Assessing the status of temperate reefs in Gulf St Vincent I: Background and methodology for assessments.

A report to the Environment Protection Authority

Anthony C. Cheshire, Jon Havenhand, Stephen J. Hall,

George Matsumoto and Alan J. Butler

Assessing the status of temperate reefs in Gulf St Vincent I: Background and methodology for assessments.

A report to the Environment Protection Authority

By

Anthony C. Cheshire[1], Jon Havenhand[2], Stephen J. Hall[2], George Matsumoto[2] and Alan J. Butler[3].

1. **Dept of Botany, University of Adelaide, Adelaide 5005.**
2. **School of Biological Sciences, GPO Box 2100, Flinders University of South Australia, Adelaide 5001.**
3. **Department of Zoology, University of Adelaide, South Australia, 5005.**

National Library of Australia Cataloguing-in-Publication data:

Cheshire, A.C., Havenhand, J., Hall, S., Matsumoto, G. and Butler, A.J. (1998). Assessing the status of temperate reefs in Gulf St Vincent I: Background and methodology for assessments. A report to the Environment Protection Agency of SA. Pp 53

Published: Department of Botany, University of Adelaide, South Australia, Australia, 5005.

Publication date: 1-Aug-1998.

ISBN 0 86396 630 6 (vI)

Table of contents

Executive Overview

Background

This report is the first report in the Reef Health series. The Reef Health project was commissioned by the South Australian Environment Protection Authority (EPA) in order to provide detailed information on appropriate approaches to the assessment of reef systems in Gulf St Vincent. The project is undertaken jointly by researchers from The University of Adelaide and Flinders University who variously provide expertise in assessment of marine ecosystems including both the benthic flora and fauna and associated fisheries species.

Scope

This report addresses the requirements under section 4.1 of the tender document in that it provides the background information necessary to develop a field based program which aims to assess the status of reefs in Gulf St Vincent. It includes a review of the literature on the biology/ecology of temperate reefs, a discussion of the concept of “reef health”, and a critical assessment of alternative assessment methodologies with specific details of the methods being employed in this study.

Objectives

The specific objectives of this report are:

1. To provide a review of the literature which details what is known about the nature of South Australian temperate reef ecosystems and how this relates to our ability to define the “health” or the “status” of these systems (Part 1). This concentrates on the assessment of biological parameters that are considered to provide longer term temporal integration of the water quality.

2. To provide details of the methodologies which can be used to assess the physical condition and the status of the biota on temperate reefs. This includes a critical assessment of these methods as they relate to the ongoing monitoring of reefs in South Australia (Part 2).

3. To provide details of the survey methodology that is being used to develop an initial assessment of the status of selected reefs in Gulf St Vincent (Part 3).

Findings

1. Methods are defined for the quantitative assessment of a number of parameters which can be used to describe the structure of biological communities on temperate reefs (including both measures of diversity and abundance). This is supported by a discussion of the taxonomic resolution which can reasonably be achieved in such surveys. Arguments are presented which conclude that species level assessments are difficult to make and, in general, are not considered either necessary or appropriate for surveys of this sort.

2. The issue of senescent condition of reefs is dealt with through a detailed discussion of our ability to gauge the "health" versus the "status" of reef systems. It is concluded that we have insufficient information to accurately define health but that we can define what would be considered the preferred states for reef systems. These in turn may be defined as goal states for the purposes of management.

3. In general, assessments of the age and life cycle distribution of benthic organisms are not possible. Detailed procedures for assessment of size structure of fish populations could be developed but these would be technically demanding and could not be generally applied by, for example, community groups.

4. A number of standard methods for assessing sediment deposition rates are available and details are provided.

5. The physical condition of reefs can be described using measures of topographical complexity and through a variety of physico-chemical measurements. Details of these are provided. It is suggested that those which relate to water quality require a sampling design that extends over greater periods than the spot measurements which can be obtained in this study. Some such data, particularly relating to turbidity, could be collected on a routine basis by community groups. Other measurements (including measurement of dissolved oxygen, suspended solids etc are more difficult and require specialised equipment).

6. Effects of introduced or exotic organisms will be considered in the context of results from the biological surveys currently being conducted. The survey methods have been developed to both identify and quantify these.

Part 1 - Background

Temperate reefs - what are they?

The term "Temperate Reef" will indicate different things to different people depending upon their background and interests. For many people the word reef conjures up images of idyllic tropical locations, a splash of colour and a multitude of corals and fishes. Extensive media exposure has reinforced this view and a large tourist industry has been built around the attractions of coral reefs. For people living in southern Australia however a reef is quite a different thing. It is generally a rocky outcrop covered in seaweeds; like coral reefs they are highly diverse environments and are good places to catch fish but overall they are not as well understood by the general community. We do not for example have glass bottomed boats or "Green Island" type resorts where people go simply to view "the reef". Rather, although our reefs are visited regularly by fishers they are largely unseen except by diving enthusiasts.

The distinction between temperate and tropical (coral) reefs is not simply one of perception. There are in fact quite fundamental differences in the structure and dynamics of these ecosystems. Temperate reefs only exist in areas where consolidated sediments or rocky seabeds provide a site for settlement and attachment of algae and sessile invertebrates. In contrast, coral reefs are largely built up by the constituent corals and algae and once established they can develop and expand upon this substrate. Furthermore, the physical and chemical environments are distinctly different. Temperate waters are cooler and nutrient levels tend to be higher than in reefs in tropical waters. Together, these factors have had a profound effect on the evolution of the biota in these regions.

Instead of being dominated by corals and sponges (many of which have zooxanthellae or other photosynthetic symbionts) the dominant biota on temperate reefs are the free living algae. Whereas many sponges, a few corals and a diverse array of other animals do exist on temperate reefs, they are rarely involved in mutualistic symbioses and are therefore largely heterotrophic in their nutrition. In essence, there is a more distinct separation between the producers and consumers on temperate reefs and consequently there are fundamental differences in many of the dynamic processes (especially in relation to trophic connections).

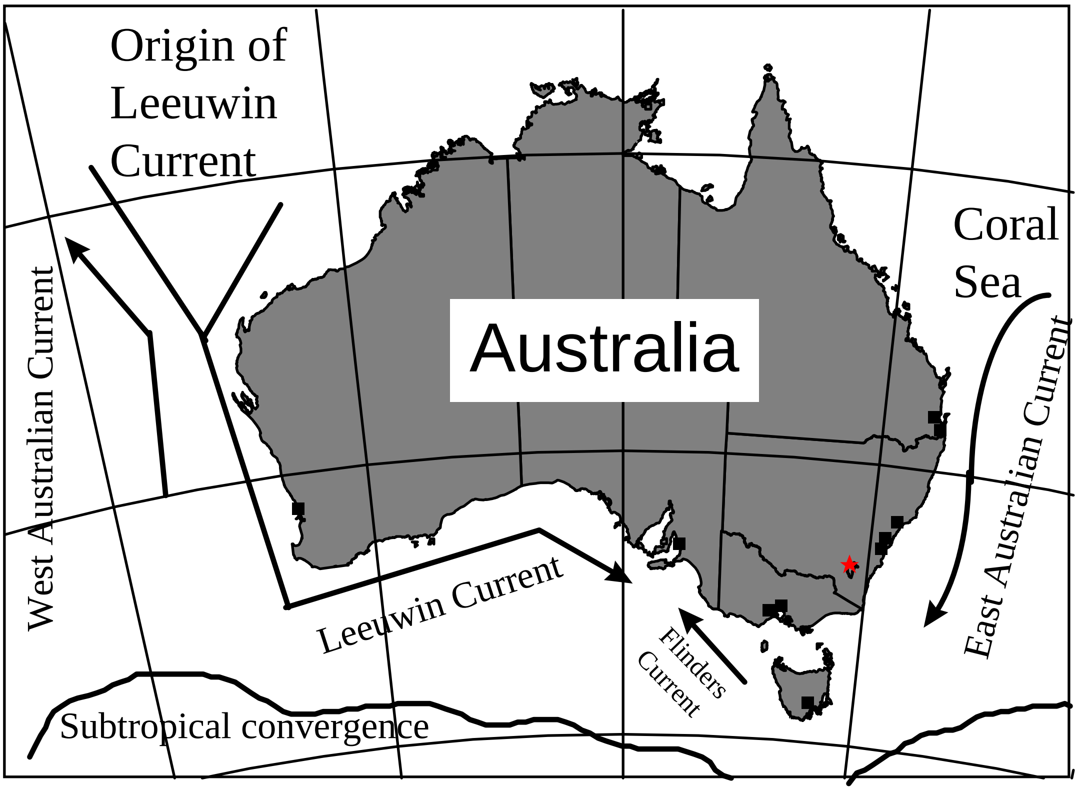
In accepting these general differences between tropical and temperate systems it is important to recognise that temperate reef systems vary considerably on a global scale. Importantly, the temperate reefs of southern Australia are unique in terms of both the species richness (there are for example more species of seaweed in southern Australia than there are corals on the

Great Barrier Reef!) and the degree of endemism at both specific and generic levels. This uniqueness can be broadly attributed to 3 principle factors including:

- the oceanographic isolation of southern Australian coasts from other temperate coasts,
- the length of our coastline at a relatively constant latitude and,
- (with respect to endemism) the fact that southern Australian coastal waters are naturally nutrient poor relative to similar temperate locations elsewhere in the world.

Oceanographic isolation has resulted from the dominance of the north-south flowing currents on both the eastern and western seaboards (East Australian Current and Leeuwin Current; Jeffrey *et al.* 1991; Figure 1). These currents bring warm nutrient poor waters south and largely isolate the southern Australian coast from the westerly flowing currents of the Southern Ocean. This in turn limits the dispersal of temperate species both to and from the southern Australian coast (although reasonably strong connections do occur across the Tasman with New Zealand; Poore 1991).

Figure 1 - Dominant features of Australia's oceanic circulation (after Jeffrey *et al.* 1990)



Low nutrient levels result as a consequence of 3 factors: the flow of nutrient poor waters from the northern tropical regions (via the north-south flows), the lack of significant upwelling zones and the slow weathering and low rainfall of the southern regions of the Australian continent. These processes act together to isolate southern Australian coastal waters from any significant additional nutrient inputs.

The southern Australian coastline represents the longest east-west running stretch of coast in a temperate region anywhere in the world and it has a wide diversity of habitats including bays, gulfs, promontories, islands and estuaries.

Together, these factors have led, over geological times (particularly since the Cretaceous - 125 mya), to an isolation of our temperate biota. Subsequent speciation has led to a very high degree of endemism (Poore 1991, Womersley 1990; Table 1). Coupled with this we have a coastal environment with a wide diversity of habitats inducing a similarly high diversity of species (Womersley 1991; Table 2).

Table 1 - Endemism and diversity of major temperate reef taxa in Southern Australia.

Taxonomic group	Diversity (# species)	% Endemic	Source
Fishes	600	85	Poore 1991
Molluscs		95	Poore 1991
Echinoderms		90	Poore 1991
Chlorophyta	124	30	Womersley 1991
Rhodophyta	800+ (currently >1000)	75	Womersley 1991
Phaeophyta	231	57	Womersley 1991

Table 2 - Comparative diversity of southern Australian macroalgal taxa (after Womersley 1991).

Region	Coast length	Temperature range	# of species
Southern Australia	5,500	Cold-warm temperate	1,155
NE North America	8,000	Arctic-warm temperate	399
Pacific North America	12,000	Arctic- tropical	1,254
Japan	6,500	Subarctic- subtropical	1,452
New Zealand	6,970	Subantarctic warm temperate	835

This unique character, with respect to both the physical/oceanographic environment and the biota in this region, has significant consequences for the understanding and management of our reefs The fundamental differences in character of southern Australian temperate reefs, and the implications this has for the underlying processes operating in these systems, make it imperative that management decisions are based upon relevant data that have been obtained from local systems. Consequently, it would be inappropriate to assume that findings from systems elsewhere in the world will be generally applicable to southern Australian reefs.

What does a reef look like

Naturally occurring subtidal hard substrata range in size from small isolated patches, such as *Pinna* shells, to large contiguous areas of rocky reef. Substantial artificial surfaces, such as jetties and piers, are also abundant particularly in metropolitan areas. This heterogeneous array of hard surfaces provide anchorage points for many species of macroalgae and sessile animals which in turn form physical habitat used by a variety of other species.

Reefs of Gulf St Vincent

Gulf St Vincent is primarily a carbonate sedimentary province in which a number of limestone reefs occur along with shell bed platforms and aeolianite dunes (Shepherd and Sprigg 1976).

In addition, a number of artificial reefs are also present comprising scuttled ships (4), tyre constructs (10) and the concrete blocks at Glenelg. There are also a number of shipwrecks that could be classified as artificial reefs. These systems have been colonised to form lush and productive ecosystems which have increased the total amount of reefal habitat in the Gulf.

Flora

Gulf reefs have a diverse flora of macroalgae conservatively numbered in excess of 500 species. Representatives of all 3 major macroalgal divisions (Rhodophyta, Phaeophyta and Chlorophyta) are common with most reefs being visually dominated by the larger brown algae. Only one species of kelp, *Ecklonia radiata,* is found on these reefs but there are many species of rockweeds (fucalean alga) commonly including species of *Cystophora* and *Sargassum*. There have been relatively few published accounts of the benthic flora of the Gulf reefs except for a comparison of algae (Collings and Cheshire 1998) between selected lower Gulf reefs and the oceanic sites surveyed by Shepherd and Womersley (1970, 1971, 1976, 1981). The major conclusion from this work was that many of the dominant (canopy) species found on reefs in the lower Gulf are the same as those found in the more exposed oceanic environments. In general the distribution of algae on gulf reefs follows a similar pattern to that proposed by Shepherd and Womersley (1970, 1971, 1976, 1981; Figure 2) for oceanic environments.

A number of unpublished theses (Collings 1989, Harvey 1990, Emmerson 1992, Turner 1995) provide details on the variability in composition and dynamics of macroalgal communities from selected sites. These have shown that community structure varies both annually (seasonal growth, shedding and recruitment) and interannually with major shifts in the dominance being reported on interannual scales (Collings 1996). Spatial variability is high with small stretches of coastline often showing more variation over small (<400 m) spatial scales than is seen seasonally over annual cycles.

These macroalgal communities are highly productive with primary production rates around 1.1% per day in winter to 2.3% per day in summer (Cheshire *et al.* 1996a, Westphalen and Cheshire in press). This gives rise to annual production figures of 20-40 kg wet weight. m^{-2}. y^{-1} (from a typical standing biomass of 3-6 kg wet weight. m^{-2}). This rate of primary production is comparable to that of a cereal crop or sugar cane stand growing under agricultural mono-culture conditions. These rates are around three times higher than those for inter-reefal seagrass systems and it may therefore be concluded that these reefs are a major source of complex organic carbon to coastal ecosystems.

Figure 2a - Schematic showing the relationship of macroalgal communities to water movement and depth on South Australian reefs (based on Shepherd and Womersley 1981).

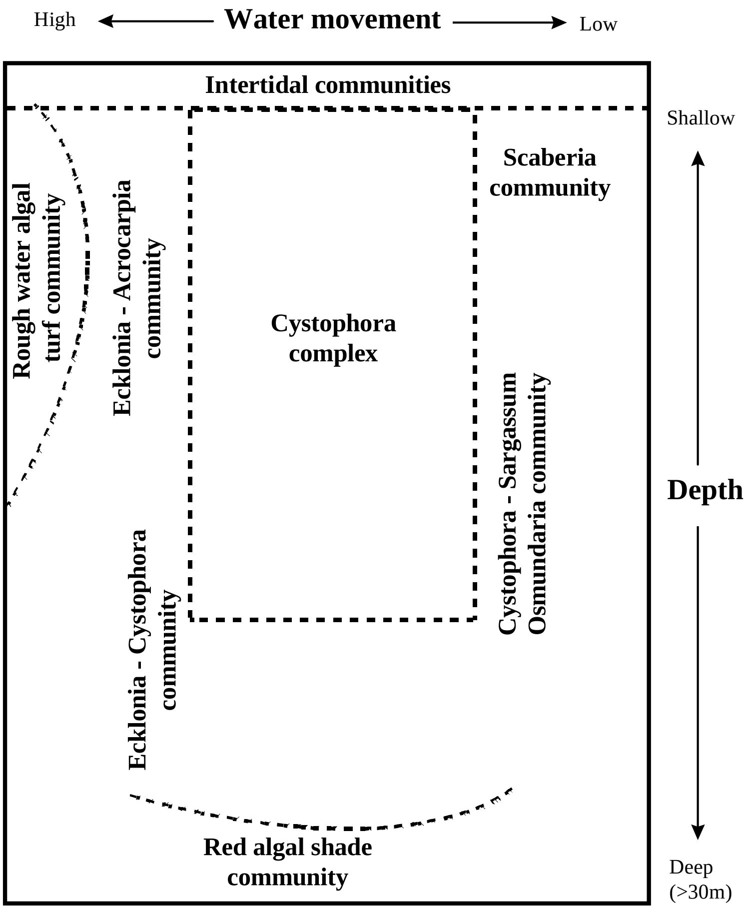
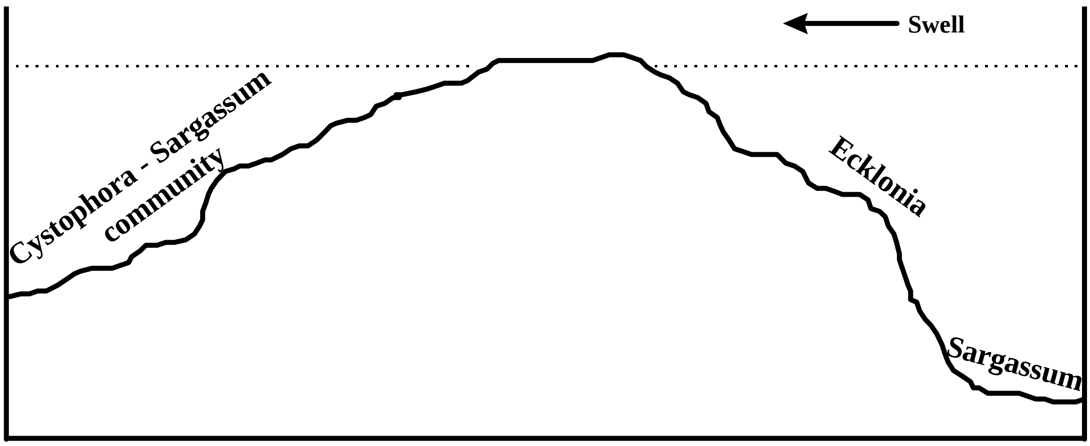


Figure 2b - Schematic showing the distribution of algae over depth and wave exposure gradients on a reef (based on Shepherd and Sprigg 1976).



Sessile/ sedentary fauna

Like the algae, the sessile (or non-motile) and sedentary (low-motility) animals of temperate reefs are characteristic of this habitat. The availability of a solid substratum presents colonisation opportunities for a variety of species that are absent from the surrounding seagrass beds and sand flats. However not all reefs provide the same sorts of habitat, and several factors are known to influence the community composition of temperate reefs.

The sessile fauna of temperate reefs are characterised chiefly by suspension feeders. On a typical temperate reef the most common groups encountered are anemones, corals and hydroids (Cnidaria), bryozoans (Bryozoa, Ectoprocta), sponges (Porifera), polychaete tube worms (Annelida), bivalve molluscs (Mollusca), and ascidians (Urochordata). These groups are well represented on the rocky reefs of South Australia, but densities and diversities of sponges, bryozoans, and ascidians are particularly high (Butler 1995, Keough and Butler 1995). In particular, the diversity of ascidians in South Australia is considerable (Kott 1985, 1990, 1992).

In contrast to the sessile fauna, the sedentary fauna inhabiting temperate reef systems are often less obvious. Characterised by unitary, rather than modular, organisms the sedentary fauna comprises a variety of taxa including herbivores, predators and scavengers. Typical sedentary herbivores on temperate reefs include sea urchins, gastropods (particularly abalone), and several species of isopod. Sedentary carnivores include species of seastars, decapod crustaceans (notably crabs), some polychaetes and gastropods. Although these species are far less numerically dominant than their sessile counterparts, their impact on the structure and dynamics of temperate reef systems may be significant.

Fish

We define mobile fauna as those taxa that cannot adequately be sampled by static point sampling methods such as quadrat counts. For the most part, this category is comprised of fish species which, although often closely associated with particular reef features, are capable of ranging over wide areas of a reef in short periods of time.

A total of 680 species have been recorded for southern Australia, many of which exploit temperate reef habitats for part of their lives. In contrast to tropical reefs, a high proportion of these temperate species are unique to Australia. Some species, such as the herring cale (*Odax cyanomelas*) are endemic to temperate Australia. Other species have more restricted geographic ranges. Within the genus *Achoerodus* (blue groper), for example, there are two species, one found in the east (*A. viridus*) and the other in the west (*A. gouldii*). The latter is the species found in eastern South Australia. Whilst very narrow geographic ranges appear to be rare, some species, including the seadragon *Phycodurus eques* have extremely limited distributions.

Although it is convenient to speak of an Australian temperate reef fish fauna, species exhibit a range of biogeographic patterns and the structure of reef fish assemblages can differ markedly between regions. Nevertheless, there appear to be functionally equivalent species exploiting the same habitats and resources in different regions. For example, territorial damselfish, large roving herbivorous species, predatory wrasse, and larger predators such as wobbegong sharks are common members of temperate reef fauna, even though the species may differ between locations (Lincoln-Smith & Jones 1995).

The temperate reef fish fauna is dominated by carnivorous taxa which either feed on reef dwelling invertebrates, or on zooplanktonic organisms above the substratum. Most of these fish species eat mobile crustaceans and molluscs, but some, such as the leatherjackets (Monocanthidae) feed on vertical drop offs and under piers and prey on a wide range of encrusting taxa. Notwithstanding the dominance by carnivores, herbivorous reef fish are by no means rare on temperate rocky reefs. Jones & Andrew (1990) estimate, for example, that 20-30% of species eat at least some algae. Moreover, in some areas herbivores dominate in biomass terms, owing to thier often large body size (Lincoln-Smith & Jones 1995). Most herbivores feed on foliose red and green algae with few exploiting the often large biomass of kelp found on reefs; the only exception to this is the herring cale (*Odax cyanomelas*) which has a specialised diet consisting almost entirely of *Ecklonia radiata*.

On a local scale, those species that do occur in the region often exhibit consistent patterns of abundance which reflect changes in either the physical or biological structure of the reef. Changes associated with depth or discontinuities in habitat type (eg at the rock sand boundary, or between the kelp and the algal turf zone) are particularly marked. Lincoln-Smith & Jones (1995) reported that the distribution of many species is determined by the topography of the rocky substratum. Greater physical complexity is associated with higher densities perhaps because refuges from predators are more abundant. The presence or absence of kelp and other large macroalgae is also a key determinant of assemblage structure for reef fishes. For example, experiments on reefs show clearly that a number of herbivorous species which feed preferentially on foliose red and green algae occur in higher densities in cleared patches within kelp beds (Jones 1992).

The life-cycle of most reef fishes includes a pelagic larval phase of between one and three months, during which time there is considerable potential for dispersal.

Reef dynamics

One feature of all ecological systems is that they are naturally variable on a range of spatial and temporal scales. For the casual observer the scales of spatial variability are easy to appreciate – a dive on a reef or video footage can easily demonstrate that reefs are not uniform and that fauna and flora are aggregated into patches of varying sizes. What is more difficult to appreciate is the temporal aspect of this variability because it is rare for repeated observations to be made at the same location over long time scales. Nevertheless there are many well documented cases of large scale natural variability, such as the removal of large kelp patches due to storms and the subsequent suppression of recruitment through changes in urchin behaviour. Alternatively, urchin die back due to disease or predation may result in a shift back to a kelp dominated system. Clearly the possibility of such variation cannot be ignored when attempting to assess the extent to which human activities are responsible for observed changes.

Algal communities

Although a significant amount of information exists on the composition of the phaeophycean (brown algae) dominated macro-algal communities typical of reefs in southern Australia, there have been few attempts to synthesise or summarise this knowledge in order to extract unifying principles relating to the dynamics of these systems. Underwood and Kennelly (1990) undertook a critical review of the literature but their focus was very much directed to what could be definitively concluded rather than what had we learnt from work to date. Similarly, Scheil (1990), reviewed the status of knowledge on macro-algal assemblages in New Zealand. The conclusion common to both of these studies was that much work remains to be done to develop a knowledge of the processes responsible for structuring these communities in Australasia. Scheil (1990) also argued, (as do Andrew and Mapstone 1987) that the nature of interactions between algae and their environment (including both the biotic and abiotic components) can only be understood with reference to the life history and phenological traits of particular species. This emphasises the need for more extensive studies on the basic biology particularly of the dominant species.

Southern Australia is notable for the diversity of the macro-algal flora (Table 2; Womersley and King 1990). This diversity has long been recognized and reported with respect to the comparative biogeography of the region but the potential significance of diversity to the ecology of the these systems has not been discussed. It is this aspect of the southern Australian systems, perhaps more than any other, that underscores our need for caution in abstracting general ecological principles from other systems. The complex nature of interspecific processes that may emerge from more diverse systems should stand as a caution against the abstraction of generalities about ecosystem processes from systems elsewhere.[1]

An important local series of studies which correlated the structure of macro-algal communities with differences in substrata, depth, light and water movement, was conducted at selected sites along the exposed rocky coasts of South Australia (and offshore islands) by Shepherd and Womersley (1970, 1971, 1976, 1981). This work concluded

- that there are consistent patterns of vertical zonation on South Australian coasts,
- that this zonation can be divided into 3 levels but
- the zones may be characterised by a variable array of species at different sites.

These studies by Shepherd and Womersley have subsequently provided the basis for a number of comparable investigations throughout southern Australia. Such studies have variously supported the observations on the existence and composition of zones for a variety of habitats (eg Farrant and King 1982, Edgar 1983, Sanderson and Thomas 1987). Other studies have noted differences in either the number or composition of the zones (see eg May and Larkum 1981, Van der Velde and King 1984) and, in response, have questioned the general applicability of Shepherd and Womersley's proposals.

The review by Underwood and Kennelly (1990) also concluded that attempts to compare and contrast the structure of subtidal macro-algal communities from different regions in southern Australia have been seriously confounded by a lack of suitable replication or a consideration of any seasonal or inter-annual changes. Thus, conclusions about variability at either local or regional scales can only be speculative until appropriate studies have been developed which address these problems.

This should not however, detract from the fact that a large proportion of the work to date has provided valuable insights on the nature of macro-algal assemblages in southern Australia. This work allows us to define a series of assemblages (Table 3) which, when considered in terms of the life history processes of the dominant taxa, are likely to be a) persistent and b) qualitatively dissimilar in terms of the structuring processes.

---

[1] See for example the work on the chemical ecology of kelp, herbivore interactions in southern Australia compared with North America (Steinberg 1989, Van Altena and Steinberg 1992).

Table 3- Phaeophycean dominated community assemblages from southern Australia.

Assemblage	Dominant taxa	Functional classification of canopy	Reference
Ecklonia	*Ecklonia radiata, Acrocarpia paniculata, Scytothalia dorycarpa, Seirococcus axillaris*	Stipitate	Shepherd and Womersley (1970, 1971, 1976, 1981)
Macrocystis	*Macrocystis angustifolia or Macrocystis pyrifera*	Floating	Sanderson (1987)
Lessonia	*Lessonia corrugata (+/- Xiphophora gladiata)*	Stipitate-foliaceous	
Durvillaea	*Durvillaea potatorum*	Stipitate	Cheshire and Hallam (1989a, 1989b)
Cystophora		Foliaceous	(Shepherd and Womersley (1970, 1971, 1976, 1981), Cheshire *et al.* (1996a)
Caulocystis	*Caulocystis uvifera* often forming mixed assemblages with *Cystophora spp.* or *Sargassum spp.*	Foliaceous	
Sargassum		Foliaceous	Shepherd and Womersley (1970, 1971, 1976, 1981), Cheshire *et al.* (1996a)
Phyllospora	*Phyllospora comosa*	Floating	
Xiphophora	*Xiphophora gladiata*	Foliaceous	

Sessile / sedentary fauna

A considerable body of literature has described the characteristics and dynamics of South Australian hard substratum systems (eg Butler, 1986, 1991; Kay & Butler, 1983; Keough, 1984a,b). It is clear from this work that although community composition varies substantially across both space and time, the scale of variation is local and large-scale community characteristics at a site remain more or less constant over relatively long periods and large distances. For example, Kay & Butler (1983) showed that although 20 - 40% of the occupants of a jetty piling may be eaten, outcompeted, or overgrown within 3 months, the overall species composition and relative abundances in these assemblages on a given jetty were roughly constant for more than two years. Similar small-scale dynamics have also been noted by Keough (1984a,b), for communities encrusting *Pinna* shells. Butler and Connolly (1996) in examining communities developing on a new jetty found that it may take a long time for this kind of large-scale "stability" to develop. Thus, these small-scale differences in community composition form a spatially and temporally dynamic mosaic. The dynamics of hard-substratum communities on jetty pilings and *Pinna* shells can be extended, with caution, to apply to communities on more expansive natural substrata such as rocky reefs (Butler 1995). To date, however, there have been no comprehensive studies to address variation in faunal community structure on South Australian rocky reefs (this contrasts with the work by Collings 1996 which comprised an extensive study of the spatial and temporal scales of variation in macroalgal communities in this region).

The rocky reefs in Gulf St Vincent are relatively isolated from each other and the majority of sessile fauna which typify these reefs reproduce by dispersive larvae. Consequently some of the species on each reef have "open" populations in which recruitment rates are independent of local adult fecundity (*sensu* Roughgarden *et al.* 1985). Other species in these populations have their own local recruitment, growth and mortality rates and then collectively behave as a metapopulation (Hanski 1992; eg. Davis & Butler 1989). For example, recruitment rates onto the pilings of jetties in Gulf St Vincent have been found to vary significantly both seasonally and interannually, but some jetties show consistently higher levels of recruitment over periods of several years (Butler, 1986, 1991). These rates will be influenced by local dynamics, such as migration between patches, so that although a species may thrive on some reefs it may be displaced on others. Rates of growth and mortality will also vary temporally, however the dynamics of the system are such that on average the populations of a given species may be maintained within a region even though local extinctions may occur (Butler & Chesson, 1990).

This variation presents problems when attempting to assess the "health" of a reef system. Clearly, some variability is natural, and might indeed be a fundamental component of the mechanisms maintaining biodiversity in the system. Our problem is to detect changes, against this background or natural variability, which represent deterioration of the system.

The factors which bring about variability in community structure of temperate reefs are well documented, and include flow rate, turbidity, shade, availability of food, recruitment, competition, and predation (see Butler 1995). These factors, and how they influence the community structure on South Australian rocky reefs are considered below.

Many of the sessile individuals on subtidal reefs require access to the water column in order to feed, hence space and access to flowing water are probably the primary factors governing the distribution of these species. Levels of suspended food are low in South Australian waters (Butler, 1995), therefore suspension and filter feeders have adapted to processing large volumes of water. Passive suspension feeders tend to be colonial species and occur commonly only in areas of relatively high flow (eg gorgonians). Erect or branching species which feed actively, but within relatively weak currents (eg bryozoans), are also found primarily in areas of high to moderate flow. In addition to their dependence on flow rates these species are especially sensitive to suspended sediment loads. If these rise too high, feeding may be compromised. Moreover, if sedimentation rates are high, the feeding apparatus may become clogged. Consequently the interaction between turbidity and flow rate plays an important role in determining local distributions of such species.

Active filter feeders, such as sponges, may supplement their feeding currents with the aid of ambient currents. Different species have different optimum conditions of water movement, for example ascidians are less dependent on ambient flow conditions than other taxa which are sensitive to changes in the suspended sediment load, and undue sedimentation can lead to clogging of the filtering apparatus and death (Rogers, 1990). Consequently in areas of low flow these species tend to inhabit near-vertical or overhanging substrata where sedimentation rates are low. Tolerances of different species vary widely, for example, the ascidian *Botrylloides leachii* is common in areas of high flow and wave surge, whereas *Ciona intestinalis* is only found in the most sheltered locations.

Keough and Butler (1995) noted that areas of high flow tend to be characterised by colonial species while in low flow areas unitary organisms dominate. Unitary species such as the bivalve molluscs *Pinna*, *Ostrea*, and *Mytilus*, are probably the least influenced by flow and sedimentation rates. These species generate sufficient internal flow to be able to grow in almost any conditions and have elaborate mechanisms to clear sediment from the filtering apparatus. *Mytilus*, for example, will often grow in dense beds on near-horizontal substrata even in areas of relatively high sedimentation rates.

Perhaps the greatest impact of water flow and sedimentation are through their effects on reproduction, larval dispersal, settlement, and recruitment. On average, areas of high flow will be exposed to greater numbers of potential settlers. However, larval settlement preferences for regions of particular flow characteristics have recently received much attention (eg Mullineaux & Butman, 1990; Pawlik & Butman, 1993), and it has become clear that many species actively select certain flow regimes (eg Wethey 1986; Havenhand & Svane, 1991). Thus, flow rate may often determine not only the numbers of larvae in a given location but also the numbers of larvae choosing to settle there. Again, turbidity and flow rate interact as the larvae of many species common to rocky reefs actively avoid settlement on upward-facing surfaces which may lead to early mortality caused by sedimentation and/or algal overgrowth (Svane & Young 1989).

Competition from algae plays a major role in determining depth distributions of sessile species on temperate reefs however secondary effects such as shading may also be important (Butler, 1995). The upper few meters of any reef system are almost invariably dominated by macroalgae, and while that canopy may substantially modify the understorey environment (and hence the associated faunal composition; Duggins and Eckman 1994) it is only below this algal zone that the sessile fauna begin to dominate.

Competition among the sessile fauna is primarily restricted to competition for space (Butler 1995). Here sub-dominant species such as barnacles and tubicolous polychaetes survive by virtue of their high recruitment rates and ability to rapidly colonise even small patches of available free space (Keough, 1984a; Butler 1991), while slower growing dominant species (eg sponges *Mycale* and *Clathria*, and colonial ascidians) may overgrow their competitors, but have low recruitment rates and are more susceptible to periodic disturbances such as storm-induced wave action. Within this dominant group, state-dependent interactions occur such that no single species is consistently dominant (Keough, 1984a). Consequently the competitive dominance of sponges and ascidians on southern Australian hard substrata is countered by disturbance and rapid recruitment and colonisation by sub-dominant species. Again, the importance of spatial and temporal variability is apparent, this time in maintaining diversity in these systems.

Fish

Many experimental studies indicate that fish taxa are an integral component of coastal reef systems in temperate waters and changes in reef habitat are almost certain to result in changes in the fish fauna. One might imagine, therefore, that this can be used as an index of reef status. However, the proximate cause of changes (ie a change in habitat characteristics) will often provide a more direct and interpretable index of change. One notable feature of the fish fauna which is particularly problematic is the considerable spatial variation that occurs between sites, within regions and the inter-annual variation that can occur even on a single reef (Kingsford, 1989). Indeed Lincoln-Smith and Jones (1995) note that ‘one of the greatest challenges in the study of reef fishes is to understand the causes of the huge natural variation we see in abundance from place to place and time to time’. With respect to the assessment of the status of reefs, this poses particular problems because detecting any signals that are indicative of undesirable trends from the noise of natural variability is likely to be difficult in short-term studies. Although changes in abundance can sometimes be associated with changes in the habitat (eg the loss of kelp due to storms), it is variable recruitment of juveniles from the pelagic larval phase that is the major source of variability (Lincoln Smith *et al.*, 1991). With respect to longer term monitoring, there are few data available from which we can document long-term trends.

The fish fauna itself can have controlling effects on other components of the system through predatory interactions. For example, Andrew & Jones (1990) have shown that the herring cale (*Odax cyanomelas*) can have a seasonal impact on kelp (*Ecklonia radiata)* stands whereby changes in the behaviour of females led to the clearing of kelp in approximately the same locations between August and October in each of three years. Although the generality of such effects remains undetermined it is clear that such behaviours have implications for the interpretation of change in temperate reef systems.

Reef Health

The analogy between the health of human beings and health of ecosystems is one that is finding growing acceptance. It is worth considering, however, how far the analogy can be pushed. Calow (1992) identifies two forms: a weak form in which the term health simply signals normality (implying of course that ill-health signals abnormality) and a strong form in which health defines a condition that is favourable (ie optimal) for the functioning of the system. In this latter case the optimal state is actively defended by homeostatic processes. Ideally, a healthy state should be the same for all reef systems since only then can objective health criteria be defined.

Calow (1992) considers the degree to which the strong form of the analogy can be applied to an ecosystem, arguing that for such a form to be valid the existence of a controlled ‘optimum’ state for a system is necessary. Control is occurring if systems remain unchanged with perturbation (ie they resist it), or if they have the ability to return to their previous state after perturbation (ie they are resilient to it). Such system behaviour can be achieved by active feedback (usually negative) control in which the system moves towards a future ‘goal state’ that is programmed into it. Alternatively, the dynamics of the interacting parts might simply lead to an equilibrium state which is not achieved by a goal directed mechanism (program) but is achieved passively - it is difficult to argue that this latter passive control is of the kind that is required if we are to accept the strong form of the analogy. Moreover, Calow argues (correctly in our view) that it is unlikely that component parts of ecosystems are programmed for active control that will lead to a ‘balanced economy’ in the ecosystem as a whole. This is because natural selection on individuals and populations will favour those that maximise command of resources even if it is at the expense of the rest of the ecosystem. Thus, the strong sense of the analogy with health in humans is flawed.

Although the strong form of the health analogy is invalid can we use the weak form usefully? In other words is there a definable ‘normal’ or baseline state which would constitute a healthy system? One approach to defining such baselines might be to list the properties of putatively pristine systems (ie those which have been unaffected by human activity). This is analogous to what happened in early medicine where physicians sought to correlate body states with conditions of health and ill-health. However, a key point to make in this respect is that the structure of the biotic components of a system (ie biodiversity in all senses of the word) varies with ‘natural’ environmental conditions (see below). Thus, to use some ecosystem state as a baseline from which to judge the effects of our activities, requires a clear specification of the relationship between structure and environmental factors. Unfortunately, we are far from possessing such understanding.

More fundamentally, we feel there is some difficulty with equating ‘normality’ or ‘health’ with the absence of human influence. This is because it implies that affected systems are inherently abnormal or ‘un-healthy’. This is not to say that the changes made to systems are desirable or morally defensible – clearly many of them are not - it is simply that there is no a priori reason why a system we have affected should be viewed as being any more or less healthy than one in which our influence is minimal. This stated however, it stands to reason that the only impacts we are likely to be able to control are those that are the result of human activities. We should therefore attempt to separate these from other impacts and control them according to our perceived common goals, whether they be commercial, recreational, aesthetic or spiritual.

There are of course inherent difficulties in making these kinds of judgements. Consider a reef that has been surveyed and shown to have rich kelp beds and a diverse fish and invertebrate fauna. In others words, the reef is in a state that most people would be happy to describe as healthy. Now imagine that before a second survey a year later (and unbeknown to the surveyor) a storm removes most of the kelp from the reef. At the same time there was a very successful recruitment of sea urchins and that these circumstances conspired such that most of the algal cover was removed and the reef became a depauperate urchin barren. Would the surveyor be correct to call the second state less healthy than the first? It is easy to see that it is less desirable from a human perspective but if the criterion for poor health is a state not engendered by human activities it clearly does not qualify. Although it is powerfully emotive to call undesirable system states unhealthy, it seems to us to be more reasonable and intellectually honest to consider reefs as being in desirable or undesirable states. This does not of course remove the difficulty of determining the controls on those states and the degree to which our actions can effect changes, but it does remove some of the hyperbole surrounding the issues.

Determining the causes of change

The link between defining states which are more or less desirable and identifying the processes which have created these states is generally beyond the scope of any short-term study. It either requires a series of long-term mensurative experiments or a series of targeted experiments designed to investigate specific processes and controls.

There are a number of examples where supposed early warnings of pathological conditions have proved deceptive. Rapport (1992) for example, cites the sudden die-back of macro-algal beds along the Finnish coast in the late 1970's. This change was first thought to be indicative of coastal wide environmental degradation resulting from eutrophication. In the mid 1980's, however, algal beds started to recover despite continued high nutrient loadings suggesting a more complex chain of events and a coastal system that was under less threat than was originally envisaged (Ronnenberg *et al.*, 1985).

Part 2 - Approaches to assessment

Assessment of ecosystems generally comprises the measurement of a variety of parameters which, when considered together, describe the physical, chemical and biological properties of the system. In any given case the choice of parameters is dependent upon the specific questions being asked and the extent to which the process of making the measurements can be allowed to impact upon the system.

Abiotic parameters

Whereas it is recognised that water quality *per se* is a major determinant of community structure it is not the aim of this work to either define water quality standards or to attempt to make measurements of water quality for comparative purposes. The objective of this work is to undertake an assessment of the physical structure of the reef system and the status of the associated biota. Spot measurements of water quality (based on physical and/or chemical parameters) are rarely meaningful and if an assessment of water quality is required this should be undertaken using an appropriately structured sampling program which deals with both the spatial and temporal scales of variability in these parameters. Therefore, we do not propose to make detailed measurements of water quality in this system. The following provides an overview of the physico-chemical parameters we intend to measure noting that apart from physical structure we are only concerned with assuring that the water quality is generally comparable across the systems we study.

Physical structure/habitat

Mapping of the physical structure/habitat can be done by diver census, 35 mm photography, video transects, and actual measurement of the substrate. Limitations of these methods are as follows:

- diver census: qualitative and highly variable
- 35mm photography: quadrats limited by size and number of quadrats, water clarity and quality of images
- video transects: (useful but limited by water clarity, image quality, and variances in distance from substrate
- actual substrate measurements: time consuming and difficult to extrapolate

Aronson *et al.* (1994) provide a clear example of one method of substrate measurement using a complexity index C, calculated as $C = 1 - d/L$, where d is the horizontal distance covered by a conformed chain (measured against the transect tape) and L is its length when fully extended (eg. Aronson and Harms 1985; Hubbard *et al.* 1990 – and others in Aronson *et al.* 1994). This chain can be any length with links approximately 15 mm long. The chain is placed against the substratum and conformed to the contours of the substratum. The index gives an estimate of topographic complexity.

Biotic parameters

Recent studies have suggested that in ecosystem assessments the time and financial resources available will not permit a detailed assessment at the level of species (and often the species are not known and so cannot be adequately defined) and that this level of detail may not be necessary. Rather, data are collected on groups of species which may represent higher taxonomic groupings (such as Families, Orders or even Classes and Phyla) or alternatively lifeforms (which group unrelated species based on the role they play within a community rather than on their phylogentic affinities). This approach of using coarsely resolved taxonomic groupings has been applied in studies elsewhere without much loss of information (Littler, 1980, Warwick 1993). Such approaches are particularly prevalent in marine studies where the species diversity is very high and comprised of many undescribed taxa.

In this report we refer to "group" which may represent any individual or collection of species which for the purposes of the assessment are being treated as a single group.

In the following discussion we provide information about a range of parameters which can be used in ecosystem assessment and a brief discussion of the use of these parameters.

Units of measure in the assessment of biota

Sedentary and sessile biota

The sessile and sedentary fauna of reefal systems have been quantified by a wide variety of methods (Coyer & Witman 1990). The most common of these are measures of presence /absence, percentage cover, or abundance per unit area.

Presence-absence

Presence-absence can be used to assess whether or not a given taxon exists within a sample area. The method is generally non-destructive and observations can be made *in-situ*. The data obtained simply report whether or not a taxon is present and there is no quantification. In some cases assessments are made on a per sample basis and this will then provide a quantitative measure of frequency (by assessing the number of samples in which a given taxa is found). In such cases however, this does not differentiate between a taxon found abundantly in all samples compared to a taxon found in low abundance in all samples.

Presence-absence data are usually collected using quadrats of known size.

Percentage cover

Cover is generally applied to assessments of biota which are sessile and involves an estimation of the proportion of any given area occupied by each taxon within a community. This method is generally non-destructive and is commonly used in vegetation analysis but is equally appropriate to the assessment of colonial invertebrates such as zooanthids, corals, sponges and some ascidians.

Whereas data can be collected on multiple levels within any one community (and percentages may consequently sum to more than 100%) this is not generally done.

A variety of techniques will provide data of this sort including line intercept transects, quadrats or visual censuses.

Abundance

Counts of organisms within samples will provide a measure of abundance generally reported as n. m^{-2}. Counts can be made *in-situ* or alternatively the community can be harvested and counts made in the laboratory. The method is quantitative but does not discriminate between large and small taxa, nor is it useful for colonial or very small organisms (due to difficulty in making or standardising counts). Abundance data are usually collected using quadrats of known size.

Biomass

Measurements of biomass (expressed as g dry weight. m^{-2}) provide one of the best indications of the relative amounts of different taxa present but are necessarily destructive when applied to any sessile organism. Further, for organisms with large inorganic components such as hard corals, some sponges and some algae, the measurements need to be adjusted to account for the non-living biomass.

Biomass measures are usually made based on harvesting all biota from within quadrats of known size.

Sampling design

Measures of abundance or cover are usually obtained for quadrats (or series of quadrats) randomly spaced along transects laid out on the substratum. Transects may run within a given habitat type and depth stratum (eg Aronson *et al.*, 1994) to provide stratified random estimates of community composition, or transects may be run across habitats and depths to provide an overview of the composition of a reef (eg Bouchon 1981). Community composition within a quadrat may be determined by direct observation (eg de Vantier, 1986), underwater photography (eg Svane & LundŠläv 1981) or underwater video (eg Aronson *et al.*, 1994).

Aronson *et al.* (1994) discuss the respective merits of these methods and suggest that video transects with subsequent point-quadrat analysis of random video frames is the most resource efficient method. Whilst lacking the optical resolution required to identify species smaller than approximately 2cm, (a criticism not true of 35mm photography), video is rapidly obtained and requires no chemical processing. This method also has the advantage of permitting the observer to review an entire sequence of video in search of rare species. However, Cheshire *et al.* (1996a,b) concluded that video assessments of soft-bottom biota in Port Lincoln South Australia did not provide data with sufficient resolution to detect anything other than gross changes in community structure and importantly failed to detect changes which were clearly evident from the data obtained by divers.

Kinzie & Snider (1978) found that several rapid assessments of percent cover yielded a far more accurate picture of the true nature of reefal systems than an intensive localised sampling regime run for the same period of time. This was primarily a result of the spatial heterogeneity of reefs, and the consequent requirement to obtain statistically meaningful levels of replication.

Mobile biota

Visual census

On shallow rocky reefs the most common means for estimating fish abundance is visual census by divers. The details of a standardised methodology for surveys on coral reefs are given in English *et al.* (1994) and it is intended that this methodology be adapted to the local situation. In brief, surveys are conducted along transect lines, with the same transects being used for the assessment of fish as for the algae and benthic fauna. Two basic assessment methods can be employed: the first records differences in the reef fish assemblage using abundance categories and the second focuses on key species to assess their abundance and population size structure.

Critical assessment of a visual census approach

One difficulty with this approach is that it requires divers who are experienced in the taxonomy of the local fish population and who are able to rapidly identify and record species during surveys. However, since the object of this study is to develop rapid and cost effective ways to assess the status of reefs, it is perhaps unreasonable to expect that the costs of employing personnel with such expertise can be borne on a continuing basis. We therefore propose to adopt the latter method and focus attention on a few key fish taxa which are of interest, either because they are subject to recreational or commercial fishery exploitation, or because they represent an important ecological component of the reef system (this latter requires specific definition but typically may relate to numerical dominance or alternatively keystone roles). The final decision about which taxa should be selected will depend on preliminary surveys.

Methods for data analysis

The methods detailed above will provide data on the "amounts" of various taxa within a community. Once obtained, these data can be analysed in a variety of ways which, in many cases, will include a comparison of samples based not solely on individual taxa but often based upon a broad comparison of all taxa present. The following provides an overview of some approaches.

Composite indices

Taxa richness

Assessment of taxa richness is generally conducted by providing a count of the number of taxa in a given area. This may be done at the level of the sample (eg taxa.m^{-2}) or at the level of an entire reef.

Taxa diversity

Diversity measures not only the number of taxa present (as in richness) but also the homogeneity of the community in terms of the relative abundances of taxa. The most commonly used diversity index is the Shannon-Weaver (or Shannon-Wiener) Diversity Index (Shannon 1948) which may be calculated at the level of the sample or at the level of the site (reef).

Evenness (Equitibility)

Pielou's index of equitibility assesses the evenness in the distribution of biota in terms of their numerical abundance. Equitibility varies between 0 and 1 and provides a measure of the extent to which a community is dominated (in abundance or biomass terms) by one or a few taxa (low value) or whether all taxa are more or less equally abundant (high value).

Field implementation

Line Intercept Transecting

The Line Intercept Transect (LIT) method provides a basis for obtaining quantitative measures on the percentage cover of sessile benthic organisms without destructive harvest. A detailed discussion of the appropriateness of the LIT method to the assessment of macro-algal dominated marine benthic systems has been provided by Turner (1995). In summary, LIT has traditionally been used only in terrestrial (eg Webb *et al.*, 1970) or marine systems (eg Reichelt *et al.*, 1986) in which one can assume that individuals are not overlapping (Muttlak and Sadooghi-Alvandi, 1993, Lucas and Seber, 1977). On the face of it this would not apply to macroalgal dominated systems in which the community is multi-layered. Nevertheless, in a detailed study based around reefal systems at West Island South Australia, Turner (1995) demonstrated that LIT was faster and just as effective in determining community structure as the more generally used destructive harvesting proposed by Littler and Littler (1985). Further, given that LIT is non-destructive it was felt that, particularly in sensitive habitats, this provided a much more responsible approach to community assessment.

Line Intercept Transecting involves placing a tape over the substratum and recording the points along the tape where the underlying biota changes (Figure 3). The resulting data set (Table 4) comprises a list of the transition points between dominant taxa/lifeforms which can be used to calculate the actual distance along each transect occupied by the various groups. From these, we can evaluate the percentage cover for each group for the transect (or portion thereof) under consideration.

Figure 3 - Schematic view of benthos showing LIT. Table 4 illustrates the corresponding data set that would be created from these data.

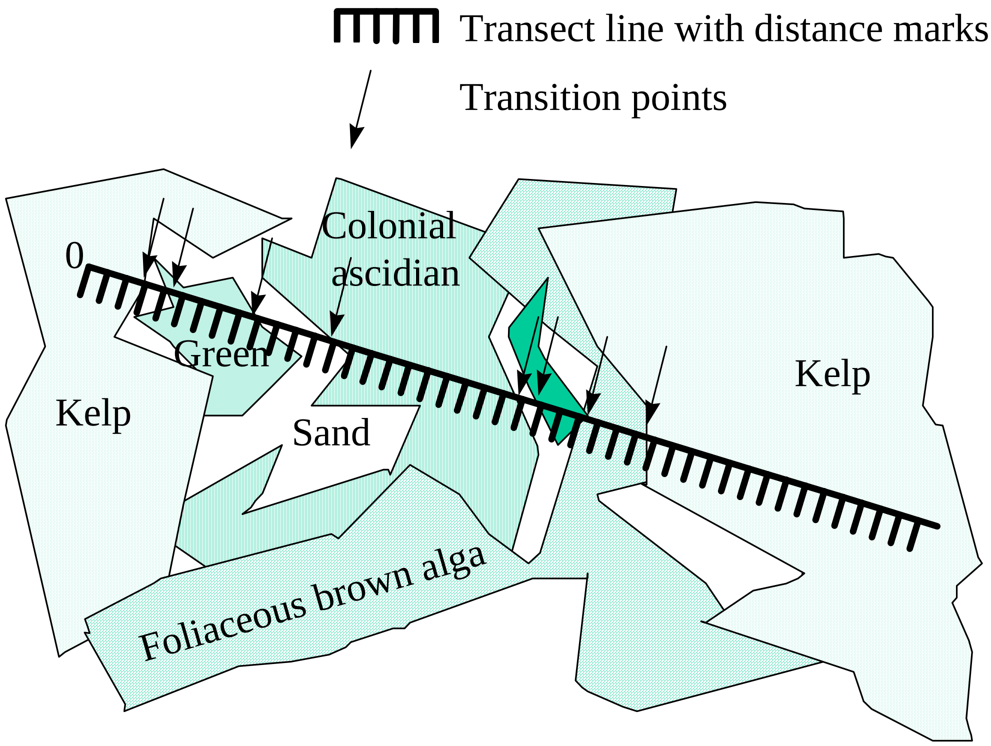


Table 4 - Typical data sheet from Line Intersect Transect. Transect - id number to provide unique identifier for each transect; **Portion** - A=0-10 m, B=10-20m; **Transition -** point on transect at which dominant lifeform changes; **Lifeform -** dominant lifeform found on transect from the last transition point to the current position on the transect.

Transect	Portion	Transition	Lifeform
1	A	3	Kelp
1	A	4.5	Sand
1	A	9	Membranous green alga
1	A	13	Sand
1	A	22.5	Colonial ascidian
1	A	23.5	Sand
1	A	26	Foliaceous red alga
1	A	29.5	Foliaceous brown alga
1	A	44	Kelp

In terrestrial systems or on coral reefs where there is little movement of organisms this method is very easy to implement. In temperate marine systems where the dominant lifeform includes macroalgae, which move about due to surge and current action, this approach can be problematical. Turner (1995) solved the problem by using the tape as a general guide for the positioning of the transect and used a 1 m weighted ruler (1.5 kg) to pin down the biota along this transect. This provided a two dimensional "snap shot" of the biota and allowed the data to be recorded in the normal way.

Destructive Quadrat Harvesting

Destructive quadrat harvesting has become the most common method for surveying subtidal marine algal systems because it minimises the amount of time spent under water (Littler and Littler, 1985). The technique involves the placement of quadrats on the substrate and lifeforms of interest within the quadrat are then harvested and placed in labelled bags. These bags are returned to the laboratory where they can be sorted into taxa and counted and/or weighed to obtain a quantitative measure per unit area (Littler and Littler, 1985).

In many cases the choice of the sample size is based almost solely on intuition and tradition rather than on a quantitative assessment of the appropriateness of the sampling regime to extract faithful information about the system (Andrew and Mapstone, 1987). The most widely used quadrat size in marine systems is a square quadrat of 0.25 metre area (Andrew and Mapstone, 1987), however there is a lot of deviation from this in the literature. For example, previous work in South Australia by Shepherd and Womersley (1970) used a 0.1 m^2 round quadrat whereas Collings (1996) used a 1 m^2 square quadrat.

In the case of the LIT the transect length was based on an extensive study by Turner (1995) which included a detailed assessment of the quantitative biases associated with this form of data collection. In the case of the quadrat data we have chosen a 0.125 m^2 quadrat based on preliminary field trials which indicated that this was the optimal size to use in combination with the LIT approach. The quadrat size was therefore chosen to provide for rapid assessment of those groups which were not frequently encountered on the LIT line.

A major problem with destructive harvesting is the resultant damage to the system (De Wreede, 1984). While destructive quadrat sampling may be an effective way of surveying an area, it's impact on that area may outweigh it's usefulness. This is particularly so in areas of high conservation status (eg areas which contain rare species), or areas which are to be repeatedly sampled (such as those subject to ongoing monitoring). While small scale disturbances are an important part of patch dynamics in marine communities (Scheil and Foster, 1986), destructive sampling is more likely to confound results and lead to changes in community structure through changes in the disturbance regime (Turner 1995). We do not believe that the impact of destructive harvesting is acceptable and this procedure will not be employed in this study. Quantitative measures will therefore be restricted to counts of abundance and estimates of percentage cover. Biomass data will not be collected.

Non destructive quadrats

Non-destructive quadrat sampling is done by placing a quadrat of given size over the substratum. All unitary organisms within the area circumscribed by the quadrat are then counted to provide a quantitative measure of the abundance of each taxa/group within that area. Results are then presented as number per unit area (generally $n.m^{-2}$).

Fish visual census

Fish counts are performed by a diver swimming for a specified distance (typically 50 m) during which time all fish within 3 m of the diver (either side and above) are identified and counted. Data are recorded on an underwater slate as the diver swims. Care needs to be taken to ensure that swims are made at the same speed (i.e. searching time is constant). A major problem with this method is in dealing with fish which are variously curious and will swim up to divers (eg. territorial leatherjackets or damsel fish) versus those which take flight before the diver gets near enough to enable an accurate identification (such as many schooling species).

Taxonomic considerations and resolution

Macro-algae

Individual non-clonal forms

Macro-algae can be broadly classified into 3 major phylogenetic groupings; red algae (Rhodophyta), brown algae (Phaeophyta) and green algae (Chlorophyta). Across these 3 divisions there is a very high specific diversity in southern Australia (probably >2,000 species) many of which (particularly in the Rhodophyta) are undescribed.

Most taxa on southern Australian reefs can be assigned by trained persons to genus (in the field) although many of the red algae require the collection of voucher specimens (and depending on reproductive status still may not be identifiable even to order).

Quantitative assessment can be performed using either biomass, abundance or percentage cover.

Encrusting, turfing and clonal forms

Algae of all 3 divisions fall into these classes; surveys cannot routinely quantify either the abundance or biomass of these taxa. If a quantitative estimate is required this would generally involve an estimation of percentage cover. By and large taxonomic resolution would be restricted to the following major groups:

- Encrusting coralline algae
- Encrusting brown algae (generally Ralfsiod crusts which also include the alternate phase of otherwise upright species such as *Scytosiphon*)
- Turfing red, green and brown algae (generally fine filamentous forms < 15-20 mm high)
- Foliaceous (<300 mm high) clonal forms (which could be classified separately) but in which individuals cannot be defined (includes taxa such as *Asparagopsis* spp.)
- Thick leathery brown and red algae including most of the dominant browns (eg. *Cystophora spp. Sargassum spp. Ecklonia radiata*) and the more robust red algae (*Osmundaria*, *Lenormandia*).

Lifeforms

Turner (1995) demonstrated that for broad scale assessment one could use lifeform classifications and still obtain valuable information on the structure of macro-algal communities on reefs at West Island (South Australia). This work classified algae into the a series of pseudo-taxa or lifeforms (Table 5).

Table 5 - Modified list of lifeforms based on those identified by Turner (1995) for classification of algal communities - these are also comparable to those identified by Littler and Littler (198?).

Lifeform	General description	Typical member taxa
Encrusting browns	Ralfsiod browns which includes the alternate phase in the life-history of some otherwise upright forms.	*Ralfsia, Scytosiphon* (gametophytes)
Encrusting reds	Comprises mainly corallinaceous red algae	*Porolithon, Lithothamnion*
Filamentous turfs	Smaller (<20 mm) filamentous forms	*Ectocarpus, Sphacelaria, Cladophora, Polysiphonia* and other ceramiaceous reds
Foliaceous browns	Bushy non-membranous browns	*Halopteris, Zonaria, Padina, Lobophora, Lobospira*
Foliaceous greens	Bushy non-membranous greens	*Caulerpa, Codium, Apjonia*
Foliaceous reds	Bushy non-membranous reds	*Plocamium, Phacelocarpus, Asparagopsis*
Membranous browns	Flattened membrane forms generally relatively transparent in general appearance	*Colpemenia, Hydroclathrus, Scytosiphon* (sporophytes)
Membranous greens	Flattened membrane forms generally relatively transparent in general appearance	*Ulva, Enteromorpha*
Membranous reds	Flattened membrane forms generally relatively transparent in general appearance	*Kallymeniaceae*
Thick fleshy browns	Generally the larger canopy dominants	*Sargassum, Cystophora, Ecklonia, Seiroccocus, Scytothalia, Acrocarpia.*
Thick fleshy reds	Generally robust persistent forms	*Osmundaria, Lenormandia*

This form of classification can be easily applied even by untrained observers and has been shown to provide good quality information on the structure of communities which is relevant to aspects of habitat quality and productivity.

Seagrasses

Seagrasses are generally not associated with hard substrata but may be found in pockets of sand on otherwise consolidated substrata. Generally these are clonal taxa but are best quantified using abundance of vegetative shoots.

Non-colonial sessile fauna

These include a variety of taxa including bivalve molluscs, solitary ascidians and some sponges. Although in reality sponges are modular ("colonial") many species form discrete entities which can be counted as individuals (eg many of the phototrophic foliaceous dictyoceratids such as *Phyllospongia* spp. and some of the axinellids such as *Cymbastela* spp.).

Colonial sessile fauna

Includes a variety of taxa including hard corals, many sponges, colonial ascidians, bryozoans, hydrozoans and zooanthids. The most effective form of assessment is to provide an estimate of percentage cover.

Sedentary fauna

Major taxa include ophiuroids, crinoids, urchins crustaceans (especially crabs), and gastropod molluscs. Assessment should provide an estimate of the number per unit area

Mobile fauna

Comprises fish and other fast moving vertebrates which, primarily because of their potential for movement they cannot be sampled using static sampling methods.

Bacteria

We do not propose to deal with microbial flora under this study.

Part 3 - Proposal for assessment of Gulf reefs

General Recommendations

Reefs vary in both space and time and that the scales of this variation is also variable. If our aim is to assess the status of a reef, we need to develop a methodology which will enable identification of major changes in community composition through both space (reef to reef) and time (months to years) without being confounded by small scale natural variability.

To address spatial patterns, it is proposed that reefal systems in Gulf St Vincent be surveyed in a stratified random pattern. Patches of substratum to be sampled are chosen randomly within a given reefal habitat (eg. depth stratum, aspect [vertical or horizontal], and exposure [seaward or shoreward]). Sessile and sedentary biota should be sampled visually by divers and only the major structural elements and known functional groups be quantified.

Abiotic parameters

Physical structure/habitat

The topographic complexity of the habitat will be quantified by determination of the Complexity Index, C, (Aronson *et al.* 1994). A fine brass or stainless steel chain 5 meters long and composed of 15 mm links will be used to determine d in the equation $C = 1 - d/L$, where d is the horizontal distance covered by a conformed chain (measured against the transect tape) and L is its length (5 m) when fully extended (eg Aronson and Harms 1985; Hubbard *et al.* 1990; - and others in Aronson *et al.* 1994). The chain is carefully conformed to the substratum at some point along the transect line.

Biotic parameters

The following is a summary of the methods proposed for the Reef Status surveys to be undertaken as a part of this program.

Sessile biota

For sessile biota including both macroalgae and sessile fauna we propose to use the modified LIT approach described by Turner (1995). A 20 metre tape is layed out as straight as practicable along a depth profile across the substrate as a guide. A weighted ruler is then used to pin the algae in place and the diver records the points along the ruler where the dominant group changes (referred to as transition points). Where there are areas in which a measure cannot be obtained (such as on an uneven substrate), that particular section is recorded as missing data.

Divers move along the tape, placing the ruler for each metre of the transect, and record the relevant details on waterproof slates. Where possible the species is noted but in all cases the lifeform must be recorded.

The most efficient method of transecting is to use two divers on the same transect line. One recording details from zero to ten metres and the other recording from ten to twenty meters. This allows for the safety of two divers in the water while making best use of each. As both divers are working on the same fixed line, they can easily find each other if the need should arise.

A competent diver with a reasonable knowledge of the species present is able to carry out the transecting procedure at a rate of 15 metres per hour. This means that two divers can complete the twenty metre transects in about 40 minutes. Obviously this period is specific to the study site, but it does serve as an indication of the time required relative to other methods.

Sedentary biota

Sedentary biota will be assessed using counts undertaken *in-situ* within a 0.125 m^2 quadrat (expressed as n. m^{-2}). Quadrats will be placed using a stratified random procedure along the 20 m LIT transect with eight quadrats per transect. Each quadrat will be placed with its lower edge adjacent to the LIT line and will be randomly located within each of the 10 m segments (0-9.5, 10-19.5).

Mobile biota

Mobile biota will be assessed using a visual census survey comprising a 50 m transect . Absolute counts or $\log_4$ abundance estimates will be made of all species present (Table 6).

Table 6 - Abundance categories used to count schooling species during fish visual transect surveys.

Abundance category	Number range
0	0
1	1
2	2-4
3	5-16
4	17-64
5	65-256
6	257-1024
7	1025-4096
8	>4096

References

Andrew, N.L. and Jones, G.P. (1990). Patch formation by herbivorous fish in a temperate Australian kelp forest. *Oecologia* **85**: 57-68.

Andrew, N.L. and Mapstone, B.D. (1987). Sampling and the description of spatial pattern in marine ecology. *Oceanography and Marine Biology Annual Review* **25**: 39-90.

Aronson, R. B., Edmunds, P. J., Precht, W. F., Swanson, D. W. and Levitan, D. R. (1994). Large-scale, long-term monitoring of Caribbean coral reefs: simple, quick, inexpensive techniques. *Atoll Research Bulletin* **421**: 1-19.

Aronson, R.B. and Harms, C.A. (1985). Ophiuroids in a Bahamian saltwater lake: The ecology of a paleozoic-like community. *Ecology* **66**: 1472-1483.

Blomqvist S. And Hakanson L. Areview on sediment traps in aquatic environments. *Arc. Hydrobiol* **91:1** 101-132.

Bouchon, C. (1981) Quantitative study of the Scleractinian coral communities of a fringing reef of Reunion Island (Indian Ocean). *Mar. Ecol. Prog. Ser.* **4**: 273-288.

Butler, A. J. (1986). Recruitment of sessile invertebrates at five sites in Gulf St Vincent, South Australia. *J. Exp. Mar. Biol. Ecol.* **97**: 13-36.

Butler, A. J. (1991). Effect of patch size on communities of sessile invertebrates in Gulf St Vincent, South Australia. *J. Exp. Mar. Biol. Ecol.* **153**: 255-280.

Butler, A. J. (1995) Subtidal Rocky Reefs. In: Underwood, A. J., Chapman, M. G. (ed.) *Coastal Marine Ecology of Temperate Australia*. UNSW Press, Sydney, p. 83-105.

Butler, A. J. and Chesson, P. L. (1990). Ecology of sessile animals on sublittoral hard substrata: the need to measure variation. *Aus. J. Ecol.* **15**: 521-531.

Butler, A.J. 1991. Effect of patch size on communities of sessile invertebrates in Gulf St Vincent, South Australia. *J. Exp. Mar. Biol. and Ecol.* **153**: 255-280.

Butler, A.J. and Connolly, R.M. (1996). Development and long-term dynamics of a fouling assemblage of sessile marine invertebrates. *Biofouling* **9(3)**, 187-209.

Calow, P. (1992). Can ecosystems be healthy? Critical consideration of concepts. *Journal of Aquatic Ecosystem Health*, **1**, 1-6.

Cheshire, A.C. and Hallam, N.D. (1989a) Methods for assessing the age composition of native stands of subtidal macro-algae: A case study on Durvillaea potatorum. *Botanica Marina* **32**: 199-204.

Cheshire, A.C. and Hallam, N.D. (1989b)Morphological differences in the southern bull-kelp (Durvillaea potatorum) throughout south eastern Australia. *Botanica Marina* **32**: 191-198.

Cheshire, A.C., Westphalen, G., Kildea, T., Smart, A. and Clarke, S. (1996b). *Investigating the environmental effects of sea-cage tuna farming. I. Methodology for investigating sea floor souring.* A report to the FRDC and Tuna Boat Owners Association. pp 44.

Cheshire, A.C., Westphalen, G., Wenden, A., Scriven, L.J. and Rowland, B. (1996a). Photosynthesis and respiration of a subtidal macroalgal community. *Aquatic Botany. (in press).*

Collings, G. and Cheshire, A.C. (1998). Composition of subtidal macroalgal communities of the lower gulf waters of South Australia with reference to water movements and geographic separation. *Australian Journal of Botany*. **46(4)** (in press).

Collings, G.J. (1989). *The relationship between canopy community structure and wave force in selected South Australian macroalgal dominated benthic communities*. Honours thesis, Department of Botany, University of Adelaide pp 138.

Collings, G.J. (1996). *Spatiotemporal variation of macroalgal communities of southern Fleurieu Peninsula, South Australia*. Ph.D thesis, Department of Botany, University of Adelaide pp 225.

Coyer, J. and Witman, J. (1990*). The Underwater Catalog: A guide to methods in underwater research*. Shoals Marine Laboratory and New York Sea Grant, New York. 72 pp.

Davis, A.R. & Butler, A.J. 1989. Direct observations of larval dispersal in the colonial ascidian *Podoclavella moluccensis* Sluiter: evidence for closed populations. *Journal of Experimental Marine Biology and Ecology* **127**: 189-203.

de Vantier, L.M. (1986). Studies in the assessment of coral reef ecosystems. In: *Human induced damage to coral reefs* (Brown, B.E. Ed). UNESCO Reports in Marine Science. **40**: 99-111.

De Wreede, R.E. (1984) Growth and age class distribution of Pterygophora californica (Phaeophyta). *Mar.Ecol. Prog. Ser*. **19**: 93-100.

Duggins, D.O. and Eckman J.E. (1994). The role of kelp detritus in the growth of benthic suspension feeders in an understory kelp forest. *Journal of Experimental Marine Biology and Ecology* **176(1)**: 53-68.

Edgar, G.J. (1983). The ecology of south-east Tasmanian phytal animal communities. I. Spatial organization on a local scale. *J. Exp. Mar. Biol. and Ecol.* **70**: 129-157.

Emmerson, L.M. (1992*). Recruitment dynamics of a fucoid dominated macroalgal community*. Honours thesis, Department of Botany, University of Adelaide pp 77.

English, S. Wilkinson, C. and Baker, V. (1994) *Survey Manual for Tropical Marine Resources*. ASEAN Australian Marine Science Project. Australian Institute of Marine Science: Townsville.

Farrant, P.A. and King, R.J. (1982). The subtidal seaweed community of the Sydney region. *Wetlands (Australia)* **2**: 51-60.

Hakanson L., Floderus S. And Wallin M. (1989) Sediment trap assemblages - a methodological description. *Hydrobiologia* **176/177:** 481-490.

Hanski, I. (1992). Coexistence of competitors in a patchy environment. *Ecology* **64**: 493-500.

Harvey, F. (1990). *A study of the interaction between the distributions of algal and gastropod communities in the intertidal zones of rocky shores of the Fleurieu Peninsula, South Australia.* Honours thesis, Department of Botany, University of Adelaide pp 119.

Havenhand, J. N., Svane, I. (1991). Roles of hydrodynamics and larval behaviour in determining spatial aggregation in the tunicate Ciona intestinalis L. *Mar. Ecol. Prog. Ser*. **68**: 271-276.

Hubbard, D.K., Miller, A.I. and Scaturo, D. (1990). Production and cycling of calcium carbonate in a shelf-edge reef system (St. Croix, USA Virgin Islands): Applications to the nature of reef systems in the fossil record. *J. Sed. Pet.* **60**: 335-360.

Jeffrey, S.W., Rochford, D.J. and Cresswell, G.R. (1990). Oceanography of the Australasian Region. In M.N. Clayton and R.J. King (Eds.), *Biology of Marine Plants.* Longman Cheshire. 243-265.

Jones, G.P. & Andrew, N.L. (1990). Herbivory and patch dynamics on rocky reefs in temperate Australasia: The roles of fish and sea urchins. *Australian Journal of Ecology*, **15**, 505-520.

Jones, G.P. & Norman, M.D. (1986). Feeding selectivity in relation to territory size in a herbivorous reef fish. *Oecologia* (Berlin), **68**, 549-556.

Jones, G.P. (1992). Interactions between herbivorous fishes and macro-algae on a temperate reef. *J. Exp. Mar. Biol. Ecol.* **159**: 217-235.

Jones, G.P. and Andrew, N.L. (1990). Herbivory and patch dynamics on rocky reefs in temperate Australasia: the roles of fish and sea urchins. *Aust. J. Ecol.* **15**: 505-520.

Jones, R.S. & Thompson, M.J. (1978). Comparison of Florida reef fish assemblages using a rapid visual technique. *Bulletin of Marine Science*, **28(1),** 159-172.

Kay, A. M., Butler, A. J. (1983). "Stability" of the fouling communities on the pilings of two piers in South Australia. *Oecologia* **56**: 70-78.

Keough, M. J. (1984a). Dynamics of the epifauna of the bivalve Pinna bicolor: interactions among recruitment, predation and competition. *Ecology* **65**: 677-688.

Keough, M. J. (1984b). Effects of patch size on the abundance of sessile marine invertebrates. *Ecology* **65**: 423-437.

Keough, M.J. and Butler, A.J. (1995).Temperal subtidal hard substrata. In. *State of the Marine Environment Report for Australia*. Technical Annex No. 1. Chapt. 7. pp13.

Kingsford (1989) Distribution patterns of planktivorous reef fish along the coast of north eastern New Zealand. *Marine Ecology Progress Series* **54**: 13-24.

Kinzie III, R. A. and Snider, R. H. (1978). A simulation study of coral reef survey methods. In: Stoddart, D. R. and Johannes, R. E. (eds) *Coral Reefs: Research Methods.* UNESCO, Paris.

Kott, P. (1985) The Australian Ascidiacea (Part 1). *Mem. Qld. Mus.* **23**: 1-438.

Kott, P. (1990) The Australian Ascidiacea (Part 2). *Mem. Qld. Mus.* **29**: 1-266.

Kott, P. (1992) The Australian Ascidiacea (Part 3). *Mem. Qld. Mus.* **32**: 375-620.

Lincoln-Smith, M.P., Bell, J.D. & Hair, C.A. (1991). Spatial variation in abundance of recently settled rocky reef fish in southeastern Australia: implications for detecting change. *Marine Ecology Progress Series*, **77**, 95-103.

Lincoln-Smith, M. and Jones, G.P.(1995) Fishes of shallow coastal habitats. In: "Coastal Marine Ecology in Temperate Australia" Pp. 240-253. NSW University Press, Sydney.

Littler, M.M. (1980). Morphological form and the photosynthetic performances of marine macroalgae: tests of a functional/form hypothesis. *Botanica Marina* **22**: 161-165.

Littler, M.M. and Littler, D.S. (eds.) (1985) *Handbook of phycological methods* New York, Cambridge University Press, Vol. 3 Ecological field methods: macroalgae.

Lucas, H.A. and Seber, G.A.F. (1977). Estimating coverage and particle density using the line intercept method. *Biometrika* **64**: 618-622.

Marshall, N. (1978). Particulate organic carbon. In: Stoddart, D. R. and Johannes, R. E. (eds) *Coral Reefs: Research Methods.* UNESCO, Paris.

May, V. and Larkum, A.W.D. (1981). A subtidal transect in Jervis Bay, New South Wales. *Aust. J. Ecol.* **6**: 439-457.

McNeill, S.E., Worthington, D.G., Ferrell, D.J. & Bell, J.D. (1992). Consistently outstanding recruitment of five species of fish to a seagrass bed in Botany Bay, NSW. *Australian Journal of Ecology*, **17**, 359-365.

Mullineaux, L. S., Butman, C. A. (1990). Recruitment of encrusting benthic invertebrates in boundary-layer flows: A deep-water experiment on Cross Seamount. *Limnol. Oceanogr*. **35**: 409-423.

Muttlak, H.A. and Sadooghi-Alvandi, S.M. (1993). A note on the line intercept sampling method. *Biometrics* **49**: 1209-1215.

Pawlik, J. R., Butman, C. A. (1993). Settlement of a marine tube worm as a function of current velocity: interacting effects of hydrodynamics and behaviour. *Limnol. Oceanogr*. **38**: 1730-1740.

Poore (1991) Crustacea Isopoda: Deep-sea Chaetiliidae (Valvifera) from New Caledonia and the Philippines. *Memoires du Museum National d'Histoire Naturelle Serie A Zoologie* **152**: 139-154.

Poore G.C.B. (1995) Biography and diversity of Australia's marine biota. In. *State of the Marine Environment Report for Australia*. Technical Annex No. 1. Chapt. 7. pp13.

Preisendorfer, R.W. 1986 Secchi disk science: Visual optics of natural waters. *Limnol. Oceanog*. **31**: 909-926.

Rapport, D.J. (1992*). What is clinical ecology? Ecosystem health: new goals for environmental management* (Ed. by R. Costanza, B.G. Norton, & B.D. Haskell), pp. 144-156. Island Press, Washington, DC.

Reichelt, R.E, Loya, A.Y. and Bradbury, R.H.l (1986) Patterns in the use of space by benthic communities on two coral reefs of the Great Barrier Reef (Australia). *Coral Reefs* **5**: 73-80.

Rogers, C. S. (1990). Responses of coral reefs and reef organisms to sedimentation. *Marine Ecology Progress Series* **62**: 185-202.

Roughgarden, J., Iwasa, Y., Baxter, C. (1985). Demographic theory for an open population with space-limited recruitment. *Ecology* **66**: 54-67.

Sanderson, J.C. (1987). *A survey of the Macrocystis pyrifera (L.) C. Agardh stocks on the east coast of Tasmania.* Report, Department of Sea Fisheries, Tasmania: Hobart.

Sanderson, J.C. and Thomas, D.P. (1987) Subtidal macroalgal communities in the D'Entrecasteaux Channel, Tasmania (Australia). *Australian Journal of Ecology* **12**: 41-52.

Scheil, D.R. (1990). Macroalgal assemblages in New Zealand: structure, interactions and demography. *Hydrobiologia* **192**: 59-76.

Scheil, D.R. and Foster, M.S. (1986). The structure of subtidal algal stands in temperate waters. *Oceanography and Marine Biology Annual Review* **24**: 265-307.

Shannon, C.E. (1948). A mathematical theory of communication. *Bell System Technical Journal.* **27**: 379-423.

Shepherd, S. A. and Sprigg, R. C. (1976). Chapter 12. Substrate, Sediments and Subtidal Ecology of Gulf St. Vincent and Investigator Strait. In: Twidale, C. R., M. J. Taylor, and B. P. Webb (eds). *Natural History of the Adelaide Region*. Graphic Services Pty Ltd, SA.

Shepherd, S.A. and Womersley, H.B.S. (1970). The sublittoral ecology of West Island, South Australia. 1.Environmental features and the algal ecology. *Transactions Of The Royal Society Of South Australia* **94**: 105-137.

Shepherd, S.A. and Womersley, H.B.S. (1971). Pearson Island Expedition 1969.-7. The subtidal ecology of benthic algae. *Transactions Of The Royal Society Of South Australia* **95**: 156-167.

Shepherd, S.A. and Womersley, H.B.S. (1976). The subtidal algal and seagrass ecology of St Francis Island, South Australia. *Transactions Of The Royal Society Of South Australia* **100**: 177-191.

Shepherd, S.A. and Womersley, H.B.S. (1981). The algal and seagrass ecology of Waterloo Bay, South Australia. *Aquatic Botany* **11**: 305-371.

Simpson, C. J. and Field, S. (1995). *Survey of water quality, groundwater, sediments and benthic habitats at Coral Bay, Ningaloo Reef, Western Australia*: a report to the Department of Conservation and Land Management. Department of Environmental Protection, Perth, W.A.

Steinberg, P.D. (1989) Biogeographical variation in brown algal polyphenolics and other secondary matabolites: comparison between temperate Australasia and North America. *Oecologia, (Heidelberg)* **78(3)**: 373-382.

Strickland J.D.H. and Parsons T.R. 1972. A practical handbook of seawater analysis. *Fish. Res. Board. Canada. Bull*. **!67** 311.

Svane, I. & T. LundŠläv (1981). Reproductive pattern and population dynamics of Ascidia mentula O.F. Mnller on the Swedish west coast. *J. Exp. Mar. Biol. Ecol*., **50**: 163-182.

Svane, I., Young, C. M. (1989). The ecology and behaviour of ascidian larvae. *Oceanogr. Mar. Biol. Ann. Rev*. **27**: 45-90.

Turner, D. J. (1995) *A comparative study of destructive and non-destructive survey techniques for use in macroalgal systems.* Honour thesis, (Botany Department. The University of Adelaide, South Australia)

Underwood, A.J. and Kennelly, S.J. (1990). Ecology of marine algae on rocky shores and subtidal reefs in temperate Australia. *Hydrobiologia* **192**: 3-20.

Van der Velde, J.T. and King, R.J. (1984). The subtidal seaweed communities of Bare Island, Botany Bay. *Wetlands (Australia)* **4**: 7-22.

Vanaltena, I.A.and Steinberg,P.D. (1992): Are Differences in the Responses Between North American and Australasian Marine Herbivores to Phlorotannins Due to Differences in Phlorotannin Structure. *Biochem Syst Ecol* **20**:493-499.

Warwick, R.M. (1993). Environmental impact studies on marine communities: Pragmatical considerations. *Australian Journal of Ecology* **18**: 63-80.

Webb, L.J., Tracey, J.G., Williams, W.T. and Lance, G.M. (1970). Studies in the numerical analysis of complex rainforest communities. V. A comparison of the properties of floristic and physiognomic-structure data. *Journal of Ecology* **58**: 203-232.

Westphalen, G. and Cheshire, A.C. (in press). Quantum efficiency and photosynthetic production of temperate turf algal community. *Aust. J. Bot.*.

Wethey, D. S. (1986). Ranking of settlement cues by barnacle larvae: influence of surface contour. *Bull. Mar. Sci*. **39**: 393-400.

Womersley, H.B.S and King R.J. (1990). Ecology of Temperate Rocky Shores. In M.N. Clayton and R.J. King (Eds.), *Biology of Marine Plants.* Longman Cheshire. 266-295.

Womersley, H.B.S. (1990). Biogeography of Australasian marine macroalgae. In M.N. Clayton and R.J. King (Eds.), *Biology of Marine Plants.* Longman Cheshire. 367-382

Glossary

carnivore	a species which predominantly feeds on animal tissues.
colonial	organisms that live together rather than as individuals. Many corals are colonial in that the polyps (which represent separate individuals) adhere to one another thus forming much larger structures than would single individuals. Reproduction is generally asexual in such species.
demersal	fish which live closely associated with the substrate (see also pelagic).
diversity	term to describe the variety of different species/taxa in a system. Various attempts have been made to define diversity some of which provide detailed mathematical formulations. It is generally used to quantify not only the number of species/taxa but also the relative abundance of those present so as to incorporate an assessment of whether all species are equally common (see also equitibility, richness).
encrusting	forms a thin layer (crust) which conforms to the substratum.
equitibility	the evenness in the numbers of different species/taxa in a system. A system with high equitibility would have more or less equal numbers of each taxa present. A system with low equitibility would have many of some taxa and few of others.
filter feeder	feed by filtering particles from the water column.
herbivore	species which predominantly feeds on plant tissues.
interspecific	generally refers to interactions between organisms of different species (eg. interspecific competition see also intraspecific).
intraspecific	generally refer to interactions between organisms of the same species (see also interspecific).
kelp	brown alga of the order Laminariales.
keystone	a species which plays a central role in controlling the structure of an ecosystem. This concept is becoming less well regarded in ecological circles as it is frequently argued that all species play such roles.

macroalgae	organisms in the Kingdom Protista. Includes taxa from three divisions the green algae (chlorophyta) the brown algae (phaeophyta) and the red algae (rhodophyta).
metapopulation	many small populations that interact to form an overall regional population.
motile/mobile	capable of movement.
mutualistic	symbiosis of mutual benefit to host and symbiont.
nutrients (plant nutrients)	comprise both macro and micro-nutrients essential to support plant growth. Macro-nutrients comprise nitrogen and phosphorus in the form of nitrates, nitrites and phosphates. Micro-nutrients comprise a range of metals in ionic forms.
pelagic	organisms which live in mid-water typically used to describe fish such as snapper or tuna (see also demersal).
phylogenetic	shared evolutionary relationships.
quadrat	a quadrat is a steel rod bent into a square which can be placed over the substratum it is used to mark out an area from which quantitative measures of abundance or percentage cover can be obtained.
richness	the number of species/taxa in a system.
sedentary	organism that is very slow moving and is often seen as being sessile even though it is capable of movement (eg. sea urchins or anemones)
sessile	organism that grows attached to the seabed.
suspension feeder	see filter feeder.
symbiont	organisms of different species that live together. Often this is for mutual benefit (see mutualism) but not always. For example, in some corals photosynthetic algae live within the coral tissue and provide carbon to the coral host. In return the algae receive inorganic nutrients (nitrates and phosphates) and protection from predators.
trophic	relationship between biota based on flow of carbon nutrition (eg. plants and herbivores are connected trophically).

weedy species	a term used to describe species which thrive particularly in disturbed environments due to their high fecundity and growth rates.

Appendix 1 - Water quality parameters

Introduction

Water quality is clearly important in determining the health status of reefs. Womersley and King (1990) define three groups of environmental factors into which water quality parameters fall, these are, dynamic, physical and chemical factors. Dynamic factors relate to water movement, including tidal movement, currents, upwelling, and wind, wave and storm action. Physical factors include parameters such as temperature and turbidity or sedimentation. Chemical factors include salinity, nutrient levels, availability of gases (eg. O_2 and CO_2), pH and pollutants (eg. heavy metals and organochlorins).

Water quality however, is not necessarily easy to measure. Most parameters are highly variable in time and space and therefore any sampling program needs to be cognisant of that variability. The following section is intended to provide an indication of which water quality parameters can be measured and how they should be measured. It should however be recognised that the sort of sampling program that one develops for looking at the biota, will in all probability not be compatible with a sampling program for looking at water quality.

Sampling for this project is structured to look at biota and consists of data collected over a short-term (approx. one month). While this is acceptable for a biotic survey, since the biota better integrates multiple factors over a longer time frame, it is not a suitable approach for a survey of water quality. Water quality data collected in this way would not incorporate the variability associated with these parameters. Therefore, although these parameters can be measured, they only provide an indication of the conditions that exist at the time of the study. If they are to be assessed in a more meaningful way a separate sampling approach is necessary.

Dynamic parameters

Water Motion

Quantification of water movement and direction is important if modeling of the reef environment is proposed. Water motion is also worth measuring for its inherent role in sedimentation, scour, feeding, and reproduction. There are a number of types of water motion that can be measured. Measuring water volume transport facilitates analysis of community metabolism and ecology. Acoustic profilers can provide instantaneous measures of the current direction and magnitude. Current meters (eg. Inter-ocean current meter, type S4) can be placed to record over either short or long periods of time the current flow and direction.

These two latter techniques can be quite expensive in terms of the equipment required and, although useful, will not be used for this study.

A qualitative measure of current flow and direction can be obtained by simply timing the drift rate of plankton or bits of algae across a known distance and then recording the direction of flow. This type of measure is handy as it requires only a compass, watch and pencil/slate. In a similar manner, particles can be video-taped and velocity can be measured at a later date. This may prove to be even easier if video transects are to be used in other portion of this study. Digital flow meters (eg. General Oceanics) can also be useful after calibration. These meters only reflect flow and not direction.

Qualitative measurements of total water motion can be determined by simply measuring how fast a substance will dissolve. This measure is qualitative as high particulate levels increase scour and dissolution rate as will any motion on the part of the substance. High variability is likely, but even these rough measurements will provide a simple measure of total water motion that can be compared both temporally and spatially.

Physical parameters

Temperature

A temperature depth profile may be taken at the time of sampling in order to provide thermocline information for that day. This type of information is not immediately useful in determining the status of a reef but should be an integral part of any long-term data set because changes over the long-term can be indicative of shifts in hydrographic conditions which may have important influences on reef systems. The accuracy of the measure should be $\pm 0.1^{o}$ C. Determination of water temperatures can be made simply by using a mercury thermometer during a dive or in water samples collected from discrete depths. Mercury thermometers, while inexpensive, are also fragile and the risk of breakage and leakage of mercury into the environment has to be considered in the design of a sampling strategy. The use of other types of thermometers (alcohol, digital, watch, etc) should be discouraged unless careful calibration is included in the protocol. The ideal system, however, is a sonde/data logger (discussed below) that can be lowered to the bottom from the boat and then pulled back into the boat. This type of system records the temperature every second and downloads the information into the data logger. This information can be printed out or downloaded into any type of computer via a RS232 connection. The system available for this project is the YSI 2000 sonde/data logger.

Turbidity

Turbidity is often assumed to be a primary component of reef health survey techniques as excessive turbidity is often correlated with higher sedimentation rates. . The YSI sonde incorporates a nephelometer in the sensor package, however, previous results have been found to be insensitive. The Secchi disk provides a very simple method of gaining a visual index of water clarity.

The Secchi disk is named after the Italian physicist Angelo Secchi who published on this technique in 1886 (in Presisendorfer 1986). The disk itself is 30 cm in diameter and is attached to a non-stretch line in the center. The disk is then lowered vertically into the water, with the disk remaining horizontal, until the disk disappears from sight. This disappearance depth is inversely proportional to the amount of organic and inorganic particles in the water.

Sediment analysis/traps

Excessive sedimentation can adversely affect the structure and function of the coral reef ecosystem by altering both physical and biological processes. There are a huge variety of shapes and sizes for sediment traps (Gardner 1980a, b; Gulickson 1982; Butman et al. 1986 - all in Coyer and Witman 1990). The basic principle behind all of the designs is to collect sediments as they sink through the water column. It has been established that simple cylinders provide the best form for sediment traps in all types of water, be it stagnant or turbulent, limnic or marine, as funnels generally undertrap and bottles overtrap the actual flux of particles in moving waters (Hakanson *et. al.*, 1989). Circular shaped traps also avoid corner effects in omnidirectionally circulating waters (Blomqvist & Kofoed, 1981). Some traps will incorporate fixatives within the sample bottles to ensure that organisms neither create or deplete sediments. Sediment traps give an indication of sedimentation rates but do not reflect the degree of re-suspension and/or horizontal transport of sediments. Horizontal sediments traps or vertical traps with baffles can provide estimates of horizontal transport.

Traps can be setup by deploying them on the first dive at a site, and then collecting them on the last dive. A two to three day interval may be sufficient to determine sedimentation rates. These traps can be constructed of PVC tubes (a minimum height:diameter ratio of 2:3 is advised). It is recommended that these traps be constructed within a weighted rack so that the entire array can be placed either onto the reef or the adjacent substrate with minimal disturbance. The rack is constructed out of rebar and weighted down with 5 kg additional weight (to minimize movement). The PVC tube(s) are then attached to the rack. Plugs (of appropriate diameter) are used to seal the tubes prior to deployment and collection. Due to the weight of the array, the racks should be lowered and recovered from the boat and not carried by the divers.

Sediment analysis examines the grain size distribution of settled sediments. Sediments are collected by cores or grabs and then sieved through a sequential sieve series to provide grain size information. Cores can be collected around a reef by using 50 cc syringes with the front end removed. The syringe is pushed into the substrate and then withdrawn; the plunger will hold the core in place. The entire core/syringe/plunger should then be placed into a labelled plastic bag and the bag sealed (Ziploc bags work well - ensure that all excess water is removed before sealing). Analysis of sediment on the reef itself can also be gathered with a syringe (intact). The material is sucked up with the syringe and the syringe is then placed into a labelled plastic bag. Sample sizes will be much smaller with this technique. Samples should be taken at random sites along the transect line.

Chemical parameters

Salinity

Salinity measurements are used to determine haloclines within the water column and to identify the intrusion of fresh water or different seawater masses into the reef system. As with temperature, there are a number of instruments for measuring salinity, the most inexpensive of which is the refractometer. These hand held devices can be purchased from commercial suppliers (approximately $300-500) and can read 0.1 %o or 0.01 %o accuracy. While inexpensive and simple, the refractometer method will not be used for this study as the refractometer represents an additional small piece of fragile equipment and requires pipettes. The preferred alternative to the refractometer is a sonde/data logger system (see temperature above).

pH

Seawater is an excellent buffering agent, but when the pH of the system is altered significantly, the chemical processes can likewise be dramatically affected. For instance, the carbon dioxide-carbonic acid-bicarbonate system is generally in equilibrium. A shift in the pH could result in a shift in the quantity of CO_2. Determination of pH in the water column should be made during the sampling procedure. Accuracy of this measurement can vary depending upon the instrumentation utilised. A pH meter is a simple method that can give accurate readings to 3 significant figures. For this study, it is suggested that accuracy to two significant figures is more than adequate as small shifts (+ 0.1) are not going to have much affect on the reef system. Since the pH scale is a logarithmic scale, a shift ± 1 unit is a 10 fold difference and this difference may have an effect on the reef system. This method, however, also involves the use of small fragile pieces of equipment. The Sonde/data logger records pH to two decimal places, and is therefore the preferred option for this project.

Oxygen

The use of an oxygen electrode (polarographic method) yields an instantaneous measure of oxygen This method is simple and can provide continuous readings. There are two types of oxygen electrodes (membrane covered solid electrodes and wide-bore dropping-mercury electrodes of which the more common is the membrane covered electrode. This type of electrode is present in the sonde systems described above.

Total Organic Carbon

Methods for determining TOC levels involve very detailed and exacting analyses and the oceanographic community has yet to agree on a standardised methodology.

Phosphorus

There are two primary types of phosphorus in seawater (reactive and total, with the difference between these two being the total organic phosphorus content). There are a number of factors that prohibit the inclusion of this parameter in the study (not the least being that the phosphorus levels are not considered to be significant - unlike the situation in the Great Barrier Reef). These factors include the rapid processing time required (within 2 hours of collection), the large sample volume required (at least 4 litres), the dedication of glassware to this analysis, the fact that standards decay and have to be generated fresh for each run, and several others (reviewed in Pilson 1978). Analytical methods to determine dissolved phosphorus can also be found in Strickland and Parsons (1972). Colorimetric analysis will normally require a spectrophotometer with a 10 cm light path and the processing of about 50 mL of seawater per sample analysis.

Nitrogen

Nitrogen levels are important in seawater systems, but levels of nitrogenous compounds are often extremely low and difficult to measure with any type of easily available equipment. Analytical methods to determine dissolved nitrogen can be found in Strickland and Parsons (1972).

Heavy metals, organochlorins

Mineralogical analysis requires the use of X-ray fluorescence; heavy metals are analysed with a Inductively Coupled Plasma Atomic Emission Spectrophotometer (except for Mercury which is analysed with a Vapour Generation Atomic Absorption Spectrophotometer); and organic analysis requires Capillary Gas Chromatography using Flame Ionisation Detection, Thermionic Specific Detection, Electron Capture Detection and Mass Spectrometry. Organotin analysis is done via an electrically heated quartz furnace atomic absorption spectrophotometer.

YSI SONDE

The YSI SONDE system is field portable and is capable of taking instantaneous measurements of dissolved oxygen, salinity, temperature, depth, and turbidity. The operation of the SONDE is simple, the unit is lowered to the bottom and then slowly raised to the surface. All measurements are logged into the data logger which can store up to 100 sites and over 80,000 readings. The System can hold 999 separate runs and all results are shown on the display during recording. Information can be downloaded directly to a printer or to a computer.

The dissolved oxygen probe is a membrane-covered electrode, self stirring and can measure dissolved oxygen levels from 0 - 20 mg/l ±0.03 mg/l.

The salinity is computed from the temperature and the uncompensated conductivity measurement and has a range from 0-50 ppt ± 0.1 ppt.

The range for depth measurements is 0-65 m ± 0.5 m.

Turbidity is measured in nephelometric Turbidity Units (NTU) with an accuracy of ± 0.05 NTU

Temperature range is -5 to 50 °C + 0.4 °C.

It is recommended that turbidity also be measured with a Secchi disk. The Secchi disk is lowered over the side of the boat within one hour of noon and the depth where the disk disappears (determined by marking the line in 0.5 m intervals) is recorded. Cloud cover (if any) and sea state should also be recorded.