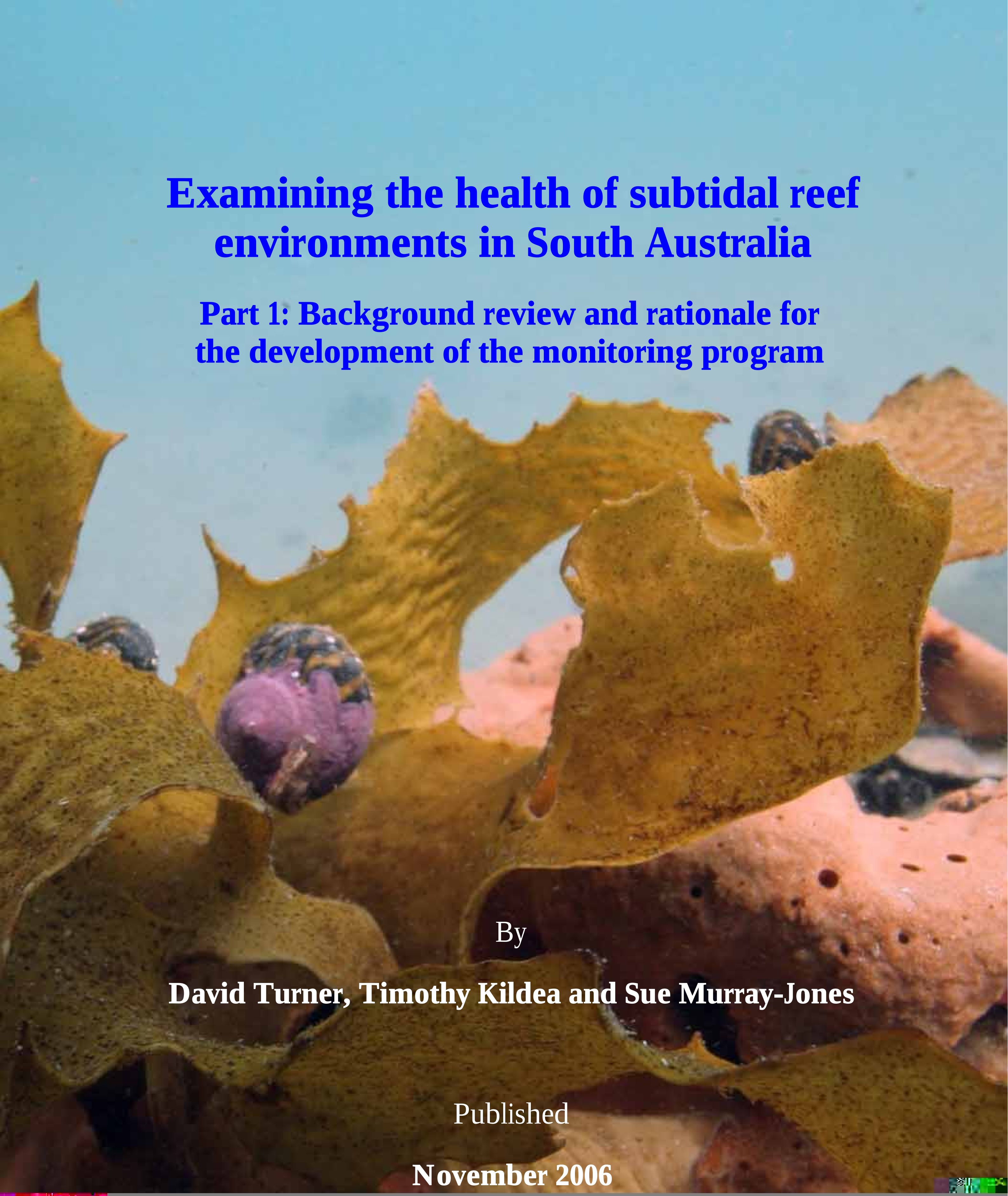

Examining the health of subtidal reef environments in South Australia

Part 1: Background review and rationale for the development of the monitoring program

By

David Turner, Timothy Kildea and Sue Murray-Jones

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Part 1: Background review and rationale for the development of the monitoring program

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More information:

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

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This report is also partially based on information contained in a previous reef health reporting series (Cheshire *et al.* 1998a, Cheshire *et al.* 1998b, Miller *et al.* 1998, Cheshire and Westphalen 2000), which had contributions from Prof. Anthony Cheshire, Prof. Steven Hall, Dr. George Matsumoto, Dr. Alan Butler, Dr. Jon Havenhand, Dr. Grant Westphalen and Mr. David Miller.

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Figures

Maps used in this text (Figure 1 and Figure 2) incorporate data that is © Commonwealth of Australia (Geoscience Australia 2003).

The Reef Health program is supported by the following agencies:

Executive summary

This report provides the following recommendations for establishing a monitoring program to assess reef health:

- Indicators should be chosen to represent important facets of the ecosystem including structural components and integral processes;
- Trigger values need to be set for the various indicators;
- Protocols must be established that outline what actions are to be taken when a trigger value is crossed, and these must have the support and backing of the relevant management agencies;
- Sampling programs should be non-destructive wherever possible to minimise the impact of repeated surveys;
- Consideration needs to be given to methodological protocols that can be adopted by volunteer divers. Alternatively compatible methods should be developed that will serve broader community use; and
- Effort should be placed into fostering greater community involvement in reef monitoring initiatives through program development and education initiatives.

Preface

The need for better ways to monitor the totality of our effects upon nature has never been felt so strongly across the globe (Rapport *et al.* 1998)

Approximately 90% of Australia's population live on or near the coast. Our marine ecosystems are important both socially and economically, and as a consequence the utilisation of the marine environment as a resource imposes a considerable pressure on these ecosystems. What constitutes 'appropriate' use of the environment as a resource is often the subject of heated debate as many activities may conflict with each other (e.g. commercial and recreational fishing; marine protected areas and aquaculture). In addition there are a growing number of physical impacts, which range from wastewater and stormwater discharges, to coastal developments such as marinas and aquaculture farms (Nicolson *et al.* 2003). The potential and actual impact of our interaction with the coastal marine environment is a growing concern, particularly to those responsible for maintaining the quality of marine habitats and the sustainability of harvestable organisms (Kingsford *et al.* 1998).

The term "ecosystem health" is increasingly being utilised in both public and scientific forums. Concerns about the "health" or condition of ecosystems and maintaining ecosystem services have slowly become a dominant focus for marine managers. This is reflected in the increasing number of statutes involved in sustainable management of marine resources and planning for coastal activities and conservation of the marine environment. In South Australia these include: *Fisheries Act 1982, National Parks and Wildlife Act 1972, Coast Protection Act 1972, Development Act 1993, Environment Protection Act 1993, Native Vegetation Act 1991, Natural Resources Management Act 2004, Water Resources Act 1997, Historic Shipwrecks Act 1981, Wilderness Protection Act 1992 and Aquaculture Act 2001*, amongst others.

Notwithstanding the extent of legislation available to manage the marine environment, there is a basic need for managers to obtain biological information, whether it is to simply describe 'what is there', or to determine whether communities are changing as a result of management decisions, or to further the knowledge of the processes that influence patterns of abundance and to predict what may happen in the future (Kingsford *et al.* 1998).

Reef monitoring along Adelaide's metropolitan coastline

Adelaide is the largest population centre in South Australia and is situated on the eastern shoreline of Gulf St Vincent. Along Adelaide's metropolitan coastline, water quality is generally considered to be of a lower quality particularly when compared to non-urban areas, with stormwater and wastewater treatment plants likely to be significant contributors (Gaylard 2004).

Reduced water quality has in turn been implicated in the declining health of seagrass communities, resulting in the loss of over 5200 ha off Adelaide since the 1940s (Seddon 2002, Westphalen *et al.* 2004).

In the mid 1990s, concern over the status of reef environments within proximity to Adelaide led the South Australian Environment Protection Authority to commission a study into the health of these habitats. The study involved staff from Adelaide and Flinders Universities and resulted in the development and evaluation of an assessment protocol and baseline monitoring (November 1996) of a number of near-shore reefs along Gulf St Vincent's eastern shoreline (see Cheshire *et al.* 1998a, Cheshire *et al.* 1998b, Miller *et al.* 1998).

The study concluded that reef communities in proximity to Adelaide's northern metropolitan coastline were in a degraded state, probably as a result of the poor water quality of the area. Also highlighted was the need for further monitoring work to assess the extent and rate of any ongoing degradation and subsequent changes to reef community dynamics (Cheshire *et al.* 1998a).

Surveys were repeated in 1999 with an expanded number of reefs, confirming the observations made in the previous study. Evidence was also found to indicate that community structures of reefs south of Adelaide were not necessarily stable and further study was recommended (Cheshire and Westphalen 2000).

Other studies have also examined aspects of reef health along Adelaide's coastline (e.g. Greig 2000, Smith 2000, Gorgula and Connell 2004, Turner 2004). Although these studies were carried out separately and often with different objectives, they shared a number of similar findings with those of the previous reef health surveys (see Cheshire *et al.* 1998a, Cheshire and Westphalen 2000).

The need for ongoing monitoring of Adelaide's metropolitan reefs is well recognised, as is the desire to expand the program to regional areas. This has resulted in the establishment of a larger program to re-examine the findings of the original reef health surveys in context with more recent studies, and to further develop the ability to assess ecological health in reef environments. Additionally, there has been more emphasis on forming a standard suite of methods for monitoring reef environments and for greater community participation in the monitoring process.

The aim of this document is to provide a review of our current understanding of temperate reef environments (within a South Australian context), and to provide a rationale for developing a quantitative method of assessment for their ecological health. The report is composed of three sections which: provide an overview of temperate reef environments in southern Australia; examine approaches used in the assessment of reef health; and discuss suitable sampling designs for establishing reef health monitoring programs.

1 An introduction to temperate reef environments

1.1 Temperate reefs – what are they?

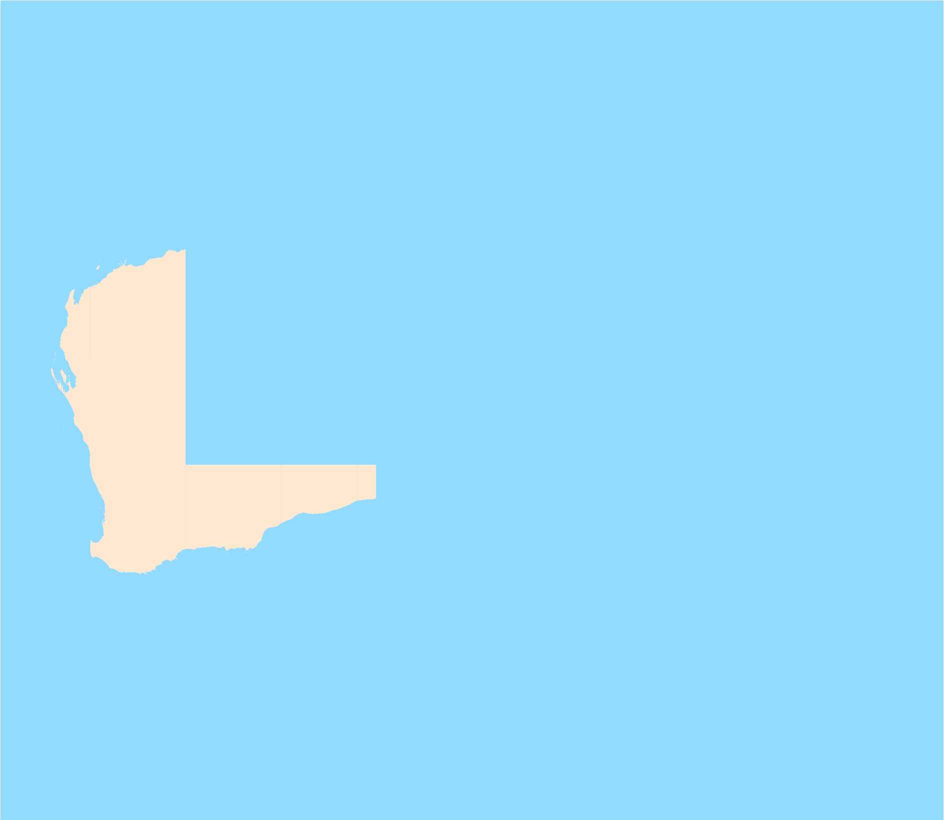
The temperate zone is generally defined as lying between the latitudes 23º 27' and 66º 33' north or south (Levinton 2001). The distinction between temperate and tropical (coral) reefs is not simply one of perception. There are in fact quite fundamental differences in the structure and dynamics of these ecosystems. Temperate reefs exist 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 substratum. Coral reefs are temperature-dependent and are generally restricted to a belt within 30° N and S latitudes (Charton 1998). Furthermore, the physical and chemical environments are distinctly different. Temperate waters are cooler and nutrient levels tend to be higher compared to reefs in tropical waters. Together, these factors have had a profound effect on the evolution of the biota in these regions (Cheshire *et al.* 1998b).

In contrast to the domination by corals and sponges seen on tropical reefs, the dominant biota on temperate reefs (at least in the photic zone) is generally macroalgae. In temperate systems, the majority of carbon fixed is via these large algae. This is in contrast to tropical systems in which the majority of carbon fixed is by the symbiotic relationship of microscopic algae living in the tissue of sponges and corals (e.g. Franklin *et al.* 1996). Hence there is a greater distinction 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).

General biogeography of Southern Australia

A number of biogeographic provinces have been identified around the Australian coastline (Figure 1), based largely on the resident macroalgal flora (Womersley 1984, Womersley 1990).

The majority of the southern Australian coastline is within the Flindersian Province, which is characterised as being transitional between cool and warm temperate. To the north of this province are the warm temperate Peronian and the southern extent of the Dampierian Provinces that extend up the east and west coasts respectively. Within the eastern part of the Flindersian province is the Maugean subprovince that encompasses Victoria, Tasmania, and extends into South Australia as far as Cape Jaffa. This area is cold temperate due to the presence of a cold-water upwelling (Womersley 1984) and is differentiated from the rest of the Flindersian Province by the presence of massive taxa such as *Durvillaea*, *Macrocystis* and *Lessonia*, that are not found elsewhere in southern Australia (Womersley 1987). Although the southern Australian coastline is mainly temperate, a slight subantarctic element exists in South East Tasmania, while parts of the South Australian gulfs have a subtropical element (Womersley 1984).



similar macroalgal communities in contrast to New South Wales, although care is required when comparing patterns derived from small-scale studies at a regional level as the patterns observed and the processes driving these patterns are not completely understood (Fowler-Walker and Connell 2002).

The sections below provide a brief summary of the types of biotic compositions observed on the subtidal reefs of southern Australia. As reef habitats are generally defined by the dominant vegetation type (which characterise a variety of different benthic assemblages, O'Hara 2001), emphasis will be placed on describing algal communities rather than the associated invertebrate and fish communities.

Macroalgal communities

Macroalgal communities tend to comprise a number of separately identifiable layers (based on size) that can coexist within a single assemblage depending on local factors (Shepherd and Sprigg 1976, Turner 1995, Turner and Cheshire 2003). Across much of Australia, the upper stratum is comprised of large brown canopy-forming taxa (Fucales and Laminariales, Shepherd and Sprigg 1976), although in colder areas (generally in the Maugean Sub-province, Figure 1), there may additionally be a floating canopy containing massive taxa such as *Macrocystis angustifolia* and *Durvillaea potatorum* (Womersley 1984). Macroalgae often coexist as multi-layer assemblages in which the structure is often dependent upon the surrounding abiotic environment (Shepherd and Womersley 1970, 1971, Shepherd and Sprigg 1976, Shepherd and Womersley 1976, 1981). Diversity appears to be maximised under open canopy conditions, i.e. those with a sparse cover of larger individuals, although some closed fucoid canopies also include a range of understorey taxa (e.g. Turner and Cheshire 2003). In contrast, diversity is low under *Ecklonia* canopies and mainly limited to smaller encrusting forms and turfing algae (Kennelly 1987, Kennelly and Underwood 1992, 1993, Connell 2003, Turner and Cheshire 2003). Systems dominated by *Durvillaea* tend to be almost monospecific with no foliaceous macroalgae existing beneath the canopy (Cheshire 1985), whereas the presence of *Ecklonia* and numerous rhodophytes under the taller *Macrocystis* canopy is not uncommon (Sanderson and Thomas 1987).

Foliaceous representatives of all three macroalgal divisions often form a stipitate (sub) canopy that may be observed below the main canopy, or alternatively as dominant in areas unsuitable for the larger phaeophycean taxa (Shepherd and Sprigg 1976, Turner and Cheshire 2003). Similarly, smaller specimens (a few centimetres tall) may exist below the larger canopy, but are also capable of forming dense stands of ‘turf’. Once established, turf beds have the ability to exclude larger taxa and thereby dominate patches of reef (Shepherd and Sprigg 1976, Kennelly 1987, Connell 2005, Copertino *et al.* 2005).

The smallest of the macroalgae may only be a few millimetres tall and are often observed as an encrusting layer on the substratum (Shepherd and Sprigg 1976). Encrusting species are also able to dominate substratum in areas less suitable for larger taxa (Dethier 1994). Alternatively, many can adapt to low light conditions and survive even when overgrown by larger taxa (Copertino 2002, Connell 2003, 2005).

Across the rocky reefs of the southern Australian coastline, different taxa dominate the upper sublittoral zone (from the mean low water mark down several metres, Shepherd and Womersley 1970, 1971, 1976, 1981). In south-eastern Australia and extending west to Cape Jaffa in South Australia, dominants include the massive genera *Durvillaea potatorum* and *Macrocystis angustifolia* on rough water coasts, while species such as *Phyllospora comosa* and *Ecklonia radiata* occur where wave force is less (Womersley 1984). West of Cape Jaffa, the upper zone tends to be dominated by *Ecklonia radiata* along with numerous genera of fucoids including *Cystophora*, *Sargassum*, and *Scytothalia* (Womersley 1984). Community composition in this region is strongly influenced by water movement. Environments influenced by calm to moderate water movement generally contain patches of *Caulocystis* spp. and *Cystoseira trinodis*, where high wave-force environments are often composed of fucoids such as *Cystophora intermedia*, particularly in the fringe zone between intertidal and sub-tidal (Shepherd and Womersley 1970, Womersley 1984). *Scytothalia dorycarpa* often dominates high-energy environments in the west (Shepherd and Womersley 1970, 1971, 1976, 1981), which is slowly replaced by *Seirococcus axillaris* in the east (Womersley 1987).

Large brown macroalgae dominate the mid-sublittoral zone, including the kelp *Ecklonia radiata* along with a host of fucoids such as *Cystophora* spp., *Sargassum* spp., *Scytothalia dorycarpa*, *Seirococcus axilaris*, *Acrocarpia paniculata* and *Myriodesma* spp. (Womersley 1984). The lower sublittoral zone tends to be dominated by a diverse range of red algae and encrusting coralline species (Womersley 1984).

In South Australia, macroalgae are generally confined to the lowest parts of the intertidal and into the subtidal, with the exception of a few encrusting brown and red taxa (Womersley 1984). However, macroalgae are found more frequently higher in the intertidal in Victoria and Tasmania (Bennett and Pope 1953, Bennett and Pope 1960). Additionally, macroalgae are mostly absent from sandy environments (Womersley 1984), although occasional individuals grow on shells and stones on the sea floor. An important exception to this is the genus *Caulerpa*, with several species able to colonise sandy habitat. Seasonal macroalgal species such as *Ulva* spp., *Enteromorpha* spp.[1], and *Porphyra* spp are often found in the mid-eulittoral, generally in late winter and spring. Lower zones within the intertidal often play host to extensive coralline algae and red foliaceous species on steeply sloping shores, whereas the conspicuous *Hormosira banksii* often dominates rocky platforms (Womersley 1984).

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[1] The genus *Enteromorpha* was recently incorporated into the genus *Ulva* (Hayden *et al.* 2003)

A number of unpublished theses (e.g. Collings 1989, Harvey 1990, Emmerson 1992, Turner 1995, Collings 1996, Pocklington 2003, Hirst 2004, Turner 2004) provide details on the variability in composition and dynamics of macroalgal communities from selected sites in South Australia. Community structure appears to vary both annually (seasonal growth, shedding and recruitment) and inter-annually, with major shifts in the dominance being reported on inter-annual scales (Butler 1995, Keough and Butler 1995). 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 (Collings 1996).

Invertebrate communities

Suspension feeders are generally the dominant sessile fauna of temperate reefs (King and Shepherd 1982). The most common groups encountered are anemones, corals and hydroids (Cnidaria), bryozoans (Bryozoa), sponges (Porifera), polychaete tubeworms (Annelida), bivalve molluscs (Mollusca), and ascidians (Urochordata). These groups are all well represented on the rocky reefs of South Australia, although the density and diversity of sponges, bryozoans, and ascidians are particularly high (Kott 1985, 1990, 1992). In contrast to the sessile fauna, the mobile fauna inhabiting temperate reef systems are often less obvious, with perhaps the exception of the echinoderms (e.g. seastars, urchins, holothurians, Kay and Butler 1983, Keough 1984a, b, Butler 1986, 1991).

Characterised by unitary, rather than modular organisms, the mobile fauna comprise a variety of taxa including herbivores, predators and scavengers. Typical mobile herbivores on temperate reefs include sea urchins, gastropods (particularly abalone), and several species of isopod. Mobile carnivores include species of urchins, seastars, crabs, prawns and crayfish, 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 can be significant, particularly the grazing species (Kennelly 1983, Sousa 1984, Jones and Andrew 1990, Andrew 1993).

A considerable body of literature has described the characteristics and dynamics of South Australian hard substratum systems (e.g. Butler 1986, 1991, Butler and Connolly 1996, Vanderklift and Kendrick 2004). It is clear from this work that, although community composition varies substantially across space and time, the scale of variation is local. Large-scale community characteristics remain more or less constant over long periods of time and over large distances. Additionally, Kay and Butler (1983) showed that although 20-40% of the occupants of a jetty piling may be eaten, out-competed, or overgrown within three months, the overall species composition and abundances of these assemblages are roughly constant when considered over a timeframe of more than two years. Keough (1984a, 1984b) observed similar small-scale dynamics with epizoic communities on *Pinna* shells. Butler (1995), in examining communities developing on a new jetty, noted that it may take many years 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 substratum such as rocky reefs (Taylor 1998).

An often-overlooked component of rocky reef communities is the epifauna. Grazing by epifauna can dramatically modify turf community structure, where it is hypothesised that up to 86% of biomass produced by turf communities is consumed by epifauna (Cheshire *et al.* 1998a).

Fish communities

Over 370 marine fish species have been recorded for South Australia (Scott *et al.* 1974), many of which exploit temperate reef habitats for at least part of their lives. In contrast to tropical reefs, a high proportion of these temperate species are unique to Southern Australia (Poore 1995).

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 (Cheshire *et al.* 1998b). Factors such as geology and topography influence the presence and abundance of fish (Harman *et al.* 2003). 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 composition may differ between locations (Lincoln-Smith and Jones 1995). 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.

The temperate reef fish fauna is dominated by carnivorous taxa, which either feed on reef-dwelling invertebrates, or on pelagic zooplankton. Most of these fish species eat mobile crustaceans and molluscs but some, such as the leatherjackets (Monocanthidae), feed at 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 and Andrew (1990) estimate, for example, that 20-30% of fish species eat at least some algae, although this figure is variable and may be much higher in some areas if omnivores are included (S.A. Shepherd *pers. comm.*). Moreover, in some areas herbivores are the dominant species in terms of biomass (Lincoln-Smith and Jones 1995). Most herbivores feed on foliose red and green algae with few exploiting the often-large biomass of kelp found on reefs; an exception to this is the herring cale (*Odax cyanomelas*) that has a specialised diet consisting almost entirely of *Ecklonia radiata* (Andrew and Jones 1990).

On a local scale, those species that do occur in the region often exhibit consistent patterns of abundance, which reflect factors in either the physical or biological structure of the reef. Changes associated with depth or discontinuities in habitat type (e.g. at the rock/sand boundary, or between the kelp and the algal turf zone) are particularly marked. Lincoln Smith and 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 in kelp beds on reefs showed that there is increase in the number of herbivorous fish species in areas cleared of kelp to uncleared areas (Jones 1992).

In a study across 50 km of the NSW coastline, the greatest variation in fish assemblages occurred at spatial scales of metres to hundreds of metres, rather than kilometres or greater (Curley *et al.* 2002). On these coastlines, the most predictable, and largest, differences in fish assemblages are generally between-habitats (Choat and Ayling 1987, Curley *et al.* 2002), although size-related changes in patterns of abundance have been documented for labrids (Choat and Ayling 1987, Gillanders and Kingsford 1998). A likely reason for differences in fish assemblages between habitats is depth and associated food availability, although other differences in diet or habitat complexity may also be important (Anderson and Millar 2004). Recruitment in fish is governed by many factors such as habitat suitability (Carr 1994), active and passive transport and larval supply (e.g. Caley *et al.* 1996, Wolanski *et al.* 1997).

1.3 Factors shaping reef communities

Reefs are structured by a variety of biotic and abiotic factors, both natural and anthropogenic. Anthropogenic factors are discussed in detail in section 1.5 (Threats to reef environments); however there is unavoidably some overlap between sections. Plants and animals occur in a very complex environment, hence the final distribution and abundance of any species is determined by interactions between any or all of the factors outlined below, and rarely by just one of them.

The following section presents a brief overview of some of the factors important in structuring reef communities. It is important to keep in mind that while good levels of correlation have been demonstrated between various factors and the ‘resultant’ biotic structure, few studies have actually demonstrated a causal mechanism (e.g. experimentally).

A noticeable and well-established feature of benthic communities on subtidal rocky habitats (and elsewhere) is that populations vary in abundance at a variety of spatial scales, from metres or less to hundreds of kilometres (e.g. Andrew 1993, Collings 1996, Fowler-Walker and Connell 2002, Vanderklift and Kendrick 2004). The factors that cause this variability in the community structure are well documented and include: depth, current velocity or water movement, turbidity, shade, availability of food, recruitment, competition, and predation – as well as interactions between these factors.

Although a significant amount of information exists on the composition of the phaeophycean-dominated macroalgal communities typical of subtidal reefs in Southern Australia, there have been few attempts to synthesize or summarize this knowledge in order to extract unifying principles relating to the dynamics of these ecosystems. Underwood and Kennelly (1990) undertook a critical review of the literature but their focus was primarily directed toward experimental work rather than including the many descriptive studies that form an important part of our understanding. Similarly, Schiel (1990) reviewed the status of knowledge on macro-algal assemblages in New Zealand. The conclusion common to both of these studies is that the processes responsible for structuring these communities are still not clearly understood. Schiel (1990) also argued (as did Andrew and Mapstone 1987) that the nature of interactions between macroalgae 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 review emphasises that more research is required on the basic biology of the dominant species found growing on temperate subtidal reefs.

Substratum

Macroalgae and most sessile invertebrates are generally confined to hard substratum. In some studies successful recruitment of algae was found to be largely independent of aspect or texture (Renaud *et al.* 1996, Collings and Cheshire 1998, Greig 2000), while in others differential recruitment was observed (Harlin and Lindburgh 1977, Vandermeulen and Dewreede 1982, Fletcher and Callow 1992). Notwithstanding, recruitment is often moderated by the presence of other organisms, which act to modify the substratum or alternatively modify another factor (e.g. reduce herbivore activity, Farrell 1991). In contrast, the presence of unconsolidated sediment over the substratum generally has an inhibitory effect (Renaud *et al.* 1996).

Oceanography

The influence of near-shore oceanographic factors is being increasingly recognized as important in structuring reef communities, particularly in the intertidal. Features such as the presence or absence of upwellings have a large influence on reef communities (Menge *et al.* 1997a, Broitman *et al.* 2001). Differences in nutrients and/or chlorophyll concentrations, forced by the upwelling of cooler, nutrient-rich waters strongly influence the local dynamics of intertidal assemblages (Bustamante *et al.* 1995, Menge *et al.* 1997a, Menge *et al.* 1997b). Similarly, offshore transport generated during upwelling events and their subsequent relaxation, is an important oceanographic process determining temporal and spatial variability in recruitment rates (Gaines and Roughgarden 1985, Alexander and Roughgarden 1996). Variability in the strength of offshore Ekman transport has been proposed to create a gradient in propagule supply of competitively

Water flow affects the reproduction, larval dispersal, settlement, and recruitment, of algae, fish and invertebrates. On average, areas of high flow will be exposed to greater numbers of potential settlers. Larval settlement preferences for regions of particular flow characteristics have been examined in a number of studies (e.g. Mullineaux and Butman 1991, Pawlik and Butman 1993) and it has become clear that many species actively select certain flow regimes, including barnacles (Wethey 1986) and ascidians (Havenhand and 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.

Flow velocities have been found to leave strong bottom-up trophic signatures on shoreline communities. In a tidal estuary in Maine, dense barnacle and mussel cover characterized high flow sites, while low flow sites had considerable bare space. High flow sites also had greater grazer and predator densities than low flow sites. Recruitment of all common organisms with planktonic larvae was greater at high flow sites, in direct proportion to the increased flux (Leonard *et al.* 1998). This is likely to be true for subtidal communities as well.

Light

Macroalgal communities have long been thought of as highly productive systems in terms of both carbon fixation and biomass turnover (Cheshire *et al.* 1996, Turner and Cheshire 2003). At the heart of this productivity is the ability of macroalgae to photosynthesise efficiently, over a range of light conditions.

Light reaching an individual alga can be quite variable and as such, many taxa are able to adjust their photosynthetic apparatus to optimise to the amount available (Fairhead 2001). The advantage of this is that individuals are able to maximise their photosynthetic rate within the prevailing light environment, although lower light conditions are generally associated with a concomitant drop in net 24-hour productivity (Cheshire *et al.* 1996). As a result, there is a lower limit beyond which carbon fixation is not sufficient to cover the individual's physiological requirements. These lower limits have been predicted using models for a range of light environments and subsequently confirmed through field observation (e.g. Turner and Cheshire 2003).

Light quality and quantity are affected by a number of factors that tend to be variable and interactive. At a broad scale, temporal factors such as time of day and season result in changes to the overall intensity of light as well as the angle at which sunlight strikes the water surface. This, along with other factors, results in variation in productivity through time (Cheshire *et al.* 1996, Fairhead 2001, Copertino 2002), with seasonal changes more pronounced at the higher latitudes (Jackson 1987, Davison *et al.* 1991).

In clear waters, light is filtered within the column at different rates depending on wavelength, such that longer wavelengths (red, orange, yellow) are lost in the shallow depths (Dawson 1966). This means that only part of the light spectrum is available for use by photosynthetic organisms in the deeper water.

Shading can also affect the amount of light reaching an individual, as a temporary (e.g. fluctuations in cloud cover) or more permanent phenomenon (e.g. position within the canopy). Algae growing beneath a canopy tend to have a highly efficient photosynthetic apparatus, so much so that the sudden loss of the canopy often causes too much light to reach the plant, resulting in pigment damage and a corresponding reduction in photosynthetic rate (Irving *et al.* 2004).

In general, on subtidal rocky reefs, algae monopolize upward-facing surfaces, whereas sessile invertebrates dominate downward-facing surfaces. The alternative states of algal versus sessile invertebrate-dominated assemblages appear to be primarily maintained by light intensity, but it is likely that light and sedimentation interact with surface orientation to maintain this pattern of habitat heterogeneity (Maughan 2001, Irving and Connell 2002, Connell 2003, 2005).

Sediment

Filter-feeding invertebrates are especially sensitive to suspended sediment loads, and 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. The ability of invertebrates to withstand high rates of sediment accumulation is related to their morphology: erect forms growing above accumulated sediments have greater rates of survivorship than prostrate growth forms, which tend to be smothered by sediments (Irving and Connell 2002).

Different species have different optimal conditions of water movement; for example, ascidians are less dependent on ambient flow conditions than other taxa, which may be 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 substratum 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 (Havenhand and Svane 1991).

Levels of suspended material are low in South Australian waters (Butler 1995), therefore suspension and filter feeders have adapted to processing large volumes of water. 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.

The amount of turbidity within the water column also changes the degree to which light is scattered by particulate matter, and hence the depth to which light can penetrate. As a result, macroalgal communities growing in clear oceanic water can extend their range deeper compared with a more turbid, low wave-energy environment (Lüning 1981, Turner and Cheshire 2003).

Biotic interactions

Many of the physical and chemical properties described above are important in defining broad habitat suitability. The presence of different organisms exploiting a given area inevitably leads to some degree of interaction, creating another level of structural complexity within the system. Interactions between living members of a system take on many forms. Even a brief inspection of the literature will reveal numerous papers describing concepts including competition between individuals, predator-prey relationships (including herbivory), and a host of defensive strategies employed by various individuals. A brief overview is given below.

Competition occurs in nature when a resource that is required by two different organisms is in short supply (Hutchinson 1959, Cale *et al.* 1989). Although in its simplest form the idea has a lot of intuitive appeal, historically it is the focus of much debate within the scientific community (see Lewin 1983). Limiting factors are likely to have different levels of importance for different species. In the case of macroalgae, the availability of suitable substratum is implicated as a major defining aspect of the system (Renaud *et al.* 1996) and it is generally accepted that competition in this regard is likely to be significant. Competition for light may also occur with successive canopies shading those below. Similarly, recruitment trials have demonstrated that the upper side of experimental plates are generally algal-dominated while the light reduced lower sides favour sessile animals (Greig 2000).

Competition, interference (Worm and Chapman 1996) or exclusion (Worm and Chapman 1998) also influence community structure. In South Australia, successional processes often lead to patches dominated by large brown macroalgae (Emmerson and Collings 1998, Turner and Cheshire 2003); however, if turf communities pre-empt the space, then they have the capacity to exclude the larger taxa and persist for several years (Kennelly 1987, Airoldi *et al.* 1995). This can also affect the invertebrate community, as mussels have been shown to be more likely to settle into turf-dominated habitats on Adelaide metropolitan reefs (Smith 2000). Moreover, the mussels have been shown to inhibit macroalgal recruitment (Smith 2000).

Competition from algae plays a major role in determining depth distributions of sessile invertebrate species on temperate reefs however secondary effects such as shading may also be important (Butler, 1995). The macroalgae that usually dominate the upper few metres of any reef system may substantially modify the understorey environment (and hence the associated faunal composition, Duggins and Eckman 1994). Clearance of dominant algae can have large effects on other biota, such as fish and invertebrates (Edgar *et al.* 2004b). Below this algal zone, 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, b, Butler 1991), while slower-growing dominant species (e.g. 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 substratum 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.

Community dynamics are often further complicated by the presence of herbivory. Fluctuating levels of herbivory may influence the outcome of competitive interactions (Gacia *et al.* 1999), and indeed may be one of the most important factors regulating distribution and abundance (see review by Lubchenco and Gaines 1981, also Benedetti-Cecchi *et al.* 2000). Furthermore, herbivory acts as a disturbance event, and may result in a total denudation of an area of substratum, both subtidal and intertidal (Chapman 1981, Schiel 1990). Under high consumer pressure, crustose algae (mostly calcified forms) replace kelps and other foliose algae as the dominant space occupiers in the low intertidal zone (Lubchenco and Gaines 1981). There is latitudinal variation in the patterns of algae/herbivore interactions (see review in Gaines and Lubchenco 1982). This is particularly noticeable for eastern Australian temperate reefs, where large areas of encrusting coralline algae are a visually dominant feature of many areas. Widely known as urchin barrens, these are maintained by the grazing of urchins and other grazing invertebrates (Fletcher 1987).

Where large sessile invertebrates such as barnacles are established, urchins are unable to graze. In NSW, cover of invertebrates on vertical substratum has been shown to be positively correlated with the density of the large barnacle *Austrobalanus imperator* (Davis and Ward 1999), consistent with the barnacle providing a refuge from urchin grazing.

For temperate fish assemblages, habitat complexity appears to affect the density and diversity of fish assemblages (Anderson and Willis 2003). Assemblages in kelp forests are more variable than those in urchin barrens (Anderson and Willis 2003). Structural complexity may also influence the foraging activities of predators and the settlement and recruitment of marine invertebrates (Keough and Downes 1982).

Recruitment dynamics

Biological traits of benthic adults include relative fecundity, spatial and temporal patterns of spawning and larval release, and parental investment. Traits of pelagic larvae include: stage of development at hatching, pelagic larval duration, vertical migration behaviour, horizontal swimming ability, and sensory capabilities (e.g. Kingsford *et al.* 2002). Together, these traits influence where and when larvae are released, where and how they are transported, their ability to move actively in the pelagic realm, and finally, their spatial and temporal settlement cues and patterns. Some direct observations of colonial invertebrate larvae (e.g. the ascidian *Clavelina moluccensis*) indicate that the majority of larvae do not disperse widely, with 80% of larvae settling within 2 m of the parent colony (Davis 1989), while other species disperse very widely (Hedgecock 1986). Physical variables such as site, oceanographic factors, topographic suitability, flow variability, etc, all interact with these biological traits to influence recruitment (Keough and Downes 1982, Underwood and Fairweather 1989, Smith and Witman 1999, Bradbury and Snelgrove 2001, Mora and Sale 2002, Sponaugle *et al.* 2002). Patterns of adult abundance and community structure in marine organisms, from local to biogeographic scales, may reflect recruitment processes. Regional-scale variation of recruitment may reflect geographic patterns in adult stock sizes or fecundities, large-scale hydrodynamic features that influence the transport of larvae (e.g. currents, upwelling), and patterns of early mortality (Hughes *et al.* 2002).

1.4 Unique Features of the Southern Australian Biota

The Southern Australian coastline has often been referred to as 'the unique south.' Reef communities found along this coastline are indeed unique, particularly when considered at a global scale, with a high diversity and proportion of endemic species (approximately 80 to 90 per cent; Table 1; Table 2). This is substantially greater than adjacent tropical systems in which only some 15 per cent of the species found are endemic to Australia (Poore 1995). For example, there are more species of macroalgae growing along the southern coast than there are species of corals on the Great Barrier Reef (Cheshire *et al.* 1998a). A number of hypotheses have been put forward to explain the high diversity and endemicity found in the region, some of which are expanded upon below.

Climate

South Australia has a semi-arid environment with Mediterranean characteristics in the southern part. Summers are generally hot and dry; with most rainfall occurring during the cooler winter months. Cool oceanic influences combine with the semi-arid climate, to produce a temperate oceanic region (Schwerdtfeger 1976). Rainfall is highest around the Mt Lofty ranges where annual median values approach 1000 mm in some areas, although typical values for the Adelaide Plains are less than 650 mm (BOM 2002). Marine habitats of Southern Australia include both cool and warm temperate regions with water temperatures ranging between 10 and 19°C. While typically there is only a 5°C annual variation at any one location, sheltered areas including the South Australian Gulfs tend to experience larger seasonal fluctuations (12 to 25°C) in water temperature (Womersley 1984).

Low nutrient levels

Southern Australian waters typically have low nutrient levels as a consequence of a number of factors. The nearshore coastal ecosystems are effectively isolated from any significant additional nutrient inputs due to the slow weathering and low rainfall of the southern regions of the Australian continent. Combined with the flow of nutrient-poor water from the northern tropical regions (via the Leeuwin and East Australian currents), the result is that South Australian species have evolved or adapted to an oligotrophic environment (Cheshire *et al.* 1998b). This process of evolution and adaptation may perhaps have been facilitated by climatic stability over the past 65 million years (Poore 1995). However, recent research has indicated the existence of a large wind-driven coastal upwelling system that forms during summer along the southern continental shelves, spanning a distance of ~800km (Kampf *et al.* 2004). Coastal upwellings occur simultaneously in three upwelling centres: off southern Eyre Peninsula, off southwestern Kangaroo Island, and along the Bonney Coast. It is hypothesised that this upwelling system provides substantial nutrient input into the near shore coastal ecosystem, evidence of which is shown by the rapid growth of phytoplankton in the region during upwelling events.

Length of coastline

On most continents, long coastlines generally traverse several latitudes rather than longitudes. As a result, their habitats (including species compositions) are strongly influenced by changes in water temperature, which occur with changes in latitude. The most extreme example is the more-or-less continuous north-south coastline of the Americas, crossing from the Arctic though the tropics and to the sub-Antarctic.

In contrast, the Southern Australian coastline lies within a narrow latitudinal range, with an approximate length of 5,500 km, making it the longest stretch of southern-facing coastline in the world. As such, the coastline provides a large area with similar physico-chemical attributes (such as temperature), but also comprises different habitats including exposed rocky shores, gulfs and bays (Poore 1995).

Diversity and endemism

This unique character, with respect to both the physical/oceanographic environment and the biota in this region, has significant consequences to 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 ecosystems. Consequently, it is inappropriate to assume that findings from other ecosystems elsewhere in the world are necessarily applicable to Southern Australian reefs.

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

Taxonomic group	Diversity (# species)	% Endemic	Source
Fishes	600	85	(Poore 1995)
Molluscs	>1000 *	95	(Poore 1995)
Echinoderms	>1000 *	90	(Poore 1995)
Chlorophyta	124	30	(Womersley 1990)
Rhodophyta	800+ (currently >1000)	75	(Womersley 1990)
Phaeophyta	231	57	(Womersley 1990)

* Taxonomic collections are far from complete and it is estimated that only a third of the total invertebrate taxa have been described in South Australia (Edyvane 1995).

Table 2 Comparative diversity of Southern Australian macroalgal taxa (after Womersley 1990)

Region	Coast length (km)	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

1.5 Threats to reef environments

There are a number of anthropogenic inputs into the marine environment that directly influence or threaten near-shore subtidal reef ecosystems, particularly the habitat-forming species such as *Ecklonia radiata* and sponges (Cheshire *et al.* 1998b). The most obvious human impacts threatening reefs include turbidity and sedimentation, nutrient enrichment, opportunistic and exotic taxa, climate change, toxicants and extractive resource use (e.g. fishing). These are discussed in greater detail below.

Turbidity and sedimentation

Increases in turbidity and sedimentation commonly result from dredging, sewage and industrial discharges, stormwater, land reclamation and erosion. In the South Australian gulfs, coastal development, effluent discharge, catchment modification, and seagrass loss have all contributed to elevated levels of sediment within the near-shore marine environment (Turner 2004). Along with pollution, sedimentation is considered to pose a major threat to marine ecosystems in South Australia (Steffensen *et al.* 1989, Cheshire *et al.* 1998a, Gorgula and Connell 2004).

Increased turbidity and sedimentation reduces the amount of light reaching algal communities, reducing photosynthesis. Deposition of sediment is not uniform but dependent on hydrodynamic conditions and the nature of the sediment. As an example, in high wave-energy environments finer sediments are quickly resuspended and only persist where entrapment occurs (e.g. in crevices or through biotic accretion, Airoldi 2003). Over prolonged periods, this can adversely affect the health of the algae. As sediment loads increase, some will begin to settle out. Sediment deposition affects reef biota through a combination of smothering, scour, and by changing the physical characteristics of the substratum (Airoldi 2003). Through these mechanisms, small-scale fluctuations in sedimentation rates have been shown to influence macroalgal community structure (Kennelly and Underwood 1993, Airoldi and Cinelli 1997, Connell 2005). High sedimentation loads can also clog the gills of sessile invertebrates, as previously discussed.

In a recent review of the effects of sediments on rocky reefs, Airoldi (2003) identified a number of common changes to community structure. Generally, organisms that rely upon sexual reproduction are more vulnerable than those using vegetative means, probably due to the lack of substratum stability and the likelihood of smothering of new recruits. In contrast, organisms with sediment-trapping morphologies, or opportunistic species and those with physical adaptations to sediment tend to do well in sediment-affected environments (Airoldi 2003).

Areas impacted by elevated levels of sediment often lose the larger canopy-forming taxa, and tend to have lower diversity, often dominated instead by turf and opportunistic foliose red algae or species with vegetative or migratory life histories (Airoldi 2003). Increases in sedimentation along the Adelaide metropolitan coastline are thought to have contributed to the transformation on many reefs from canopy to turf-dominated macroalgal assemblages (Turner and Cheshire 2002, Connell 2003, Gorgula and Connell 2004, Turner 2004, Connell 2005).

Salinity

Decreases in diversity have been reported under conditions where salinity is less than (Middelboe *et al.* 1998), or greater than (Kendrick *et al.* 1990) the average for open marine systems. Linked with fluctuations in salinity is the presence of fresh water or marine inflows that can also contribute nutrients, suspended matter and pollutants to the near-shore environment.

Nutrient enrichment

Low nutrient concentrations are a major factor limiting plant (and algal) growth (Cosser 1997). Conversely, algal blooms and excessive epiphyte growth are often observed in eutrophic waters. Nutrient availability also stimulates phytoplankton growth that in turn promotes an increase in filter-feeding organisms such as sponges, tubeworms and mussels (Brodie 1997). Increased phytoplankton growth may lead to blooms, which are capable of increasing turbidity and may result in toxic water conditions, although toxin production is limited to only a few phytoplankton and cyanobacteria species.

We are only recently beginning to understand the effects of increased nutrient loads on subtidal reef assemblages in temperate waters. Declines in abundances of some species of fish and invertebrates, and a decline in the species richness of fish assemblages have been demonstrated around a subtidal outfall (Smith and Witman 1999). A recent study at West Island, South Australia, showed that an increase in nutrients had interactive effects with grazers and canopy cover. The loss of canopy-forming algae can be a precursor to nutrient-driven changes of benthic assemblages (Russell and Connell 2005). In the presence of kelp, no effect was detected on macroalgal assemblages when ambient nutrients were increased; however, when nutrients were increased in the absence of kelp and when grazers (mostly molluscs) were present, foliose algae dominated the community. In the absence of kelp and grazers however, and with increased nutrients, filamentous-forming algae dominated space. Steneck *et al.* (2002) believe herbivory is the greatest threat to kelp forests and, although they were principally referring to urchins, the results from South Australia (Russell and Connell 2005) show that combined herbivory and nutrients have the potential to change macroalgal assemblages and reef structure.

In general, turf-forming algae are known to form more extensive habitat on subtidal rock adjacent to urban than non-urban coasts of South Australia (Gorgula and Connell 2004). In experimental trials, the addition of nutrients to the water column had the largest influence on the growth of turf-forming algae, while increased nutrients plus increased sedimentation together were sufficient to explain variation in turf formation between urban and non-urban habitats in South Australia (Gorgula and Connell 2004).

Urchin barrens are common in New South Wales and Tasmania but are rare and not extensive in South Australia except in parts of mid to upper Spencer Gulf (S.A. Shepherd, pers. com.). Russell and Connell (2005) postulate that this may be due to Southern Australian waters having typically low nutrients and therefore ecosystems are more strongly influenced by bottom-up inputs instead of top-down interactions. It is possible that increases in nutrients so that they are no longer limiting may allow top-down interactions to play a more important role in structuring the reef, allowing trophic cascades to begin. Alternatively lower urchin numbers may be due to lower larval supply and or increased predation.

Toxicants

The substances in the marine environment that are of most concern are those that are persistent and toxic even at low concentrations. Many inhibit growth or recruitment and are often associated with urban and stormwater runoff, and industrial discharges. Some chemicals may also bind to fats leading to bioaccumulation in organisms. The degree to which any chemical accumulates in an organism depends on the chemical and the organism itself; however, it may be as high as 500,000 times greater within the organism than in the surrounding seawater (Bryan 1979, Edgar 2001).

Suspension feeders are at the greatest risk of having high concentrations of toxicants as they filter large quantities of water and so accumulate the toxicant. Algae are also likely to have high concentrations of toxic substances due to their large surface-area-to-mass ratio. Bioaccumulation up the food chain is of particular concern. Both carnivorous animals and particularly filter feeders eat many times their own body weight in prey, all potentially containing the toxic substance. Heavy metals, for instance, can cause cancer, behavioural disorders and other problems in a broad range of mammals, including marine mammals (Irwin *et al.* 1997), and can adversely affect human heath if built up in the tissues of fished species (Olsen 1983). The toxicants that are of most concern are heavy metals, tributyltin, organochlorine pesticides, dioxins and polychlorinated biphenyls. While South Australian waters are not polluted by world standards, high metal levels have been found in water, sediments (Anon 1996, 2000), fish (Edwards *et al.* 2001) and dolphins (Butterfield 2003) in the Port River system in Gulf St Vincent, and from upper Spencer Gulf in sediments, seagrasses (e.g. Ward 1987), and fish and molluscs (Edwards *et al.* 2001).

Extractive resource use

Extractive resource use is capable of instigating change in subtidal reefs, with the most common use being fishing. Fishing is known to have numerous effects on the species targeted, including reducing average size, fecundity, and behavioural changes (Tegner and Dayton 1999, 2000, Shepherd and Baker in prep.). In cases where levels of exploitation are high, effects can be severe, with fisheries being in decline worldwide (Tegner and Dayton 1999), and nearly one in four collapsing between 1950 and 2000 (Mullon *et al.* 2005). Worldwide it is estimated that up to 90% of large predatory fish have been lost (Myers and Worm 2003).

Current figures for fish stocks managed by the Australian Government indicate that fourteen (19%) species are considered ‘overfished’ with the status of a further 40 species (54%) being uncertain (Caton and McLoughlin 2004). It is disturbing to note that this represents an increasing trend towards ‘overfishing’ in the past decade in spite of changes to management (O'Brien 2004). In South Australia, most species of commercial interest are considered to be ‘fully exploited’ with a further two classified as ‘overfished’ (Nicolson *et al.* 2003). For coastal reef fish species in Gulf St Vincent, the greatest impact appears to be through recreational fishing activity and primarily through rock fishing (Shepherd and Baker in prep.).

In addition to affecting the targeted species, fishing also has cascading effects onto other marine biota. Probably the best documented of these is the formation of urchin barrens as a result of the removal of predators of urchins, such as sea otters in California (Fanshawe *et al.* 2003) and lobsters in New Zealand (Shears and Babcock 2003).

Closures of reefs to extractive industries such as fishing can have widespread ecosystem effects, and result in dramatic changes in the abundances of both macroalgal and fish species (Edgar and Barrett 1997, Shears and Babcock 2002).

Research at Leigh Marine Station, New Zealand, where a Marine Protected Area was declared 25 years ago, showed major community changes after fishing was banned (Shears and Babcock 2003, Parsons *et al.* 2004). Between 1978 and 1996 benthic communities shifted from being dominated by sea urchins to being dominated by macroalgae. This was a result of a trophic cascade thought to be an indirect effect of increased predator abundance. Densities of sea urchins have continued to decline in shallow areas of the reserve and after 25 years of protection, all sites classified as urchin barrens in 1978 were dominated by large brown algae. Lower densities of grazing molluscs were also found at reserve sites, and are thought to be responses to changes in habitat structure, representing additional indirect effects of increased predators (Shears and Babcock 2002, 2003).

Other extractive industries such as sand or mineral extraction can also impact on reefs. As an example, a study into the impact of sediment plumes, associated with near-shore sand mining on Adelaide's southern metropolitan coastline, demonstrated a considerable level of degradation on Noarlunga and Horseshoe Reefs (Turner 2004).

Coastal development

The majority of Australians live near the sea. In coastal cities such as Adelaide, seaside suburbs are almost entirely developed with little of the natural coastal system remaining. From an ecological point of view, coastal development such as housing, marinas, aquaculture operations, industry, boat ramps and wharves, and dredging for various purposes has caused widespread impacts, like overfishing, can make an ecosystem more prone to invasion by opportunistic species (e.g. Levine 2000, Harris and Tyrrell 2001).

2 Approaches to the assessment of health in reef environments

A problem associated with biological studies is dealing with the enormous complexity of ecosystems and how the information gleaned from the respective studies can be fed back into the environmental decision-making process. One method of representing the state of the environment in a simplified way is to use biological indicators. These are variables, or indices that integrate and characterise information embodied in comprehensive data sets, which are often not directly measurable. Biological indicators attempt to encompass the multiple webs of actions, reactions and interactions observed in ecosystems and therefore attempt to represent extremely complex conditions in a strongly-condensed form (Müller *et al.* 2000).

Through the use of biological indices, patterns over time or space can be evaluated to determine the effectiveness of environmental protection and management measures in maintaining ecosystem health and services (EHMP 2004). A key feature required from the indices is the ability to detect changes in the ecosystem whether natural or man-made, and to predict the ecosystem's response to these changes.

Budget size is generally a strong driving factor in determining the design of monitoring programs and as a consequence biological indices are often considered as a powerful and cost-effective tool in providing information about the state of an ecosystem, although it is important to take into consideration that extensive studies may be required to determine which indicators are suitable for specific ecosystems and these studies in themselves come at a cost. Overall, it is important to be aware that the development of biological indicators to assess the state or health of an ecosystem requires careful thought, particularly when considering the extent of the complex biological processes that exist in the subtidal marine environment (Fairweather 1999).

The following sections provide a discussion on what defines "ecosystem health" and what biological indices are available in marine temperate ecosystems to provide information about the "state of the environment".

2.1 Definition of ecosystem health

The term "ecosystem health" has increasingly been utilised in both the public and scientific forum, and has become a popular goal for environmental managers (Xu *et al.* 2004). Whether the scientific community has truly accepted the term is a point of conjecture, but since the early 1980s the notion has grown to the extent that there are a number of international journals (e.g. Ecosystem Health, Journal of Aquatic Ecosystem Health) devoted to the subject.

The health metaphor has evolved over many years and has existed as a variety of different analogies (e.g. GAIA: the concept of the earth as a living organism as proposed by Lovelock, Lovelock 1979). In the late 1970's the notion of health was extended to ecosystems from medical science when it was considered that many human-dominated ecosystems were highly 'stressed' and thus ‘dysfunctional’ (Rapport *et al.* 1998, Xu *et al.* 2004). It was proposed that as dysfunctional ecosystems degrade, their ability to supply services was substantially impeded and thus the capacity of the environment to sustain economic activity and human health was therefore greatly reduced (Rapport *et al.* 1998). The implication is that the state of the environment or ecosystem directly influences the health of human society; therefore ecosystem health is analogous and contributory to human health (Rapport *et al.* 1999).

Rapport *et al.* (1998) consider that the definitions of ecosystem health are all closely aligned with concepts of stress ecology, where health is considered in terms of ‘system organisation, resilience and vigour’ (Rapport *et al.* 1979, Rapport *et al.* 1998) and an absence of symptoms of ‘ecosystem distress’ (Costanza 1992). An ecosystem is healthy if it is free from “distress syndrome… stable and sustainable… active and maintains its organisation and autonomy over time and is resilient to stress” (Haskell *et al.* 1992).

The concept of ecosystem health is subjective; judgements are made against what we expect a healthy ecosystem to be like (Fairweather 1999), and often this is very dependent upon a mixture of scientific, social and political objectives (Fairweather 1993). It is powerfully emotive to call undesirable system states “unhealthy” (Cheshire *et al.* 1998b). It is perhaps more reasonable and intellectually honest to accept that ecosystem health is in fact just a desired state. The real problem with the analogy is that it appears to be very difficult to succinctly define the properties that make an ecosystem healthy.

Cheshire *et al.* (1998b) suggest that one approach is to define the normal or “baseline” state, by listing the properties that constitute a putatively pristine ecosystem (i.e. one that has been unaffected by human activity), analogous to early medicine where physicians sought to correlate body states with conditions of health and ill-health. However they provide a caveat to this argument, stating that to use some ecosystem state as a baseline from which to judge the effects of human activities requires a clear specification of the relationship between structure and environmental factors. The problem is that we are far from possessing such understanding for southern-temperate Australian reefs (Cheshire *et al.* 1998b).

Rapport (1992) argues that it is far easier to recognise the symptoms of an ecosystem in ‘distress’, than to determine what represents ‘good’ health. The symptoms of an ecosystem in ‘distress’ include (Rapport *et al.* 1995):

- A shift to smaller organisms
- Reduced diversity with loss of sensitive species
- Increased dominance by weedy and exotic species
- Shortened food chain lengths

- Altered energy flows and nutrient cycling
- Increased disease prevalence
- Reduced stability (ecosystem is constantly changing)

Listing the symptoms of distress allow for some form of value judgement to be made about the state of the environment, but it still requires basic knowledge of the ecosystems considered. Additionally, most of these symptoms require prior information against which to measure change.

Fairweather (1999) argues that, for environmental assessment, we should adopt a whole-system approach, as there is a strong limitation in our understanding of human effects on ecological functions and processes. That is, to determine whether an ecosystem is healthy or not, processes should be targeted rather than species. Hence, there is a need to gain knowledge of ecological function to assess whether a system is 'dysfunctional'. In the sphere of environmental assessments, ecosystem health should focus on the effects *per se* to a system rather than concentrate on levels of pollutants or other things that may or may not be the cause of the problem (Fairweather 1993).

It is clear that the definition of ecosystem health is fraught with problems, often running the risk of becoming an anthropomorphic cliché. Overall the health metaphor is probably more useful for public communications and bringing together disparate disciplines concerned with different aspects of environmental change (Fairweather 1999). Perhaps the definition is best left in a simplistic form, by describing ecological health as a function of the key processes that operate to maintain a stable and sustainable ecosystem (Dennison and Abal 1999).

2.2 Indices for assessing reef health

A general problem of all ecological analyses and environmental decision processes is the dealing with the enormous complexity of the investigated ecosystem (Müller *et al.* 2000). Defining an appropriate set of indices for assessing reef health is a step towards dealing with this problem.

Indicators need to represent extremely complex conditions in a strongly condensed form. Therefore indicators should be correct from an analytical point of view; they should do justice to statistical demands, such as whether they are quantifiable, reproducible, valid, or sensitive; and should have a high qualitative ecological foundation (Müller *et al.* 2000). The indicators should not only take into account pragmatic arguments and but also adhere to scientifically rigorous protocols.

Monitoring protocols are normally ecosystem specific and are generally built upon a knowledge base of the system that is to be studied. In developing a set of parameters or indices to assess an ecosystem, a whole suite of physical, chemical and biological properties can be considered. In any given case the choice of parameters or indices 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 an ecosystem (Cheshire *et al.* 1998b). Time and financial resources are other factors taken into consideration when indices are chosen, as are issues of accuracy and repeatability, as well as underlying environmental variability.

Indices can be broadly placed into three major groups. These are:

1. Water quality indices
2. Ecotoxicological indices
3. Biological indices

The following section will discuss the appropriateness of these indices in assessing reef health and what indices have been considered in past studies, particularly those that are applicable to Southern Australian temperate reefs.

Water quality indices

Traditionally water-quality parameters are often utilised to assess ecosystem health, as generally they are cost effective and comparatively easy to acquire, particularly with the development of a wide range of *in situ* sensors. Data collected are usually assessed against predetermined "trigger values". If the parameters measured exceed the prescribed trigger value, this indicates that the water quality is poor and that the health of the surrounding ecosystem may deteriorate. The trigger values are values that have been developed from water quality data collected over a number years, which take into account seasonal and annual changes in the natural variability of the parameters measured, for a particular ecosystem (ANZECC 2000). Trigger values for a suite of water-quality parameters and the method used for determining these values are presented in documents such as the Australian and New Zealand Guidelines for Fresh and Marine Waters, Volumes 1 & 2 (ANZECC 2000). The Environment Protection Policy (EPA 2003) provides a revised series of trigger values for South Australian aquatic ecosystems. Ecosystems are broadly placed into three categories; marine, freshwater and estuarine.

The majority of the water quality parameters measured provide information on the physical and chemical attributes of the water column, rather than the biological attributes. The parameters commonly measured include:

- Turbidity, dissolved oxygen, salinity, pH, secchi depth and water temperature
- Phytoplankton biomass (correlated from chlorophyll *a* concentration)
- Nutrient concentrations – e.g. total nitrogen (TN), total phosphorous (TP), oxides of nitrogen (NO_x), ammonium (NH_4^+) and filterable reactive phosphorus (FRP)

- Heavy metals (e.g. zinc, cadmium, copper, lead)
- Delta N measurements (ratio of N^{15} to N^{14})

Water quality is a major determinant of community structure. This is very evident when considering nutrient concentrations for near-shore coastal ecosystems. In South Australia, near-shore coastal ecosystems are generally considered to be nutrient poor, particularly with respect to dissolved nitrogen (Jeffrey *et al.* 1990). However, nutrient enrichment has occurred along many near-shore ecosystems due anthropogenic inputs such as wastewater treatment discharges, stormwater and industrial and agricultural effluents. Opportunistic algal species are able to utilise these nutrients (e.g. *Ulva lactuca*) and the resulting blooms have displaced local species (Steffensen 1976), diminished light availability to benthic plants and macroalgae, and caused fluctuations in pH and dissolved oxygen, which have stressed or eliminated species sensitive to these changes (Herbert 1999).

The Environment Protection Authority of South Australia has monitored nutrient, bacteria and heavy metal concentrations along the metropolitan coast as part of their ambient water quality monitoring program (Gaylard 2004), with samples being collected monthly in the vicinity of jetties. The main findings from the study were:

- Elevated concentrations of ammonia observed along the length of metropolitan coast;
- Elevated concentrations of oxidised nitrogen adjacent to Glenelg Waste Water Treatment Plant, Torrens River, Patawalonga outlet and numerous storm water drains;
- Elevated concentrations of nickel and zinc;
- Elevated concentrations of chlorophyll *a*.

Inferences can be made from these results on how biological process may be affected by elevated concentrations, such as bioaccumulation of heavy metals in fauna and flora having adverse effects on biological processes. The problem remains though, that unless the water-quality indices are linked with other studies examining the biological processes, the conclusion drawn from the results remain subjective. For example, measuring nutrient levels in the water body provides knowledge about ambient concentrations but it provides no knowledge about which organisms are utilising the nutrients. Also, nutrient measurements may be low because of fast uptake by organisms.

Exceeding water-quality trigger values suggests that problems exist with the health of the ecosystem, but the description of the ecosystems outlined in the policies (ANZECC 2000, EPA 2003) are very broad (e.g. ‘marine’). The policies do not take into account natural variability that may exist in the chemical (e.g. metals, nutrients) and physical (e.g. water motion, turbidity, temperature) parameters along the South Australian coast.

Water-quality parameters are relatively easy to collect but the data obtained are not always very meaningful. Most parameters are highly variable in time and space and any sampling program therefore needs to take into account this variability. Caution should be taken with spot measurements of water quality. 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 (Cheshire *et al.* 1998b).

Ecotoxicological indices

A growing number of studies incorporate ecotoxicological assessments into the decision-making process of environmental management (Crowe *et al.* 2004, den Besten and Munwar 2005). Ecotoxicological studies are generally used as effect-based studies to determine the consequence of an action (such as dredging or desalination plant discharge) on ecosystem health. Tests on organisms can be chemically oriented, focusing on the mode of action of a toxic compound, or be ecologically oriented, aiming to link cause and effect observed in the field (den Besten and Munwar 2005). Ecotoxicological testing is perhaps not a true indicator of health as it is generally used for risk-based assessments, but it is briefly discussed as the species chosen are used as indicator species to make judgemental decisions on how an ecosystem reacts if the ambient conditions are modified.

Most environmental monitoring is concerned ultimately with effects of environmental contamination on populations and communities of organisms (den Besten and Munwar 2005). Due to the high cost and complexity of sampling biological communities, surrogate biological indicators or concentrations of toxic contaminants are often the only variables measured. Assumptions are then constructed on the level of impact for populations or communities (Crowe *et al.* 2004). Such indicators can be powerful tools, but contamination does not necessarily result in an obvious impact, particularly if the toxins function on a chronic timescale, with gradual degradation of biological activity (Paine *et al.* 1996). Also, it is often inferred what the impact will be on natural populations and communities and the effectiveness of the test is not always evaluated in the field (Crowe *et al.* 2004). Tests are often based on a few individual species, many of which are not found naturally in the ecosystem examined (e.g. juvenile trout are often used to test chemicals that are potentially released into the marine environment).

There are exceptions and ecotoxicological studies are tending more towards using local species, particularly in Southern Australia. These species include *Hormosira banksii* (Kevekordes and Clayton 1996, 2000, Kevekordes 2001), *Ecklonia radiata* (Burridge *et al.* 1996) and *Mytilus* spp. (Crowe *et al.* 2004), all of which are found on temperate reefs; however species are generally chosen on the basis of whether they are able to survive laboratory manipulation (germinating gametes in petri dishes in the case of *Hormosira banksii*) or are capable of being deployed in the field (e.g. mussels).

There is still the question of how contaminants affect different trophic levels within an ecosystem. The contaminant may not affect the test organism, but may affect other organisms in the food web, and thus change the ecosystem over a period of time. If there is more than one chemical involved there may be a synergistic response, in which the combination of chemicals is more toxic than if the test organisms are exposed to them separately. In summary, ecotoxicological studies have their place in assessing the risks of chemical contaminants on ecosystem health, but the limitations of the studies need to be clearly defined.

Biological indices

Biological assessment may be used to assess the ecological health of an ecosystem. It provides a direct measure of ecosystem health rather than, for example, the episodic sampling of fluctuating water quality. There is, however, no generally agreed method for choosing what to measure in an attempt to determine whether an ecosystem is healthy or not (Underwood *et al.* 2000).

It is difficult to quantify what is suitable as a broad-scale indicator of ecosystem health. Impact studies often select a particular species or a group of species to indicate when an impact (e.g. dredging, sewage discharge) has had an adverse effect on the community. The decision to choose which species to use often requires extensive knowledge of an ecosystem, and the ecology of the species. Equally, to demonstrate that the population ecology of one species is like that of many others requires that someone investigates them all in detail (Underwood *et al.* 2000). There is no current agreement about how to choose indicator species, despite widespread discussion about the types, features and properties of species (Underwood and Peterson 1988, Keough and Quinn 1991). More often than not, a species is chosen because it is common and easily identified.

Macroalgae have been used as indicators for pollution for decades (Levine 1984), particularly in the northern hemisphere. As indicators of health, they have several intrinsic advantages: they are sessile and therefore can be used to characterise one location over time; they are easily collected in abundance at many localities; and they readily accumulate compounds present within the waters of their environment (Levine 1984).

At a community level, abundance of macroalgal taxa has been taken as an indication of environmental conditions (Eklund and Kautsky 2003). Investigations of community structure can be revealing but there are disadvantages inherent in this approach (Levine 1984). Sampling, separation and enumeration of organisms are time consuming and require expertise. Adequate taxonomic keys are not always available, dependent upon location, or require specific training to utilise (Levine 1984). Care is also required with the interpretation of the results because cause and effect relationships for species distributions are intricately associated with natural processes. In reviewing studies of Southern Australian rocky reefs, Underwood and Kennelly (1990) described the extent of knowledge of the processes responsible for structuring macroalgal communities as limited, albeit progress has been made in a number of areas since this publication.

An alternative method to seeking a particular species to provide an indication of the state of biological function is to assess ecosystems based on functional groups or lifeforms (Steneck and Dethier 1994). Using this approach, often unrelated species are grouped based on the role they play within a community rather than on their phylogenetic affinities. This method has been used extensively with success for coral-reef surveys throughout Asia (English *et al.* 1994), and has been adapted to surveying reefs in South Australia (Cheshire *et al.* 1998b). The concept is based on the assumption that instead of an impact affecting individual species, it is more likely to affect a whole suite of life forms or functional groups.

The results from a survey of South Australian reefs suggested a shift in communities dominated by robust brown algae to foliaceous red algae in some areas (Cheshire *et al.* 1998a). The change in community structure was correlated with changes in exposure to water motion and anthropogenic influences. This study suggests that perhaps it is more important to look at patterns and processes controlling ecosystems than trying to determine how an impact will affect individual species. This is an approach that Fairweather (1999) considered in determining the health of New South Wales estuaries, and he suggested that biological indicators should focus on ecological processes rather than on traditional structural measurements.

An example of an ecological process that could lend itself as an indicator is recruitment or immigration of organisms into a community (Fairweather 1999). Recruitment (defined as the number of juvenile macroalgae per metre squared) was successfully used as a biological indicator to determine the impact of a sediment plume generated from a sand dredge, on reefs off the metropolitan coast of Adelaide (Turner and Cheshire 2002, Turner 2004). The study tracked the phaeophycean-dominated macroalgal assemblages over a four-year period and demonstrated that there was a significant reduction in recruitment of a number of taxa for areas impacted by the sediment plume, which was then followed by a slow recovery; however this method proved to be very labour intensive.

It is clear that there is no easy solution in choosing an indicator to determine the ‘health’ of an ecosystem. Whether the indicator is a species, a functional group or a biological process, there will always be some countervailing arguments within the scientific literature. In assessing ecosystem health we need to build from our knowledge of the ecosystem. The chosen indicator species should allow us to differentiate between natural variability inherent within ecosystems and population or community structural shifts due to anthropogenic influences. It is a difficult quest that will always be dependent upon constraints in resources both financially and technically. The final hurdle is perhaps succinctly defining what our expectations are for an ecosystem that can be described as “healthy”.

3 Establishing a monitoring program for South Australian reef environments

The use of a system of indicators in the assessment of ecosystem status needs to be placed within the context of a monitoring program that is both pragmatic and scientifically robust. In this section, some of the important considerations for designing a monitoring program are discussed.

3.1 Designing a reef monitoring program

The need for ecological assessments

Ecological assessments are made as a basis for addressing the status and or functioning of the system in question. In order for results to be meaningful, biological assessments need to be based on a sampling design that is appropriate for the objectives of the study. As such, it is generally necessary to tailor studies to the particular circumstances. Notwithstanding this, it is useful to identify a number of broad categories of assessment (based on Kingsford *et al.* 1998):

- Baseline studies – used to define the present state of the system;
- Impact studies – identify the type and magnitude of a change resulting from a particular perturbation;
- Monitoring studies – involve repetitive sampling and should be sensitive enough to detect efp5h .aheoul9

such, the monitoring program would need to comprise a group of methods that together encompass a number of different organisational levels within the system.

Sampling design

Sampling is a necessary compromise because it is pragmatically impossible to undertake a complete census. In designing a monitoring program, Keough and Mapstone (1995) identify two opposing forces that influence the design of the program. The first is the requirement for sampling to be able to detect changes (especially impacts) at an acceptable level. This is juxtaposed against the need to minimise the costs associated monitoring so as to be feasible to undertake.

Sampling produces an estimate of the actual value, and the precision of this estimate will depend on the amount of variation associated with the population in question and the intensity of the sampling. Increases in variability will reduce the precision of the estimate, which is an important consideration because it increases the size of the minimum detectable change.

Sampling programs need to be well planned if they are to achieve the objectives of the study. While specific design requirements will vary depending on the issues to be addressed, Kingsford and Battershill (1998a) provide a number of guidelines for establishing a sampling program (see below):

- All levels of sampling should be replicated;
- Be aware of small scale variation (e.g. time of day, within site);
- Consider the eventual statistical analysis during the design phase;
- Replicate samples should be statistically independent;
- Consider the biology of the organisms being studied; and
- Effects can only be demonstrated through comparisons with control sites.

The placement of replicate samples also depends on the nature of the study, with stratified approaches often being used. As an example, samples may be placed randomly or haphazardly within a predetermined habitat type and/or along a depth contour. Alternatively, samples may be taken across different habitats and or depths to provide an overview of the environment (Kingsford and Battershill 1998a). It has also been shown that several rapid assessments of percent cover yielded a far more accurate picture of the true nature of reef systems, than an intensive localised sampling regime run over the same period of time (e.g. Kinzie and Snider 1978).

3.2 Units of measurement for the assessment of biota

Taxonomic resolution

There are often practical limits to sampling resolution; in particular the formal identification of taxa to species for many macroalgal and invertebrate groups requires expert knowledge. For this reason studies often trade a coarser level of taxonomic resolution for increases in sampling efficiency (i.e. each sample is completed more rapidly, allowed a greater number of replicates to be made). The use of functional forms has often been used for rapid assessments and is an integral part of the standard methodologies used for coral reefs systems (English *et al.* 1994).

Steneck and Dethier (1994) argue that the use of functional groups (or operational taxonomic units) can be justified for macroalgal systems because relatively few attributes are of overriding importance in structuring the assemblage. Functional classifications have also been used effectively for surveys involving intertidal (Metaxas *et al.* 1994) and subtidal (Cheshire *et al.* 1998a, Cheshire and Westphalen 2000, Turner and Cheshire 2003) macroalgal assemblages.

Classifications at species and family levels produced similar results for freshwater macro-invertebrate assemblages (James *et al.* 1995, Wright *et al.* 1995, Chapman 1999), although the studies also found that observations became blurred at the level of order. Coarser resolution was also found to reduce differences in holdfast fauna between sites (Anderson *et al.* 2005).

Sessile and sedentary biota

"Sessile" and "sedentary" refer to biota that are either fixed to the reef and cannot move, or move very little (e.g. macroalgae, sponges, ascidians, mussels). As such, the patterns of distribution of these organisms fluctuate less than for more motile species including fish. These types of biota have been quantified using a wide variety of sampling formats (see reviews in Littler and Littler 1985, Coyer and Witman 1990, English *et al.* 1994, Kingsford and Battershill 1998b), with the most common employing quadrats, transects and visual censuses.

Quadrats

Quadrats of all shapes and sizes have been used in ecological studies and a number of authors have explored the efficiency of the method (e.g. Bormann 1953, Weigert 1962, Brummer *et al.* 1994). Within a quadrat, assemblages may be determined through direct observation, or indirectly through photo or video.

Currently there are limits to the resolution that can be obtained by remote techniques, although the method is certainly suitable for picking up broad changes in community composition (Miller *et al.* 1998). Such methods do, however, have a number of pragmatic advantages by reducing diver effort and thus reducing sampling costs. In addition, techniques involving three-dimensional photographic methods are allowing for greater accuracy (e.g. Roerslett *et al.* 1978, Svane and Gröndahl 1988) and produce an archival record that can be re-examined.

The most common measures obtained using quadrats are presence/absence, percentage cover, and abundance (counts or biomass) per unit area.

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 in each sampling unit 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 of occurrence (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 (Cheshire *et al.* 1998b).

Cover measurements are generally applied to an assessment of sessile biota and involves an estimation of the proportion of any given area occupied by each taxon within a community. This method is also 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 (Cheshire *et al.* 1998b), or for other sessile organisms as an alternative to counts. A variation on this theme is the use of point intercept quadrats, which are now widely used in rapid assessments in both Tasmania (e.g. Edgar *et al.* 1997) and Victoria (Edmunds and Hart 2003). In addition to quadrats, cover measurements of this type are also collected using line intercept transects and visual censuses (see below).

Counts of organisms within samples provide a density measure of abundance, generally reported as number per unit area. 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, Cheshire *et al.* 1998b).

Measurements of biomass (expressed as dry weight per unit area) provide one of the best indications of the relative amounts of different taxa present but are necessarily destructive when applied to any sessile organism. Furthermore, for organisms with large inorganic components such as hard corals, shelled molluscs, some sponges and some algae, the measurements need to be adjusted to account for the non-living biomass (Cheshire *et al.* 1998b). Biomass measures are usually made based on harvesting all biota from within quadrats of known size. The method also minimises field time given that identification and measurement of the connected material can occur in the laboratory. These factors have led this technique to be widely adopted as a standard sampling method (Littler and Littler 1985).

Transects

Transects are lines laid according to a predetermined protocol (e.g. randomly, following a depth contour or in a particular direction, etc). Sampling may either take place directly along the line (as in line intercept transects) or employ a second format (e.g. quadrats, visual census, belt transects), that use the transect line as a guide.

Line intercept transects (LITs) were originally used in terrestrial studies (Cottam and Curtis 1956, Webb *et al.* 1970), and have also been used effectively in marine environments (Marsh *et al.* 1984, Reichelt *et al.* 1986). LIT is now a standard method for surveying tropical marine ecosystems and is used throughout the world (English *et al.* 1994). The methodology has also been used in temperate, macroalgal-dominated systems and (with some modification) found to be an efficient technique for obtaining quantitative assessments (Turner 1995). In South Australia, several studies have made use of the technique for rapid, non-destructive benthic sampling (e.g. Cheshire *et al.* 1998a, Cheshire and Westphalen 2000, Turner and Cheshire 2003).

Very mobile biota

Visual census

On shallow rocky reefs the most common means for estimating the abundance of fish and large motile invertebrate is visual census by divers. This method has been used extensively in the literature and has been standardised in tropical marine environments (English *et al.* 1994). Similar standardisations are also occurring in temperate Australia (Edgar *et al.* 1997, Edmunds and Hart 2003). Surveys are generally conducted along transect lines recording information on the size and abundance of the organisms within a certain pre-defined distance of the line.

There is inevitably some degree of bias relating to differential visibility of different taxa and observer behaviour (experience and subjective decision making). However, a recent assessment of these biases found them to relatively consistent amongst trained divers, meaning that compensation can be applied (Edgar *et al.* 2004a). Notwithstanding, visibility needs to be adequate in order to apply this method.

3.3 Concluding remarks

The concept of ecosystem health is subjective and what we mean by health in a South Australian context is difficult to define without further work. Furthermore, without a clear definition of health, it is difficult to identify a clear set of indices to base assessments upon.

Reasonable information is, however, available for many aspects of local reef environments and in the last few years a number of studies have begun to address the types of processes that structure these communities. Furthermore, the previous Reef Health studies also provide a solid foundation on which to further develop the program.

relating not only to the assessment of health, but also broader principle of reef ecology. At least a subset of the data collected should also be compatible with that obtained through the previous Reef Health work to allow for temporal comparison.

In establishing the monitoring program to assess the health of reef environments, a number of pragmatic issues need to be considered. These include limitations in the current knowledge of these environments, and also the level of resources required to make meaningful assessments. As a starting point the choice of 'suitable' indicators and appropriate thresholds may need to be based on a combination of, that obtained through reviews of the literature, along with input from experts.

Following on from this, knowledge limitations need to be addressed through carefully planned studies of important components of reef environments. It is likely that as understanding evolves, so will the types of indices used, and the level of confidence that may be attached to assessments of reef health. As such this is likely to be an iterative process, and monitoring programs need to be designed with the capacity to be modified where necessary.

As part of this process, there needs to be a degree of involvement from a range of stakeholders and management agencies. Both to assess the suitability of the developing protocols and also to establish the sorts of responses appropriate to survey outcomes.

Ecological monitoring is expensive and the level of resources made available for reef assessments will place restrictions on how comprehensive studies will be. In particular, it is likely that sampling will be intermittent, necessitating that the chosen indices need to be robust enough to provide quantitative information on system state based on "snapshots". For this reason it will be necessary to use biological indices rather that simply measuring water-quality parameters.

The role that the broader community can play in ongoing monitoring of reef environments also needs to be carefully reviewed. Trained volunteers already make a valuable contribution to biological monitoring in a number of fields (e.g. birds, frogs), and volunteer monitoring already exists for tropical marine environments.

In temperate Australia, Reef Watch monitoring programs exist in South Australia and Victoria, while Western Australia and Tasmania have other programs with similar objectives. Experience gained from these programs demonstrates that trained volunteers do have the ability to collect good quality data for reef environments and that this information can be used to greatly increase the overall monitoring effort.

Recommendations

It is recommended that in establishing a monitoring program:

- Indicators should be chosen to represent important facets of the ecosystem including structural components and integral processes;
- Trigger values need to be set for the various indicators;
- Protocols must be established that outline what actions are to be taken when a trigger value is crossed, and these must have the support and backing of the relevant management agencies;
- Sampling programs should be non-destructive wherever possible to minimise the impact of repeated surveys;
- Consideration needs to be given to methodological protocols that can be adopted by volunteer divers. Alternatively compatible methods should be developed that will serve broader community use; and
- Effort should be placed into fostering greater community involvement in reef monitoring initiatives through program development and education initiatives.

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