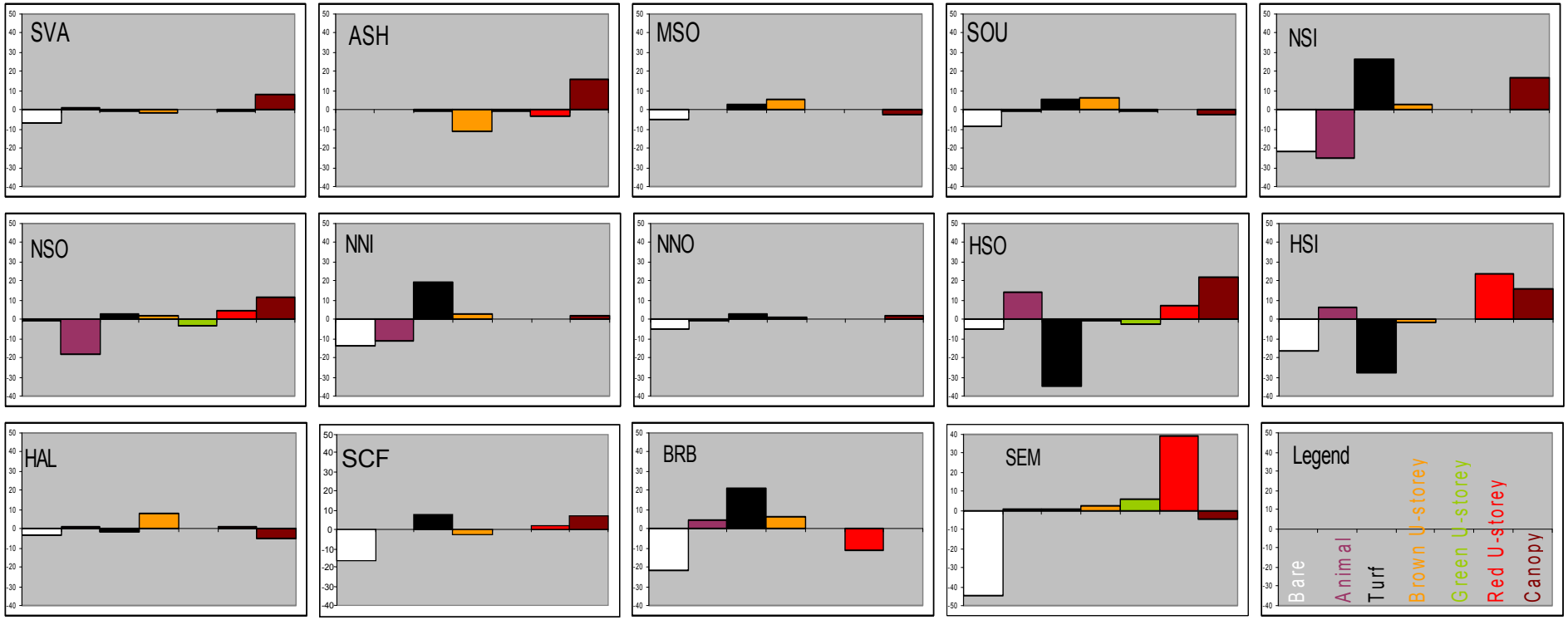

Figure 9. Composition change across all sites common to both the 2005 and 2007 surveys in terms of functional groups along LITs. Legend for X axis is presented at bottom right. Abbreviations for site are as presented in Table 2.

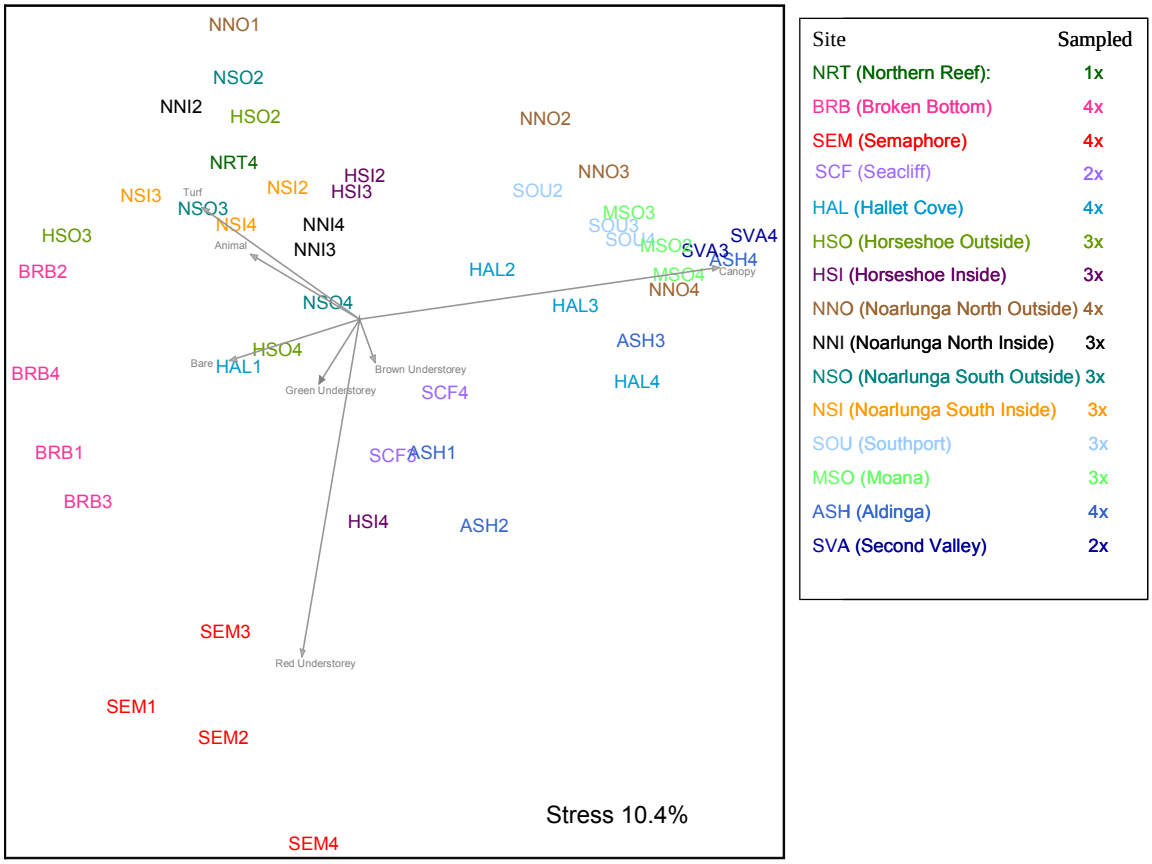

Figure 10. nMDS ordination of all reefs surveyed as part of the Reef Health studies on the Adelaide / Fleurieu coast since 1996. The numeral following the site label indicates year of survey: 1= 1996, 2=1999, 3=2005, 4=2007. The legend indicates the number of years that each site was sampled.

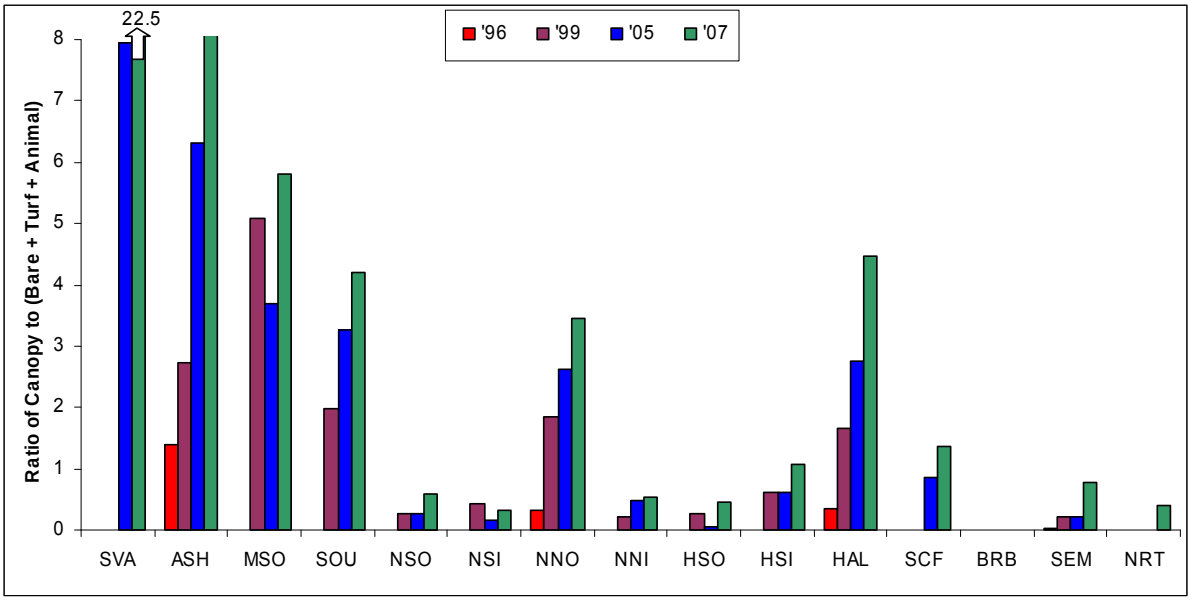

Figure 11. The ratio of canopy cover to combined turf, bare and animal cover at each. Some sites were surveyed on all four occasions, other far less. Site abbreviations as described in Table 2.

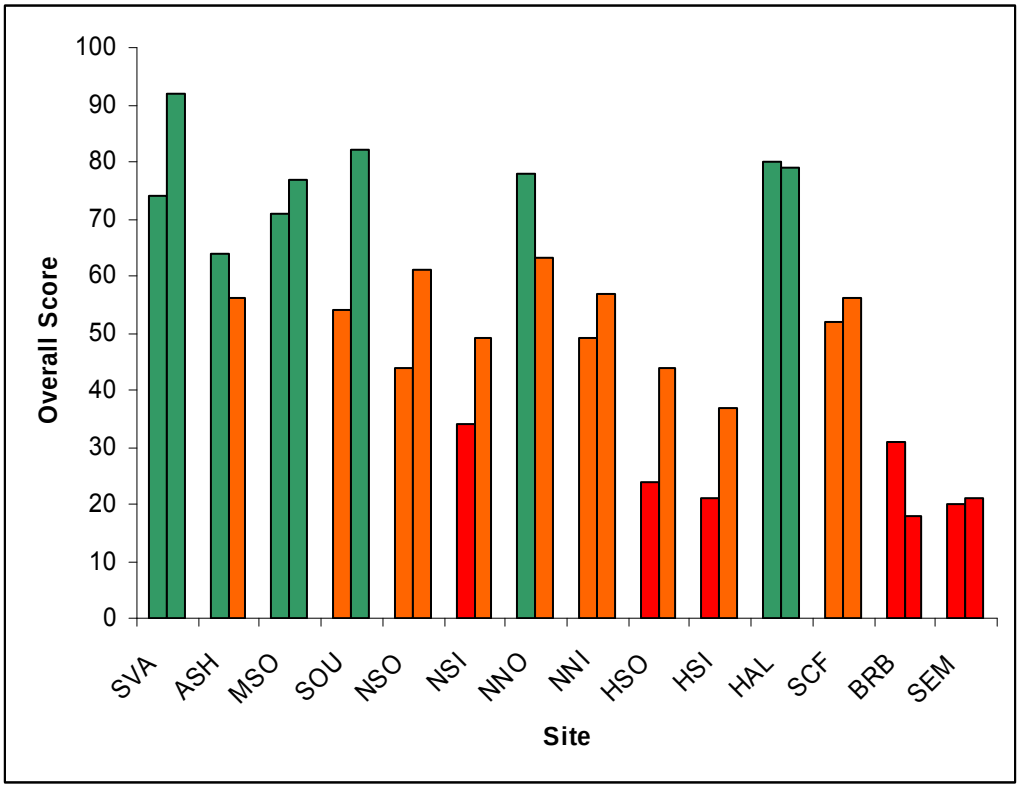

Figure 12. Comparison of overall reef health index of each site between 2005 and 2007. Colours of bars indicate the status of the reef (good, caution or poor; green, orange or red), and the left hand column of each pair represents the 2005 status; the right hand column represents the 2007 status. Site abbreviations are as described in Table 2.

Across all sites there was an average improvement in condition of 7 points (Table 6). On average, the absolute change in overall score (i.e. without regard to direction of change) was 12 points. This varied substantially between sites, from 1 point at Semaphore to 28 points at Southport. Taking direction into account, the greatest decrease in condition was evident at Noarlunga North Outside (-15 points), whilst the greatest increase was seen at Southport (+28 points). These changes should be viewed in the context of the range of scores exhibited across all sites, which was 18 (Broken Bottom) to 92 (Second Valley).

Changes in the individual indices used to compile the overall score were, however, far greater. It was not uncommon for a site to demonstrate changes of more than 60 points in an individual index (Table 6). It is worth noting that many sites demonstrated changes of >50 points for indices involving the abundance of mobile animals (invertebrates or fish). Wrasse abundance changed by more than 50 points at nearly half of all sites. In contrast, canopy cover varied by more than 10 points at only 4 sites and never varied by >42 points. This is also reflected in the average absolute change figures at the bottom of Table 6.

Taking into account the direction of change (improvement or deterioration), it is evident that substantial improvements have been seen, on average across sites, in those indices reflecting site-attached fish abundance, and particularly wrasse abundance (Table 6). Wrasse abundance changed by 39 points (of a possible 100) averaged across all sites. In contrast, macroalgal species richness has decreased markedly.

Table 6. Changes in reef health indices 2005-2007. Green status implies an improvement whilst red implies a decline and black implies no change. Bolded values indicate a change of greater than 50 points in a given index. The use of "*" indicates that a null score was involved. A null score in 2005 is shown by the asterisk preceding the number and vice versa if a null was scored in 2007. An indication of the temporal variability across all sites is provided where a null score is not encountered at any site, both in terms of magnitude (Avg Absolute Change) and including direction (Average Change).

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	Abundance / size of wrasse	Presence of invasive taxa	Evidence of high sedimentation	Richness of mobile invertebrates	Richness of macroalgae	Score	Change in Status
Semaphore	0			0*	5	7	-14	0*	0	-5	-21	1 (20->21)	**Poor->Poor**
Broken Bottom	0	*0			**-61**	0	-7		0*	0	-26	-13 (31->18)	**Poor ->Poor**
Seacliff Reef	23				0	21	0			0	-22	4 (52->56)	**Caution->Caution**
Hallett Cove	-1				**53**	0	27			-15	**-65**	-1 (80->79)	**Good -> Good**
Horseshoe Inside	42			0*	28	0	24		0	5	-7	16 (21->37)	**Poor->Caution**
Horseshoe Outside	16	0*			20	0	24		0*	21	-15	20 (24->44)	**Poor->Caution**
Noarlunga North Inside	-7		44*		0	0	**100**		*0	11	-9	8 (49->57)	**Caution->Caution**
Noarlunga North Outside	0				10	0	0		*0	-26	0	-15 (78->63)	**Good->Caution**
Noarlunga South Inside	10	*2	0*	0*	-7	0	**95**		*0	5	8.5	15 (34->49)	**Poor->Caution**
Noarlunga South Outside	29				**51**	0	5			16	-2	17 (44->61)	**Caution->Caution**
Southport	0				3	36	**66**		0*	10	0	28 (54->82)	**Caution->Good**
Moana Outside	0				0	**-60**	**74**		0*	-47	0	6 (71->77)	**Good -> Good**
Aldinga Shallow	0				49	-28	**-54**			-15	0	-8 (64->56)	**Caution->Caution**
Second Valley	0				0	28	**61**			15	0	18 (74->92)	**Good -> Good**
Average Change	8				10	0	29			-1.8	-11	7	
Avg Absolute Change	9				20	13	39			14	13	12	

3.3 Comparison of scientific and community survey data

A subset of the reefs surveyed during the 2007 Reef Health program was also surveyed by the community-based Reef Watch organisation to allow a comparison of the situation as viewed by professional marine scientists and a community-driven group with tutelage from experts in the fields of taxonomy and sampling methods.

There is a marked similarity between the relationships between sites as determined by the Reef Health (professional) program and Reef Watch (community) program. Whilst sites

do not fall out in exactly the same place (indicating identical composition), ordination of the average composition of the sites common to both sets of surveys, on the basis of functional groups along LITs, identifies a similar pattern in both sets of data (Figure 13). This becomes particularly evident when the ordination is separated to display only Reef Watch sites in one graph and only Reef Health sites in another (Figure 14). In both sets of data, Second Valley is found toward the top of the ordination, Hallett Cove lower and to the left and Seacliff to the bottom left. Most Noarlunga sites then form a group in the bottom right. Only a single Noarlunga site in each dataset is found outside this group, a fact which, in both cases reflects an unusually high preponderance of canopy for this group (see also Figure 15).

Noarlunga North Outside demonstrates a tight clustering of transects (Figure 16), indicating that the average obtained for the whole site is reasonably representative. However, the transects of Noarlunga South Inside are widely scattered, indicating that the biota varies a great deal across the four transects, and consequently, the average, as plotted in Figures 13 and 14, has a great deal of uncertainty associated with it (more than any other site). For this reason, the discrepancy between the situation as described by Reef Health and Reef Watch is unsurprising. The same cannot be said of Noarlunga North Outside, which the Reef Health surveys identified as having a high degree of similarity between transects.

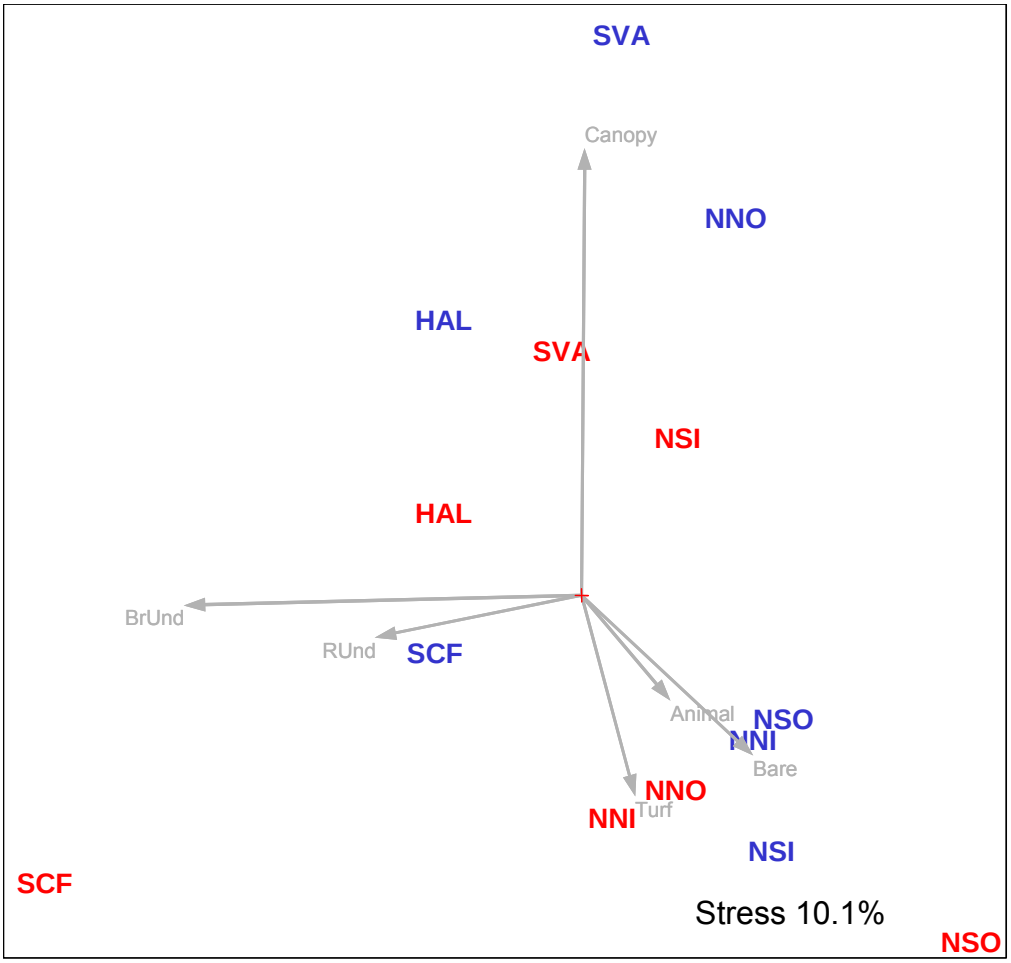

Figure 13. nMDS Ordination showing the biotic relationship (i.e. similarity) between average composition of reefs as surveyed by the Reef Health program (in blue) and by the Reef Watch program (in red). Abbreviations for sites follow Table 2.

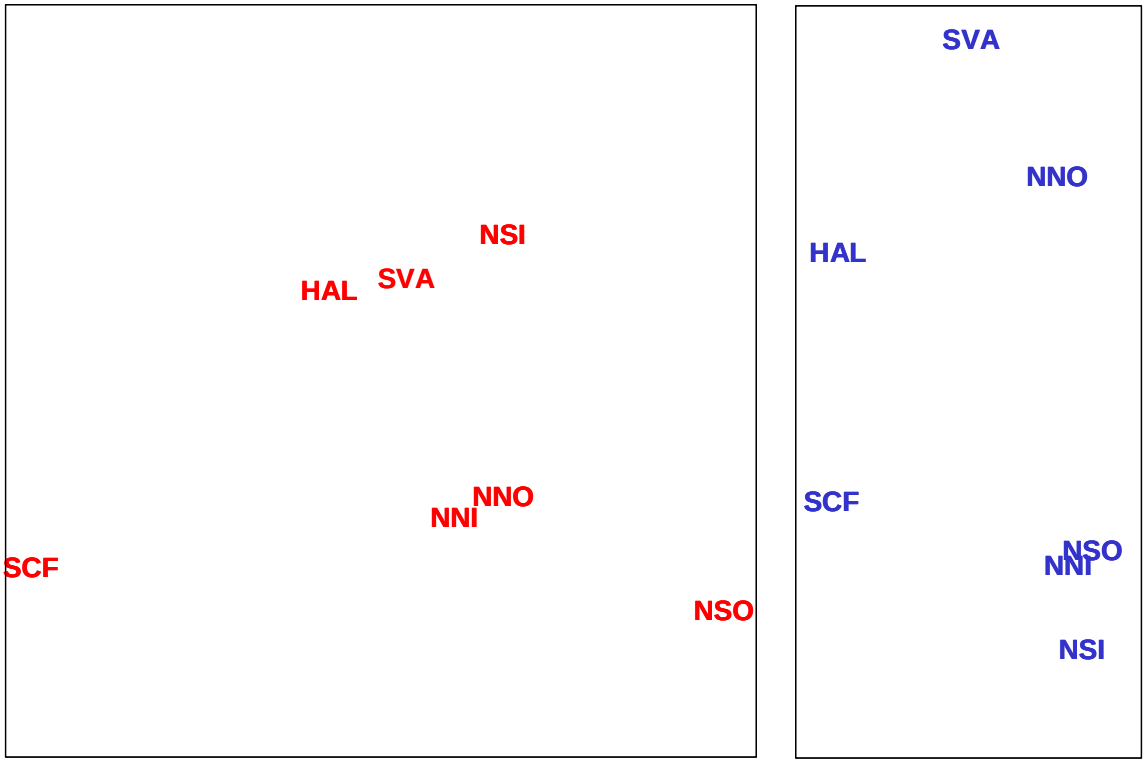

Figure 14. Biotic relationships between sites as demonstrated by Reef Watch (in red) and Reef Health (in blue). This is simply a different visualization of the analysis demonstrated in Figure 13, with the two sets of data separated. It is based on the same analysis and the axes have not been altered in any way. Abbreviations are as described in Table 2.

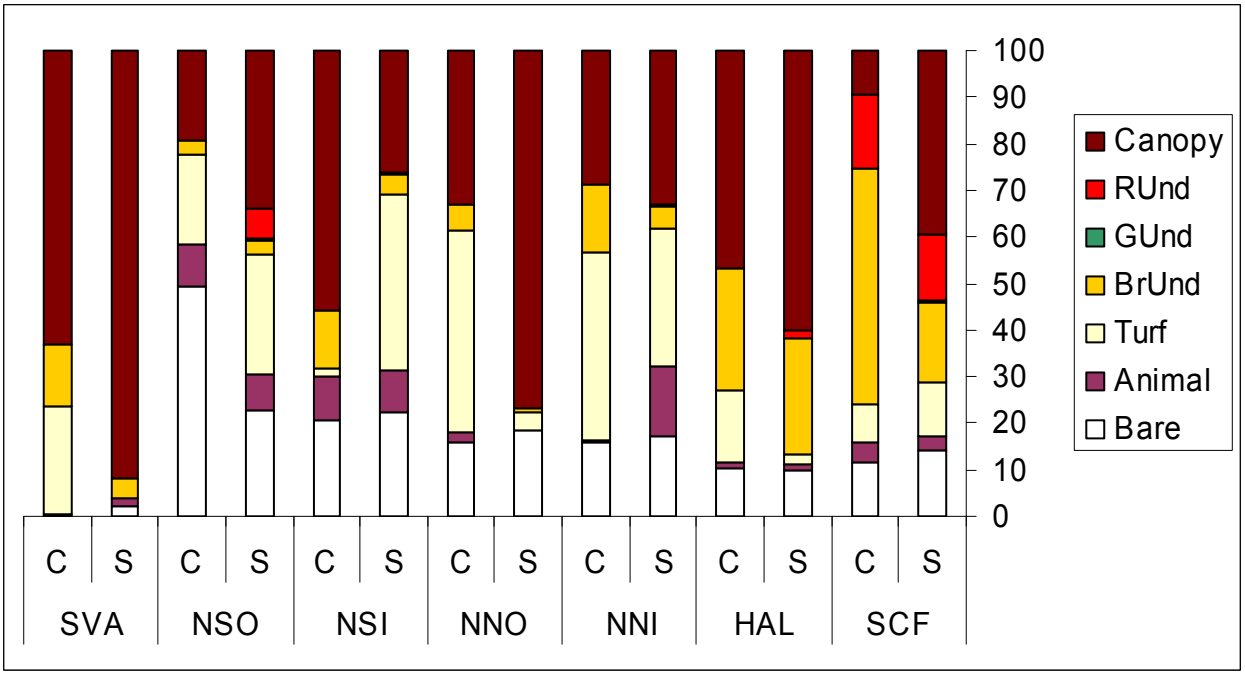

Figure 15. Comparison of the average cover (%) of LITs of each functional group at each site common to both the Reef Watch (C) and Reef Health (S) surveys from both sets of surveys.

In contrast to the similarity noted between the assessments of reefs made by Reef Health and Reef Watch on the basis of the LITs, assessment of the reefs via the use of indices (which take in a greater range of variables than the LITs which form the basis of the ordinations) revealed substantial differences (Table 7). Only three of seven sites were classified with the same grade (Good, Caution or Poor), and on average the scores differed by 25 points. This is a substantial difference considering that across the entire range of sites common to both surveys, the Reef Health scores differed by only 36 points (Reef Watch scores varied by 69 points). The difference between Reef Health and Reef Watch scores varied between 1 (Noarlunga South Inside) and 52 (Noarlunga North Outside).

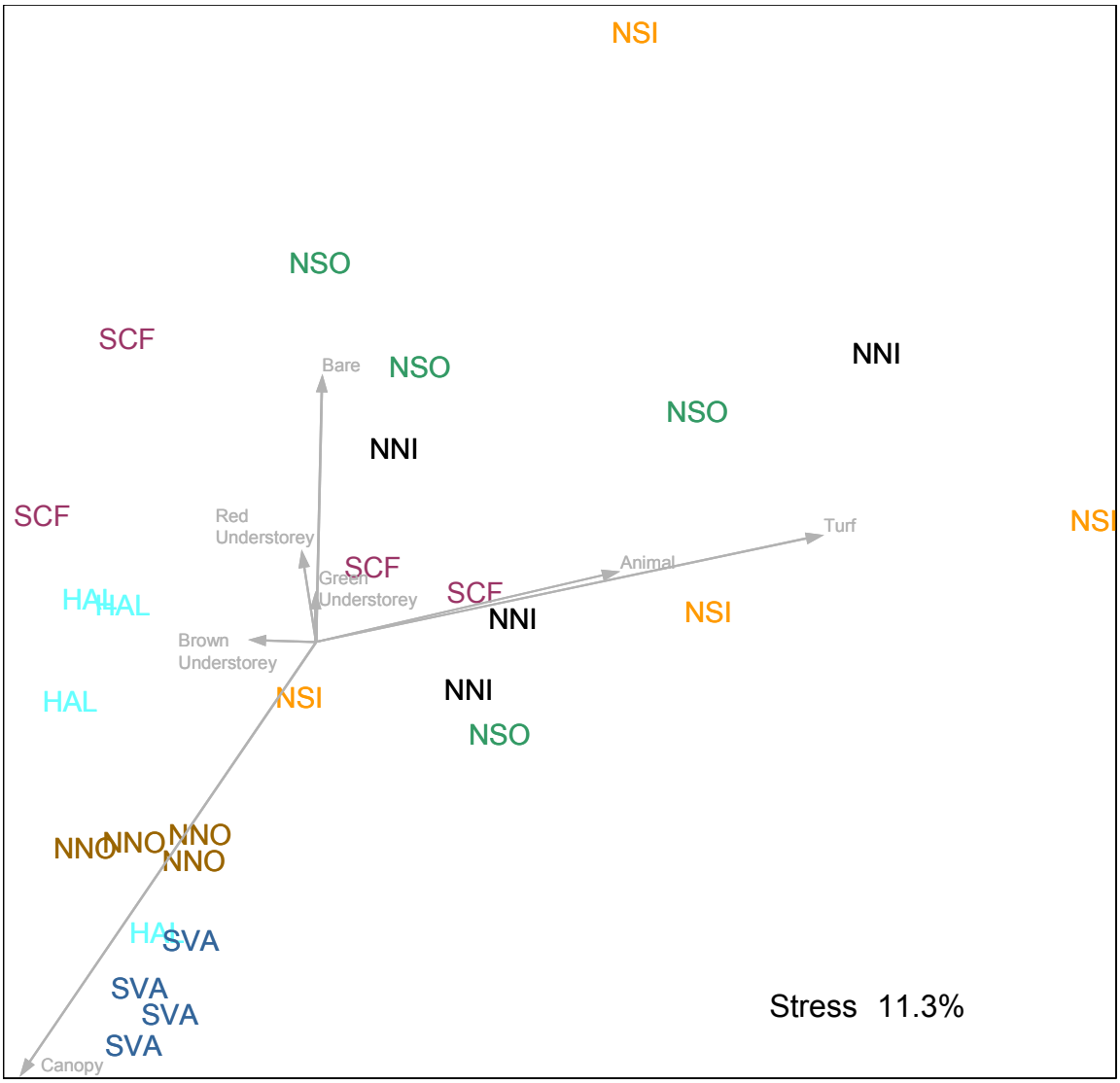

Figure 16. nMDS ordination showing the biotic relationship (i.e. similarity) between composition of transects of sites common to both the Reef Health and Reef Watch programs utilising the Reef Health LIT data. Abbreviations for sites follow Table 2.

Table 7. Reef Health index and status as assessed by Reef Health and Reef Watch programs. "Traffic light" indicators are provided to grade reefs according to a 3-point scale of "Good" (>65), "Caution Required" (35-65), and "Poor" (<35) and colour coded accordingly. The individual indices used to calculate the overall score are provided also. Note that Reef Health scores are not identical to those provided earlier as less attributes have been measured in order to make Reef Health and Reef Watch comparable (macroalgal richness and high sedimentation have been removed).

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	Richness of mobile invertebrates	Overall Score	Status
Seacliff RH	48				100	42	39		28	**51**	Caution
Seacliff RW	0				18	0	26		5	**10**	Poor
Hallett Cove RH	99				100	100	100		34	**87**	Good
Hallett Cove RW	67				100	100	100		28	**79**	Good
Noarlunga North Out RH	100				100	100	0		44	**69**	Good
Noarlunga North Out RW	33	0			11	42	0		13	**17**	Poor
Noarlunga North In RH	33				100	100	100		31	**73**	Good
Noarlunga North In RW	22	0			100	100	42		23	**48**	Caution
Noarlunga South Out RH	33				100	100	49		47	**66**	Good
Noarlunga South Out RW	0			0	11	77	100		21	**35**	Caution
Noarlunga South In RH	10	2			59	100	95		47	**52**	Caution
Noarlunga South In RW	90				83	63	0		18	**51**	Caution
Second Valley RH	100				100	63	100		42	**81**	Good
Second Valley RW	100				8	94	100		31	**67**	Good

4 Discussion

4.1 Current status of Adelaide's reefs

There is a broad delineation into two groups of a "southern" set of canopy-dominated sites and a "central" group of sites dominated by a more diverse array of functional groups. It might reasonably be assumed that this represents a natural gradient, probably associated with a decrease in water movement from south to north (although a low level pollution gradient is possible). However, Broken Bottom and Semaphore stand out from this general pattern, supporting far less canopy than is evident at other "northern" sites, such as Seacliff and Northern Reef, which are more similar to the central sites.

In past surveys (Cheshire et al 1998, Cheshire and Westphalen 2000, Turner et al. 2007), Semaphore and Broken Bottom demonstrated a biota that was severely depauperate in canopy algae, and it was postulated that this was a result of the polluting influence of metropolitan Adelaide. However, these reefs represented the northern extreme of a south – north geographic gradient which could be naturally associated with a decrease in canopy algae. Thus, the evidence for an urban influence was weak as it was confounded by a natural gradient. However, the inclusion of the Northern Reef, north of these two metropolitan reefs, has allowed us to see what a relatively unimpacted northern reef should look like. Whilst this reef has less canopy than the other unimpacted sites, defining a northern extreme of the natural south – north gradient, it makes it clear that Broken Bottom and Semaphore stand out from this. Thus, we now have much better evidence for an anthropogenic effect on these reefs.

As a caveat, it should be acknowledged that the two impacted reefs, at depths of 8 – 10 m, are relatively deep. However, they are within a couple of metres of the depths of most other reefs and Seacliff Reef, at 12 m depth is deeper again. Thus, it is considered unlikely that depth is the cause of the apparently degraded state of Broken Bottom and Semaphore reefs.

It is evident from the ordination of each of the LITs from which the average is derived, that there is considerable small-scale variability (Figure 5: represented by a wide scattering of the four transects representing a single site). Indeed it is clear that whilst reasonably good definition is evident between the broad groups, the sites within a group are poorly defined. The exceptions to this are the transects of Broken Bottom and Semaphore, which form well defined groups representing the respective sites. Whilst some other sites demonstrated less variability between transects (in particular the southern sites), there

was little difference between the sites, so despite the tight grouping of transects, there was still overlap between similar sites. This strengthens the argument that the two purportedly impacted sites are indeed quite different to the remainder of the sites.

It is worthy of note that whilst Broken Bottom and Semaphore are characterised by an extremely depauperate canopy, they are not particularly similar to one another; Broken Bottom is characterised by a dominance of turfing algae, whilst red algal understorey dominates the reef at Semaphore.

There is some similarity between the health indices devised by Turner et al. (2007) and the general pattern noted in the ordination of the LIT data. With a few exceptions, the southern (canopy dominated) group were classified as being in good condition; the central (moderate canopy) group were classified as "caution" and the two putatively impacted sites were both classified as "poor". The three exceptions were Aldinga, which was classified as "caution" rather than "good", Noarlunga North Outside which was also classified as "caution" rather than "good", and Northern Reef, which was classified as "poor" rather than "caution" which might have been expected on the basis of its position in the LIT ordination.

Noarlunga North Outside only just fails (by 2%) to achieve a "good" rating and it is evident that its unexpectedly low rating was caused by very low scores for the individual indices for mean abundance / size of wrasse and high sedimentation (score of 0 in both instances). It could be argued that high sedimentation is not so much an indicator of health as a physical characteristic of the site. However, it was retained for the current analysis on the basis of consistency with the previous study and because turfing algae are known to trap sediment; thus it could represent an indicator of poor condition in addition to a forcing. Nevertheless, if it were not for this contentious index, this site would have received a good classification. Aldinga receives a low score on the basis of a low score for wrasse and mobile invertebrate predators. The fact that these are mobile individuals (particularly for the fish) may be an indication of a temporally variable situation, although it should be noted that this site had poor mobile predator abundance in 2005 also (Turner et al. 2007). Also, Shepherd and Baker (2008) demonstrated a tight relationship between relief and abundance at this location, indicating the possibility that a site with low relief was chosen. Alternatively, sampling effort may not be great enough for these measures, and more sampling may be required at a single point in time.

Northern Reef achieves a poor score on the basis of a variety of features – canopy macroalgal cover, turfing algal cover, abundance of both site attached fish and wrasse, and evidence of high sedimentation. At a site defined by the LIT data as being of average composition, this finding probably says more about the suitability of the indices than about the real "health" of the site. As mentioned previously, the sedimentation index is dubious, as it may really be more of an indicator of physical conditions than of biotic response. Fish, being mobile, are likely to vary considerably in abundance over short timescales not relevant to habitat quality. Thus it is possible that indexes involving mobile fauna are not accurate enough because of a lack of temporal replication. Also, it is widely regarded that blue throated wrasse are unlikely to occur naturally as far north as Northern Reef anyway. Finally, areal cover of macroalgae demonstrates an interesting point with regard to scaling. A comparison of Northern Reef indicates that it has approximately 25% cover of canopy macroalgae. This provides it with the very low index score of 7 (of 100). By comparison, Hallett Cove has approximately twice the cover, but an index score of 99 (of 100), so a score 14 times higher is registered for a cover only twice as great. Conversely, Broken Bottom had almost zero canopy (0.15%) and yet scored only 7 points less in terms of canopy. The implications of this discrepancy reflect on the appropriateness of the index and will be taken up in Section 4.4, comparing index calculation and ordinations as methods of assessing reef condition.

To summarise the biological patterns demonstrated along the eastern coast of Fleurieu Peninsula, there is a natural south – north gradient which is largely reflected in a decrease in macroalgal canopy. This decrease is complemented by an increase in the cover of other functional groups, predominantly red and brown understorey and turfing algae. Numerous authors have commented on the complementary nature of canopy and turfing species (e.g. Gorgula and Connell 2004, Wernberg 2006). Superimposed on this natural gradient is an apparently anthropogenic influence which causes Broken Bottom and Semaphore to stand out with an unexpectedly depauperate canopy. The natural south – north gradient probably reflects a trend of decreasing wave force and indicates a tendency for higher wave force regions to support greater canopy cover.

On the basis of demonstrated effects elsewhere, Turner et al. (2007) propose a variety of anthropogenic causes for the unexpectedly poor condition of metropolitan reefs. These include high sedimentation rates, turbidity and eutrophication associated with urban runoff, industrial sources and wastewater treatment plants (e.g. Gorgula and Connell 2004, Troell et al. 2005), and fishing pressure (e.g. Jackson et al. 2001, Knowlton 2004,

Shepherd and Baker 2008). Interestingly, recent quantification of sedimentation (Fernandes et al. 2008) identified the reefs near the Onkaparinga River (particularly Noarlunga and Southport) as suffering far greater sedimentation than the more northerly reefs of Broken Bottom and Semaphore. This was attributed to minimal cross-shore advection and the greater distance from shore of the latter reefs. They did observe, however, that while sedimentation was lower at Broken Bottom and Semaphore, the nature of the sediments was distinct. Specifically, sediments at Broken Bottom and Semaphore were finer, had greater nitrogen content and N:P ratios, and high $\delta^{15}N$ signatures, all of which suggest a significant effect of wastewater / industrial effluent in this region.

Whilst Turner et al. (2007) did not pinpoint the causal mechanisms, it is worth noting that they were not alone in voicing concern about the degraded state of ecosystems within Gulf St Vincent. Similar concerns have been raised by numerous authors with regard to both reef communities (e.g. Steffensen et al. 1989, Cheshire et al. 1998, Cheshire and Westphalen 2000, Turner and Cheshire 2002; Gorgula and Connell 2004) and seagrass meadows (e.g. Neverauskas 1987, Seddon 2002, Westphalen et al. 2005, Bryars et al. 2006, Collings et al. 2006a & 2006b, Fox et al. 2007). This study has not further distinguished the cause of the problems, but it has, by repeating the surveys in different years, made it clear that the patterns are long term and not transient (Figure 11). Furthermore, through the inclusion of a reef to the north of those suspected to be impacted (Northern Reef), it has allowed a picture to be obtained of what a relatively unimpacted northern reef should look like, and highlight the fact that Semaphore and Broken Bottom represent unusually impacted sites, rather than simply an extreme of the natural south - north gradient demonstrated in the biota of the reefs.

4.2 Temporal change in reefal communities

There is a striking similarity between the pattern demonstrated by the reefal communities surveyed in this study and the same communities two years previously (see Turner et al. 2007). In both instances there is a distinct group of southern sites, comprising Second Valley, Aldinga, Moana, Noarlunga North Outside Southport and Hallett Cove. A separate group, consisting of central reefs is defined between these canopy dominated southern reefs and the quite depauperate northern reefs (excluding Northern Reef which is not considered metropolitan, and was not surveyed in 2005). This pattern is consistent across the two years with few exceptions. The most obvious exception is Horseshoe

Reef, the condition of which, in 2005 raised concerns (Turner et al. 2007). Indeed, the outer side of the reef (HSO) appeared similar to the more depauperate sites in 2005 (Figure 6). In 2007 this reef appeared more similar to the central sites, indicating an improvement in condition. However, concerns do exist that this reef varies substantially on small scales and it is not possible to rule out spatial confounding.

Despite the similarity in general pattern across years, it is clear that there has been a broad trend towards increasing macroalgal canopy and a decrease in the amount of turfing algae, animals and bare substrate. Those sites with high canopy cover tended to change least and those with lower cover and more representation from other functional groups changed the most (Figure 10). This may be because, if the composition of the sites is heading towards a canopy dominated community (such as characterises Second Valley) then there is less room for change in those sites which originally supported high canopy.

The availability of data from surveys of a range of sites in 1996 and 1999, in addition to 2005 and 2007 has allowed an assessment of longer-term change (although it has to be noted that the 1999 and 1996 surveys were conducted 4 months earlier, in November). In some cases this extends to 11 years. Such a long-term dataset is invaluable considering the relatively slow speed often evident in marine ecosystem recovery (Knowlton 2004, Wernberg 2006). Whilst the relativity of sites to each other has remained fairly consistent, there has been a clear general trend in the less impacted sites to become more canopy-dominated, with less bare substratum and cover by animals and turfing algae (Figure 11). A number of reefs have, over time, developed very similar, canopy-dominated communities (Figure 10). The tendency towards this canopy dominated state has occurred at sites demonstrating a wide range of compositions. Some sites (Second Valley, Moana, Southport) were always relatively canopy-dominated and have changed only minimally. However, several sites (Noarlunga North Outside, Hallett Cove and Aldinga) have evolved to this state from compositions that were markedly different to the current (2007) state, and from each other (Figure 10). Interestingly, several of the other sites (Seacliff, Noarlunga North Inside, Noarlunga South Outside, Horseshoe Reef Inside and Outside), despite not converging all the way to the same composition of biota demonstrated by the southern (canopy dominated) sites, have shown a trend in this general direction — towards greater canopy and less turfing algae. While there is some evidence for this trend at Broken Bottom and Semaphore, it is conflicting and therefore weaker. Certainly, the amount of bare space has reduced, but

this has been associated with only a miniscule rise in canopy at Broken Bottom and a slight *decrease* at Semaphore. Furthermore, the decrease in bare space at Broken Bottom has been offset by an increase in turfing algae. Thus whilst the pattern is relatively consistent, it is least clear at the two reefs closest to metropolitan Adelaide.

The changes evident across time in the LIT data are reflected to some degree in the fact that there was a 7 point increase in overall reef health across all sites. Six sites displayed negligible (<10 point) change across 2005-07, two decreased by >10 points and six increased by >10 points. These changes were driven primarily by an increase in abundance of wrasse, abundance of site-attached fish and areal cover of canopy. There was a decrease in macroalgal richness, but this is probably the result of a more intensive taxonomic search in the 2005 survey, using voucher specimens, whilst in 2007 voucher specimens were not obtained, and many species were identified only to higher order taxonomic groups (e.g. to the subgenus level of *Sargassum*, rather than to species level). Furthermore, as invertebrate and macroalgal quadrats were not carried out in 2007 (as was the case in 2005), the lower sampling effort may be reflected in lower species richness. Macroalgal richness in particular is likely to be affected because the macroalgal quadrats, absent in 2007, actually targeted understorey species, often missed by LITs. Thus the apparent increase in overall health is a conservative estimate, as similar levels of taxonomic scrutiny and sampling effort may have improved the macroalgal richness score in 2007.

It is instructive to identify the individual indices that contributed substantially to the large changes in the overall health index seen at some sites. Noarlunga North Outside, which decreased by 15 points overall, was influenced heavily by achieving a zero for sedimentation in 2007, whilst it was null in 2005. In effect, this is a decrease of 78 points for this character, which, as mentioned earlier is dubious as an indicator of biotic health. Sedimentation was also involved in the increase noted at Horseshoe Reef Outside, as was a decrease in turf to the point where it was considered "null" rather than 0 as an index. All other sites demonstrating large changes (>10 points) in overall score (Second Valley, Southport, Noarlunga South Outside, Noarlunga South Inside and Broken Bottom) did so in large part because of large changes (>50 points) in the individual indices for either the abundance of site-attached fish or the abundance of wrasse. These mobile fauna are likely to be temporally variable, and as such, some of these large swings could be due simply to the particular time that was sampled. There is ample evidence that the sighting of even site-attached fish is likely to be variable in time, depending on species, food

abundance (Shepherd et al. 2008), age (Shepherd 2006), and time of day (Shepherd and Clarkson 2001). Given the single point in time used to survey a reef, its representativeness of the reef state for that particular year is unknown, and according to literature evidence, unlikely to be high. Thus changes in fish-related indices across two years are likely to be confounded by the lack of temporal replication within the year. Furthermore, the adequacy of sampling effort for even one point in time remains an unknown.

Despite the misgivings above, because of the wide range of indices and sites being utilised in the model, there is still justification for accepting the overall trend toward better "health" of the reefs in general, although individual sites may be modelled less accurately.

Previous reports comparing the state of the reefs across time (Cheshire and Westphalen 2000 for the 1999 surveys, Turner et al. 2007 for the 2005 surveys) have admitted that apparent temporal change may actually be the result of small-scale (10s of metres) differences in location. Whilst the same GPS locations were used for position in 2005 and 2007 because measuring temporal change was a stated aim, to some degree the same caveat must be made here as permanent markers were not left after 2005. Despite the "noise" introduced by this uncertainty, it is unlikely to have caused the systematic bias required to produce the uniform response seen across the entire suite of sites of this study, and therefore a real trend toward improved reef health is considered likely.

Whilst there have been a range of studies which have demonstrated change in reefal communities as a result of anthropogenic intervention such as the introduction of marine reserves (e.g. Edgar and Barrett 1997, Babcock 1999, Parsons et al. 2004) or removal of introduced species (e.g. Piazzi and Ceccherelli 2006), this study is unusual in that it documents positive change without the rationale of a specific pulse anthropogenic event. Nevertheless, it is evident that over the course of more than a decade, a relatively consistent change in the composition of the reefs has been evident. Such a positive change may reflect a successional change toward a locally stable state after a disturbance (e.g. Leinaas and Christie 1996, Choi et al. 2002, although Platt and Connell 2003 indicate this is rare) or it may be a direct response to a changing physicochemical environment.

Data from the National Tidal Centre indicate that there was no dramatic sea surface temperature event in the period immediately prior to 1996 which might have acted as a disturbance (Figure 17). Rather, maximal temperatures in these years seem no different

to the years following. Similarly, there has been no consistent trend in temperatures across the period of study (1996-2007). This would indicate that sea temperatures were not responsible for either a disturbance or a gradual change since this period. The possibility remains that an event occurred before the beginning of sea temperature monitoring in June 1992, the effects of which are still being felt.

It is possible that large-scale storm events before the first survey may have caused a severe disturbance to the community (e.g. Renaud et al. 1997, Underwood 1998). Turner (2004) demonstrated, using a relative wave height model based on windspeed, direction and fetch, that in 1994 there was a period of particularly high relative wave exposure, which may have led to physical disturbance. However, whether such a disturbance would have been responsible for damage to the community from which it takes up to 11 years to recover is debatable.

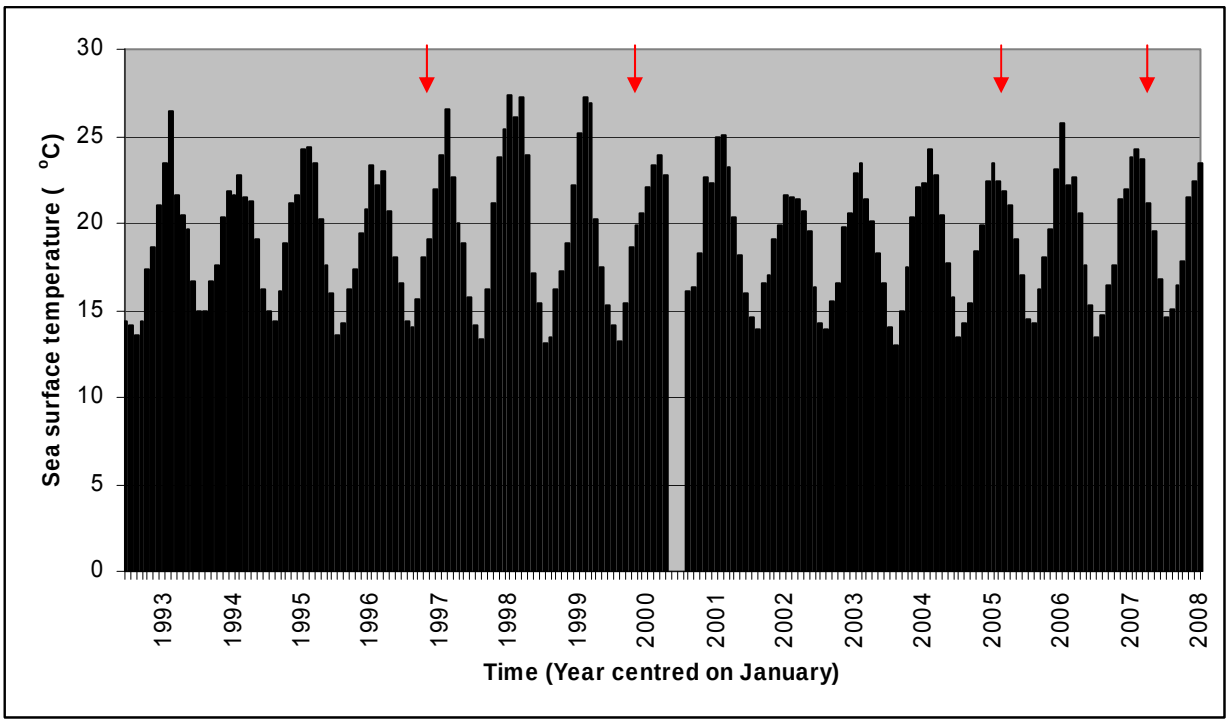

Figure 17. Maximum monthly sea surface temperature measured at Port Stanvac across the period 1992 to 2008. Data provided by the National Tidal Centre. Red arrows indicate surveys.

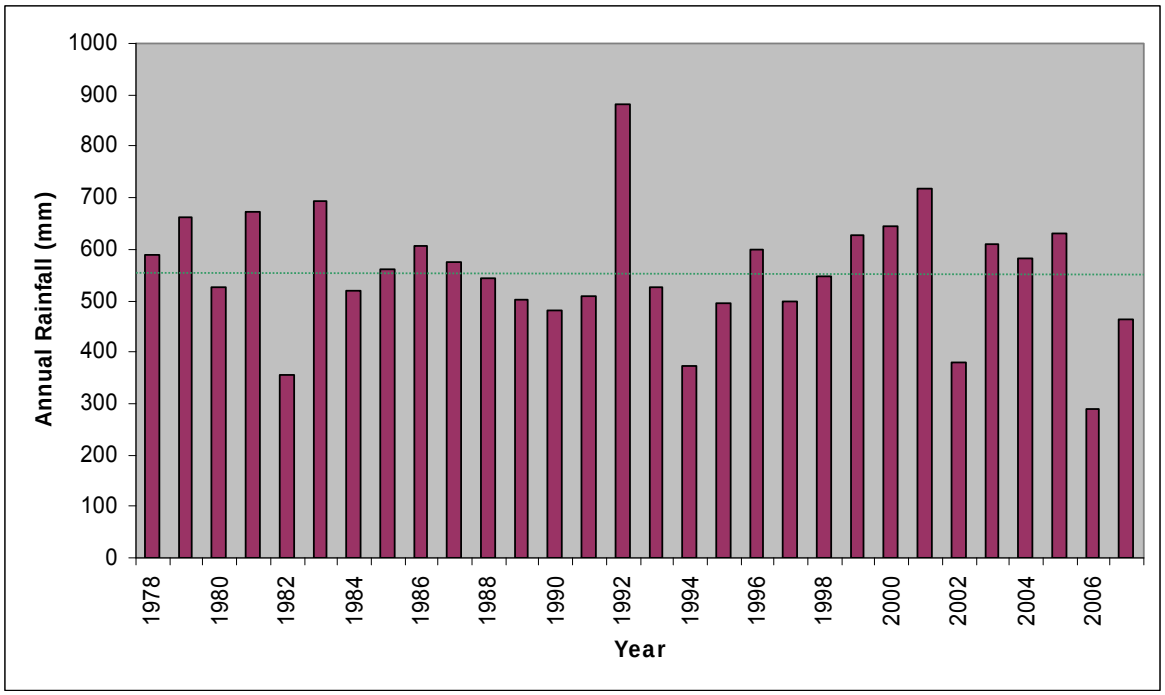

Figure 18. Annual rainfall measured at Kent Town (Adelaide) by the Bureau of Meteorology. The green line represents average rainfall since 1978.

Also worthy of consideration is a generally decreased runoff of stormwater due to the dry conditions of the past 10 years (Figure 18). This might be associated with a decreased sediment load and subsequently less sedimentation on the substrate and turbidity in the water column. These factors have been associated with poor environmental condition (e.g. Gorgula and Connell 2004, Troell et al. 2005). If it is assumed that run-off (and consequently sediment load) are related to rainfall, then the data do not support the notion of a decreased sediment load since 1996. Annual rainfall in the years following 1996 has been approximately the same as in previous years (Figure 18). Of the 11 years, six have been above the average (since 1978) of 551mm. Thus, in reality, the period has not been associated with particularly low rainfall. However, there was one year of extremely high rainfall (1992), and it is possible that this represents a disturbance event. The likelihood of this is unknown.

Another source of sedimentation and turbidity was the dredging events associated with the beach sand replenishment program until 1997 (Cheshire et al. 1999). Since then dredging has been restricted to the maintenance (or deepening) of boating channels. Thus it is quite possible that the higher levels of dredging in the mid 1990s acted as a disturbance, from which the reefal communities are recovering. However, no attempt has been made to quantify the changes in turbidity associated with the cessation of these

operations, nor to assess the relative scale of other programs such as boat channel maintenance or the deepening of the Outer Harbour Swing Basin. Nevertheless, the potential effects of the sand replenishment dredging were raised in earlier reports, and the current work does not produce contradictory evidence.

Whilst Cheshire and Westphalen (2000) considered sedimentation to be the likely cause of the degraded state of some of these reefs, they acknowledged eutrophication as another important possibility. In light of the findings of the Adelaide Coastal Waters Study (Bryars et al. 2006, Collings et al. 2006a, Fox et al. 2007), which experimentally tested for and discovered significant negative effects on seagrasses of nutrients at concentrations similar to that found off the Adelaide coast, this now has to be taken quite seriously. Thus, another possibility which bears consideration is that eutrophication has decreased over the period since the 1990s. Gorgula and Connell (2004) demonstrated that an increase in nutrient levels led to an increase in turfing algal species at the expense of those species which typically constitute the canopy. Furthermore, this study was conducted in a nearby area, at West Island on southern Fleurieu Peninsula. Thus, many of the same species are likely to be involved and responses may be similar.

Data collation and modelling by Wilkinson et al. (2005 and 2006) indicates that since the mid 1990s, total nitrogen input from metropolitan wastewater treatment plants (Bolivar, Glenelg, Christies Beach) has reduced markedly, from 2573 tonnes yr^{-1} to 680 tonnes yr^{-1}. This represents a reduction of 72% at Bolivar, 37% at Glenelg and 39% at Christies Beach. SA Water estimates that across this time, nitrogen load has decreased from approximately 2970 tonnes yr^{-1} to under 1000 tonnes yr^{-1} (Figure 19). Phosphorus discharge has reduced over this period by 48%, 70% and 23% at Bolivar, Glenelg and Christies Beach respectively. Whilst causation has not been proven, we have demonstrated a potentially important correlation across time between anthropogenic nutrient discharge and the status of Adelaide's reefs.

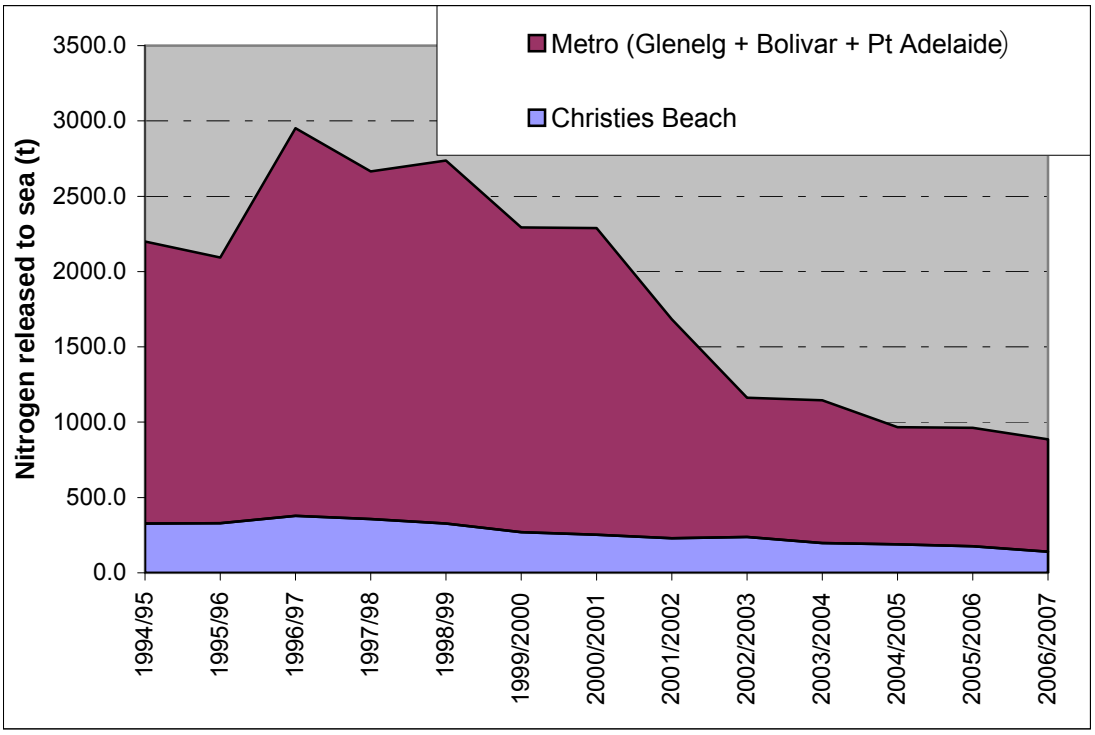

Figure 19. Total nitrogen discharged annually by the wastewater treatment plants of metropolitan Adelaide. Data provided by SA Water.

Stronger evidence of this link may have been provided if ambient water nutrient levels reflected the decrease in wastewater inputs. Whilst data from the South Australian Environmental Protection Agency indicates that this is not the case (Figure 20), this is not altogether unexpected, as ambient water nutrient levels reflect the status after a wide range of processes, including uptake, have occurred. Ambient levels are thus a reflection of what is *left* of the inputs after they have been acted upon by a range of complicated processes.

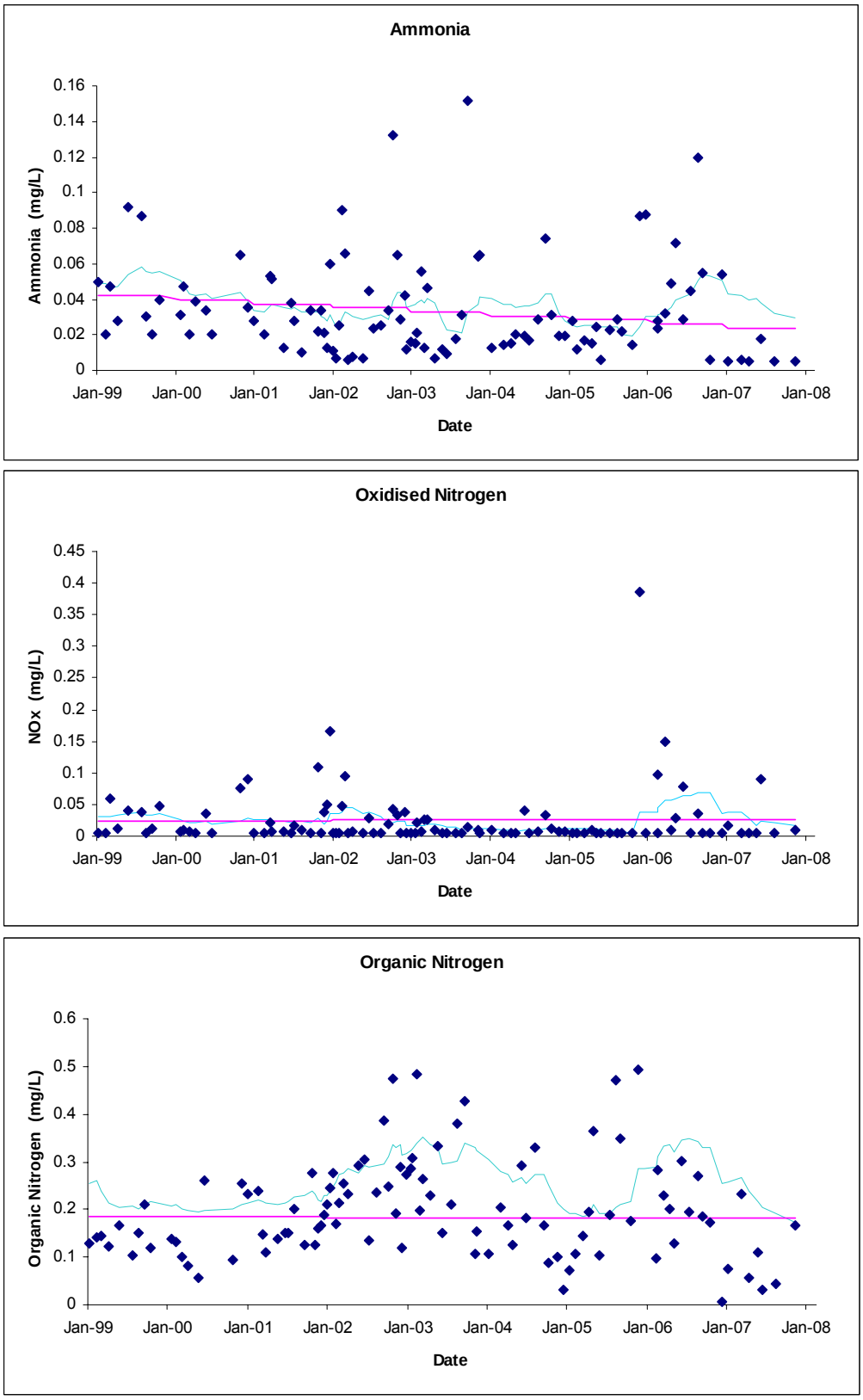

Figure 20. Ambient concentrations of nitrogen as ammonia, oxidised nitrogen and organic nitrogen off Grange jetty from 1999 to 2007. The blue line represents a 12 point running average for smoothing purposes. The pink line represents a line of best fit of the annual averages. In no case did this represent a significant linear relationship (P= 0.15, 0.85, and 0.87, R2= 0.267, 0.005 and 0.0003 respectively). Data from S.A. EPA. (http://www.epa.sa.gov.au/nrm_map_mb.html).

Notwithstanding the possibility that sedimentation has had an effect on the status of the reefs, this correlation between reef health and wastewater inputs is a heartening sign that improvements to wastewater treatment plants may be having positive effects on the biota. This finding is in keeping with the experimental observations of Gorgula and Connell (2004) who demonstrated independent positive effects of sedimentation and nutrients on turfing algae. It does not indicate that improvements to wastewater treatment should cease, nor that there is room for increased dredging activity, as there are reefs on the Adelaide metropolitan coast, which are still clearly degraded. Rather it is a sign that reef degradation may have been caused by eutrophication and/or sedimentation, that improvements to wastewater discharge and/or dredging activity can reverse the situation, and that future improvements are likely to have a positive impact on Adelaide's reef ecosystems.

4.3 Efficacy of community-based monitoring

Turner et al. (2006) identified the potential for community-based organisations to play an important role in monitoring the status of reef ecosystems. Such a role has been effectively played in monitoring of the physical environment (Devlin et al. 2001, Leys et al. 2001, Nicolson et al. 2003, Becker et al. 2005) and of flora and fauna (e.g. Owens 2000, Hartup et al. 2001, Greenwood 2003, Stewart-Koster et al. 2003, Evans and Hammond 2004, Sadlier et al. 2004, Berkes et al. 2007).

Community-based monitoring initiatives are increasingly being applied to aquatic and coastal areas (Cuthill 2000). In aquatic habitats, both the flora and fauna have been studied, as well as water quality (Fleming and Henkel 2001, Fore et al. 2001, Engel and Voshell 2002, Sharpe and Conrad 2006) and some initiatives, such as Waterwatch, operate across Australia. Estuarine community monitoring programs (Arundel and Fairweather 2002) have been put into place, as have marine monitoring, both in Australia (Hodgson 1999, Barrett et al. 2002, Wheeler 2003) and throughout the world (e.g. Davies et al. 2001, Pattengill-Semmens and Semmens 2003, Whitelaw et al. 2003, Goffredo et al 2004).

In particular, Turner et al. (2006) recognized that the Reef Watch Community Environmental Monitoring Program, established in 1997, represented an ideal vehicle for

ongoing monitoring of South Australian reefal ecosystems. Based at the Conservation Council of South Australia, Reef Watch has carried out more than 800 surveys at a variety of locations across South Australia. Reef Watch operate a training system to teach divers skills in identification of fish, invertebrates and algae, as well as sampling technique. Only after this training are divers eligible to participate in Reef Watch surveys. Maintenance of skills is aided by annual identification workshops facilitated by scientific experts and a variety of on-line quizzes and tutorials. Briefings by instructors before dives are also of assistance.

By harnessing the willingness of volunteers to participate in this activity, an enormous amount of information can be obtained. Conversely, both Turner et al (2006) and Bischoff (1997) point out that for some, concerns exist that the data are likely to be of a poor quality, and therefore of little use, or that the time spent teaching volunteers would be better spent actually collecting data. However, after the training provided by instructors and scientific experts, it was concluded by Turner et al. (2006) that overall, volunteers participating in the Reef Watch program demonstrated a reasonable level of competency. Nevertheless, "to be fully credible, a community monitoring program needs to demonstrate that the data collected are precise and reproducible" (Turner et al. 2006). Thus, one of the principal aims of the Reef Health 2007 surveys was to make comparison with the data collected at the same sites by Reef Watch to obtain an indication of the efficacy and accuracy of this community-based monitoring.

The similarity of the relationships demonstrated by the ordination of LIT data between sites as described by the Reef Watch and Reef Health datasets indicates that community-based monitoring of reefs has great potential. Whilst identical composition was not recorded for any site by the Reef Health and community-based Reef Watch programs, and the reef health indices showed little similarity, there are a number of reasons why this might be the case, and they do not invalidate the concept of utilising the community to monitor the status of reefs.

Firstly, it has been identified that consistently lower amounts of canopy were recorded by Reef Watch volunteers. This is mirrored by consistently higher scores for brown understorey. Such a difference might be explained by a methodological difference in the way data were recorded — Reef Watch volunteers recorded "small brown algae" where the alga was under 10 cm in height, whereas Reef Health recorded the taxa to the lowest level possible. Whilst small brown algae were classified as brown understorey, some of

that would have been juvenile canopy plants, in particular, members of the genus *Sargassum* which lacked the large fertile laterals that they carry for much of the year. Womersley (1987) indicates that most *Sargassum* species are left with only a basal rosette after their reproductive period. In this state they would have been identified as brown understorey by Reef Watch, but recognised and recorded as canopy plants by Reef Health, helping to explain the discrepancy noted above.

Another, possibly more substantial source of error, is geographic location. Reef Watch volunteers were unable to use the GPS co-ordinates used by the Reef Health team (usually as a result of having to rely on shore based divers rather than boat access), and as such had to rely on conducting a survey in the same broad area. This could have resulted in surveys being conducted in regions separated by a distance of some hundreds of metres. The problem is, to a limited degree, lessened by analysis at the level of the site rather than transect, but pairs of transects were separated by only 40 m, which might be significantly less than the distance between the two groups of surveys. Indeed, at one reef where approximate locations were used (Seacliff), the actual locations were >350 m apart. Unsurprisingly, the Reef Watch and Reef Health estimates of the health of this site varied greatly, in terms of both the ordination of LITs and Reef Health index. Medium scale spatial variation could thus be mistaken for a difference caused by methodology (community v professional survey teams).

Collings (1998), in a study based at Cape Jervis (on Fleurieu Peninsula), demonstrated that great variability was associated with separation by only a few hundred metres. Similarly, Edgar and Barrett (1997) identified a high degree of variability on a small transect to transect scale in a study of marine reserves. Furthermore, assessment of the individual transects of this study, which were usually all within 20 m or so of each other, often reveals substantial variability on this small spatial scale — variability that in most cases swamps any inter-site differences (Figure 16). In all instances where Reef Health reports have involved calculation of temporal change at given reefs (Cheshire and Westphalen 2000, Turner et al 2007), the caution has been added that it was not *exactly* the same location which was surveyed, and that apparent temporal change may actually be small scale spatial variation. While spatial repeatability may explain much of the apparent variability between the two methods, it does indicate a concern if repeated monitoring is the aim.

In this instance, we have chosen the data from the Reef Health program to benchmark the community-collected data against. This may not be entirely appropriate, as it assumes that there is no error or variability in the Reef Health data. While it can be reasonably assumed that Reef Health, with its professionally trained scientists will be more accurate, it would represent over-confidence to claim it was without error. Furthermore, ordination of the individual transects indicates that the estimation of the "average" composition of each site has substantial variability associated with it. No attempt was made on this occasion to assess the accuracy of Reef Health data by resurveying exactly the same transects.

Another major source of variability, which may introduce further discrepancy between community-based (Reef Watch) data and that collected by the Reef Health program, is short-term temporal variability. This is a factor which is more likely to influence the reef health indices than the ordination of the LITs, as the LITs do not involve analysis of the motile fauna such as fish. As indicated in the earlier section dealing with the 2005 – 2007 change, the presence of fish (even site-attached fish) is highly variable with time. Given that no attempt was made to synchronise the timing of the Reef Health and Reef Watch surveys, this is likely to introduce substantial variability to the analysis of the reef, which will be expressed as discrepancies between the methods. The great discrepancy between the sources of data collection in the assessment of overall reef health index (in light of the similarity in patterns demonstrated by the ordination of the LIT data) is a strong indicator of the importance of short-term change on indices. This will be dealt with further in the following section.

Finally, whilst not representing a cause of variation between the two collection methods, it is unfortunate that the range of sites sampled by Reef Watch during 2007 was not wider. With the exception of Second Valley, all sites are to be found within a relatively small geographic region. More importantly, Reef Watch was unable to survey reefs representing the other end of the spectrum – the apparently degraded reefs of Broken Bottom and Semaphore. To be able to compare the two sets of data across the entire spectrum of sites would have been instructive. Clearly there are constraints placed on organisations staffed by volunteers, with a range of their own commitments in addition to reef surveillance. Nevertheless survey site prioritisation may represent an issue in the future.

The success of community-based monitoring in this instance reflects that demonstrated in a wide variety of ecosystems (see earlier references). Sharpe and Conrad (2006) identified a series of factors common to the limited number of groups in Nova Scotia which had been successful in establishing a long-term monitoring record. These were

- Management by a steering committee composed of members from the community, academia, the government and private sector;
- Adequate long term funding;
- Access to scientific expertise in data collection and interpretation procedures;
- A good communication program for both the community and the volunteer data collectors; and
- Engagement of politicians and managers by the volunteers.

It is unsurprising that the Reef Watch program has proven successful given that it has, in large part, fulfilled each of these criteria. However, it is worth noting that future success will depend on an ability to maintain this record and a demonstrable program of quality assurance (see Hodgson et al. 1997).

Potential improvements to the program

The clear similarity in terms of most relationships between sites as indicated by the LIT data is a very strong indication that Reef Watch volunteers represent an extremely valuable asset in the important drive for ongoing marine monitoring. However, this study has identified areas which, if addressed, would greatly improve the value of the collected data.

Specifically, small to medium scale spatial variability (which confounds important comparisons across time) must be avoided. It is of markedly less value to conduct surveys without the accuracy of GPS locations, and it is further suggested that permanent markers (to which transect tapes can be temporarily attached and pulled taught) should be put in place for the long term at a variety of sites.

It has become evident that further taxonomic skills are necessary to clarify the situation for *Sargassum* species so that they are routinely classified as canopy rather than brown understorey, on the basis of their stature at certain times of the year. A herbarium could be maintained for each of the permanent sites containing examples of all or most species, or at least photographs thereof, along with an instruction as to which functional group

they should fall into. A site-specific "photographic herbarium" could form an invaluable aid for a pre-survey briefing, and would probably increase the interest level of the participants. Furthermore, as participants in the program change, it is important to maintain and document the level of accuracy through a series of benchmarked transects where teams are asked to resurvey a variety of transects which are scored concurrently by an instructor. Instructor benchmarking could be conducted using a series of photo transects. These activities should ensure that change seen across years is not simply due to changes in personnel.

Over the course of the Reef Health program and previously conducted scientific surveys such as Cheshire et al. (1998) and Cheshire and Westphalen (2000), a relatively good picture has been composed of the state of a wide variety of reefs within South Australia. It would be useful to prioritise sites for survey to maintain a relatively consistent historical record across a range of sites. For instance, rather than resurvey a particular site annually, it may be wiser to survey that reef every second year and incorporate another reef of different composition (such as the inner metropolitan reefs - Broken Bottom and Semaphore) or location. Obviously, this needs to fit within the capabilities and aspirations of a volunteer organization, but a logical plan of critical surveys would make a good foundation. Beyond these, *ad hoc* surveys of reefs should be considered a bonus. Finally, it is important that the plan be a flexible one, able to be adapted to new information as it is discovered, or when important developments occur in particular areas.

Finally, one activity which has not been incorporated into Reef Watch reef surveys is the compilation of photo-transects, whereby participants pull a tape transect taut, and take a close-up photograph of each 50 cm section, with some overlap. A similar approach has been used on coral reefs (Aswani et al. 2007). Care would have to be taken to ensure that the tape was not entirely obscured by canopy algae (a problem not faced on coral reefs), but this is not an insurmountable problem. This activity has a number of advantages: Digital underwater cameras (and consequently photography) have become much less expensive and therefore more common. The training required to pull a taut transect line between two permanently fixed points and then take a photograph of each 50 cm segment is far less onerous than the identification skills currently required. It may therefore appeal to more (or at least a different group of) people. Obviously a person skilled in identification is then required to analyse the photo-transect, but importantly, the photographs are a permanent record, so where important change is apparent, it is

possible for direct comparison of photo-transects by the same operator, eliminating the possibility of a mistake on the previous occasion. Photo-transects, created by stitching together the individual 50 cm sections could then be displayed via the internet and comparisons made side by side of selected transects by any interested persons. A permanent visual record of change is then available. From a researcher's point of view, the benefit of such an activity in the future is clear when one considers how useful such a record of the past would have been.

4.4 Comparison of LIT ordination and index calculation methods

Interpreting the ordination of LIT data requires an understanding of what a relatively healthy reef should look like, and ideally such a reef should be included in the analysis. If reefs move away from this state, it can be assumed to represent change for the worse. Obviously this requires appropriate pristine reference sites, which in turn requires an understanding of the factors likely to affect composition. As an example, the 2005 survey included a number of sites on Yorke Peninsula and some of these were classified as being in poor condition, despite being exposed to relatively low human influence (Turner et al. 2007). This is likely to be because the reference "healthy" state described a reef exposed to good water movement (unlike the low water movement conditions experienced by the purportedly poor reefs) rather than because the reefs were actually "unhealthy". It is only with time, and surveys and analyses such as represented by the Reef Health project that we can build up a picture of what should be expected at a pristine site in a given environment. For example, the Northern Reef has been used in this instance to indicate what a relatively "pristine" reef should look like. Having made the assumption that our reference sites are appropriate, analysis of reef composition is relatively simple.

Conversely, the Reef Health Index is not simple. It is able to produce a single figure with which to assess the status of the reef, and it does so utilising a wider range of features than associated with the LIT ordinations. However, it should be considered a "work in progress". It is a complex model, with many assumptions which, over time, will need to be refined. While this renders the current outputs as speculative, this does not invalidate the concept. Indeed, the index probably has the potential to be of great use to managers needing to assess and summarise reef condition. The process of refining and improving this model requires regular application to identify where improvements can be made.

Calculation of the individual indices relies upon a mathematical relationship, which defines a value for the index on the basis of measured figures. These relationships, whilst qualitatively logical, often have little quantitative rationale. For instance Figure 2 demonstrates a generalised relationship between a raw score and the index value, with a lower limit (below which there is no effect on index score), an upper limit (above which there is not effect on index score), and a linear positive relationship between raw score and index score between these limits. There is intuitive logic in the qualitative relationship – for example, even in a pristine macroalgal bed, you would not expect 100% cover – patchiness is an inherent part of a natural ecosystem. Thus, it might make sense to have an upper boundary set at 60% (as in this study), above which any increase fails to increase the index score. However, whether 60% is the appropriate value is debatable. Similarly, is condition best described using a linear relationship? If only a single index were used, this would be less important – at least ordinal relationships could be ascertained. However, as the overall index is constructed from an average of a series of these indicators, the composite index may be inaccurate. Similarly, there is currently some weighting in terms of importance of the individual indices in the calculation of the overall index. For example, some indices are measured on a scale of 0 to 100 points (such as canopy cover) whilst others are scored on a zero to 50 point scale (such as turf). Whilst it is not suggested that all factors should be equally weighted, relative weighting is an issue that may need to be revisited in the future. Furthermore, many of the factors measured (if not all) are interrelated in complex ways. Babcock et al. (1999) and Parsons et al. (2004) demonstrated the strong trophic interactions between predatory fish, urchins and algal cover. However, these interactions may vary according to locality (Fowler-Walker and Connell 2002, Anderson and Millar 2004). Thus, while calculation of the indices may also help us understand these trophic interactions for our own waters, the understanding itself may help refine the model. Whilst considerable thought from a variety of experts has gone into deciding which indices are appropriate for inclusion in the overall reef health index (Turner et al. 2007), this should not preclude either new indices, or deletion of old ones as our understanding grows.

The above summarises some of the issues that bear further thought in the future as our knowledge grows. They do not necessarily represent concerning issues, merely ones which may be addressed at a later point in time. At this stage, no attempt has been made to address these in detail. However, two aspects of the Reef Health index do require attention and will be dealt with here.

Firstly, this report has identified the shortcomings in the method for quantifying reefal fish populations as an issue confounding long term temporal change. Methods more appropriate to such a dynamic component of the ecosystem will need to be developed. These would have to incorporate a longer-time scale, possibly multiple surveys arranged around critical points of time and tide, or alternatively, use of video technology might allow long surveillance periods that could be subsequently assessed at high speed. It is also suggested that a greater amount of spatial replication may be helpful (i.e. more transects spread across the reef). Visibility bias would have to be taken into account in some manner such as a correction factor for visibility, but this is certainly not an insurmountable problem . Furthermore, an assessment of adequate sampling effort for a single point in time needs to be made. It is worth noting that Reef Watch, with a large group of divers particularly interested in fish, has the capacity to overcome this problem.

Secondly, the application of null scores for some indices at some sites does not have a consistent effect on overall score. The logic behind the mechanism of a null score is as follows: When a negative feature, such as mussel cover is being considered, a relationship is used whereby the possible index scores range from 50 at low amounts of mussel cover (15% see Figure 21), down to 0 at high amounts of cover. If mussel cover is less than 15%, then it is considered a "normal" situation, and the "negative index" should be discounted – therefore the site gets a "null" score for the mussel index. This simply entails averaging over one less index. Unfortunately there is a mathematical inconsistency – at any site where the composite (overall average) health index is less than the score obtained from the best possible mussel cover index, it is an advantage to have the mussel cover scored rather than having a null. For example, consider a reef which has a composite score of 35. If it has no mussel cover (a "perfect" health situation) then a null is awarded for mussel cover and the composite score of 35 is unaffected. If, however, it has 15% mussel cover (surely a worse health situation), then it scores 50 for the mussel index and the overall index is dragged *up* above 35. Thus the effect of a null score can be counterintuitive. In a general sense, there is an inconsistency in the effect of a null score, depending on the overall score of the site and the possible scores for an individual index. This inconsistency must be rectified in future incarnations of the index.

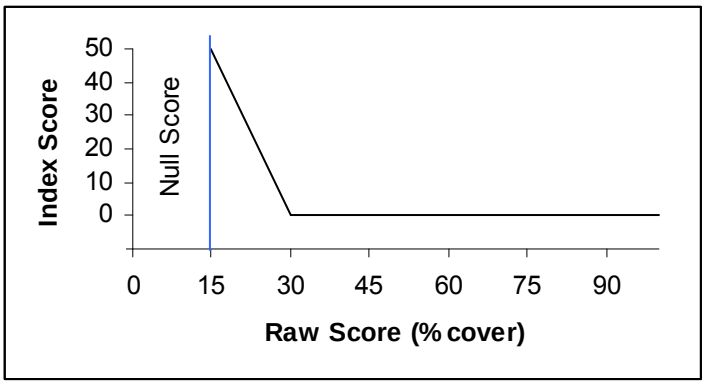

Figure 21. The relationship between raw mussel bed cover (as a percentage) and the index score. Note that cover of <15% gives a null index score, whilst any cover greater than 30% gives a zero index score.

A similar inconsistency is caused by the selection of thresholds to scale between. It was noted in Section 4.1 that a site with approximately 25% canopy cover (Northern Reef) scored only 7 points, barely any more than a site with virtually no canopy (Broken Bottom). However, a site with twice the canopy cover (Hallett Cove) scored 99 points. There may be good reasons for choosing thresholds, as identified in Turner et al. (2007), but results like this may be an indication that fine-tuning is required. This is not unexpected, given the experimental nature of these indices (Turner et al. 2007).

On a more general note, some experts have questioned the use of the term "health" to describe the status of ecosystems and our ability to "calculate" it (Calow 1992). They do so on the basis of the fact that characterisation of the health of an ecosystem must involve subjective decisions concerning the features to measure, the way in which those features are measured and then combined, and what values are to be considered healthy (e.g. Wood and Lavery 2000). Whilst these are very relevant points and need to be addressed carefully, we argue that the concept of human health has a similar set of concerns, and despite these, it retains great usefulness. Ecosystem health has the potential to be equally useful despite its subjective nature. However, it will need careful development.

Whilst the Reef Health index provides a useful set of insights into the status of South Australian reefs, it is in the early stages of development, and as such more store should be put (at this stage) into the conclusions of the multivariate analysis of the line intercept transect data. In the current study, the conclusions are broadly similar. However, where discrepancies arise, it is considered that the more reliable analysis is to "let the data tell it's own story" in the simplest way possible. While there are obviously assumptions and

models utilised for the multivariate analysis, it is considered at this stage that they, unlike the reef health index, are widely tested, understood and used. The added potential of the more complicated index calculations determines that the technique should be retained and refined. In this instance, the conclusions drawn from the ordination of LIT's are considered to be more robust than the indices and allow comparison with earlier studies and should continue to form the backbone of the reef health analysis. However, different approaches to those used here, or modifications, in particular the inclusion of fish and invertebrate data into the dataset used for ordination, also deserve attention in the future.

5 Conclusions and Recommendations

From the standpoint of repeatability, it is heartening to see that a broadly similar pattern emerged in the assessment of the reefs of Adelaide and Fleurieu Peninsula in 2007 as was the case in 2005. There were two relatively well-defined groups, corresponding roughly to the northern and southern sites, with the two inner metropolitan reefs (Semaphore and Broken Bottom) demonstrating distinct biota.

Importantly, this set of surveys included a site (Northern Reef) more northerly than either of the two reefs considered anthropogenically impacted by Turner et al. (2007). This allowed an assessment of the hypothesis that Semaphore and Broken Bottom simply represented the northerly extreme of a natural gradient (which is geographically the case). This work has made it clear that Broken Bottom and Semaphore are in worse condition than would be indicated by a natural geographic gradient. Anthropogenic impact is a likely cause.

Taking a long-term view of the situation was possible due to the availability of datasets from surveys of reefs common to the current study, which were undertaken in 1996 and 1999. There has been a relatively consistent trend toward a more canopy-dominated state, with less turf and bare areas. This change could be interpreted as an improvement in the health of the reefs. Whilst the cause of this change has not been positively identified, it mirrors improvements in effluent treatment that have seen marked decreases in the nitrogen and phosphorus load discharged to the sea. This may represent evidence that decreasing nutrient loading is likely to result in the improvement of some degraded reef systems. However, the current poor status of inner metropolitan reefs is an indication that whilst efforts to improve effluent treatment for environmental reasons have been a laudable success, further improvement is still required.

The use of the community-based Reef Watch organization to monitor the status of the reefs of South Australia appears to have great promise. Very similar patterns are evident in the biotic relationships between reefs as determined by Reef Watch data and by the (professional) Reef Health project. There are certainly discrepancies between the two datasets, but the broad pattern is quite similar, despite the fact that the range of sites in common does not encompass the extremes of the biotic variability (i.e. Reef Watch did not survey the northern, degraded reefs).

At this stage, the production of the overall reef health index represents a work in progress. Nevertheless, its potential demands persistence. Whilst a high degree of

variability is evident between 2005 and 2007 in the overall index of the sites, and between the data collected by professional and community-based organizations, this can be largely attributed to inadequate sampling methods for the mobile fauna and effects of the scaling of raw scores to index scores (especially in the case of "null" values). Neither of these represent insurmountable problems. More appropriate methods have been developed for the census of mobile fauna, and could be utilised, albeit requiring a greater amount of time than is the current case. Similarly, recalibration and reconstruction of the model used to calculate reef health indices on the basis of a better understanding of the way the model works will improve the accuracy of this assessment. Such an understanding is best gleaned through applying the model (as has been done in the current study) to identify its strengths and weaknesses. Finally, it is worth pointing out that because of the experimental and variable nature of the reef health index, the most useful interpretation will come from an assessment of both the average and variability in the index (and its individual component indices) across a series of points in time as opposed to a snapshot view at any one point in time.

The ideal of establishing the "health" status of a reef is a laudable one. It undoubtedly faces challenges on several fronts: The models which provide "scores" for each index rely on an understanding of the relationships between biota and environment that is continually evolving, but in several instances (particularly the fish-related indices) is in its infancy, and likely to be specific to individual species rather than to broad suites of organisms. The methods used to acquire data for these methods still demand further validation (is sampling effort great enough in both time and space to adequately assess biota across the reef?). Overarching these considerations is the question of whether it is useful to determine a single figure to indicate health, or whether it is an unnecessary oversimplification which should be avoided by addressing each of the individual indices without any attempt to mathematically combine them. The concept of assessing reef health is a valuable one which deserves persistence and attention to these challenges in order to refine it for the future.

It is recommended that the monitoring of the condition of reefs should continue, utilising the demonstrated expertise of the Reef Watch organization, in conjunction with ongoing professional input for advice and analysis where necessary. Modifications to the methodology which will improve the utility of this data include

- Permanently marked transects to better highlight temporal change;

- Compilation of regularly consulted herbaria for each site;
- Incorporation of permanent "photo-transects";
- More accurate methods for mobile fauna census, including more transects in different parts of the reef and assessment of temporal variability;
- Monitoring of a broader range of reefs (if necessary sacrificing the frequency of resurvey);
- Continued improvement to the model utilised for the calculation of the reef health index based on the outcome of future surveys and understanding. This must address the construction of the individual indices, the method of combination used to provide an overall index and a determination of the appropriateness of providing a single figure for environmental "health".

It must be recognised that such a program will necessitate an improvement in the provision of resources to allow community-based monitoring initiatives to fulfil the goal of accurately assessing the status of the State's reefs.

Importantly, this study has provided evidence to suggest that the status of the reefs of Adelaide and Fleurieu Peninsula has generally improved since the cessation of the dredging associated with beach sand replenishment, and over the same period that the nutrient loading of wastewater has decreased. While direct causal links have not been established, these improvements are to be applauded. In light of the demonstrably degraded nature of metropolitan Adelaide's northern reefs, it is clear that continuing improvements to water quality in the future are critical to the rehabilitation of these systems. It also serves as a warning that the expansion of Adelaide's footprint must be associated with improvements to the way that we deal with land-based discharges if further impacts are to be avoided.

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Appendix A: Sampling Methodology (adapted from Turner et al. 2007)

These methods were originally developed and described in Turner et al. (2007). Essentially the only difference between the protocol utilised in 2005 and the current study is the lack of the benthic quadrats utilised in the earlier study. Therefore the methods are reproduced here from that report.

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 systematic, after haphazard choice of starting point, 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 8, and annotated with additional information where appropriate.

Table 8. 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 9).

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 9), a sample was also collected for later confirmation.

Table 9. 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[1]

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 descent 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 and the tally were recorded in the final column (Table 10). A scale marked on the margin of the survey form was used to help calibrate size estimates.

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

Table 10. 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 recorded 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[2]

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.

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

Counts (but not sizes) of all larger non-sessile invertebrates (>5 cm), along with cryptic or sedentary fish (Table 11) were recorded on the data sheet. Smaller and more numerous taxa, along with sessile invertebrates such as ascidians, were recorded using the LIT method described later.

Table 11. 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)[3]

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 were 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

[3] Method based on Turner (1995)

the point at which there was a transition to another taxon (Figure 22). The code for this taxon and transition point was then recorded on the data sheet (Table 12) 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.

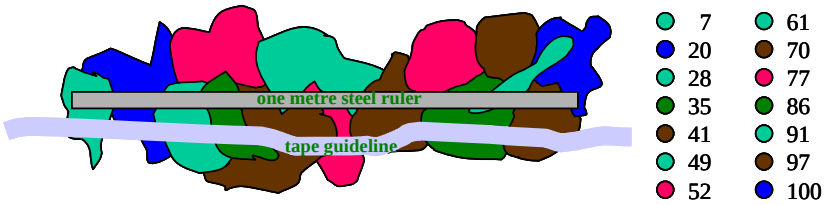

Figure 22. An example Line Intercept Transect and resulting data.

Table 12. 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.

Appendix B: Site descriptions for reefs included in the 2007 surveys

Site descriptions given here (Table 13) are generally based on information collected during the 2007 field survey program. In total, 15 sites were surveyed using the methodology described in Appendix A. Measures of exposure and relief are subjective estimates.

Table 13. Site description for reefs surveyed in the eastern Gulf St Vincent during 2007. Empty cells indicate a lack of data

Reef	Description	Composition	Relief	Exposure	Dominant biota
Northern Reef	Broken Bottom horizontal reef	Limestone	0.5 m	Low	Sparse *Caulocystis* and *Sargassum* canopy, red understorey and sponges
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*
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*
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

Reef	Description	Composition	Relief	Exposure	Dominant biota
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	Sandstone	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 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.
Aldinga	Aldinga reef is comprised of a series of gently sloping rock platforms with occasional prominent outcrops. 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*
Second Valley	Sloping undulating reef with some boulders	Schists	1 m	Low	*Sargassum, Cystophora* and *Ecklonia* abundant

Appendix C: Reporting codes used during data analysis

Table 14. 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
Canopy 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*
Brown Understorey	BRFOLI	Brown foliaceous algae	*Halopteris, Cladostephus, Lobospira*
	BRLOBE	Brown lobed algae	*Zonaria, Padina, Lobophora*
	BRMEM	Brown membranous algae	*Scytosiphon*
Red Understorey	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*
Green Understorey	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 15. 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*	*Rhycherus filamentosus*
Aracana ornata	*Meuschenia hippocrepis*	*Stigmatopora nigra*
Austrolabrus maculatus	*Neoodax balteatus*	Stinkfish spp.
Blennidae spp.	*Nesogobius* spp.	Syngnathidae spp.
Bovichtus angustifrons	*Norfolkia clarkei*	*Tetractenos glaber*
Bullseye undifferentiated	*Notolabrus parilus*	*Tilodon sexfasciatus*
Cheilodactylus nigripes	*Notolabrus tetricus*	*Trachichthys australis*
Chelmonops curiosus	*Odax acroptilus*	*Trachinops noarlungae*
Clinidae spp.	*Odax cyanomelas*	*Vincentia conspersa*
Cnidoglanis macrocephalus	*Omegophora armilla*	Wrasse spp.
Cochleoceps spp.	*Parablennius tasmanianus*	
Cristiceps australis	*Parapercis haakei*	
Diodon nicthemerus	*Parapercis ramsayi*	
Dotalabrus aurantiacus	*Paraplesiops meleagris*	

Table 16. 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*	