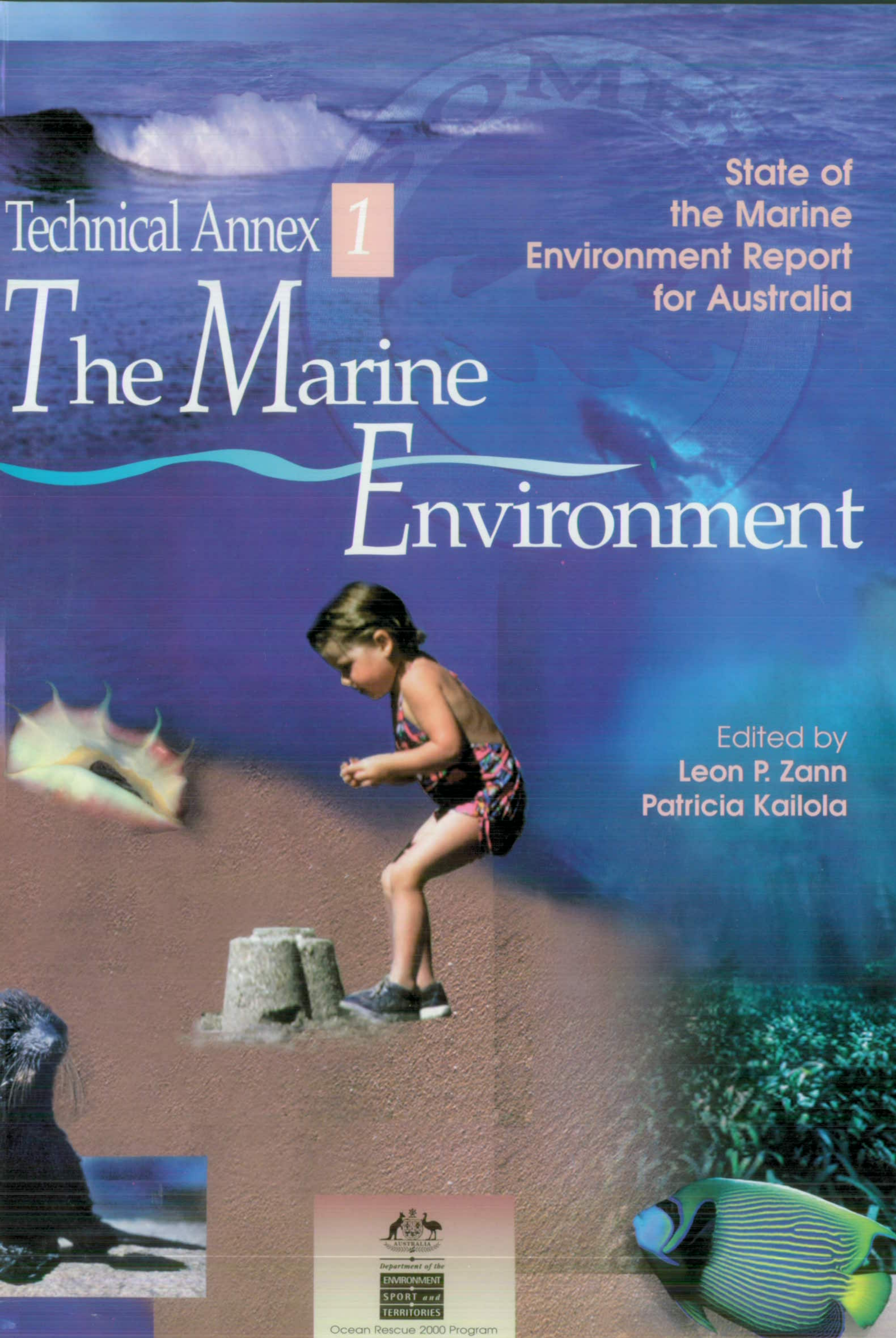

Technical Annex 1

The Marine Environment

State of the Marine Environment Report for Australia

Edited by
Leon P. Zann
Patricia Kailola

Department of the
ENVIRONMENT
SPORT and
TERRITORIES

Ocean Rescue 2000 Program

AUSTRALIA'S 200 MILE EXCLUSIVE ECONOMIC ZONE

(excluding Macquarie, Heard, and Macdonald islands)

Original map by CANDATA reproduced from Australian Fisheries Resources, Bureau of Resource Sciences, 1993.

The State of the Marine Environment Report for Australia Technical Annex: 1

The Marine Environment

Edited by
Leon P. Zann
Patricia Kailola

Great Barrier Reef Marine Park Authority
Townsville, Queensland, Australia

Ocean Rescue 2000
Department of the Environment, Sport and Territories, Canberra

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Introduction

Australia is an island continent and the sea is very important to Australians. A quarter of the population lives within three kilometres of the coast, some 86% live in the coastal catchments, and two-thirds reside in coastal towns and cities.

Australia's coastline, including that of Tasmania, is almost 70 000 kilometres in length. Australia's seas are vast in size and have a rich and unique marine biota. Australia's newly proclaimed 200 mile Exclusive Economic Zone (EEZ) is over 11 million square kilometres in area, and is one of the largest in the world. It spans 33° of latitude (58° including the Antarctic Territory), and encompasses all five ocean climate zones.

The sea has great economic value to Australia. Coastal and marine tourism, fisheries, marine transport, and offshore petroleum are estimated to be worth around $17 billion per year.

Our view of the sea has changed greatly over the past 40 years. In the 1950s the sea was regarded as the last frontier. In the 1960s it was seen as the solution to the increasing resource depletion on land. By the 1970s there were early concerns about the vulnerability of coastal waters. During the 1980s these deepened as some fisheries and marine ecosystems began to decline.

In 1990 the Group of Experts on the Scientific Aspects of Marine Pollution (GESAMP), reporting to the United Nations on the health of the world's oceans, concluded that 'chemical contamination and litter can be observed from the poles to the tropics and from beaches to abyssal depths', and if allowed to go unchecked, this would lead to 'global deterioration in the quality and productivity of the marine environment. We fear, especially in view of the continuing growth of human populations, that the marine environment could deteriorate significantly in the next decade unless strong, coordinated national and international action is taken'.*

Ocean Rescue 2000 program

Because of growing concerns in Australia on the state of Australia's marine environment, the Commonwealth Department of Environment, Sport and Territories established the Ocean Rescue 2000 program in 1991 to promote the conservation and sustainable use of the marine and coastal environment. Ocean Rescue 2000 builds on existing marine conservation and management programs and is part of the national strategy for Ecologically Sustainable Development.

The principal objective of the program is to develop and implement the Australian Marine Conservation Plan which is to guide the use and management of Australia's marine resources. Other objectives include ensuring adequate baseline and monitoring information on the marine environment, activities and management, and ensuring its accessibility to decision-makers and managers; fostering an educated, informed and involved community; and developing and implementing a national representative system of marine protected areas.

The program consists of the following elements:

- National Representative System of Marine Protected Areas;
- Australian Marine Conservation Plan;
- State of the Marine Environment Report for Australia (SOMER);
- National Marine Education Program;
- National Marine Information System; and
- Marine and Coastal Community Network.

The State of the Marine Environment Report

The State of the Marine Environment Report (SOMER) is the first comprehensive, scientific description of Australia's marine environment. It was undertaken primarily to provide baseline information for the proposed Australian Marine Conservation Plan. It has also provided information for the Commonwealth government's new national State of the Environment reporting program which will report in 1995.

The Commonwealth Department of the Environment, Sport and Territories commissioned the Great Barrier Reef Marine Park Authority (GBRMPA) to prepare SOMER. The Authority has over 15 years experience in research and management of the Great Barrier Reef, the world's largest multi-use marine protected area, and its expertise is being increasingly sought for marine environmental management, both nationally and internationally.

SOMER describes in detail the major marine ecosystems and their states; the major uses of the marine environment and their effects; the general issues and threats affecting the marine environment; the condition or health of the marine environment; and marine environmental management and conservation. SOMER examines habitats and communities from the shore to the ocean depths.

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*GESAMP: (IMO/FAO/UNESCO/EMO/WHO/IAEA/UN/UNEP) 1990, The state of the marine environment, UN Regional Seas Reports and Studies No. 115, UNEP.

The SOMER Process

The production of SOMER was a great challenge. Australia's marine environment is vast and covers a great range of climates, ecosystems, habitats and human influences. More significantly, it is very incompletely known. Long-term scientific information on the marine environment, essential to accurately assess its condition, is very scattered, or lacking altogether in many areas.

The topics covered in SOMER were initially identified by a workshop of experts from marine science, resource management and industry. The GBRMPA appointed a senior marine scientist to coordinate the project and produce the reports. An expert Advisory Committee assisted and advised the coordinator in the identification of expert authors and reviewed the technical papers and reports produced. These commissioned technical reviews were also subject to a process of open scientific peer review. The 83 technical papers thus produced provided the source material for the main reports, the *State of the Marine Environment Report for Australia: Technical Summary* and the non-technical overview *Our Sea, Our Future.*

Much of the information collected for SOMER is unpublished. Because of the scientific value of this information, a range of papers is being published. This volume contains 16 papers on the physico-chemical environments, the nature and status of major ecosystems and habitats, the status of the better-known marine megafauna, and special considerations in the management of marine organisms.

Leon P. Zann
SOMER Coordinator

Acknowledgments:
SOMER is the result of the efforts of 134 scientists and technical experts, 14 members of the Advisory Committee, and around 160 external reviewers. Production of this volume: D.C. Sutton and Jim Campbell.

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SOMER process

topics identified by workshop

expert scientists commissioned to produce **technical papers**

papers peer reviewed

Selected papers published:
SOMER Technical Annexes 1 & 2

technical papers, condensed & management implications identified

reviewed by authors, SOMER Advisory Committee and relevant govt agencies

combined to produce technical report:
SOMER Technical Summary

major findings and issues identified & collated

reviewed by SOMER Advisory Committee and relevant govt agencies

published as non-technical report:
Our Sea, Our Future. Major findings of SOMER

major findings summarised in brochure:
Our Sea, Our Future. Summary of SOMER

applications of SOMER:

- ***Australian Marine Conservation Plan***
- ***National Marine Education Program***
- ***Marine Environment Conference (Uni Queensland, 1995)***
- ***National State of the Environment Report (1995)***

Contents

The oceanography of Australian seas

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Introduction

The study of physical oceanography encompasses the description and understanding of all physical processes occurring in the ocean. The magnitude of the processes ranges from those extending over many thousands of kilometres and several years for large scale ocean circulation processes, to those covering only centimetres and lasting for as short as seconds for small scale processes. In the space available here, all aspects of physical oceanography cannot be adequately covered. For example, discussions about possible sea level rise (Church et al. 1991) or climate variability studies (Ramage 1966, Meyers et al. 1991) are not included; and a description of the weather affecting the Australian region can be found instead in the 'Manual of meteorology' produced by the federal Bureau of Meteorology. Instead, this report concentrates on introducing the main areas of physical oceanography relevant to Australian waters and directing the reader to literature references for more detailed explanations.

Temperature and salinity in the upper ocean

The upper ocean consists of the top few hundred metres where changes occur on time scales of a few days to many months.

There are many factors controlling the physical properties of the upper ocean. Among the most important are incoming and outgoing radiation, precipitation and evaporation, sea-ice formation and melting, mechanical energy inputs and mixing processes. Some factors tend to make the water column more stable (more strongly stratified) by reducing the density of the upper layers. For example, incoming radiation tends to heat the surface layers through absorption and they are then less dense than the layers beneath. Precipitation, river runoff and sea-ice melting provide fresher water to the surface layers, also rendering them less dense as a result of the reduced salinity. Although vertical mixing is somewhat inhibited by a stably stratified water column, strong winds and wave action tend to mix the surface layers through mechanical action. This action creates a relatively homogeneous 'mixed layer' of less dense fluid in the upper 100 m of the ocean. The confluence of water masses at zones of convergence (ocean fronts) can also generate substantial vertical mixing.

In the tropics, water temperature decreases rapidly with depth. The region of maximum temperature change is called the *thermocline*. The main thermocline is centred at approximately 500 m depth in the tropics and mid-latitude oceans, while a seasonal (summer) thermocline often exists at approximately 100 m depth in mid-latitudes. While temperature and salinity may change with depth, the combined effect of the two usually is such that density increases with depth. The level of maximum salinity change with depth is called the *halocline*, while the level of maximum density change with depth is called the *pycnocline*.

Evaporation, outgoing radiation, cooling by contact with colder air or the formation of sea-ice tend to render surface layers denser than those below by virtue either of a lower temperature or an increased salt content (salt is rejected into the sea below during sea-ice formation). Denser fluid overlying less dense fluid is an unstable situation and it causes the water column to overturn, generating substantial 'convective' mixing. Since the density of the ocean usually increases with depth, the newly formed denser fluid at the surface usually is not dense enough to cause the whole water column to overturn. Instead, a convective mixed layer of some tens of metres depth is formed. In Antarctic waters, convective mixed layers of approximately 500 m depth form in winter.

The distribution of water properties in the world's oceans is outlined by Sverdrup, Johnson and Fleming (1942) and by Pickard and Emery (1982) and the annual and seasonal climatologies are now available in digital and microfiche form (Levitus 1982).

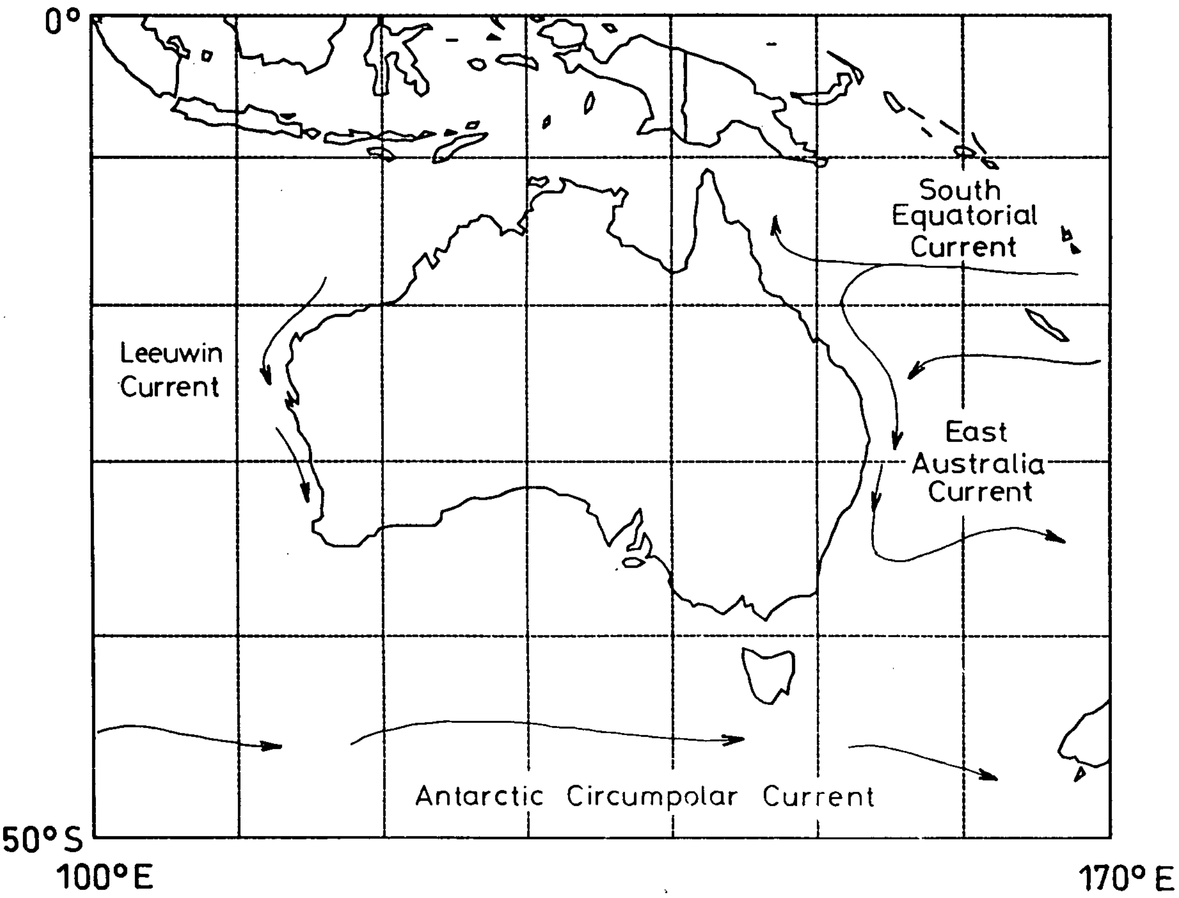

Figure 1: Major oceanic currents affecting the Australian coastline.

Large scale ocean circulation

Wind-driven circulation

The overall result of westerly winds in the mid-latitudes and easterly winds in the tropics is to drive the ocean currents in the major ocean basins in large closed circulation patterns called gyres. Gyres rotate anti-clockwise in the southern hemisphere (Pickard & Emery 1982). In the Southern Ocean, which extends all around the Antarctic continent, the prevailing westerly winds drive the Antarctic Circumpolar Current to the east although there is evidence of strong variability (Morrow et al. 1992).

Because the earth is spherical and rotating, ocean currents tend to intensify toward the western boundary of the ocean basins. These strong but highly variable currents are called western boundary currents, and their variability is often sufficiently strong to mask the general poleward direction of their flow. The East Australia Current (EAC) is an example of a western boundary current (Figure 1). In the western Pacific Ocean, the vast number of shallow areas containing comprised of coral reefs and islands have the effect of weakening the western boundary current effect — although at times the EAC flows very strongly off Australia's eastern coast. This current flows south from the Coral Sea (Church 1987), hugging the continental slope until the region of Forster (central New South Wales) where it tends to run further offshore. Once or twice a year the EAC strains itself into loops in the Tasman Sea off the New South Wales coast. These loops become detached from the current and form EAC eddies, warm disc-shaped water parcels having a diameter of 200–300 km, and extending 1500–2000 m into the ocean (Nilsson & Cresswell 1981, Bennett 1983). These eddies may be seen from satellites fitted with infrared detectors (Cresswell & Legeckis 1986).

Off the west coast of Australia the deep ocean currents are highly variable and do not necessarily flow to the north as might be expected from a traditional gyre model (Cresswell & Golding 1980, Bye 1983). The reasons for this are not clear but are generally thought to be associated with flow through the Indonesian Archipelago from the Pacific Ocean to the Indian Ocean (Godfrey & Weaver 1991), which tends to drive a fairly persistent westward flowing current off north-western Australia (Holloway & Nye 1985). Instead, the Leeuwin Current flows southward in the deeper waters off the Western Australian coast, and this is occasionally strengthened by winds. The Leeuwin Current is, however, primarily driven by *thermosteric* effects associated with the extremely warm waters which come from the region of the North West Shelf (Weaver & Middleton 1989). It is a seasonal current, reaching its maximum strength about mid-year (Cresswell 1991, Smith et al. 1991). It is also highly variable both in its overall structure and in the detail of its turbulent nature (Batteen & Rutherford 1990).

Winds blowing parallel to a coastline can produce upwelling or downwelling effects through the surface *Ekman layer transport*, which transports surface waters perpendicular to the wind direction. In particular, coastal upwelling is induced whenever a wind blowing parallel to the coast has the deeper ocean to the left of the direction of the wind vector. The earth's rotation causes a surface Ekman layer to transport surface waters offshore, and these must be replaced by the upwelling at the coast of colder, nutrient-rich waters from below. Wind-driven upwelling does not appear to be persistent along any particular section of the Australian coast, but happens from time to time at many locations (eg Rochford 1975, Schahinger 1987, Griffin & Middleton 1992).

Thermohaline circulation and water masses

In this section the principal characteristics of water mass types and the thermohaline circulation in the oceans around Australia are outlined. In the tropical Indian Ocean and western Pacific Ocean, surface temperatures often exceed 28°C with reduced solar radiation causing a gradual decrease in temperature at higher latitudes. More rapid reductions in temperature occur at the Subtropical Convergence (40–45°S) and again at the Antarctic Convergence (55–60°S). Tropical sea surface temperatures in the deep ocean change little with season while a typical seasonal variation in most non-tropical oceanic Australian waters is approximately 5°C. On the continental shelf of north-western Australia the summer heating provides a substantial warm pool of surface waters, and this provides the source region for the Leeuwin Current. Surface temperature maps for February and August for the world's oceans are shown by Sverdrup, Johnson and Fleming (1942: Charts II and III) and by Pickard and Emery (1982: Figures 4.3 and 4.4), and here for Australian waters in Figures 2a (February) and 2b (August).

Surface salinities change little with the seasons being between 34 psu and 35 psu (Practical Salinity Units) in the tropics and at latitudes greater than 40°S, and between 35 psu and 36 psu at mid-latitudes. Influences of Antarctic sea-ice melting on surface waters are not observed much further north than the Antarctic Convergence.

Below the surface waters to about 800 m depth lies South Pacific Central Water (to the east of Australia) and Indian Central Water (to the west of Australia) extending in latitude from the equatorial waters to the Subantarctic Convergence.

Low salinity, cold Antarctic surface waters meet subantarctic waters at the Antarctic Convergence. The resulting dense mixture sinks and flows slowly northward below the Central Water at an average depth of approximately 1000 m. This colder, lower salinity Antarctic Intermediate Water is seen in both the Indian and Pacific oceans as far north as the equator (England 1992). Below the Antarctic Intermediate Water, and flowing southward at depths between 1000 m and 3000 m in the Pacific, lies Pacific Deep Water. Pacific Deep Water is originally of Atlantic/Antarctic origin, and deep waters of the Indian Ocean have similar properties to those of the Pacific. The formation of Antarctic bottom water in the Weddell Sea and at other continental margins of Antarctica (Middleton & Humphries 1989) results in a dense, cold water mass which sinks to abyssal depths off Antarctica and flows slowly northward contributing to the properties of the deep water masses in the Atlantic, Pacific and Indian oceans.

Pickard and Emery (1982) presented temperature–salinity diagrams for the South Pacific and Indian oceans (Figure 7.13) and north–south vertical sections of water properties in the central Pacific (Figure 7.34). The general description of water masses and currents of the world's oceans given by Sverdrup, Johnson and Fleming (1942) is both lucid and generally valid.

Continental shelf circulation

Wind-driven circulation

Weather patterns change on both seasonal and synoptic (one to three weeks) time scales. In shallow continental shelf waters, where the water column has much less mass than in the deep ocean, wind stress can generate substantial currents in a day or so. These winds might change direction on seasonal and/or synoptic time scales, each time accelerating continental

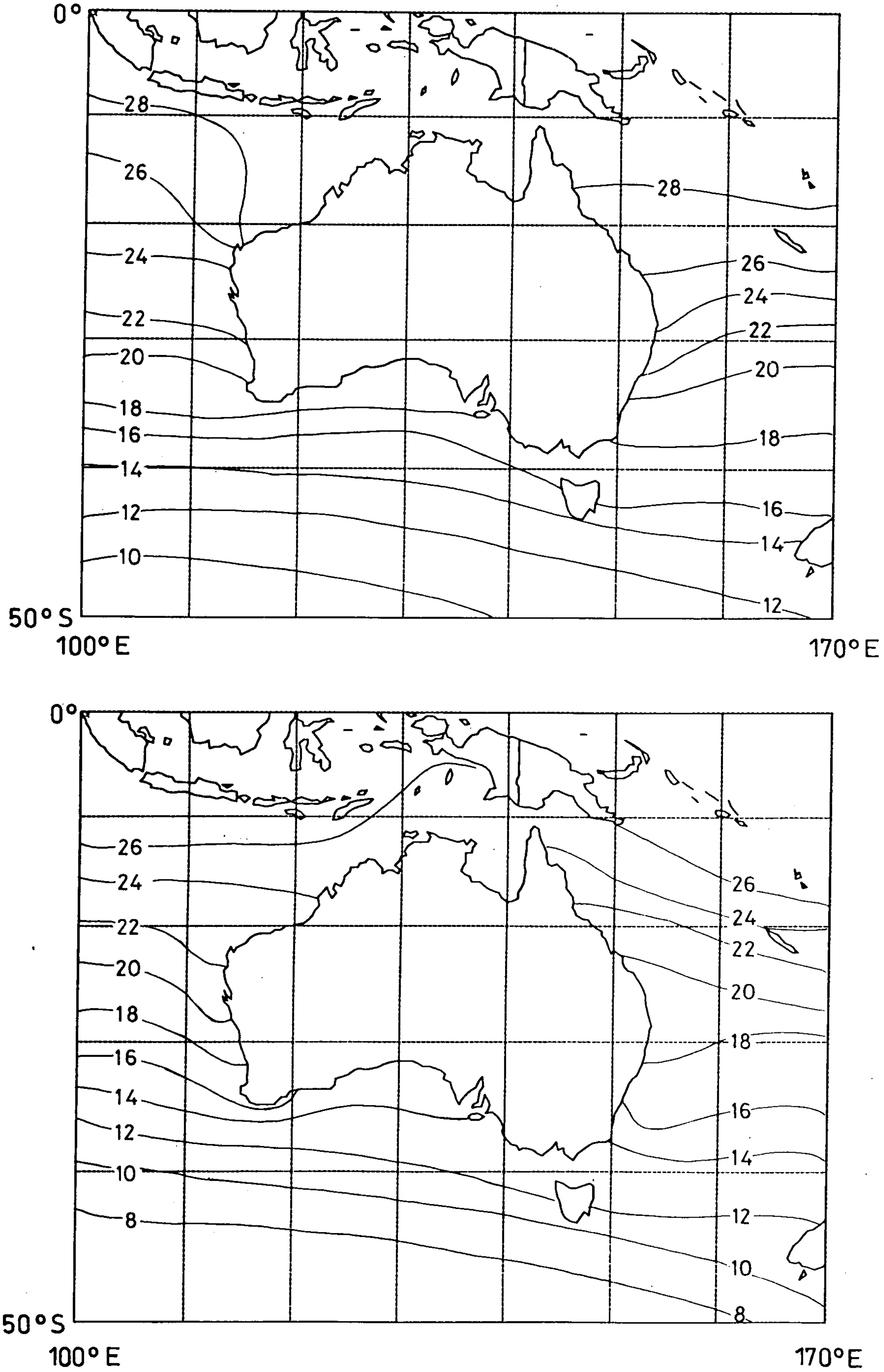

Figure 2: Sea surface temperature maps of Australian waters in February (top) and August (bottom).

shelf waters to flow along the coast. The current pulses forced by the synoptic-scale wind reverse direction in response to the along-shore component of the wind stress, but the pulses themselves propagate in a wave-like manner along the continental shelf with the coast to the left of the direction of propagation. Such propagating current pulses are called *continental shelf waves* or *coastal trapped waves.*

Continental shelf waves propagate at speeds of about 4–10 m a second, with faster speeds on wider continental shelves. The associated currents may be 0.1–0.2 m per sec, and over a two-week time period can transport particles several hundreds of km along-shore. Wind-driven continental shelf waves have now been identified as a primary source of current variability in practically all Australian continental shelf waters such as the Great Barrier Reef (Andrews 1983, Middleton & Cunningham 1984, Burrage, Church & Steinberg 1991, Cahill & Middleton 1993), the New South Wales shelf (Freeland et al. 1986), the Great Australian Bight (Provis & Lennon 1981), Bass Strait (Middleton 1991, Middleton & Viera 1991) and the North West Shelf (Webster 1985). It is likely that much of the variability observed by Godfrey, Vaudrey and Hahn (1986) off the south coast of Australia is also due to wind-driven continental shelf waves. On most Australian shelves there are mixed contributions of locally wind-forced shelf waves and freely propagating shelf waves (generated by wind-forcing farther upstream in the sense of continental shelf wave propagation), enhancing the difficulty of current prediction. A schematic diagram of surface and sub-surface circulation patterns for continental shelf waves in the southern Great Barrier Reef reproduced from Griffin and Middleton (1986) is shown in Figure 3. While this diagram is strictly valid only for the area for which it was drawn, the general features are typical and the complexity of the flow structure is well illustrated.

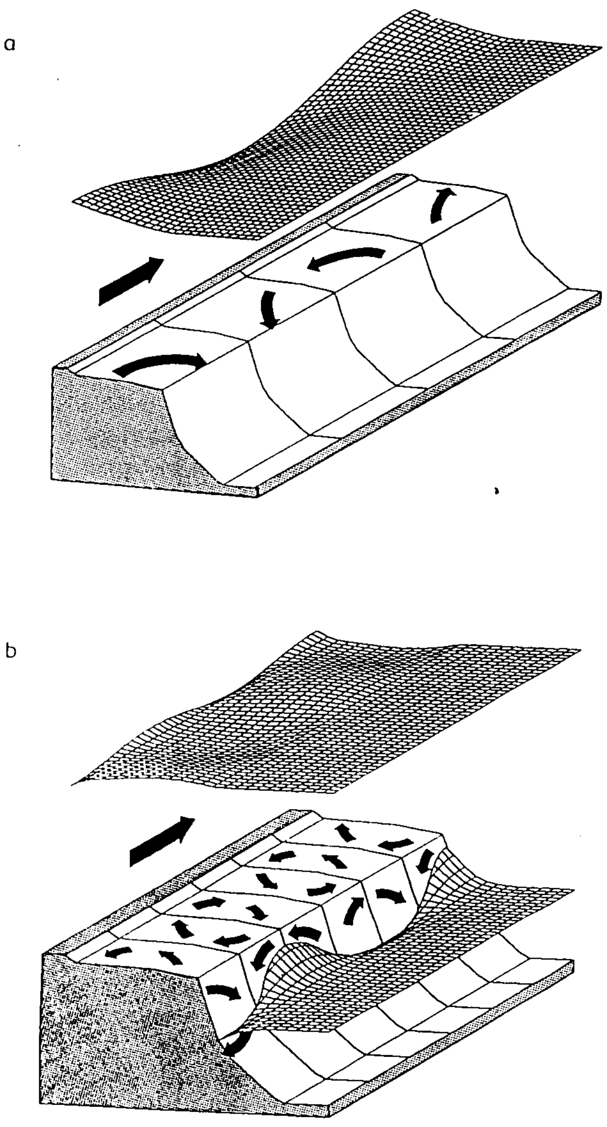

Figure 3: Schematic diagram of surface (a) and subsurface (b) circulation patterns for continental shelf waves in the southern Great Barrier Reef.

An upwelling of nutrient rich waters from the deeper waters of the continental slope is associated with that phase of the continental shelf wave which has current pulses flowing with the coast to the right of the current direction. At times this upwelling is strong enough to bring extra nutrients into the euphotic zone. Continental shelf waves may have a modal structure whereby waters both deeper and further offshore oscillate in opposite phase to the nearshore surface waters.

In some parts of the world, seasonal winds are sufficiently strong to produce consistent alongshore coastal currents and persistent upwelling. A few persistent upwelling zones appear to exist off the Australian coast (Andrews & Gentien 1982, Schahinger 1987) although more areas may be identified once a history of sea surface temperature pictures taken from satellites is acquired.

Between November and May cyclones are a common feature of Australian tropical waters. They can generate strong currents and high mean sea levels (Fandry & Steedman 1989, Hearn & Holloway 1990) as well as high seas.

In general, wind-driven continental shelf circulation patterns have along-shore currents which are approximately 10 times faster than across-shelf currents. This feature is essentially a result of the presence of the coast.

Deep ocean effects on circulation

In many cases the offshore deep ocean currents flow in the opposite direction to the local-wind-driven continental shelf currents, and in these cases there is often a (changing) zone of demarcation. Inshore of the zone the currents are primarily wind-driven while offshore from the zone the currents are primarily driven by deep ocean effects. In such circumstances, prediction and interpretation of current speed and direction can be extremely difficult.

The situation described above is often true off the New South Wales coast, and analyses of current data from the Australian Coastal Experiment (Freeland et al. 1986) was hampered by the existence of the strong southward-flowing East Australian Current (EAC). The EAC is a strong, highly versatile current which is fed by the South Equatorial Current in the Coral Sea (Burrage 1993) and which hugs the continental shelf as it flows poleward. It tends to separate from the coast near Cape Hawke (New South Wales) and often forms large eddies and meanders. The Leeuwin Current, flowing southward along the outer continental shelf of Western Australia, appears to be primarily driven by deep ocean effects (primarily the flow from Pacific to Indian oceans via the Indonesian Archipelago) but strongly affects shelf circulation along the entire west coast (Godfrey & Ridgway 1985, Weaver & Middleton 1989).

On occasions the different temperature characteristics or turbidity of shelf and ocean waters enable easy identification by airborne or satellite infrared scanners (Cresswell et al. 1983).

Tidal circulation

The apparent motion of the sun and the moon around the earth at regular intervals is a result of the forces of gravitational attraction. The balance between gravitational and centripetal forces for each celestial body is such as to result in a tidal response not only on the side of the earth nearest the celestial body but also on the farthest side. The response consists of both sea level variations of several metres (tides) and associated ocean currents (tidal streams). The apparent orbits of the sun and the moon around the earth are, however, neither circular, nor co-planar, nor aligned with the earth's equatorial plane. These irregular orbits cause a change in the tidal forcing and the ocean response with time. To allow for these variations it is usual to consider the tidal forcing as arising from many fictitious celestial bodies, each of which has a perfectly circular orbit, and each of which produces a perfectly regular response. The combined effect of all constituents gives the whole tidal response. Amplitudes and phases of the contributions to the tidal height field are given in the *Australian national tide tables*, while a practical guide to overall effects of tidal currents often appears on relevant navigational charts.

Because of the presence of the continents, the tides cannot follow the celestial bodies around the world as the earth rotates. Instead, tides propagate in the deep ocean around amphidromic points (places where the ocean surface does not vary at all). Tides tend to propagate clockwise around amphidromic points in the southern hemisphere, and anticlockwise in the northern hemisphere.

For narrow continental shelves such as that on the east coast of New South Wales the tides have ranges much the same as those occurring in the deep ocean (approximately 2 m). For very wide continental shelves such as the North West Shelf off Western Australia or the region offshore from Mackay in Queensland, the tidal oscillation is nearly resonant and the spring tidal range is approximately 10 m (Holloway 1983, 1984, Middleton, Buchwald & Huthnance 1984). Although sea level variations due to tides can dominate on wide shelves, the sea level fluctuations associated with wind-driven currents may contribute substantially to the sea level variability on narrow shelfs, or during neap tides. In the complex coral reef systems offshore from Mackay, tidal currents in channels can reach 4 m per sec (8 knots).

To determine the tides for a particular region, it is usual to make measurements of tides at hourly intervals for periods of several months and at several locations. Amplitudes and phases of constituents, for both tidal currents and heights, are computed from the data and, since the tidal response for each constituent is quite regular, predictions for future times can then be made with a high degree of reliability. The *Australian national tide tables*, produced annually, contains predictions for a representative selection of coastal locations.

Some peculiar features of tides on the Australian coast are their large ranges (over 10 m) on the North West Shelf and in Broad Sound, the 'Dodge' tides in Spencers Gulf where the tides are absent for several days, and their ability in the Great Barrier Reef to draw nutrients into the shallower waters (Thompson & Golding 1981).

Computer models are sometimes used to produce a synoptic view of tides in a region such as the

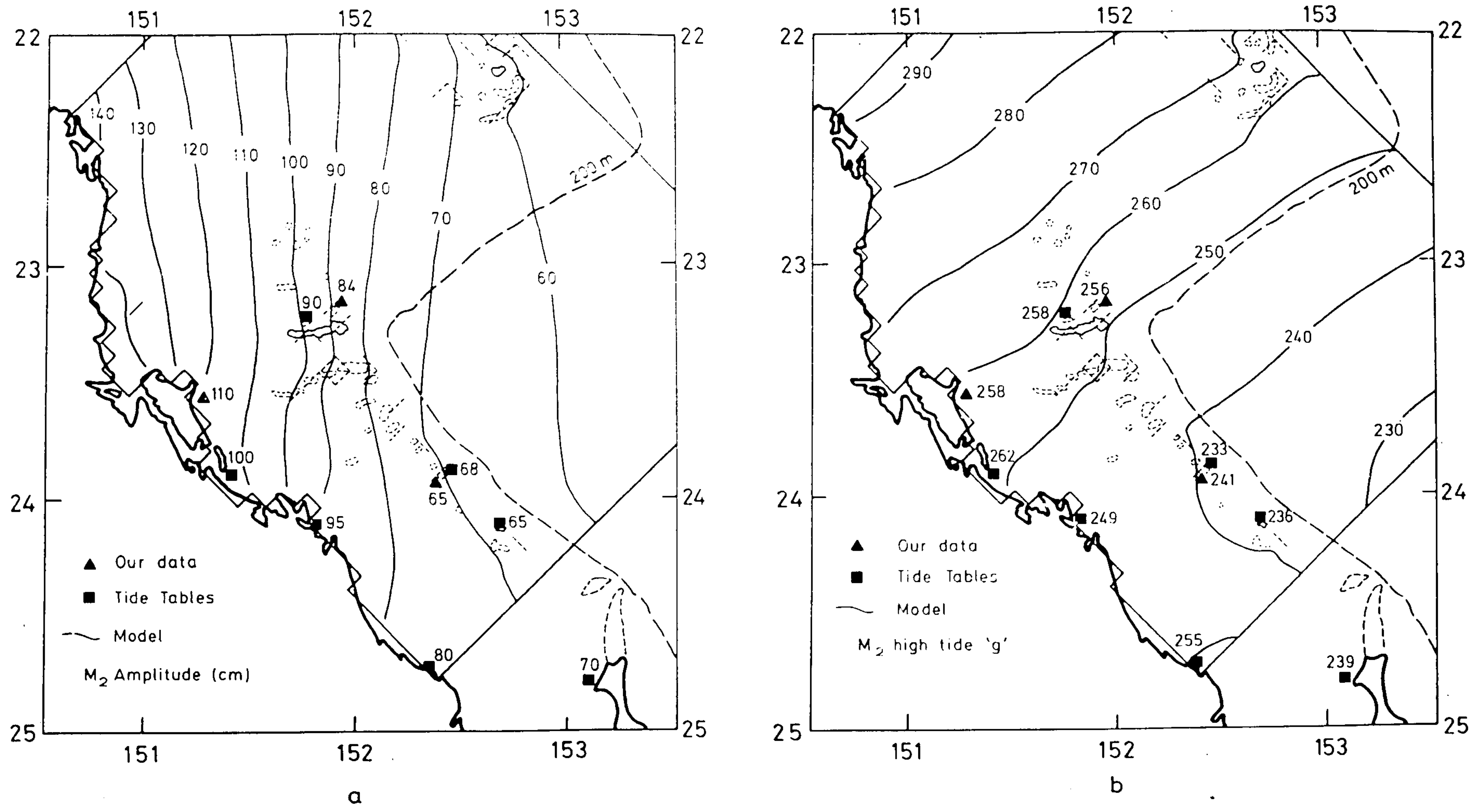

Figure 4: Comparison of a numerical model's predicted M_2 tidal height (a) and Greenwich phase lag (b) fields with the author's (Griffin, Middleton & Bode, 1987) own and previous observations

Gulf of Carpentaria or Bass Strait. A good example of the presentation of tidal data, and of comparison with a numerical (computer produced) model of the tidal flow in a continental shelf region is given by Griffin, Middleton & Bode (1987, Figure 4), reproduced here as Figure 4.

Flow in embayments and semi-enclosed shelf seas

On outer continental shelf regions current flows tend to be directed primarily alongshore and are usually consistent, both in speed and direction, for distances of several hundred kilometres alongshore. However, coastlines are not straight: they are convoluted to varying degrees with promontories and embayments of various scales. These topographic features modify the regular outer-shelf flow, occasionally resulting in closed or semi-closed nearshore recirculation patterns. These 'eddies' may be several kilometres in diameter, and may have very strong currents flowing in various directions, depending on the exact location at which current measurements are made (Deleersnijder, Norro & Wolanski 1992, Middleton, Griffin & Moore 1993). They can serve to confuse interpretations of prevailing current directions when their existence is not appreciated. For example, measurements at a location a few kilometres directly offshore from a headland might show persistent offshore flow due to the redirection of a general alongshore flow by the headland, while a more general shoreward flow might occur some distance upstream or downstream of the headland. Such circulation features might be important to the local biology or to sediment transport. Robinson (1982) provides a useful review of the types of flow which might be expected. An example of a study of wind-driven flow in a semi-enclosed embayment is that undertaken by Steedman and Craig (1983) in Cockburn Sound, Western Australia.

With topographic scales of several hundred kilometres the flows are strongly controlled by the coastline and somewhat insulated from the open ocean. For example, wind-driven flows in the Gulf of Carpentaria (Forbes & Church 1983) and Bass Strait (Fandry 1983) are both extremely complex, and predictive capability for these regions has only been achieved through the use of numerical models.

The degree of isolation from open ocean waters afforded in some embayments and gulfs may allow water masses of distinctive properties to form. Evaporation during the dry South Australian summer renders the upper reaches of Spencer Gulf and Gulf St. Vincent extremely saline with salinities exceeding 48 psu (Practical Salinity Units) in Spencer Gulf and 42 psu in Gulf St. Vincent (Nunes & Lennon 1986). Although the temperatures are warmer than oceanic temperatures, the effect of the substantially increased salinity is to produce a water mass of density somewhat greater than that of the continental shelf waters outside the gulf. The dense, high salinity water flows seaward along the bottom of the gulfs as a gravity current, and cascades off the continental shelf into deeper oceanic waters (Nunes & Lennon 1987). During winter, surface cooling of Bass Strait waters causes them to be well mixed (Baines & Fandry 1983), but as a result of the limited depth of Bass Strait (approximately 100 m), waters cool substantially more than those in the adjacent deep ocean. The cool, dense Bass Strait waters then cascade down the continental slope into the Tasman Sea (Godfrey et al.1980, Villanoy & Tomczak 1991).

Conclusions

In writing this summary, many simplifications have been made in order to present the essence of the physical processes of importance to Australian seas. The simplifications have been made not only in the properties of each individual process, but also in the range of processes considered. In seeking additional knowledge of physical processes the reader is offered a selection of key references on each topic. These have been chosen because they are recent and relevant.

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The technical paper by Dr. J. Middleton was reviewed by Dr. G. Cresswell, CSIRO Division of Oceanography, Hobart and Dr. A. Pearce, CSIRO Laboratories, North Beach, Western Australia

Marine geology and sedimentology of the Australian continental shelf

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Introduction

Australia's continental shelf covers approximately two million km^2, an area equal to about 25% of the continent's land surface. Most of Australia's petroleum is derived from sedimentary deposits located on the continental shelf and the sea over the shelf contains some valuable fisheries. A growing tourist industry relies on a healthy marine environment: yet this same environment receives millions of tonnes (t) of eroded topsoil, fertiliser, sewage and industrial waste each year.

What do we know about Australia's continental shelf, about its origin, history and geology? What are the important factors controlling the functioning of its natural sedimentary systems? This chapter attempts to answer these questions as well as to explore the implications of shelf sedimentology for environmental monitoring and management.

Physical setting

The continental shelf is the sea floor which surrounds Australia (Figure 1). It is shallow, generally less than 200 m in water depth. Geologically, the continental shelf is similar to the rest of the continent in that its foundation is comprised of granitic crustal material — unlike the deep ocean bed which is underlain by basalt (Figure 2). Strictly speaking, the shelf extends from the beach (*foreshore*) environment, across the seaward dipping, low gradient (about 0.1°) *shoreface* to an offshore location where it rapidly changes in slope and is known as the *shelf break*. From there, the seabed forms a steeper *continental slope* which grades into the *continental rise* and *abyssal plain* at great depths (Figure 2). The continental shelf is thus bounded inclusively by the shoreface and the shelf break (Figure 2). The shoreface is generally a zone of active sediment reworking, delimited at its offshore margin by the so-called *fairweather wave base*, which is the downward limit of effective wave-induced sand movement during normal sea conditions.

In a global context, the depth of the shelf break (20–550 m water depth and defined as 200 m by international convention) and the width (2–1500 km) exhibit a wide variability. In the case of Australia (Figure 1), the shelf break is located about 10 km offshore from Fraser Island (east coast) and North West Cape at Exmouth (west coast) but at over 500 km distance offshore on the Arafura Shelf in the north. In vertical profile, most of Australia's shelf is a smooth, flat surface which dips gently seawards (Figure 2). However, some parts are rimmed by shelf edge barrier reef systems (eg the Great Barrier Reef shelf).

Geological history

The shape of the Australian continent owes its origin to the rifting apart of a much larger super-continent known as Gondwanaland. About 65 million years ago Australia was still a part of Gondwanaland, connected along its southern margin to what is now Antarctica (Veevers 1984). Rifting and seafloor spreading caused the splitting away of Australia and its last connection with Antarctica — at the southern edge of Tasmania — was severed about 50 million years ago. Australia moved northward, pushed along by the sea floor-spreading 'conveyor belt'. About 10–15 million years ago, the northern margin of Australia collided with the Pacific Plate causing tectonic uplift, volcanic activity and eventually the creation of New Guinea.

Throughout Australia's geologic history — both prior to and after its separation from Gondwanaland — rivers and ice have eroded the mountains and transported sediment to the continental shelf, where sedimentary layers several km thick have accumulated (Figure 2). Australia's major oil and gas fields are located

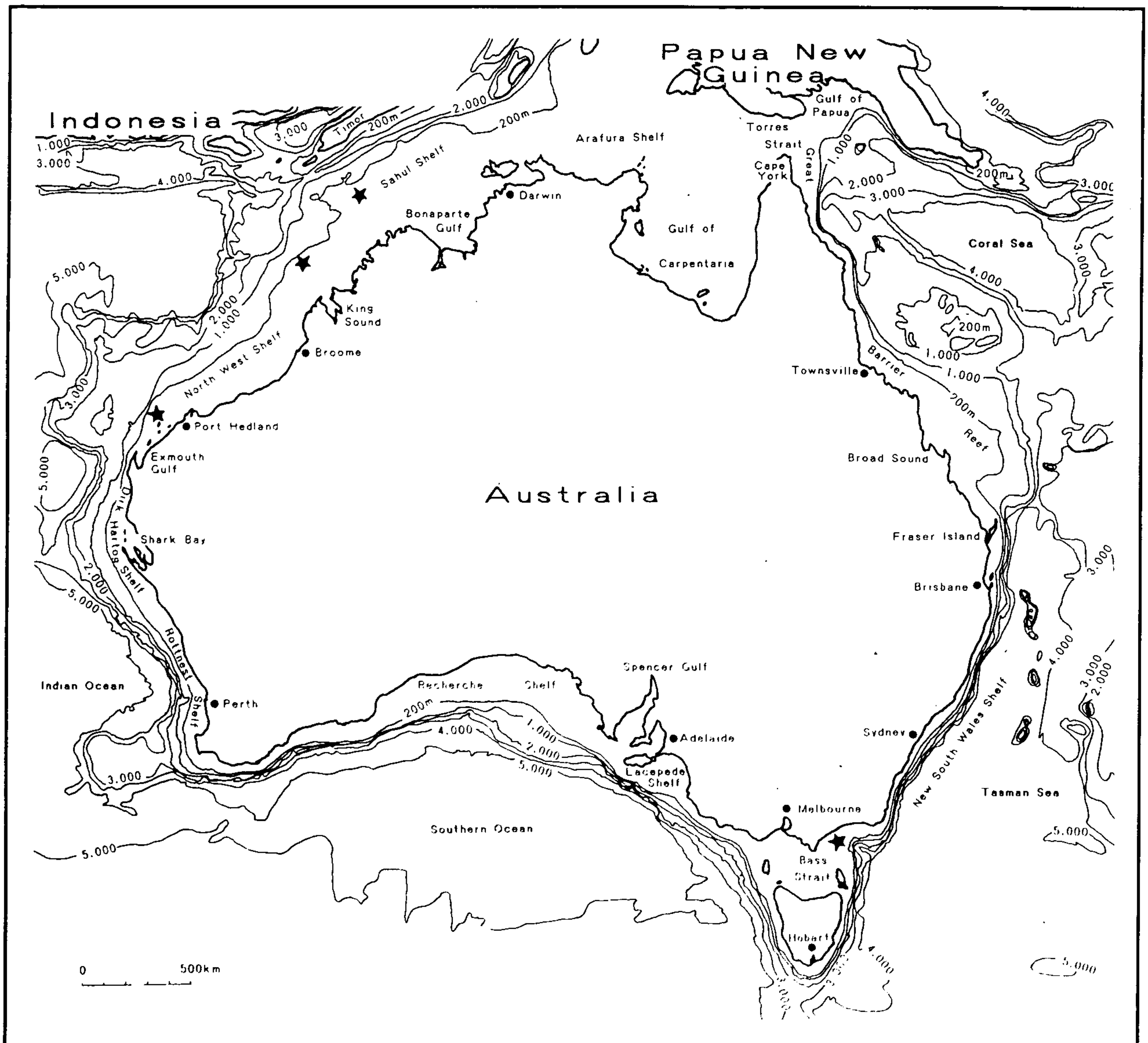

Figure 1: Location and regional bathymetry of Australian shelf seas. The locations of major oil and gas fields are indicated (stars).

within these marginal, sediment-filled basins (Figure 1).

Hence, the continental shelf as we see it today has evolved over many millions of years. Its surface morphology has been modified by processes related to sea level change, the rate and type of sediment supply and the energy available to erode, rework and disperse sediment. In the following sections the relative importance of each of these factors is assessed with reference to the Australian continental shelf.

Sea level change

Sea level rise (*transgression*) and fall (*regression*) relative to a given coastal province is the function of three different (though often contemporaneous) processes:
(1) the melting and/or formation of polar ice caps causing a change in the ocean's volume with consequent *eustatic* sea level changes;
(2) the loading of the earth's lithosphere with sediment, water or ice resulting in deformation causing *isostatic* changes; and
(3) the collision or rifting apart of continental plates or the subduction of oceanic crust beneath continental margins causing vertical *tectonic* movements.

The present position of eustatic sea level is high in relation to the record over the past 150 000 years (Figure 3). During previous lower sea level stands, coastlines would have occupied positions on what is now the outer continental shelf or upper slope. Therefore, the morphology of the present outer shelf and upper slope is partially the product of past, low sea level coastal sedimentary processes and partially the product of modern sea level shelf processes.

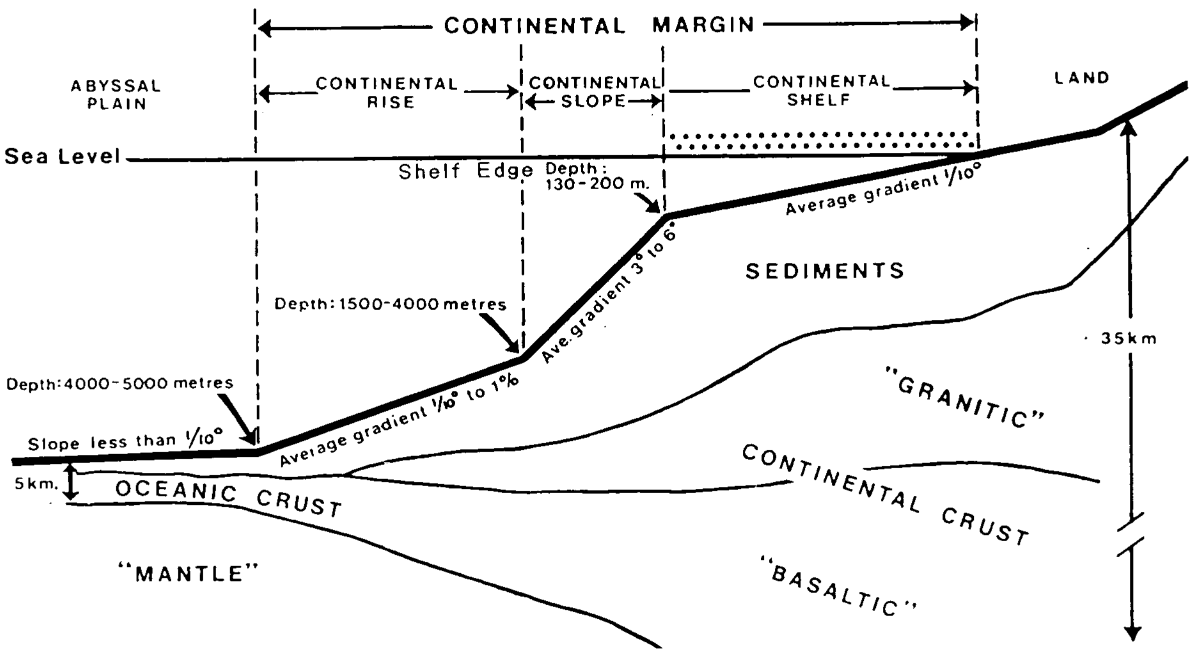

Figure 2: Diagram showing a typical shelf cross-section, illustrating the average gradients and positions of the continental shelf, shelf edge (depth 130–200 m), continental slope, continental rise and abyssal plain. The general geological composition of the shelf is shown. It has a granitic basement with overlying sedimentary basins and underlying oceanic crust.

It has been shown in Australia that the lowering of eustatic sea level to at least −130 m occurred during the last glacial episode, 15 000–20 000 years ago (Chappell et al. 1983). Over the last 150 000 years, eustatic sea level has oscillated many times around 40–80 m below present sea level (Figure 3). Available sea level data indicates that the most recent sea level rise began about 15 000 years ago, reaching a maximum rate of about 2 cm per year between about 11 000 and 12 000 years ago. This high rate was followed by lesser rates of transgression until the position of the present sea level was reached about 6500 years ago (Figure 3).

Isostatic changes in relative sea level have caused only minor (about 4 m maximum) fluctuations around Australia in the Holocene period (Nakada & Lambeck 1989). Similarly, changes in sea level over the past 20 000 years or so due to tectonic effects are thought to be limited on the Australian continent which is considered to be 'stable' relative to other continents. Over longer time intervals (more than 20 000 years) there is evidence of tectonism in parts of Australia. Although the collision boundary in New Guinea forms a northern, tectonically active zone, the effect of this zone does not appear to extend southwards into the Cape York region of Australia in terms of late Pleistocene to Holocene sea level change.

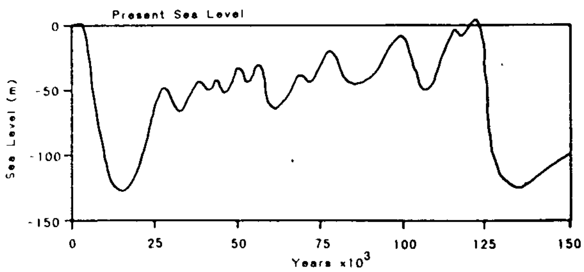

Figure 3: Sea level curve showing the effect of *eustatic* (ie related to the melting of polar ice caps) sea level change over the last 140 000 years.

Land bridges and shelf lakes

During Pleistocene periods of lower relative sea levels (Figure 3), the shelf was exposed subaerially and the Australian mainland was 'joined' by dry land to several of the adjacent large islands such as Tasmania and New Guinea (see Figure 4). Such 'land bridges' are considered to have facilitated the migration of animals and humans in the late Pleistocene 'ice age' (eg Peterson 1991). The shallow seas comprising the Sahul Shelf, Gulf of Carpentaria, Torres Strait and Bass Strait were thus subjected to erosion and sedimentation by rivers and wind on several occasions during the past 150 000 years (Figures 3 and 4). The basins in Bass Strait, Bonaparte Gulf

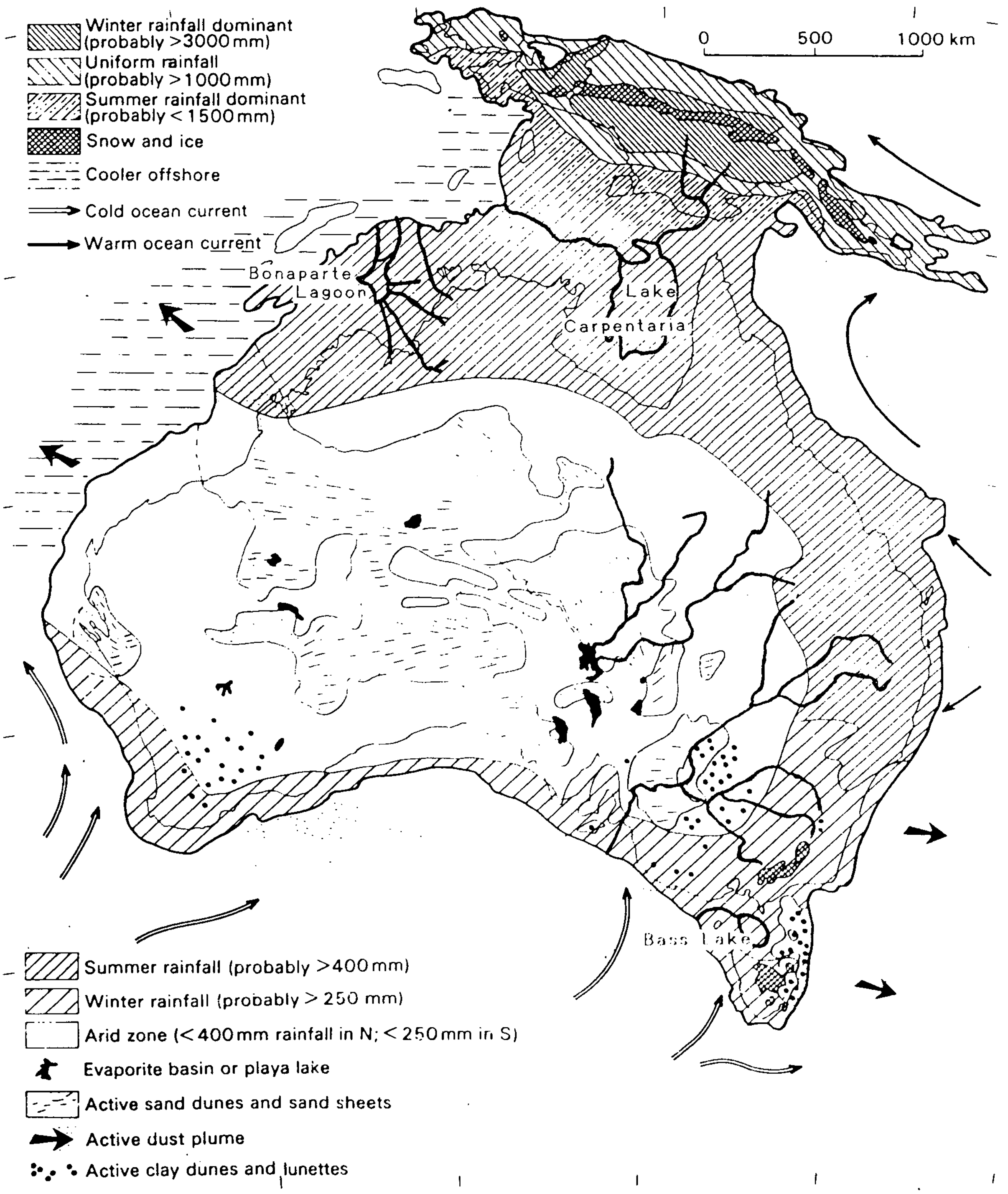

Figure 4: Morphoclimatic map of Australia and Papua New Guinea at about 18 000 years ago (after Williams 1984). The approximate positions of lakes and lagoons in Joseph Bonaparte Gulf, the Gulf of Carpentaria and Bass Strait are shown.

and the Gulf of Carpentaria are considered to have been the sites of large fresh to brackish water lakes and lagoons during these periods of emergence (Van Andel & Veevers 1967; Blom & Alsop 1988; Jones & Torgersen 1988). At the peak of the last ice age, when sea level was about 130 m lower than at present (Figure 3), rainfall on the continent was also less than now and dust storms swept terrestrial clay into the adjacent oceans (Figure 4). Cores obtained from the deep ocean basins surrounding Australia contain discrete layers of these terrestrial clays (Thiede 1979).

Shelf sediment supply

In describing shelf sediments, the most widely used terminology is that proposed by Emery (1968) whose classification has a genetic basis. It distinguishes *authigenic* (chemically precipitated), *organic* (biologically produced shells, planktonic tests, etc.), *residual* (weathered from underlying rock), *relict* (sediments deposited in a different low sea level environment, such as a beach ridge, river delta or subaerial dune) and *detrital* (material presently supplied by rivers, glaciers or wind). Since most relict sediments were reworked during the post glacial transgression and/or are presently being reworked by modern shelf processes, the term *palimpsest* was later proposed to differentiate sediments which exhibit 'petrographic attributes of an earlier depositional environment and, in addition, petrographic attributes of a later environment' (Swift et al. 1971: 343). On the Australian shelf, residual and authigenic sediment types are generally minor, in terms of total volume, than are the other types named above. Detrital and organic (*biogenic*) sediment types are generally dominant although relict and palimpsest sediments may be locally important.

Supply of detrital sediment to the Australian shelf

How much sediment is supplied to the Australian shelf each year by rivers and wind? The answer to this question is unknown: indeed, estimates of the supply of detrital sediments to the Australian shelf are rare in the scientific literature. The central portion of Australia, covering an area of 3.62 million km^2 (or 47% of the continent's land surface) is either desert or contains drainage which flows into inland basins and does not reach the sea. About 40% of the Australian continent is mantled by wind-blown sand and dust storms most probably supply sediments to the shelf. The amount of wind-blown sediment actually reaching the shelf is unknown.

The best known studies of river sediment supplies are those of Belperio and Searle (1988) for the rivers of north-east Queensland draining into the Great Barrier Reef province, and by BMR

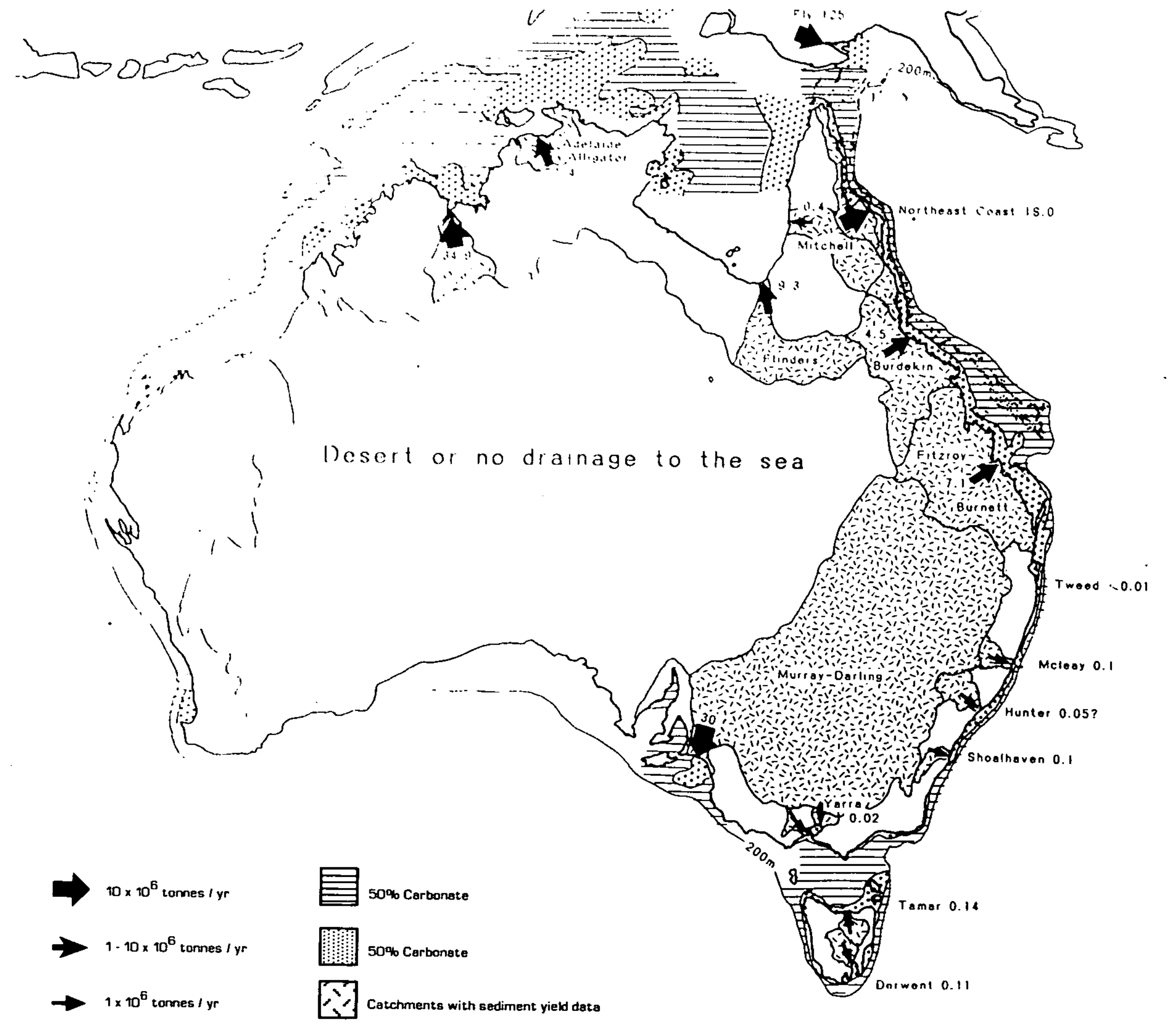

Figure 5: Location of shelf areas where surficial sediments contain greater than or less than 50% calcium carbonate. River catchment areas and estimated total sediment loads supplied to the coastal

(1989) and Jansen et al. (1979) for the Murray River. For the rest of Australia, only isolated studies have been carried out in small catchment areas often in relation to dam construction (eg the Ord and Snowy rivers), or in assessments of the effect of forestry and agriculture practices on river sediment loads (summarised in Figure 5). In Australia, river sediment loads are particularly affected by extreme rainfall episodes. The sediment discharged into the sea during one extreme flood event may exceed greatly the amount supplied over several years of 'normal' river conditions. This situation implies that, in order for accurate assessments to be made, river sediment loads must be monitored over long time intervals spanning perhaps tens of years before accurate sediment loads may be determined (Rieger & Olive 1988).

The sediment loads supplied to the coastal zone by many rivers are trapped in estuaries; hence, little or no sediment may reach the deeper waters of the continental shelf. In general, coarse sand-sized grains do not escape from any of the estuaries along the southern coast of Australia, with a few minor exceptions (eg the Shoalhaven River in New South Wales: Roy & Thom 1981). For instance, all of the sand supplied by the Murray River is trapped within an estuarine lake (Lake Alexandrina) and only a portion of the fine-grained silts and clays are exported to the continental shelf. River catchments in the seasonally wet northern parts of Australia generally have a higher sediment yield (to the order of 100–300 t/km^2/yr) than those in the south (to the order of 10–30 t/km^2/yr). Nevertheless, data are unavailable for about 50% of catchments (Figure 5).

The catchments with data on sediment yields (Figure 5) indicate that the total river discharge of sediment to the coastal zone is 106 million t/yr. The amount of sediment discharged by the Ord River (Western Australia) (34.9 million t/yr) is considered to have been affected by overgrazing (Wasson 1987): a pre-European settlement discharge being probably about 10% of this (ie 3.5 million t/yr). Extrapolating the available sediment yield data to the surrounding catchments, an estimate of the pre-European settlement discharge from Australia is in the order of 150 million t/yr (average sediment yield of 38 t/km^2/yr). This sediment load is a very small amount by world standards. The sediment load of the continent of Africa is 530 million t/yr in comparison, and that of South America is 1800 million t/yr (Milliman & Meade 1983). On the Australian continental shelf, therefore, fluvial (river-based) supply has in the past provided only relatively small point sources of sediment.

Shelf sediment distribution

The distribution of sediment types has been mapped for about 70% of Australia's continental shelf and no investigations have been made of the remaining 30% (Figure 5). Details of different sediment distribution patterns are not provided here, but in general, carbonate content in sediments exceeds 50% on the outer shelf, whereas inner shelf sediments commonly contain more than 50% non-calcareous sediments (both detrital and palimpsest) (Figure 5). In southern Australia this non-calcareous inner shelf sediment is generally coarse-grained, relict or palimpsest quartz sand. For example, the large patches of sediment having more than 50% carbonate on the inner parts of the New South Wales, Lacepede and Rottnest shelves are composed mostly of relict or palimpsest quartzose sands. In northern Australia, the inner shelf, non-calcareous sediments are commonly *terrigenous* silts and clays supplied by rivers. An example of this is in the Great Barrier Reef province (Belperio & Searle 1988).

Outer shelf carbonate-rich sediments are also commonly relict in origin. On the New South Wales Shelf and on the North West Shelf, for example, the carbonate-rich sediment is a mixture of relict and recent biological grains. In contrast, Holocene reefal carbonate buildups in the Great Barrier Reef commonly exceed 10 m and locally are as much as 30 m in thickness. However, the spatial significance of reefs is not great since only about 5% of the Great Barrier Reef shelf is actually covered with reefs (95% of the shelf is characterised by inter-reefal carbonate and terrigenous sediments). The biological organisms producing carbonate-rich sediments vary with latitude, reflecting the ocean's temperature. Hence, whereas hermatypic corals and the calcareous green alga *Halimeda* characterise tropical shelf sediments north of 24°S, bryozoa dominate the cooler water sediments along the southern margins of Australia (particularly south of latitude 38°S: Marshall & Davies 1978). Because of the diverse origins of shelf sediment grains it is difficult to quantify with any certainty the rate at which biological sediment is accumulating on the continental shelf. Sediment cores and seismic profiling studies have shown that sediments deposited during the last 6500 years or so (related to the present Holocene sea level; see above) commonly have a thickness of about 1 m. Given the combination of detrital, relict and palimpsest content averaging 50% with the remaining 50% having been biologically produced over the past 6500 years, and a shelf area of 2 million km^2, a crude estimate of biogenic supply is approximately 150 million t/yr. In other words, detrital river input and biogenic carbonate input of sediment are roughly equal in terms of gross supply to the continental shelf.

Shelf energy regimes

Once they have been introduced into the continental shelf environment, sediments respond to and attain equilibrium with different *energy régimes* related to the nature of currents characterising a given area. Shelf currents may occur at a number of different spatial and temporal scales: the scale of turbulence (0.2–5 sec); the scale of wave orbital currents (5–20 sec); the scale of tidal currents (6 h); and the scale of storm events (6–10 days). Superimposed upon these 'events' will be other currents, such as wind-driven ocean currents, which may flow at a steady rate for months without changing significantly in speed or direction. Accordingly, shelves may be classed relatively as 'high' energy or 'low' energy based on the degree of physical reworking of sediments by currents. In addition to such current reworking, sediments are also mixed and reworked by the activities of burrowing organisms, a process known as *bioturbation* (Aigner 1985).

The current regime on any specific section of the continental shelf is the product of a combination of different current types, although one type may dominate locally. A widely used classification scheme of shelf types is that which emphasises the role of different currents (Swift 1972, 1976). Thus, sedimentation processes on a section of continental shelf may be dominated by: (1) swell waves and storm currents (80% of the world's continental shelves); (2) tidal currents (17%); or (3) intruding ocean currents (3%).

Storm dominated deposits of the Australian shelf

Currents produced during storm events — either as tropical cyclones or temperate storms — dominate in the erosion and transportation of sediment over 82% of the Australian shelf surface area (Figure 6). The energy expended and the amount of sediment transported during one storm event may equal many months (or years) of non-storm background processes. Even on highly dynamic tidally influenced shelves, the effect of a storm is to initiate sediment movement at even greater water depths and at greater rates in shallower depths than is experienced under normal conditions. Each year, storm-dominated shelves may experience less than one or as many as four or five storm events that cause sediment transporting flows.

Sediments on the southern parts of Australia's continental shelf are arranged in zones parallel to the coast, reflecting the dominance of currents related to ocean swell and storms (Figure 6). Studies which have documented the development of shelf storm deposits include those of the Rottnest Shelf (Collins 1988), the Lacepede Shelf (James et al. 1992) and the New South Wales Shelf (Davies 1979). In these locations, long-period swell waves cause nearly continuous reworking of sediments on the inner shelf, winnowing away fine-grained sediments and leaving a sorted sandy sediment in the current affected depth zone. Muddy sediments are deposited below this depth; they are accumulating on the middle New South Wales Shelf in 60–130 m depth, and on the higher energy Lacepede Shelf are deposited in water deeper than 140 m.

Tropical cyclones are the cause of storm events in much of northern Australia (Figure 6). They are associated with atmospheric low pressure systems and attain mean wind speeds of at least 63 km/h. The sections of the Australian shelf most frequently affected by cyclones are the North West Shelf with up to 25 cyclones per decade and the Great Barrier Reef with up to 15 cyclones per decade. These shelf areas are not affected greatly by swell generated in a major ocean basin. The swell pattern is represented in significant wave height data obtained from satellites: whereas significant wave heights are less than 1.5 m for 50–70% of the time in northern Australia, they are larger than 3.5 m for 30–50% of the time along much of southern Australia (see Figure 6).

Tropical cyclones induce strong currents that erode and transport sediment over a wide area. Current measurements obtained in the Gulf of Carpentaria during one tropical cyclone, recorded near-bottom currents with hourly average speeds up to six times larger than background current speeds (Church & Forbes 1983). One cyclone in the Great Barrier Reef province is reported to have caused the erosion on the mid-shelf of a sediment layer averaging 6.9 cm thick and to have transported sediment a minimum distance of 15 km (Gagan, Chivas & Herczeg 1990). Modelling studies by Hearn and Holloway (1990) on the North West Shelf have shown that, under the influence of tropical cyclones, strong westward flowing coastal and inner shelf currents are established between the eye of the cyclone and the coast. Such cyclone-induced currents are clearly a significant factor affecting sediment movement on cyclone-dominated shelves, but they may also influence the long-term (net)

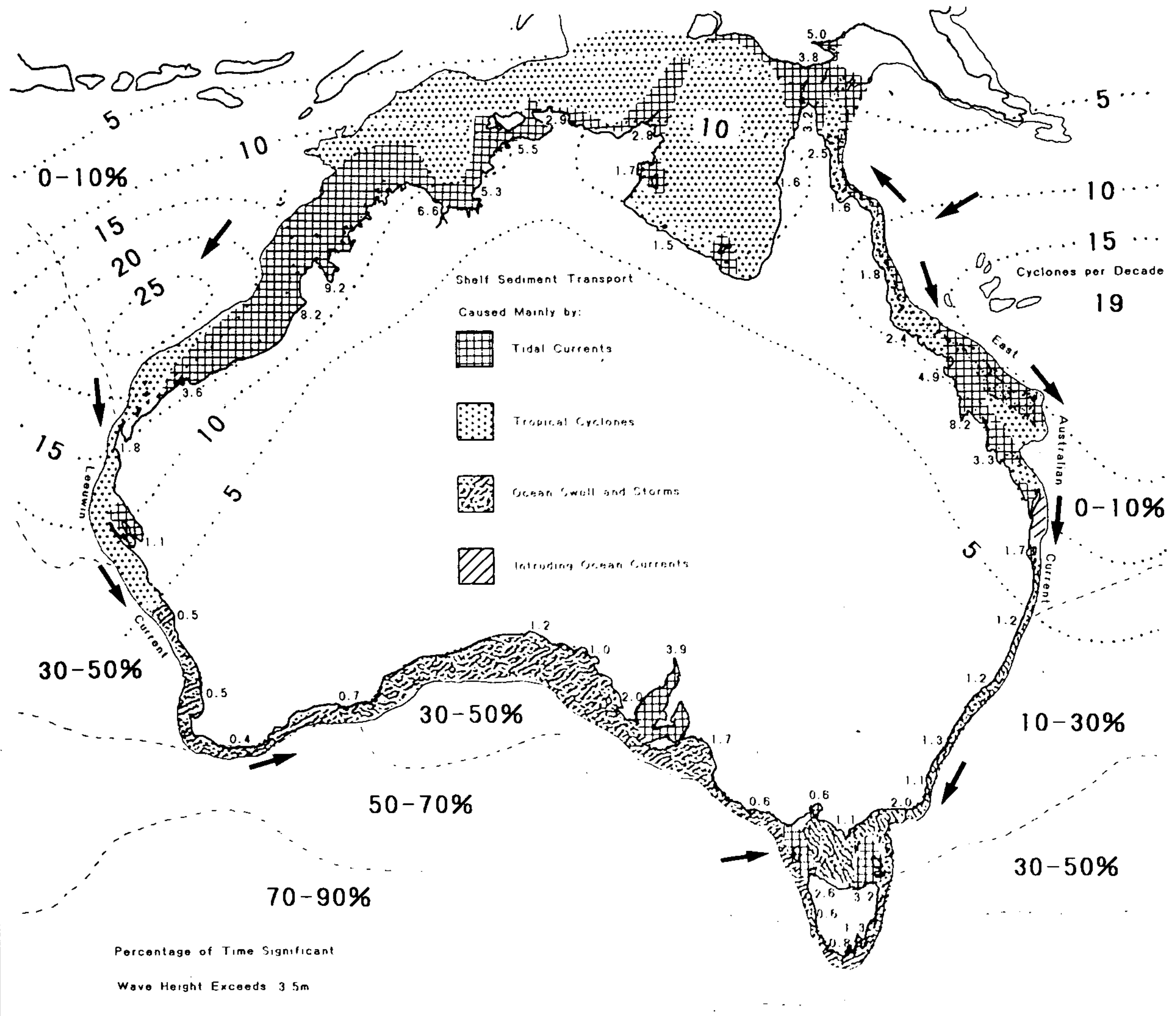

Figure 6: Division of the Australian shelf into regions in which sediment transport is caused mainly by tidal currents (17.4% of the shelf area), currents derived from tropical cyclones (53.8%), ocean swell and storm generated currents (28.2%) and intruding ocean currents (0.6%). Contours of tropical cyclone frequency per decade are from Lourensz (1981) and contours for significant wave height percentage exceedence are from McMillan (1982). Mean spring tidal ranges indicated along the coastline are derived from the Australian National Tide Tables. The location and direction of flow of major ocean currents are indicated.

sediment movement on some otherwise tidally-dominated sections of the shelf (Figure 6, and see below).

Tidally dominated deposits of the Australian continental shelf

Tidally dominated shelves occur generally where the mean spring tidal range measured along the coast exceeds 4 m (*macrotidal*). Around Australia, tidal ranges greater than 4 m occur along the north-western coastline between Port Hedland and Darwin and reach a maximum range of about 9.2 m in King Sound (Figure 6). The southern Great Barrier Reef coastline is also macrotidal, with a tidal range of 8.2 m in Broad Sound. In the Fly River estuary (western Gulf of Papua) tidal ranges are 5 m.

Tidal currents are also an important sand transporting agent in *mesotidal* (2–4 m tidal range) areas such as Torres Strait, Moreton Bay (Queensland) and Bass Strait. Tidal currents are able to dominate sand transport in *microtidal* areas (tidal range less than 2 m) in restricted cases where coastal geometry affords shelter from ocean generated swell and wind-driven currents. Such is the case in many bays (eg Shark Bay), the approaches to some major ports (eg Port Phillip Bay) and in partly enclosed gulfs (eg Spencer Gulf, Gulf St. Vincent and the Gulf of Carpentaria). Tidal currents are accelerated as they flow over and between shelf edge barrier reefs; thus sediment transport is affected by tides along the shelf edge over much of the Great

Barrier Reef (Figure 6). In total, sediment transport is dominated by tidal currents on about 17.4% of the Australian continental shelf.

Tidally dominated shelves exhibit discrete zones of seabed scouring and erosion. Such are associated with an area where tidal currents reach a maximum speed (Figure 7). Seabed sediments are arranged in a divergent pattern demonstrated by an increasing supply of sand of decreasing grain size as the distance away from the scour zone increases (Johnson et al. 1982). Such diverging bedload transport patterns are known as *bedload parting zones*. Examples of areas where tidal currents have produced such bedload parting zone facies are found in the Torres Strait and Gulf of Carpentaria regions (Figure 8). In these areas, tidal currents are accelerated as they flow through constricted channels located between islands and reefs (Harris 1988). The zones of maximum tidal current speed are sometimes related to *tidal amphidromic points*. These are a type of standing wave node in which sea level change is small but current speeds are large; two such amphidromic points are found in the Gulf of Carpentaria (Figure 8). Sedimentary facies are arranged as shown in Figure 7 with respect to tidal current maxima associated with the Torres Strait, Groote Eylandt and Mornington Island areas.

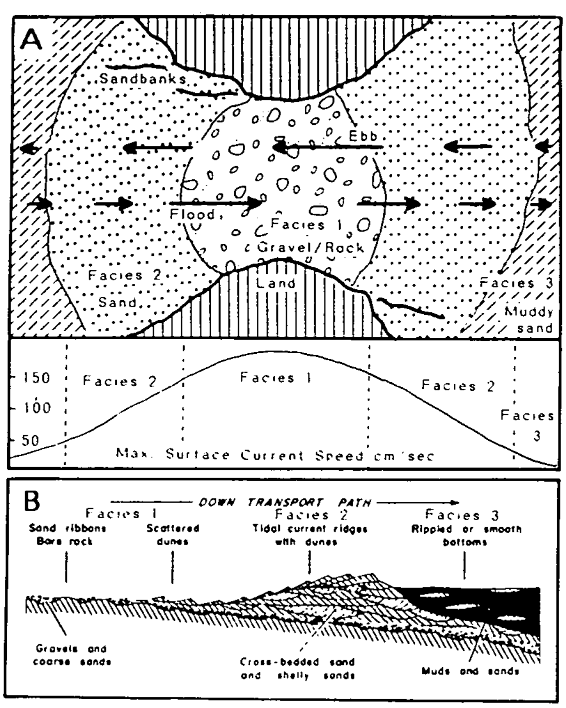

Figure 7: Idealised diagrams showing: (1) the spatial distribution of sedimentary facies related to an area of accelerated tidal flow (bedload parting zone) located in a constricted channel, with characteristic maximum surface current speeds; and (2) a cross section showing bedforms and relative thicknesses and content of deposits (after Allen 1970).

Shelves dominated by intruding ocean currents

Sediment transport and dispersal controlled by intrusive ocean currents affects only a little more than 1% of the Australian continental shelf (Figure 6). The only location where such currents are known to dominate sediment movement around Australia is the shelf offshore from Fraser Island, where the southward-flowing East Australian Current intrudes. Sidescan sonar and seabed photographs obtained during the 1980 cruise of the German research vessel *Sonne* to the northern New South Wales – southern Queensland region, show the development of large submarine dunes at depths as great as 80 m, but more typically in the 40–50 m depth range. Dune morphologies indicate a general southward transport of sand, thought to be related to the effect of the East Australian Current. The Leeuwin Current may have a similar effect over parts of the Western Australia shelf, but further research is needed to confirm this.

Environmental indicators

Monitoring sedimentation

An understanding of the impact of humans on sedimentation patterns is of considerable interest in the coastal zone, where various harbour and beach engineering projects have altered such things as natural sediment transport patterns and caused coastal erosion etc. (see Bird, this volume). However, few studies in Australia have considered the effect human activities have had on shelf sedimentation patterns. The use of continental shelf sediments as a record of pre-historical environmental conditions may prove to be a tool of fundamental importance to organisations charged with monitoring the effects of human-induced environmental change. Such changes include those related to:

(1) river discharge of sediment (in terms of total load, sediment composition and nutrients) as it is affected by mining, deforestation and agriculture; and

(2) the nature and mass of biogenic carbonate sediment input as it is related to oceanographic factors (eg temperature, turbidity and nutrients).

Examples of the forecasting use of sediments are shown below in corals, Fly River delta deposits and sediment accumulation over long time periods.

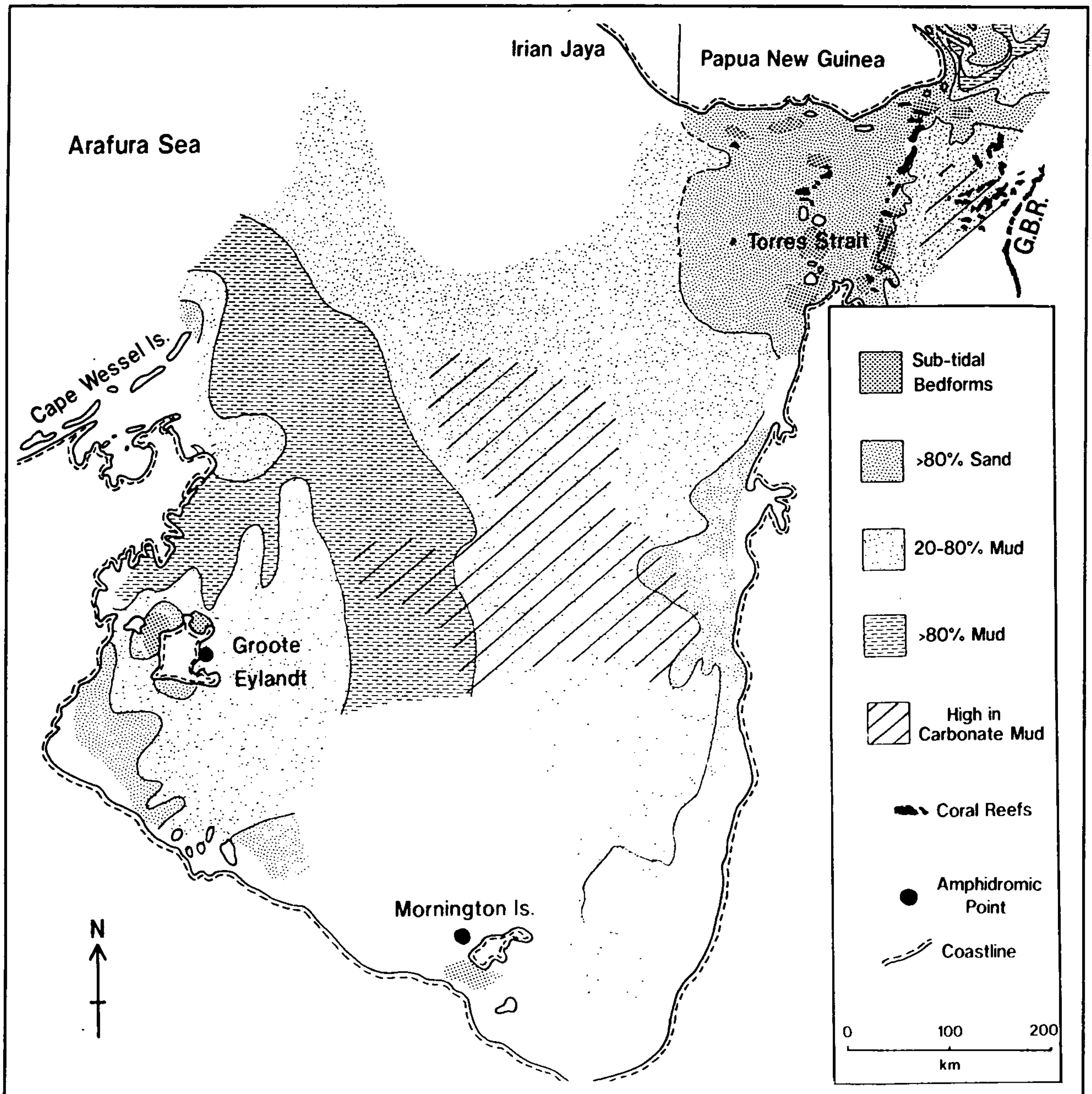

Figure 8: Sedimentary facies of the Gulf of Carpentaria and Torres Strait area. At locations of high tidal energy, subtidal bedforms occur. Note that these are related to the positions of tidal amphidromic points near Mornington Island and Groote Eylandt. Tropical cyclones dominate in the movement of sediments in other parts of the Gulf. An area of sediments having an increased carbonate mud content is found in the central Gulf.

Corals exhibit seasonal density banding that can be observed in X-ray photographs. The banding is apparently related to seasonal variations in coral growth similar to tree rings. The exact cause of the bands is unclear, although seasonal variation in water temperature, light intensity, nutrient availability and/or water turbidity have been suggested as possible factors. The spacing and character of coral bands, including isotopic and trace element signatures, are potentially useful as environmental indicators (Barnes & Lough 1989).

In order for sediments to contain a useful record of short-term environmental changes (ie on the scale of years) both rapid deposition (more than 1 cm/yr?) and a minimal amount of reworking (bioturbation) must occur. Sedimentation in the Fly River delta of the Gulf of Papua is rapid (on the order of 2–10 cm/yr) and core samples of deposits contain seasonally spaced laminations (*varves*) (Harris et al. 1993). These Fly River deposits contain information on spatial and temporal variations in the concentrations of heavy metals such as copper and zinc (Baker &

Harris 1991). Deposits formed by rapidly migrating bedforms, such as occur in the entrances to Moreton Bay and Port Phillip Bay could also contain potentially useful sediment records.

Sediments accumulating over long time intervals (hundreds to thousands of years) can also be used as environmental indicators. Aharon and Chappell (1983) discuss the use of carbon and oxygen isotopes from coral (and bivalve) samples to deduce the history of global ice volume and oceanic temperature in the Holocene.

As noted above, basic descriptive information on sediments is unavailable for about 30% of the Australian continental shelf (Figure 5) and the present input rate of sediments by rivers to the shelf is unknown. The 'information gap' widens further when consideration is given to baseline data on environmentally important indicators such as the heavy metal content of sediments. In many industrialised areas around Australia, heavy metals have been introduced into the marine system where they accumulate in fine-grained sediment depositional zones.

More importantly, perhaps, is the question as to whether or not total river input of sediment to the continental shelf has changed significantly since European settlement. Recent studies concerning loss of soil due to logging and agriculture suggest that increases of sediment loads by a factor of between ten and one hundred or more may commonly result (eg Doeg & Koehn 1990). A 50% increase in the sediment load of the Fly River (Figure 5) has occurred due to the operations of a single mining company (Harris et al. 1993). Nutrients bound in topsoil and supplied to the continental shelf by rivers are also a significant factor in eutrophication problems identified in the Great Barrier Reef area (see Brodie, this series).

Effects of offshore activities

Human activities which involve interaction with shelf sediments include demersal trawl fisheries, offshore dumping of dredge spoil, sand mining and aggregate extraction, urban sewage, oil spills, the construction of oil pipes and platforms and telecommunications cables. In the case of most construction activities, shelf sediments are disturbed in a restricted area and usually on only one occasion. In some cases, scour of the seabed by strong tidal currents may endanger unburied pipelines (eg on the North West Shelf). Oil spills and oil introduced through urban run-off, ships' bilges, etc. has caused elevated levels of hydrocarbons in shelf sediments (eg the Great Barrier Reef: Smith, Bag & Sin 1987; Sydney ocean outfall: Nichols & Espey 1991). The dumping of dredged spoil at sea has been carried out at a number of locations around Australia, including most of the major shipping ports (see review by Harris, Baker & Cole 1991).

Fishing for bottom-dwelling prawns and scallops involves dragging a net or dredge through or across the sediments. Since such operations are carried out over wide areas of the Australian shelf, their impact on the erosion and reworking

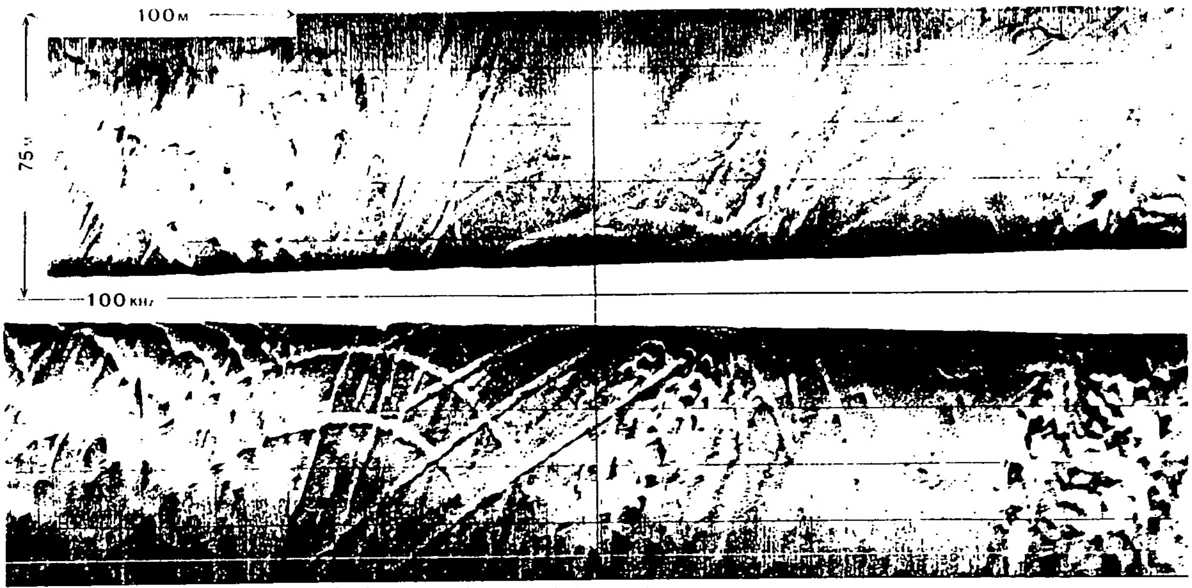

Figure 9: Sidescan sonograph showing elongate fish trawler marks curving across the seabed in Moreton Bay (from Harris et al. 1992). Such marks have been identified also in Torres Strait and probably occur wherever prawn trawling operations are carried out.

of shelf sediments is potentially great. Sidescan sonar images obtained from Torres Strait and Moreton Bay by the Ocean Sciences Institute (University of Sydney) document the occurrence of elongate parallel sets of 'grooves' which are attributed to prawn trawlers (Figure 9). The ecological impact and the spatial and temporal distribution of trawl marks around Australia are unknown.

Existing monitoring programs and databases

Existing monitoring programs which involve the measurement of continental shelf sediment properties in relation to other environmental parameters are few in number. The Torres Strait baseline study carried out by the Great Barrier Reef Marine Park Authority and the Sydney Water Board in its ocean outfalls project are the only programs in Australia that have a strategy involving repeated shelf sediment sampling.

Databases known as 'geographic information systems' (GIS) incorporating information on shelf sediments and bathymetry, are being prepared by: the CSIRO Division of Wildlife and Ecology, Canberra; the CSIRO Division of Fisheries (Cleveland and Hobart); the Australian Geological Survey Organisation (formerly the Bureau of Mineral Resources) in Canberra (their map series for offshore resources being prepared will be available also as a GIS); and the Ocean Sciences Institute in cooperation with the Defence Science and Technology Organisation and the Royal Australian Navy Hydrographic Office in Sydney.

Conclusions

The Australian continental shelf has a complex history involving the rifting apart of Gondwanaland followed by collision and uplift to create New Guinea over a time period of many millions of years. The sediments now found on the surface of the shelf were deposited partly during the Pleistocene period when sea level was lower than at present, and partly under the conditions of the present high sea level stand. Sediments are mostly biologically produced calcium carbonates on the outer shelf and river-derived quartzose sands and muds on the inner shelf. Under natural conditions, mean sediment accumulation rates are typically very slow (about 0.02 cm/yr). Sediment types have not been mapped for about 30% of Australia's continental shelf and the accurate determination of river and biogenic sediment supply to the shelf is yet to be carried out. The mobility of shelf sediments is controlled by combined swell waves and storms (28% of the shelf), tropical cyclones (53%), tidal currents (17.4%) and ocean currents (0.6%). Future research should be directed at:

- obtaining basic descriptive data on sediment types from uncollected areas (ie the southern and western shelves);
- obtaining baseline data on heavy metals and other contaminants from representative shelf environments;
- a better understanding of sediment supply and physical processes of shelf sedimentation; and
- studies of the sediments as records of environmental change.

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The technical paper by Dr. P. Harris was reviewed by Dr. J. Marshall, Australian Geological Survey Organisation, Canberra, and Prof. J. Middleton, Centre for Marine Science, University of NSW.

Marine ecosystems: hard and soft shores

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Introduction

Hard and soft seashores ring our island continent and form the interface between sea and land.

It is beyond the scope of this report to describe in detail all of the research on these habitats. However, we can discuss rocky, sandy and muddy seashores from the viewpoints of recent ecological research, our understanding of how the ecological communities in these habitats function, and how this knowledge is used in environmental decision making. Rocky shores are the main hard substrata of headlands along the open coasts and sometimes within estuaries (Bird 1984). Sandy beaches also characterise many of our open coasts and are extremely important for Australia's recreational beach use. Mudflats and sandy tidal flats exist along the shortest amount of coastline (although they may cover large areas), being defined as any mostly unvegetated soft-sediment shores within moderately calm waters. They may often however, be contiguous with seagrass or mangroves (see below).

In this report we focus on research published in the peer-reviewed literature and exclude wherever possible, unpublished research contained in theses and less accessible reports. We also limit our discussion to those hard and soft shores not considered in other parts of the total report. Our presentation focuses upon intertidal and nearshore habitats, thereby largely excluding discussion on subtidal rocky reefs (see Keough and Butler, this volume). In particular, we do not discuss coral reefs and the habitats of their lagoons (including sandy sediments and beachrock); nor the vegetated parts of soft (especially muddy) shores such as seagrass beds, mangrove forests and saltmarshes. As we emphasise below however, these divisions are artificial from an ecological point of view (Fairweather & Quinn 1993).

The distribution of shore types

A first step in examining the state of the environment of hard and soft shorelines is to understand their extent, distribution and characteristics. Estimates of coastline vary enormously because of the fractal nature of the coasts; the estimate tends to increase as our measuring abilities increase and the measuring step subsequently is decreased. Nevertheless, a number of geographers (eg Galloway & Bahr 1979) have attempted to estimate the extent of different coastline types around Australia. Fairweather (1990a) used these estimates to assess the 'availability' of different habitats for study in the various States (Table 1).

The different sorts of seashores are not evenly distributed around our coastline (Table 1). The geomorphologic reasons for this are complex: shore types depend on a source of substratum (eg parent rock or sediment) and the forces of waves, currents and winds that alter them (Bird 1984). The basic distinction worldwide (Cooke & Doornkamp 1990) is between cliffed coasts and shore platforms on the one hand and depositional coasts and beaches on the other. High energy shores lose sediments as they are transported away by water movement and so erode to parent rock material (if available). Australia's high-cliffed coasts are the prime example of this landform. In contrast, shores of lesser energy are characterised by smaller and smaller particle sizes as the available energy wanes. Thus, sandy shores are subject to less energy (both in water but also wind) than are most rocky coasts, but they have more associated energy than have muddy shores (with the finest sediments). The sand deposited upon beaches is often blown further inland where it forms not only dune systems but also acts as a source of coarser particles for estuarine soft-sediment shores. Tidal flats with muddy or more sandy sediments are found mainly in either shallow and therefore calm embayments (eg gently sloping sea floors where wave energy is minimal) or within estuaries. Under these conditions, the ebb and flow of the tide over large expanses of shallow ground tends to vertically accrete fine sediments. Within estuaries, the bottom is exposed to strong

Table 1: Coast types as percentages of coast lengths. Data given for mud alone and for mud and mangrove shores combined because the mudflats associated with adjacent mangrove forests are probably under-represented by the scale of mapping used to generate these estimates (i.e. any coast with mangroves would be counted as that type even where mudflats extend seaward of mangroves). The percentage remaining from the total is coral reefs and estuary mouths. Based on data in Fairweather (1990a).

	Coast Type				
State	**Rock**	**Sand**	**Mud**	**Mud /Mangrove**	**Total**
Queensland	4	36	1	19	59
Northern Territory	9	40	6	48	97
Western Australia	19	49	2	19	87
South Australia	33	59	1	8	100
Victoria	26	64	7	9	99
Tasmania	29	68	2	2	99
New South Wales	33	65	1	2	100
Total	18	47	2	19	84

Table 2: Ecological studies as percentages of the total (729) papers published in 1980-7 on the three types of coastline addressed in this present paper. Based on data in Fairweather (1990a).

State	**Papers dealing with**			**Total**	**Number of**	**% of Total**
	Rock	**Sand**	**Mud**		**Papers**	**Papers**
Queensland	2	7	4	13	321	44
Northern Territory	0	26	26	54	19	3
Western Australia	20	13	6	39	124	17
South Australia	45	14	2	61	51	7
Victoria	32	3	14	49	59	8
Tasmania	40	5	3	48	40	8
New South Wales	43	7	6	56	200	28
Total Papers	176	67	44			
% Papers	24	9	6	36		

current regimes as the estuary flows but also to a major source of sediment from erosion of the land further up the estuary and river system. Much sediment in estuaries is trapped and consolidated by vegetation, particularly mangroves and saltmarshes (Bird 1984).

The world's coasts have been classified in terms of wave types. According to that classification, the north and south coasts of Australia appear to be dominated (Cooke & Doornkamp 1990) by low energy, small wave regimes, while the west and east coasts have mainly swell waves. West-coast swells typically have more energy than have east-coast swells (Cooke & Doornkamp 1990).

Both tidal flats and saltmarshes are more extensive under macrotidal conditions (Cooke & Doornkamp 1990). Mud accumulates only under very sheltered conditions, whereas sand can be deposited under a variety of conditions (Cooke & Doornkamp 1990), just as it can be transported easily by both water and wind due to its size range of particles. Muddy sediments dominate the north coast of Australia, either under extensive mangrove stands (Table 1) or as wide, nearly flat shores experiencing enormous tidal ranges (eg around Broome on the north-west coast). Sandy beaches are common in all States but along the east and west coasts of the continent swept by current systems like the East Australian Current in the east and the Leeuwin Current in the west, beaches up to 150 km in length are separated by rocky headlands. Rocky coasts are most widespread in the south of our continent, as epitomised by the Great Australian Bight and much of the Tasmanian coastline.

Our understanding of hard and soft shores as ecosystems

The next step in assessing the state of hard and soft shores in Australia is to evaluate Australia's research effort and hence our understanding of the dynamic ecosystems along these shores. Assessing the effectiveness of marine research is difficult because research activity may not correlate with research quality *per se*. Nonetheless, Fairweather (1988a, 1990a) assessed marine ecology in Australia by reviewing all papers published in the period from 1980 to 1987. He found that 22% of the 729 papers dealt with rocky reefs, 10% with sandy bottoms and 6% with muddy bottoms. Thus, less than 40% of this published output came from research on coastal types which made up more than 65% of our coastline (Tables 1, 2). In constrast to this under-representation are studies of coral reefs (31% of papers, at least 14% of coastline), mangroves (13% and 17%, respectively) and seagrass (16%, proportional extent unknown but small). These data suggest that unvegetated, soft-sediment shores have been under-represented in Australia's recent research output in marine ecology, a trend not unique to Australia (Wilson & Browne 1991).

There is evidence that research is concentrated in some States and institutions within those States. For example, the University of Sydney (see Underwood in press) tends to dominate rocky shore research (Table 3), followed by the universities of Western Australia and Tasmania, and Melbourne and Monash in Victoria. The vast majority of this research has been based in the Sydney region at the Cape Banks Marine Scientific Research Area (see Westoby 1991). Only recently is rocky intertidal research being actively pursued in some States, like Queensland and New South Wales (non-urban areas) (eg Fairweather 1991). The relatively high levels of research on rocky and unvegetated soft shores by universities compared to other institutions (Table 3) probably reflects their different research missions: the non-university institutions have concentrated on fisheries biology and non-experimental approaches to large-scale problems whereas university marine ecologists have more recently focused on field experiments at smaller scales.

Research on hard shores

The biota and their patterns of zonation on many of Australia's rocky shores were described in a classic series of papers in the 1940s, 1950s and early 1960s (Dakin, Bennett & Pope 1948, Bennett & Pope 1953, Endean, Kenny & Stephenson 1956, Womersley & Edmonds 1958, Bennett & Pope 1960). More recently, rocky shores have become a focus for ecological research worldwide. They provide a two-dimensional benthic structure (excluding explicit consideration of the infauna of algal and mussel beds and 'pelagic' organisms in rock pools) and the dominant animals on these shores usually have a hard outer surface (shell or exoskeleton), are either sessile or slow-moving and have life cycles commensurate with either the research careers of ecologists or the funding cycles of research. These attributes make quantifying population dynamics feasible and render rocky shores very amenable to field experimentation (Connell 1972). The main emphasis of field experiments has been on factors controlling the upper and lower limits of the distribution of organisms (ie intertidal zonation) and

Table 2: Breakdown of research on rocky, sandy and muddy shores published during 1980-7. Data from a subset of papers surveyed as described in Fairweather (1988a, 1990a): numbers are percentages: they don't necessarily sum to 100% because of omitted minor categories and the lack of mutual exclusivity between some categories. See Fairweather 1990a for descriptions of the categories.

Shore type	Rocky	Sandy	Muddy
Papers surveyed	98	34	11
Universities			
Sydney	51	0	0
Queensland	2	26	0
WA	9	6	0
Tasmania	7	0	9
Melbourne	4	0	36
Monash	4	0	0
NSW	1	9	0
Wollongong	1	3	0
Adelaide	0	9	0
Others	6	9	0
Other institutions			
CSIRO	2	15	9
State authorities	9	21	36
Overseas	1	15	9
Others	1	0	0
Geomorphological setting			
Open coast	85	47	0
Embayment	10	38	73
Estuary	5	12	27
Tidal height			
Intertidal	79	65	64
Subtidal	22	41	45
Methodology			
Descriptive	97	100	100
Process	66	47	64
Experimental	54	18	18
Object of study			
Animals	83	94	91
Plants	33	0	0
Habitats	5	6	0
Ecological processes	15	9	18

subsequently how biological and physical factors structure assemblages of species on rocky shores. With few exceptions (eg Black et al. 1979, Underwood 1975, 1981, review by Underwood in press), there have been no attempts in recent years to quantify distribution patterns of local species, either within or between shores. Despite the recent experimental emphasis in marine ecology, there is still a need to measure the temporal and spatial consistency of zonation patterns. Such measurement rarely has been done, either in Australia or overseas. Similarly, the geographical limits of the ranges of most species on hard or soft shores at the spatial scale of individual rock platforms or tidal flats, and their temporal consistency, are not known — even for common species — except from where individuals have been collected for other purposes (eg environmental impact assessments) or in general terms from early natural history accounts (see references above). For example, there are numerous rocky shore species that have a geographic limit to their distribution along the coast between Melbourne and Sydney (eg *Lepsiella vinosa, Morula marginalba, Austromytilus rostratus*), yet their precise distributional boundaries are unknown. These boundaries should be resolved by a systematic survey of our coastline (ignoring State boundaries!).

One component of ecological research on rocky shores for which local information is particularly lacking is the ecology of populations and assemblages of algae. There are a number of algal taxonomists in Australia but few ecologically-orientated marine botanists working on rocky shores. The life histories of some algal groups have been extensively studied (eg Clayton & King 1991 — includes some Australian contributions) and patterns of recruitment and small-scale spatial variability of micro- and macroalgae and their interactions with sessile animals have been described for one site in New South Wales (review by Underwood & Kennelly 1990). Much of the available ecological information on macroalgae has come from zoologists examining herbivory (eg May, Bennett & Thompson 1970, Steinberg 1989) or ecologists examining general patterns of community organisation in intertidal and subtidal reefs (review by Underwood & Kennelly 1990). There is an urgent need for research on the population ecology of the common intertidal algae, particularly of those species that provide structural habitat for other biota and/or are susceptible to human disturbance — such as *Hormosira banksii* (Brown, Davies & Synnot 1990, Povey & Keough 1991). Such research should attempt to link the available information on life histories (Clayton & King 1991) with field data on patterns of recruitment, growth and mortality. Extent of dispersal also is not known for most Australian algae. An examination of genetic relationships among algal populations, both within and between shores (as has been done for some species of invertebrates: Ayre 1990) would provide indirect evidence of the amount of exchange between separate populations. Data on dispersal ability are essential to assess the ability of algae to recover from natural and anthropogenic disturbances.

In contrast to algae, our knowledge of the ecology of rocky shore invertebrates is considerable. The population dynamics (ie population structure, population and individual growth rates, mortality and reproductive cycles) of most of the common gastropods of the south-east coast of Australia have been described (reviews by Underwood 1979, in press) but the spatial extent of these studies is very uneven and varies considerably between species — as is the case for most marine research on any continent. For example, some species of limpet (eg *Cellana tramoserica*) have been studied on shores in New South Wales and Victoria, whereas data on most species are restricted mainly to one shore in New South Walers (ie the Cape Banks Scientific Research Area). Information on other groups (eg polychaetes, non-cirripede crustaceans, echinoderms) is less extensive, although some excellent studies are available (Underwood in press). There have also been numerous experimental studies in Victoria, Western Australia and particularly Cape Banks (New South Wales), on the processes of intra- and inter-specific competition and predation and their influence on the structure of rocky intertidal assemblages (reviews by Underwood 1985, in press, Underwood & Kennelly 1990). These studies have been important internationally for ecologists' understanding of the processes structuring marine benthic communities (Underwood & Denley 1984). In particular, the studies have shown that the role and nature of competition among mobile herbivorous gastropods is fundamentally different from that of sessile invertebrates and that spatial and temporal variations in recruitment may be an overriding influence on the zonation patterns of organisms and subsequent community structure on rocky shores. It is crucial for testing the generality of these ideas that studies are commenced in other parts of Australia which are directly comparable to those published from the small number of locations described above. This research has yet to occur.

Patterns and variability in recruitment of marine invertebrates have attracted much research attention in recent years (Underwood & Fairweather 1989) on both small (Denley & Underwood 1979, Underwood, Denley & Moran 1983) and large spatial scales (Caffey 1985). Locally, temporal and spatial patterns of recruitment of barnacles on New South Wales shores have been intensively examined (Denley & Underwood 1979, Caffey 1985) and these studies have led in turn to experimental studies on processes affecting settlement and post-settlement survival (Jernakoff 1983, Underwood, Denley & Moran 1983) and their consequences on other species (eg Fairweather 1988b). These Australian studies on barnacles have shown that the high degree of variability in recruitment not only influences the barnacles' population structure but can also determine interactions with the other component species within the habitat. These studies have made a major contribution to a general understanding of the complexities of settlement and recruitment of intertidal organisms. In constrast, we need data on the patterns of recuitment, and their causes, for other species — either sessile (eg mussels on southern shores) or mobile (eg gastropods; but see Quinn 1988) and comparisons of recruitment patterns of species with different capacities for dispersal.

A logical consequence of varying degrees of dispersal is that populations of intertidal animals will show different levels of genetic similarity. There have been a number of Australian studies on gastropods and potentially clonal animals such as anemones, corals and seastars (review by Ayre 1990; also Watts, Johnson & Black 1990), but rocky shore studies have been spatially restricted (primarily to Western Australia). Ayre (1990) indicated that for those species examined, the degree of genetic exchange between sexually reproducing populations is high with no apparent genetic subdivision of populations, although there was considerable fine-scale heterogeneity. Comparing the genetic variability between populations of species with different degrees of dispersal is one important direction for extending this type of research.

There is little information available on vertebrates (fish and birds) that use rocky shores: indeed, we probably know more about the activities of humans as predators on shores near urban centres (see review by Quinn et al., this volume) than we do about fish and birds. Some studies on the population ecology of intertidal gastropods have identified fish and birds as potential predators (Parry 1982), although others have argued that non-human vertebrate predators are not major influences (eg Underwood 1978 regarding fish on New South Wales shores). Basic, presently unavailable information — such as on the abundances of fish and birds on rocky shores, their patterns of feeding and their diets — would provide a basis from which experimental studies (eg predator exclusion) could be initiated. Research on the trophic links between vertebrates and benthic rocky shore biota is clearly required to resolve the role fish and birds (and humans) play in structuring rocky shore assemblages.

Research on soft shores

There are no cohesive theories about any aspects of the ecology of soft sediment intertidal assemblages (Posey 1987), largely because of a paucity of experiments investigating the mechanisms controlling the biota (Hairston 1989, Warwick, Clarke & Gee 1990, Wilson 1990). This situation contrasts with generalisations we can make about rocky shore communities due to the experimental effort put into understanding that habitat. Rocky shores are clearly different habitats from soft-sediments and therefore are likely to have different processes reigning over their ecology (Hairston 1989). The unique features of soft-sediments shores and their assemblages (see Table 4 for comparison of rocky, sandy and muddy shores) include:

an obviously three-dimensional structure to the habitat (ie depth is an important variable);

the grain size, depth and chemistry (especially oxidation state) of the sediments which exert profound influence on the types of organisms living within it, as does sediment mobility whether by storms, currents or bioturbation;

a large size range of organisms which include some that ingest the sediment matrix (obtaining nutrients from organic mater within) as well as live on or within it;

a lack of large attached plants, especially in exposed locations like sandy beaches (and this explains the paucity of botanical studies in these habitats; see Table 3), with most primary producers being microscopic; and

often a contiguity of habitat with other types such as seagrasses, mangroves, saltmarsh and open ocean which facilitates the movement of organisms among them — both between and within different stages of their life cycles.

Of the complete Australian literature on marine ecology during eight years of the 1980s, Fairweather (1990a) identified unvegetated soft bottoms as the least studied benthic habitat in the country. Overseas, tidal flats are known to be important because they have high biodiversity

Table 4: Some characteristics of shorelines made up of the three different substrata.

Characteristics	Rock	Sand	Mud
Substratum			
Sediment size	hard, continuous pebbles to boulders	coarse grain size	fine grain size
Mobility	immobile but erodable	reworked by waves, abrasive	reworked by currents (infrequent)
Burrowed?	rarely bored but bio-eroded by grazers	mainly interstitial meiofauna, some large burrowers	many burrowing organisms of range of sizes
Primary production and main energy sources	micro- and macroalgae, plankton	nutrient poor SiO_2 no rooted plants wrack, plankton, some diatoms in surface layers	lots of organic matter, productive surface microalgae, plankton, seagrass, some macroalgae
Anaerobic?	no	rarely	yes, at quite shallow depths
Organisms			
Size range of common taxa*	small to large	mainly small	small to medium
Appearance	epifaunal from obvious to cryptic#	infaunal	mainly infaunal
Dominant/ common taxa	gastropods, mussels oysters, barnacles, echinoderms, cnidarians, flatworms, annelid tubeworms, sponges, ascidians, bryozoans, isopods, amphipods, sipunculans, at least 5 divisions of algae	crustaceans, annelids, molluscs, nematodes	various worms amphipods, gastropods, clams, crabs, wading birds, fish, many others
Rare taxa		colonial animals, rooted plants, echinoderms	colonial animals, angiosperms

* Size range goes from meiofauna (=small) to megafauna (=large), excluding transients # some smaller species are infaunal within mussel or tubeworm beds

(Warwick 1993), are good indicators of environmental health (due to providing the ecosystem service of improving water quality), their biota are important as food for prawns, fish and birds, and they serve as settlement sites for larvae (these last two factors make them integral to our recreational and commercial fisheries). Scientific information about Australian tidal flats is essential to our understanding of how these habitats function as ecological units, which in turn is used to manage human activities in estuaries. At present, requests for information about how these ecosystems work can be met with only general statements, untested ideas or assertions about overseas concepts untried here. As well as providing much-needed ecological knowledge for environmental decision making, any research on tidal flats will stimulate scientific activity in an ecosystem that has been neglected in this country, more so than on any other benthic marine habitat.

Quantitative surveys of subtidal soft-sediment benthos have been done in Australia (eg Poore &

Rainer 1979), including detailed examinations of spatial and temporal variation (Jones 1987, Morrissey et al. 1992). There has also been some quantitative sampling in Australian tidal flat habitats which have described the biotic assemblages present (eg Rainer 1981), although most have been related to the presence of some type of effluent discharge (eg Dorsey 1982). In addition, foreign scientists have recently established research programs here, including sampling on sandy beaches (eg Dexter 1983a, 1983b, 1984, McLachlan & Hesp 1984a, 1984b, McLachlan 1985, Warwick, Clarke & Gee 1990, Dexter 1992). However, there have been few studies which have gone beyond this necessary step of description to examine experimentally the factors responsible for such community structure. With the exception of the large research program of Peterson and Black in Western Australia (see below) and the very recent experimental work of Walters and Moriarty (1993) on microbenthos in seagrass, there has been no research program in temperate Australia using experiments to test our ideas about these ecosystems. This situation is in stark contrast to those in the United States of America (eg Peterson 1977, 1979, Wilson 1990, Woodin 1991), Europe (eg Reise 1985) and South Africa (eg Branch & Pringle 1987) where there is a strong experimental tradition of studying tidal flats. The results from these studies demonstrate that a variety of interactions, including competition and predation, can vary from intense to non-existent and that adult-larval interactions can be important for some species (Wilson 1990, Woodin 1991).

It is a reflection of the lack of interest in community processes in unvegetated soft sediment habitats in Australia that the only experimental research program mounted in this country was instigated by an overseas ecologist on sabbatical leave in Western Australia (see eg, Peterson & Black 1986, 1987, Black & Peterson 1987, 1988, Peterson & Black 1988a, 1988b, 1991, Peterson 1991, 1992). The field work in these studies was based on excellent natural history observations of the assemblages present as well as the ideas and experimental techniques previously employed in the United States of America (eg Peterson 1977, 1979, Peterson & Andre 1980, Peterson 1982). They demonstrated, for example, that food limitation in suspension feeders was possible on tidal flats and that competition for food was often more important than competition for space. This experimental program also revealed that the causes of observed patterns in sizes and shore levels were more complex that was suggested by simple inference from sampling. The program has been extremely productive in terms of published output (cited above) and the increase in our understanding of these neglected ecosystems is likewise immense. So it is clear that experimental techniques successfully used overseas are applicable here and further elaborations of these methodologies overseas (eg Wilson 1990, Woodin 1991) illustrate the opportunity for developing and testing theory appropriate for Australian soft-sediment communities. Besides this program, the only papers using manipulative field experiments on Australian tidal flats have been a spatially confounded predator exclusion study (Kent & Day 1983) and recent meiobenthic manipulations in seagrass (Walters & Moriarty 1993).

Thus, in Australia, soft-sediment research has been neglected in favour of studies on coral reefs, rocky seashores and vegetated habitats within estuaries and it has been slow in moving from purely descriptive studies (ie species lists and zonations schemes) to quantitative or semi-quantitative attempts at description of communities (often linked to depth zonation). We have a sparse understanding of trophic interactions on sandy beaches (eg Robertson & Lucas 1983) and hence the flows of energy in soft-sediment ecosystems. There have been few Australian studies that have quantitatively monitored the dynamics of soft-sediment biota through space and time, despite recent work showing that a hypothesis-testing rationale can be applied to sampling studies in these habitats (Warwick, Clarke & Gee 1990, Morrissey et al. 1992, Morrissey et al. 1992).

Linkages among different habitats

A full understanding of these shorelines requires that we acknowledge links among different ecosystems. Fairweather and Quinn (1993) argued that exchanges of water and biota are common between offshore and onshore areas of our oceans, along coasts washed by the same seas and across a variety of habitats. There is evidence that the larvae of intertidal or nearshore animals (eg barnacles, crabs, fish) on the west coasts of the United States of America are transported across the continental shelf to recruitment sites (eg Shanks1988 and references therein; Farrell, Bracher & Roughgarden 1991, Roughgarden et al. 1991) by oceanographic processes such as upwellings. Similar data on the movement of larvae are lacking for Australia. Input from the terrestrial environment, by run-off from adjacent land or through rivers (and therefore estuaries) can provide nutrients, energy and matter for coastal habitats (Nixon 1980). Again, such information is entirely lacking for Australian shores. These linkages clearly have important implications for the management of our shores (Fairweather & Quinn 1993), such as integrating management of marine and adjacent terrestrial and freshwater habitats. The unit of management should be at the scale of the catchment, although this has been rarely achieved anywhere in the word, let alone in Australia. The methodology for studying these linkages requires working at a larger scale than is usual for intertidal research. For example, recent advances in remote sensing of water movement may allow us to measure the dispersal of larvae of intertidal biota. In addition, monitoring intertidal populations on a range of shores spread across a geographic region may reveal large scale patterns of recruitment that are consistent on some spatial and temporal scales (eg Peterson & Summerson 1992). These linkages imply that the potential for routinely monitoring intertidal habitats at the scale of individual shores or greater needs to be investigated. For example, aerial photography of intertidal macroalgal, mussel or seagrass beds through time would provide data on changes in these assemblages at a spatial scale relevant to management authorities.

Research needs for environmental issues

Intertidal habitats are the most accessible marine environments to humans (and their wastes) and are therefore undergoing considerable anthropogenic change (Fairweather 1990a), from pollution, reclamation for terrestrial uses, vegetation clearing, harvesting of organisms, exotic species, dredging, etc. There is a moderately large amount of recent information in Australia on the methodology for measuring environmental impacts on marine environments (eg recent issues of the *Australian Journal of Marine and Freshwater Research* Vol. 42(5) 1991 and the *Australian Journal of Ecology* Vol. 18(1) 1993). In contrast, published applications of these techniques to specific impacts on Australian intertidal habitats are rare. The effects of domestic sewage on rocky shores have been described near at least two cities in south-eastern Australia (Borowitzka 1972, Fairweather 1990b, Brown, Davies & Synnot 1990), with comparable changes in algal assemblages: a reduction in large brown algae (eg kelps, *Hormosira banksii*) and an increase in filamentous, primarily ephemeral, red and green algae (eg *Ulva* species). The direct effects of sewage on intertidal invertebrates is unknown. The effects of industrial pollutants have not been examined for many seashores in Australia, except where it is confounded with domestic sewage outfalls. McGuinness' (1990) work on the effects of oil on invertebrates in mangroves and saltmarshes in Botany Bay gives an indication of what experiments may be possible. The detrimental effects of TBT (tributyltin, from antifouling paints) on intertidal gastropods (inducing a debilitating sex change in females known as 'imposex') are well known overseas and now have been documented here (Wilson, Ahsanullah & Thompson 1993).

Recreational activity of humans in marine environments is increasing, particularly our use of intertidal habitats (see Quinn et al., this volume). Humans collect plants and animals for food and bait and walk over much of the shore (both hard and soft) during both active extractive activities (eg fishing, collecting) and more passive recreation (eg sunbaking, swimming). Measurements of the spatial and temporal scales of this type of human activity and experimental studies on its effects should be a research priority of marine ecologists in Australia. For example, recreational fishing is growing as a human activity and a large proportion of anglers obtain their own bait from pumping ghost shrimps or worms from sand flats and collecting algae, crustaceans, gastropods, bivalves and ascidians from rocky shores. These are potentially very pervasive activities because these organisms are removed from their populations. This results in the population's reduced abundance as well as indirect effects in its recruitment and the abundance and recruitment of other species by trophic cascades, habitat facilitation, etc. (Fairweather 1990c). In soft sediments, there are also the detrimental effects of obtaining the animals, which in turn disrupts any layering of the sediment (including grain-size sorting and physicochemical conditions) as well as destroying the homes and bodies of many other burrowing organisms (and so the 'bycatch' is very great) (Quinn et al., this volume).

From a conservation perspective, it is unusual to find, for any shoreline in Australia, a listing of rare species living on it or an estimate of the shoreline's local biodiversity. The biogeographic limits and affinities of particular taxa are better known through the efforts of marine taxonomists, but we lack the extensive knowledge of biotic assemblages on hard or soft shores to allow proper planning for conservation. For example, where along the east coast of Australia should we place reserved areas to maximise the species within them by taking advantage of overlapping biogeographic provinces? We do know that mudflats and (to a lesser extent) sandy habitats are extremely important as feeding grounds for migratory wading birds protected under international treaties (ie the Chinese Australian Migratory Birds Treaty and the Japanese

Australian Migratory Bird Treaty). It is also clear that a number of habitats among these shore types are grossly polluted (such as many urbanised estuarine mudflats) or are regularly disturbed by recreation (eg most sandy or rocky shores near cities).

With the notable exceptions of studies on recreational collecting and trampling of algae and invertebrates (Fairweather 1991, Povey & Keough 1991), there are no experimental studies of the effects of human impacts on Australia's shoreline biota. This is despite numerous recommendations in the Australian literature that such experimental studies be done (Underwood 1990, Keough & Quinn 1991, Peterson 1993, Walters 1993) and the obvious success of experimentation in advancing other areas of ecology. The recent studies of impacts from recreational activity on seashores (see review by Quinn et al., this volume) constitute a good example of the sorts of applied research that need to be addressed more widely along the coast of Australia.

Examined in the light of Peters' (1991) arguments about how ecology should be more socially responsible, there have been profound shifts in the attention of seashore ecologists who are turning toward applied problems rather than purely examining patterns and processes in nature. There are two related approaches to this shifting emphasis. One is a change from a specialised, pure ('because it's there') research program toward using the intertidal habitat as a test system for ideas applicable in a wider range of habitats and on larger scales (such as general ecological theory or environmental impact assessments, eg Underwood 1991, 1993). Another is to make a rather bigger jump to testing the specifics of the real-world problem (ie harvesting, recreational trampling or pollution) with the *in situ* system (ie the seashore) (Fairweather 1990b, McGuinness 1990, Keough & Quinn 1991, refer papers in Quinn et al., this volume). The advantage of the second approach is that the studies are done at the appropriate scale and with the relevant organisms (rather than by testing theory with different organisms not of interest to the public, etc. or at a micro-scale which is more tractable). The most obvious advantage to this approach is that the direct outcome is research findings that lead to specific management actions at an applicable scale. Similar arguments have been made by Schindler (1987 — that lake acidification studies be done at the scale of whole lakes) and Walters (1993 — more generally about the importance of research at scales relevant to management).

Conclusions

This brief review has illustrated four important characteristics about the research done on our hard and soft shoreline.

1. Some aspects of rocky shore ecology in Australia (eg population dynamics of gastropods and experimental investigations of their interactions) are well understood and have provided direction for much overseas work; but local studies have been on a restricted spatial scale (mainly on one extensive shore near Sydney in New South Wales). Comparable research programs in other locations are needed to test generalisations from previous studies.

2. Our knowledge on other habitats is very sparse, particularly for unvegetated soft sediments (both mud and sand), despite these habitats comprising a majority of the coast. Research on the ecology of tidal flats — in particular experiments testing hypotheses about benthic assemblage structure — are desperately needed.

3. Although Australian marine ecologists have been involved in the design and evaluation of methods for environmental impact assessments, there has been little application of these innovative techniques to environmental problems on local shorelines.

4. Experimental studies, including both manipulative experiments at small spatial scales and designs that treat environmental impacts as experiments at larger scales, should be a priority for environmental research. Such experiments are the only way to test specific hypotheses about the mechanisms by which intertidal environments change.

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Temperate subtidal hard substrata

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Introduction

Hard surfaces are common features of many subtidal habitats along most of the coastline of temperate Australia. Natural substrata range from large rock surfaces, to small patch reefs and boulders and to small biogenic surfaces such as bivalve shells. Near major cities there are extensive artificial surfaces, such as piers, breakwaters and various marina facilities.

Hard surfaces generate important spatial heterogeneity in subtidal areas and provide attachment space for a wide diversity of sessile organisms. Some of these organisms, especially large brown algae, also contribute important physical structure for reef habitats which are in turn used as habitat by a wide range of mobile animals, particularly fish.

These habitats are important to humans in a number of ways. They form the base of substantial recreational fisheries, especially down the east coast of Australia (Kingsford, Underwood & Kennelly 1991) and are popular in summer for diving and snorkelling activities. Just as in coral reef habitats, the sessile organisms on hard surfaces are potential sources of important natural products (eg van Altena & Miller 1989, Davis, Butler & Altena 1991).

The animals and plants in this habitat are extraordinarily diverse in global terms. Many groups, such as red and brown algae (Womersley 1990), ascidians (Kott 1985, 1990a, 1990b), bryozoans (Bock 1982) and crustaceans (Poore 1990) have much higher species richness in southern Australia than in comparable temperate habitats elsewhere. Often, these groups are more diverse in temperate areas than in the tropics (eg bryozoans, red algae). It is also possible that the processes in these local temperate reef habitats are different from those occurring outside Australia or Australasia. Steinberg (Estes & Steinberg 1988, Steinberg & van Altena 1992) has suggested that levels of herbivory and plant defensive compounds differ between Australia and North America, and Jones and Andrew (1990) and Holbrook, Schmitt and Ambrose (1990) have discussed the roles of herbivorous fish and sea urchins in the dynamics of subtidal algal communities in different geographic regions. While some of these lines of evidence are convincing, Underwood and Fairweather (1986) have cautioned that broad-scale comparisons may be confounded by methodological differences.

Our knowledge of temperate subtidal hard substrata is considerably less than that of corresponding intertidal areas, largely because subtidal areas are less accessible and have a shorter history of experimental approaches. The scarcity of quantitative descriptions of the flora and fauna has been highlighted elsewhere (Underwood & Kennelly 1990, Underwood, Kingsford & Andrew 1991, Kennelly & Underwood 1992), and we will not deal with it in much detail other than to emphasise that general statements must be treated with considerable caution.

Approaches: experimental versus observational

As with other habitats, ecologists have taken a variety of approaches to understanding reef systems. We divide the approaches broadly into two categories: observational and experimental.

The observational approach has a long history, developing from simple qualitative description to elaborate methodologies that involve formal sampling (Andrew &

Mapstone 1987 review the considerations in sampling properly) and attempts to describe recognisable 'communities', 'assemblages', 'biocoenoses', etc, sometimes with the aid of multivariate statistical techniques. Such approaches seek explanations for the observed patterns by noting correlated variables, such as depth, light intensity, turbidity and exposure to wave action. An example of this approach for British subtidal reefs is Hiscock 's (1985).

A neglected aspect of descriptive (or observational) approaches is the need to follow particular sites *through time*, in order to gain not only a static picture but an understanding of the dynamics of the assemblage. When this has been done (eg Kay & Butler 1983; Underwood, Kingsford & Andrew 1991, Kennelly & Underwood 1992) it was found that, although the overall composition of the assemblage was fairly constant over time, substantial small-scale change was constant. An appreciation of this phenomenon is crucial background to experimental studies of processes, to assessing the state of the system and ultimately to making management decisions.

The experimental approach begins with observations of patterns and correlated variables. From them develop possible explanations for the patterns — call these hypotheses — and then tests of the hypotheses by means of manipulative experiments, generally carried out in the field. There need to be *controls* in such experiments, the fundamental idea being to compare the system with and without some alteration to the variables under study.

Both approaches have their place, but we would argue that to *understand* the working of the system — hence to understand the effects of human activities on it and be able to manage it — experiments are essential. There is a large literature arguing this case both by those in the tradition of experimenting on few variables at a time (eg Underwood 1985, 1990) and by those in the tradition of pattern-recognition using powerful multivariate methods, who nevertheless recognise the distinction between recognising patterns and assigning causes (eg Gray et al. 1990, Warwick & Clarke 1991, Clarke 1993).

There has only been limited work of either sort on Australian temperate subtidal reefs, and its geographic extent is small. Studies that allow us to identify the important processes in these habitats are much rarer than the habitat descriptions, and experimental work has been done mainly in the gulfs of South Australia, in Port Phillip Bay, Victoria and off the coast of central New South Wales only. Because of the different quality of the information coming from the studies, we discuss them separately below, and summarise the major studies showing processes on Table 1.

Dominant assemblages - patterns

Descriptions of the floral or faunal assemblages occurring on subtidal hard substrata are geographically patchy. For some areas, such as central regions of South Australia, reasonably comprehensive surveys have been performed that allow the major habitat types to be mapped (eg Shepherd & Sprigg 1976). Elsewhere, including much of the Great Australian Bight and exposed coasts of western Victoria, there is almost no published information. The early descriptions were largely the result of work by Shepherd and Womersley (Shepherd & Womersley 1970, 1971, 1976, Shepherd 1981, Shepherd & Womersley 1981, Shepherd & Sprigg 1976) who provided qualitative descriptions of many reefs in southern Australia. The most useful broad-scale quantitative data come from the studies along the coast of New South Wales by Underwood and colleagues (Underwood & Kennelly 1990, Underwood, Kingsford & Andrew 1991, Kennelly & Underwood 1992) who focused on plants and mobile invertebrates. Most of the available data consist of single surveys, so there is little information about temporal variation on short (seasonal) or long (annual) time scales. We know of almost no long time series for whole assemblages (but see Kay & Butler 1983 and Kennelly & Underwood 1992) and the few long-term data sets for single species are based on commercially exploited species (McShane, Beinssen & Foley 1986, Shepherd 1990, Prince & Shepherd 1992). As an example, Colman, Keough, Quinn, Gwyther and Smith (1991) described the temporal and spatial extent of all known long-term subtidal monitoring programs in Victoria, a process that occupies little space in their report. It is not our purpose here to provide a comprehensive review of these data: this is currently being done in various forms by a range of State authorities such as the Land

Table 1: Extent of knowledge about processes controlling structure of assemblages on temperate subtidal hard substrata in Australia. Data are drawn from only experimental ecological studies (see text). This table does not cover information not yet published in refereed journals and books (departmental reports, theses, etc).

Location	Habitat	Organisms	Time/Space	Process	Reference
Edithburgh, central SA	P	A	10/1	CPWDN	Kay & Keough 1981 Kay & Butler 1983 Keough 1984b Davis et al. 1991 Butler 1986, 1991 Keough 1983
Edithburgh	R[1]	A	3/1	CPD	Keough 1984a Chernoff, 1987 Pitcher & Butler 1987
Gulf St Vincent, SA	P	A	1/5	CDW	Butler 1986, 1991
Rapid Bay, SA	P		10/1	CPW	Kay & Butler, 1983 Keough & Butler 1979
Gulf St Vincent, SA	PR	*Clavelina* (=*Podoclavella*) *moluccensis*	3/2	DCW	Davis 1987a,b, 1988a,b Davis & Butler 1989
NSW-SA	R	K	4/many	NP	Steinberg & van Altena 1992 Steinberg 1989
Portsea, PPB, Vic	P	A	5/1	CPW	Russ 1980, 1982 Fletcher & Day 1983
Popes Eye PPB, Vic	R	*Parma victoriae*	1/1	C	Jones & Norman 1986 Jones & Norman 1986
Central NSW	R	KAUM	3/1	CP	Fletcher 1987
Central NSW	R	KUA	3/1	DCWP	Kennelly 1983 Kennelly 1987a,b,c, 1989
Central NSW	R	*Didemnum moseleyi*	2/1	CDW	Stocker 1991 Stocker & Underwood 1991
WA	R[2]	K	2/1	CD	Kirkman 1981, 1984

Key:
Habitat: P, pilings, other human-made structures; R, rocky reef; Organisms: A, sessile animals; K, canopy-forming plants, typically kelp bed; M, motile animals; U, understorey algae (where one or few species of these are named); Process: C, competition; D, dispersal; G, genetics; H, human impacts; L, light; N, natural products, chemical ecology; O, oceanographic features including upwellings, terrigenous nutrient inputs; P, predation; T, turbidity; W, wave action, other disturbance, patch dynamics; Time period: given in years if a pattern-descriptive study; Space: given as number of separate locations (eg different bays).

1. Small bivalve shells as substrata (*Pinna bicolor, Chlamys asperimma*)
2. Problems with design (see Underwood & Kennelly 1990)
PPB = Port Phillip Bay

Conservation Council (Victoria), and the Department of Primary Industries (South Australia).

The lack of knowledge referred to above is a serious impediment, but it can be placed into context by noting that similar complaints have been raised about the quality and quantity of information available for subtidal reefs in other parts of the world, including such well-studied areas as central California and parts of Chile, Canada and New Zealand (Foster 1990, and other papers in Chapman & Underwood 1990).

One of the major problems in producing any comprehensive description of assemblages is the level of taxonomic resolution available. For many surveys, the taxa that are reported depend on the taxonomic expertise readily available to the investigators. This expertise may not correspond to the patterns of species richness among higher taxa at a particular site (Keough & Quinn 1991). The causes of these problems are well known. For many taxa, the majority of species remain undescribed, even though the taxa are acknowledged to be speciose. In other taxa, it is possible that only a few systematists in Australia are capable of identifying the organisms to generic or specific level, and specimens (eg sponges, amphipods, red algae) must be sent to these specialists. This has the effect of either increasing the budget of the work due to consulting costs, creating a bottleneck as the relevant systematists are overwhelmed by the size of the task, or forcing the investigators to classify organisms only to higher taxonomic levels. We note that a number of authors have advocated the use of higher taxa, usually in the form of functional groups (Jackson 1979, Choat & Schiel 1982, Littler & Littler 1984) and have demonstrated the utility of this approach (Keough 1984a, 1984b, Butler 1986, Warwick 1988, Butler 1991). This approach was claimed to be unsuccessful in Kennelly's and Underwood's (1992) study, because it hid the species-specific patterns of variation present in this assemblage. However, some of their pooled higher groups were very broad (eg sessile animals, mobile invertebrates) and may have consisted of a number of what other authors would consider to be functional groups.

The oldest general scheme for describing spatial patterns was devised by Shepherd and Womersley (1970, 1971) using concepts developed from intertidal zones of rocky shores. They recognized three important zones, based mainly on depth/exposure combinations, and characterised by the algal groups that dominate each zone. A detailed summary of this scheme is provided by Womersley and King (1990). Other authors (reviewed in Underwood, Kingsford & Andrew 1991; and see Sanderson & Thomas 1987) have found it necessary to erect a number of other classes of habitats and assemblages.

Habitat-forming plants

The dominant feature of many open coast reefs is the presence of large, canopy forming brown algae. These plants, primarily kelps such as *Ecklonia, Macrocystis* and *Durvillea,* occur from shallow subtidal depth to 15 m or so, their actual depth distribution varying with the transparency of the water to light. Beneath their canopies is an assemblage of smaller plants, primarily red and brown algae, and a range of animals. In deeper water, these large algae give way to smaller, turfing species, primarily red algae, mixed in with sessile animals. Red algae and animals may also be important in shallower water, where light, wave action, or sedimentation restrict the presence of canopy-forming brown algae. Womersley and King (1990), Underwood and Kennelly (1990), and Underwood, Kingsford and Andrew (1991) provided very good overviews of this topic. An important feature in these shallow-water habitats is the presence of 'barrens', areas dominated by encrusting coralline algae and high densities of herbivorous invertebrates.

We urge some caution in interpreting this discussion of spatial patterns. A number of recent authors have raised doubts about the utility of broad-scale generalisations about spatial patterns (Foster 1990, Schiel 1990, Underwood, Kingsford & Andrew 1991, Kennelly & Underwood 1992). Underwood, Kingsford and Andrew (1991) cautioned that simple habitat classifications are not generally useful. Some habitats — especially *Ecklonia* forests and barrens — exist as small interspersed patches, so a broad classification at a large spatial scale will be forced to ignore the small-scale patchiness. Such a classification, using 'average' or 'most common habitat' descriptors, would have no predictive value at smaller spatial scales. The patterns that initially appeared to exist may have been the artifact of a small data base. In other areas, especially the west coast of the United States of America, the strength of generalisations has weakened as more detailed quantitative information has become available (Foster 1990).

Diversity/taxonomy: knowledge and gaps

The best-studied groups of algae are the greens and browns, less so the red algae. They are especially diverse in southern Australia with high levels of endemism at the species level (Womersley 1990). Even though they have received much attention, they are still far from being described completely. Importantly, many species cannot be identified in the field, thus placing serious limitations on our ability to make rapid, comprehensive floral surveys. It is important to retain voucher specimens as a basis for the names reported in such surveys.

Small-scale distributions

The structural complexity of the algal assemblages varies with the degree of exposure to water movement. In general, lower energy environments have canopies that are less complex structurally than high energy environments. Low energy environments may have no macroalgal canopy, or one dominated by species of *Sargassum* or *Cystophora*, while kelps provide the structure on more exposed coastlines. This variation is related to some combination of wave action, light and sedimentation, and it is hard to separate these factors.

Biogeography/large-scale patterns

At larger spatial scales, there are geographic differences in algal assemblages, particularly in the dominant brown algae. These differences appear more related to latitude than longitude. Down the east coast of Australia, the dominant canopy-forming species are *Ecklonia radiata* and *Phyllospora* species. These taxa — but particularly *Ecklonia* — are common within Port Phillip Bay and Western Port in Victoria, central South Australia and in south-western Western Australia (Kirkman 1984, Clayton & King 1990). At greater latitudes — particularly open coasts of Victoria, south-eastern South Australia and Tasmania — other kelps are found, including *Macrocystis angustifolia*, *M. pyrifera* (Tasmania only) and *Durvillea* species. *Macrocystis* and *Durvillea* do not extend westwards far beyond the Victoria–South Australia border.

These distributional differences are important for understorey plants and animals, because the kelps are morphologically very different from each other; and one of the most important effects of kelps is thought to be their action in attenuating water movement on exposed coasts (Eckman, Duggins & Sewell 1989, 1990, Eckman & Duggins 1991). The hydrodynamic environment provided by *Macrocystis* is very different from that provided by *Ecklonia*. We would therefore expect major structural differences between the kelp forests of southern Victoria and Tasmania, and those from lower latitudes. The direct role of kelps in fish recruitment, and their indirect influences on benthic communities, have been studied most intensively along the west coast of the United States of America, but such experimental work has not been done in Australia.

The barrens habitat is rare from east Gippsland through to Western Australia. Jones and Andrew (1990) attributed this to the presence of particular sea urchins. They suggested that *Centrostephanus rodgersii* is the primary species responsible for maintaining barrens along the eastern coastline while the other common urchin, *Heliocidaris erythrogramma*, cannot create or maintain barren habitats. We note that Sanderson and Barrett (1989) suggested that *Heliocidaris* is capable of maintaining barrens, although the evidence for such a suggestion is unclear. The dominant urchins in open coast reef environments vary from *C. rodgersii* and *H. erythrogramma* in New South Wales, to *Heliocidaris* alone in South Australia, to a mixture of *Heliocidaris*, *Tripneustes* species and *Echinometra mathaei* in south-western Western Australia (Jones & Andrew 1990). Jones and Andrew suggested that *Echinometra* is also capable of maintaining an environment with patches of barrens.

Animals

In open coast environments, sessile animals occur in greatest abundance in well-shaded parts of reef, such as vertical rock faces, underhangs and caves and undersides of boulders. They are relatively uncommon on upward-facing rock surfaces. The dominant sessile animals are modular or colonial, including sponges, ascidians, soft corals and hydroids, and bryozoans. In more sheltered environments animals may occur in more open habitats. These distributions presumably reflect the reduced growth rates of plants in those areas. In low energy environments, the sessile animals are predominantly solitary or unitary; these assemblages correspond to those referred to as 'fouling communities'. The numerically dominant groups are barnacles, solitary ascidians, bivalve molluscs and polychaetes.

Mobile animals occur in most areas, but are not well described. Most of these organisms use the structure provided by the plants, or are herbivorous, but in most cases the exact nature of the relationship is unclear. Some associations may be facultative and others obligate. Work overseas suggests a major influence of the physical structure of the macroalgal assemblage on mobile animals in influencing recruitment (eg Carr 1989, Eckman, Duggins & Sewell 1990) or interactions between established organisms (Holbrook, Schmitt & Ambrose 1990, Schmitt & Holbrook 1990a). The most prominent mobile animals are reef-associated fish, sea urchins, and other commercially-exploited species such as abalone and lobster.

Diversity/taxonomy

The degree of taxonomic resolution is more uneven than for the plants. Despite considerable systematic effort, some groups such as amphipods and sponges have a large number of undescribed species. In other taxa, very common species remain either undescribed or in a state of systematic confusion (eg *Celleporaria* species: see Bock 1982, also Hutchings 1982, Plate 24.2). Some of these groups have very high species richness in temperate Australia. The larger mobile groups, particularly the fish, echinoderms, and molluscs, are much better known.

Unfortunately, the smaller mobile organisms (crustaceans, opisthobranch molluscs) and the majority of the sessile animals (especially some colonial ascidians and sponges) are almost impossible to identify in the field. It is unlikely in the near future that we will be able to produce detailed descriptions of the fauna of particular locations. Most of the existing literature lumps these groups into higher taxa, often corresponding to phyla or classes; so we can provide only very broad ideas of biogeographic patterns.

Small-scale distributions

As we mentioned earlier, there is a scarcity of detailed quantitative descriptions of faunal assemblages. The best details come from studies in a couple of areas only. A range of papers provides descriptions at levels ranging from species to functional groups. These data, from central South Australia, come from a variety of natural and artificial surfaces. The other major data set is from rocky reefs in central New South Wales. At one of the most intensively studied sites, Edithburgh (South Australia), the composition of the assemblage of sessile animals varies considerably with the nature of the substratum. Large, stable surfaces are dominated by sponges and tunicates, while smaller, isolated surfaces have many bryozoans and unitary organisms. The presence of this small-scale variation makes it difficult to interpret or generalise from spot censuses at other localities.

Biogeography/large-scale patterns – variation among sites

Different taxa show degrees of large-scale variation, ranging from local variation in structure of assemblages to the consistent presence of some widely-distributed taxa. Many animal groups are mixtures of species with restricted and wide distributions.

Given the taxonomic limitations, it is difficult to comment on sessile organisms. Many sessile animals do have restricted spatial distributions (eg Kott 1985) but there are some broad similarities in the best-studied sessile assemblages (Portsea in Victoria; Edithburgh and Rapid Bay in South Australia). Even at these sites, the dominant species at a particular site may be rare or absent from others (eg Kay & Keough 1981, Russ 1982).

Among mobile animals there are similar patterns. In addition to the sea urchin example discussed earlier, there are some regional differences in other echinoderms. For example, the Tasmanian asteroid fauna includes a number of species that are restricted to that region (Dartnall 1980). The major reef-associated fish also vary; Jones and Andrew (1990) provided representative lists of herbivorous fish from temperate Australia and New Zealand; and other families show some differences, both longitudinally and latitudinally (Last, Scott & Talbot 1983; Hutchins & Swainston 1986).

A major cautionary note can be derived from Butler's (1986) description of sessile assemblages at a number of piers in South Australia. He found substantial differences between assemblages at piers separated by as little as 5 km, with different species dominating the assemblages. Kennelly's and Underwood's (1992) is another of the few studies covering such spatial scales and reporting important differences at small scales. In attempting to assess gaps in our knowledge, or provide an overview, it must be remembered that we are integrating a range of studies that are spatially and temporally well separated and such integrations may be of limited use if there is a large amount of variation on smaller spatial scales. This situation is particularly important where there are large areas with only one or a few quantitative descriptions of the fauna or flora (eg Western Australia: see Kirkman 1984; Hatcher 1989).

Processes

Subtidal reefs are important fish habitats, and this is perhaps the most widely perceived attraction to sport divers. In dimly lit places such as caves, beneath overhangs, on vertical rock faces and the pilings of well-shaded jetties and other human-made structures, brown and green algae become rare, even the red algae being reduced in variety and abundance. The space becomes dominated by sessile animals. In addition to the fish, in both plant- and animal-dominated assemblages there are many motile, benthic animals, only a few of which have been studied (Keough & Butler 1979, Edgar 1982, Keough 1984a, Fletcher 1987, Andrew & Underwood 1989, Watts, Johnson & Black 1990, Andrew 1991, McShane & Smith 1991, Steinberg & van Altena 1992).

Ecological processes that affect these assemblages of plants and animals are mentioned briefly below, with emphasis on aspects that could be influenced by human activities.

Modularity

In common with most plants, many of the sessile animals on reefs are *modular* or *colonial*. They reproduce asexually as well as sexually, producing replicates of a basic unit, or module. Sometimes asexually-produced offspring live separately, these organisms being referred to as *clonal* (Hughes 1989). Often however, the replicated modules remain attached to each other and form a *colony*. The best-known examples are the tropical reef-building corals, but the capacity to form colonies is an important ingredient in the ecology of the rocky-reef fauna in the temperate zones. It is displayed by the dominant groups of sessile animals on temperate reefs: sponges, bryozoans, ascidians, cnidarians. Modularity or coloniality is not found in some animal groups such as molluscs or fish, which are *unitary* animals. The distinction between modular and unitary organisms makes a profound difference to their ecological roles. Modular animals are often damaged rather than killed by predators, wave action or competitors; and mortality from various risks is often much less in large than in small colonies — so after surviving a risky youth, old colonies may live for a long time. Data on the population dynamics of such organisms are scanty, despite some recent studies (Davis 1987a, 1987b, Davis & Butler 1989, Klemke & Keough 1991, Stocker 1991, Stocker & Underwood 1991) partly because most ecological research projects are too short-term. Indeed, this is true even for some unitary marine organisms (eg Butler 1987). The lack of such data inhibits the assessment of the state of, and possible changes in, ecosystems.

Dispersal and recruitment

No animal or plant lives forever and no site remains habitable forever: offspring of some kind move away from the parent and become established in a new site, later becoming a 'recruit' to the population. The distances moved however, and the numbers lost during this dispersal, differ greatly between kinds of organisms. Some molluscs, for example, have larvae capable of travelling huge distances across oceans, though the frequency with which they achieve this is unknown. At the other extreme some colonial animals are known as poor colonisers of newly-created space, having few and short-lived larvae. The dispersal actually achieved by a given kind of animal depends on the type of propagule (crawling, swimming, etc), the time it spends as a larva and the interaction between its behaviour and the movement of waves and currents. This means that its study needs to be pursued in several ways and involves collaboration between biologists and physical oceanographers. There is much to be learned about dispersal of different kinds of organisms and it is an active research area in Australia. One clear fact is that recruitment of many species is highly variable (Keough 1983, Caffey 1985, Butler 1986). In the context of this report, the importance of both dispersal and the variability of recruitment is that they influence the dynamics of assemblages. They determine whether and how a given reef will be reoccupied by a particular species after some local catastrophe (one example of which is localised overfishing; eg Prince 1992, Prince & Shepherd 1992) and may determine the normal dynamics of populations and assemblages in subtle ways (Butler & Chesson 1990).

Genetic structure

The genetics of marine populations of both plants and animals are poorly known and potentially both very interesting and important. Because many of the organisms are clonal and modular, the structure of populations can be very different from that of typical, sexually reproducing unitary animals on land. There may be a great deal of local adaptation, such that organisms of the same species from different reefs are genetically different, differ somewhat in their ecology and could not easily be transplanted to supplement damaged areas. There is evidence of such possibilities (Davis 1989, Ayre 1990) but much more work is needed. The work may be important for plans for reseeding depleted stocks of commercial species.

Competition

Amongst sessile organisms competition for space is an important process in determining the composition and dynamics of assemblages. It has been studied (Russ 1980, Kay & Keough 1981, Russ 1982, Kay & Butler 1983, Keough 1984a, 1984b) and occurs by various mechanisms with varying consequences. It commonly interacts with processes such as predation and water movement. Thus for example, the interaction between competitive properties of different animal groups and their capacities for larval dispersal appeared to explain the relationship between assemblage composition and patch size in the studies of Kay and Keough (1981) and Keough (1984a, 1984b), competitive abilities interacted with the presence or absence of predatory fish to influence the composition of assemblages studied by Russ (1980) and Keough (1984a, 1984b). Competitive effects have often been implicated as major effects on canopy and subcanopy algae, acting on established plants or newly settled individuals (eg Reed 1990a, 1990b). Established plants may also inhibit further colonisation (Kennelly 1987c). Competitive effects such as these may strongly influence the ability of an assemblage to recover from a disturbance (Dayton et al. 1992). Again, most of the experimental studies of competition between macroalgae are derived from the northern hemisphere or New Zealand, with a handful of data from Australia.

Mobile animals may also compete. There are few studies from temperate subtidal environments in southern Australia, but a volume of work from kelp-dominated habitats in other parts of the world highlight interactions between vertebrates and molluscs (eg Schmitt & Holbrook 1990b, Holbrook & Schmitt 1992 and references therein).

Competition — and hence the structure of assemblages that are valued by divers — could thus be changed by human activities that either alter physical disturbance rates or fish densities.

Predation

The effects of predators on sessile organisms have been extensively studied in the intertidal zone. Similar approaches (manipulative experiments, excluding or fencing-in predators) have been used subtidally (Kennelly 1983, 1991). They showed some effects of predators; though the experiments have usually been small or sporadic. From observations, we know that smaller and more cryptic predators do exist in such assemblages but there are no studies of their effects (an example is nudibranchs feeding on sponges which are dominant competitors for space).

'Predators' — herbivores — on algae are potentially important in structuring plant assemblages. Three major groups of herbivores have been studied in central New South Wales habitats — sea urchins, molluscs and fish. The most important of these seem to be urchins, which are capable of causing a change from a kelp canopy to barren areas (Fletcher 1987, Jones & Andrew 1990). Herbivorous molluscs such as *Cellana tramoserica*, are more common in barren areas and presumably take advantage of the changes induced by the urchins. Most of the herbivorous fish, especially those of the genus *Odax*, forage within kelp areas rather than over the barrens. While these fish can have localised effects on the abundance of canopy-forming plants, Jones & Andrew suggested that their effects are more localised and ephemeral than are those caused by urchins.

In these habitats anything that affects the population structure of urchins, including natural (Andrew 1991) or anthropogenic events, may have profound effects on the canopy-forming species, and ultimately on the rest of the assemblage. Similarly, understanding the processes that determine the distribution of urchins may help to understand the large-scale patterns. We must emphasise, again, that these conclusions apply only to the New South Wales coast. We know of no similar processes along the southern coastline.

Wave action

Observations indicate that wave action is highly important in determining the kind of community that occurs at a site. Some of the reasons are understood — for example, different sessile suspension feeders feed in different ways and are favoured by different rates of water movement; certain organisms are more robustly constructed than are others and so can withstand wave damage. There is, however, a dearth of experimental field work placing these pieces of knowledge in context and enabling assessment of the importance of water movement relative to other factors. Certain studies have shown that 'disturbance' is significant. For example, when Kay and Butler (1983) monitored a sessile assemblage on pier pilings for a long period, they found that nearly half of the space on the pilings was occupied by different species after three months, and that some of this turnover was due to colonies being stripped from the piling surface by wave action during storms. On the other hand however, it is argued (Sebens 1985) that such findings are typical of studies on pilings and that on rocky substrata the only organisms that persist are robust enough and firmly enough attached to be little affected by wave damage.

Physical disturbance may also remove canopy-forming algae in a manner similar to that effected by sea urchins. In contrast to clearings made by herbivory, recolonisation or increased growth of sub-canopy algae follows a disturbance (Kennelly 1987a, 1987b). It is possible that these algae can inhibit the recovery of kelps by preventing their recruitment (Kennelly 1987c). Apart from Kennelly's studies on the New South Wales coast, there is little other information from other areas.

There needs to be more empirical study of wave action effects. In its absence, it is reasonable to predict that alterations to wave action (eg its amelioration by constructing breakwaters) will alter sessile assemblages.

Turbidity

Turbidity of coastal waters can be increased by human activities on land. Turbidity reduces light and thus obviously favours shade-tolerant plants and non-phototrophic organisms. It may also cause subtle changes to the interactions between organisms, because different kinds of suspension-feeding animals have different feeding mechanisms, some more susceptible to clogging by sediment than others. There are however, no Australian studies experimentally distinguishing the effect of this factor from other factors commonly associated with it (increases in nutrients, presence of toxic pollutants).

Light

Solar radiation is absorbed much more strongly in water than in air. Hence the intensity of radiation decreases very rapidly with increasing depth in the sea and the rate of reduction differs at different wavelengths; thus, the attenuation is least at a wavelength of about 465 nanometres (nm) — blue — and increases rapidly in both the ultraviolet and infrared directions. Depending on such factors as turbidity, the details of the rate of attenuation differ between different water bodies. But plants growing at increasing depths or on vertical or undercut surfaces, must be adapted not only to limited light but also to a different spectral composition: ie generally speaking algae are progressively eliminated in the order green, brown, red as depth increases. For example, although we mentioned earlier that the south-eastern South Australian coast is characterised by kelp beds (brown algae), Shepherd (1981) studied a community of red algae at 15 m depth off that coast. The best study of the effect of variation in light is Kennelly's (1989) in which he investigated combined effects of sand scour and shading in understorey algae. Human activities that alter light, predominantly by altering turbidity, will have profound effects on plant distributions and hence on other organisms. Because some plant assemblages grow only in shallower waters, and shallow rocky reefs occupy only a small proportion of Australia's coastal waters, human activities could potentially reduce the amount of his particular habitat.

Oceanographic features

Biological communities are strongly influenced by oceanographic features such as upwellings and fronts (review: Kingsford 1990). Knowledge of these is improving in some parts of Australia. Oceanographic features influence the assemblages on rocky reefs by altering the physico-chemical conditions (eg by raising cold, nutrient-rich water that favours different plant communities, as in the south-east of South Australia: Womersley 1984) and by influencing the dispersal of organisms. For management purposes, there is a great need for a better understanding of the effect of oceanographic features on dispersal. For example, there are fairly stable oceanographic fronts for part of the year in the South Australian gulfs and Investigator Strait which are likely to influence the dispersal of animal larvae. However, those influences have not yet been studied.

Linkages

There are interactions between communities on rocky reefs and those of adjacent habits. Fish sheltering on reefs may feed over neighbouring seagrass beds. The seagrass, via its detritus, may be providing much of the food source for the reef organisms. Rock lobsters in New Zealand have been observed to move hundreds of metres away from shelter on feeding excursions (McKoy 1983). In some areas, rock lobsters migrate along the seafloor, so that some reefs may be 'sources' of these animals and others 'sinks'. Although the potential connections between habitats have been noted and others can easily be postulated, they have not been studied experimentally or quantitatively. We cannot therefore, predict the effects of altering those relationships (for example, the effects on neighbouring seagrass or sandy-bottom areas of creating an artificial reef have not been documented, even though a number of such reefs have been created).

Substratum characteristics

The spatial patchiness of sublittoral reef assemblages has been noted above. The size and spatial distribution of patches of substratum is a potentially important factor in determining the sort of assemblage that will persist on it (Kay & Keough 1981, Keough 1981, Keough & Butler 1983, Keough 1984a,1984b, Kennelly & Underwood 1992, Underwood, Kingsford & Andrew 1991). Aside from a long-standing interest in fouling communities on artificial substrata, there have been few studies of the influence of substratum type — eg different rock types (but see Caffey 1982).

Possible and known effects of human activities

Although they are subject to many of the same anthropogenic activities as most other marine habitats, we generally know much less about the effects of those activities on subtidal reefs (see eg Colman et al. 1991 for Victorian examples). Our information is largely confined to a few case studies, generally from near major cities. The general shortage of studies of processes, discussed above, makes it difficult to make reasonable predictions about the likely effects of some human activities.

The most serious potential effects are those on habitat forming species, predominantly large algae. Loss of these species is likely to have dramatic effects on species that depend on the structure provided by these habitat-formers. An accurate prediction of the effects of their loss depends on an understanding of the relationship between habitat-formers and other species. In most cases, it is not known whether any present associations represent facultative or obligate relationships, and only careful experimentation (eg Carr 1989) can distinguish between these alternatives.

Most of the meagre data base concerns changes in adult populations, although some recent literature has focused on the vulnerability of propagules to various toxicants (eg Raimondi & Schmitt 1992, Kingsford & Gray in press). These effects are potentially important for non-dispersing species, including commercial ones such as abalone.

Point discharges

Point source discharges that are likely to influence subtidal reefs include sewage outfalls, thermal effluent and pulp mills. Their effects are likely to vary with the local hydrodynamics: on open coast reefs, toxicants may be rapidly dispersed and diluted, while in lower-flow environments, it may be possible for these discharges to be retained locally at high levels. As with most such discharges, mainly only localised high levels are noticed. Furthermore, different geographic areas may have very different hydrodynamic regimes. In order to understand the fate of toxicants or the scale on which biological changes may occur, it is often necessary to construct numerical models of circulation. Numerical models for different coastal regions may exhibit very different behaviour (Keough & Black in press). Hence construction of a model for a new locality may require the collection of considerable site-specific data as a general model cannot well be applied. This requirement greatly limits our ability to make general statements about the impact of point discharges.

Of this general class of activities, it has been claimed that sewage outfalls to reduce the local abundance of canopy-forming brown algae in Victoria (references in Colman et al. 1991). Some overseas literature has reported similar dramatic effects, although the effects usually have been the result of sustained input of barely untreated effluent (National Research Council 1990). The effects include loss of canopy forming algae and, in Kaneohe Bay, Hawaii, loss of corals and benthic invertebrates and their replacement by algae and sponges.

Thermal effluent, combined with increased turbidity, has also reduced the abundance of habitat-forming kelps in California (Murdoch, Fay & Mechalas 1989).

Effects of other activities include accumulations of toxicants in the tissues of organisms such as bivalve molluscs (Viarengo & Canesi 1991). These effects have rarely been linked to population changes and we cannot at this stage predict their consequences. It is generally difficult to extrapolate from local effects to more widespread population changes, because of the wide dispersal of propagules of many marine organisms, and the difficulty in predicting the amount of larval exchange between populations (Keough & Black in press).

Non-point discharges

Effects of more diffuse discharges are less clear. The now widespread reports of the effects of tributyltin (TBT) on molluscs (Spence et al. 1990) suggest that some population declines are possible. No effects have been reported on other benthic animals and we know of no well-documented reports of other effects from diffuse sources.

Sedimentation

There are anecdotal reports of the damaging effects of sedimentation, but no convincing, published local data.

Collection

Changes in abundance of commercially exploited species are dealt with elsewhere in this volume, and we will not cover them here. Similarly, collection of animals and plants from intertidal rocky shores is the subject of another review (Fairweather & Quinn, this volume). Anecdotal reports highlight the removal of animals as a part of various recreational activities. Line- and spear-fishing, and collection of molluscs and crustaceans for food all potentially affect local populations. The effects of these processes is generally not known. In southern Australia, M. Turner and M. Norman (pers. comm.) made one of the few attempts to determine whether the level of recreational activity by divers affected populations of reef fishes. They found suggestive evidence for blue-throated wrasse, but generally concluded that the level of variability precluded detection of any but very large effects (see also Keough & King 1991). Schaap and Green (1988) had similar problems with variability in their comparison of reefs under different fishing pressure in Tasmania.

Other organisms including echinoderms, molluscs, crustaceans, gorgonian corals and fish, are collected for aquaria or souvenirs. Although we have observed such collection frequently, there are no quantitative data about the impact of these activities.

Introductions

Introduced species are a major part of many assemblages on subtidal hard substrata. They are especially common on artificial surfaces in harbours, and prominent taxa include barnacles, ascidians, bryozoans, and polychaetes (eg Russ 1977). In more open coast environments these species are less abundant and have not often been reported as major components of the biota (but see Sanderson & Barrett 1989, Sanderson 1990). There are some instances of species that are presumed to have been introduced via ballast water, and which may have important effects (Jones 1991). In most cases, however, the impacts of these introduced species are unknown and the environments around harbours are subject to a wide range of other human activities. It would be very difficult to isolate any effect of introduced species *per se*. There may also be problems in even identifying introduced species in many taxa, because the endemic species are so poorly known.

Summary

- There exist few detailed quantitative descriptions of the flora and fauna of temperate subtidal reefs, particularly studies including substantial temporal and/or spatial components.
- Those few studies are concentrated in very few geographic regions.
- Descriptions are hindered by poor systematic knowledge of many taxa and/or the difficulty of identifying benthic invertebrates and many algae in the field.
- The results of descriptive studies suggest that broad habitat classifications are not especially useful, because they obscure considerable smaller-scale variation.
- Studies identifying processes which produced observed patterns are the most useful for understanding the patterns or for predicting the impact of novel stresses on these assemblages.
- There are few such studies, and they took place on natural rocky reefs near Sydney and on artificial substrata in Gulf St Vincent in South Australia (and to a lesser extent, Port Phillip Bay, Victoria).
- While some predictions can be made about the effects of some human activities on subtidal reef assemblages, the predictions should be viewed with extreme caution given the limited basis for their construction.
- The prediction of change, and identification of appropriate management activities, requires much more detailed biological information than is currently available; and a priority should be the identification of the process that most influence the structure of assemblages.

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The status of Australian estuaries and enclosed marine waters

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Introduction

Historically, Australian estuaries have been the major points of human settlement. With European settlement, they formed a focal point for ocean and river shipping which was soon followed by storage, processing and supply facilities. Adjacent lowlands proved suitable for early agricultural activities while the estuaries themselves were used as efficient waste disposal systems. Ironically, the natural productivity of these estuaries ensured the supplies of fish and shellfish to sustain the very settlements that now threaten that productivity. With increasing populations along Australia's coastline and with a corresponding increase in recreational usage and urban development, all coastal resources are coming under more pressures (Yapp 1986). This pressure however, is particularly evident in estuaries near large urban centres.

Many studies have emphasised the significance of estuaries to the commercial and recreational fishing industries (see review by Hutchings & Saenger 1987, Saenger 1991). Not only are estuaries popular fishing grounds (PA Management Consultants 1985, Neumann & Hundloe 1986) but their seagrass, mangrove, saltmarsh, mud flat and sand bar habitats provide productive feeding, spawning and nursery grounds necessary for the continued viability of fish and invertebrate populations. There is a distinct need for the adoption of sensible management strategies and for the identification of research priorities. These steps will ensure that the economic value of estuaries can be utilised while they maintain conservation values not only for fish but for all plants and animals dependent on the estuarine environment.

Before sound decisions can be made in relation to estuarine management and research, it is necessary to assess the resource and, using a consistent approach, review relevant available knowledge (United States Department of the Interior Fish and Wildlife Service 1970, Field, Alexander & Broutman 1988). Such an estuarine inventory has recently been completed for Australia (Bucher & Saenger 1989, 1991) with the support of the Australian National Parks and Wildlife Service and the Australian Recreational and Sport Fishing Confederation. The following account draws mainly on the information from that inventory which contains records for 783 estuaries and enclosed marine waters around Australia as shown in Table 1. For that inventory, estuaries were defined as semi-enclosed bodies of water and their adjacent wetlands that have input from both marine tidal inundation and terrestrial runoff. Limitations to this applied definition are that (1) the permanent water body must be wide enough to be represented on a 1:100 000 topographic map by double lines for at least 1 km upstream of its seaward entrance, and (2) the estuary must represent the discharge point of a catchment of at least 15 km^2. All other terms and categories as used below are also defined in Bucher and Saenger's inventory (1989).

Usage

As noted above, Australian estuaries are being increasingly used for a range of activities even though there is high public awareness of the importance estuarine wetlands hold for conservation. Attitudes to estuarine wetlands were surveyed in northern New South Wales during 1986 (unpublished survey by the Centre for Coastal Management based on 423 random interviews). The results showed that over 80% of respondents thought that coastal wetlands were important, many believing that the prime value for the wetlands was in providing habitats for fish and birds. Nearly 40% of the respondents regularly used wetlands for a variety of purposes (fishing, boating and aesthetic appreciation) and nearly 70% felt these systems were threatened by (in order) urban development, pollution, drainage and mining.

Table 1: The distribution of estuaries and enclosed marine waters by State, biogeographic zone and climate.
Biogeographic zones are based on the scheme adopted by the Australian Committee of the International Union for Conservation of Nature and Natural Resources for describing marine and estuarine protected areas.

State	No. of estuaries
Queensland	307
New South Wales	81
Victoria	35
Tasmania	63
South Australia	15
Western Australia	145
Northern Territory	137
TOTAL	783
Biogeographic zone	
Gulf of Carpentaria	136
North-east coast	192
Central east coast	34
Lower east coast	66
Bass Strait	57
Tasmanian coast	41
South gulf coast	9
Great Australian Bight	6
South-west coast	13
Lower west coast	6
Central west coast	2
North-west coast	42
North coast	179
TOTAL	783
Climate	
Tropical	415
Subtropical	170
Temperate	198
TOTAL	783

All estuarine uses have some resultant impact although clearly, some are less intense than others (Table 2). With future increases in usage, it will be essential that some of these activities are regulated, while others will need to be relocated to non-estuarine areas if estuarine values are to be maintained in terms of their productivity and the services they provide (Saenger 1991).

Table 2: Estuarine uses in order of decreasing environmental impact and pollution. (Based on United States Department of Interior Fish & Wildlife Service 1970).

1. Urbanisation
2. Industry
3. Waste disposal
4. Transportation
5. Agriculture & forestry management
6. Mining
7. Power generation
8. Recreation
9. Pest control
10. Defence
11. Water supply
12. Fish & wildlife management
13. Commercial fishing
14. Research
15. Sanctuaries

Estuarine composition and status

Variability of Australian estuaries

Australian estuaries occur over a wide range of geological and climatic conditions and consequently display a great variety of form. The largest estuary has an area of 1120 km^2 (Victoria River, Northern Territory) and the smallest covers only 0.11 km^2 (Merrika River, New South Wales). Enclosed marine waters range in area from 46 km^2 for Raffles Bay, Northern Territory to Shark Bay in Western Australia which is the largest at 13 887 km^2. Associated catchment areas range from 15 km^2 (five unnamed estuaries in Queensland) to the Murray-Darling Basin in South Australia which is 1 062 530 km^2.

Table 3 shows the total areas of open water, intertidal mud/sand flats, mangrove, seagrass and saltmarsh areas in the estuaries of each State. Western Australia contains the most open water and mud/sand flat habitat (53% and 46% respectively of the national total), presumably because many areas experience large tidal ranges. Queensland and the Northern Territory support the most mangroves and saltmarsh (a combined total of 78% of Australia's mangroves and 76% of saltmarsh) while Victoria has the most extensive seagrass beds so far recorded (58% of the total, mostly recorded from Western Port and Corner Inlet).

Based on the total mangrove areas (11 617 km^2) as determined by Galloway, Story, Cooper & Yapp (1984) it is worth noting that 70.5% of Australian mangrove areas are associated with estuaries and only 29.5% occur on open shorelines. The percentage of estuarine saltmarshes is likely to be smaller than that of mangroves but no data are available.

The supply of fresh water to Australian estuaries is highly variable both in amount and regularity. Rainfall on estuarine catchments range from an arid 242 mL p.a. (northern Spencer Gulf, South Australia) to 4321 mL p.a. (Tully and Hull rivers and Maria Creek,

Table 3: Distribution of estuarine habitat types by State (km^2) [n/a: not available]

State	**Open water** (incl. subtidal seagrass beds)	**Intertidal flats** (incl. intertidal seagrass beds)	**Mangroves**	**Seagrass** (subtidal + intertidal beds)	**Saltmarsh area**	**Total estuarine**
Qld	4092.8	1574.4	3423.9	67.7	5321.9	14413.0
NSW	1323.1	n/a	106.8	152.8	57.2	1487.0
Vic	2682.5	443.8	40.5	346.4	124.9	3291.7
Tas	1824.8	274.3	n/a	n/a	37.0	2136.1
SA	759.5	218.5	111.0	n/a	83.7	1172.6
WA	17824.7	2890.7	1560.8	11.3	2965.0	25241.2
NT	5186.7	821.5	2952.4	22.5	5004.8	13965.5
Total	33694.1	6223.2	8195.3	600.6	13594.5	61707.0

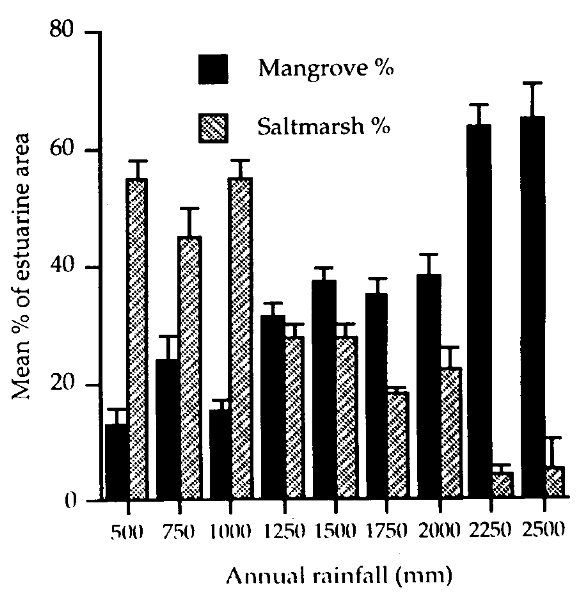

Figure 1: Mean percentage of tropical and subtropical estuaries covered by mangrove and saltmarshes with annual rainfall categories. (Error bars are Standard Errors)

Queensland) (Bureau of Meteorology 1977). Runoff coefficients estimated for each catchment cover almost the entire range of values, ie from 0.01 (Port Davis Creek, South Australia) to 0.96 (Little Henty River, Tasmania). The supply of saline water is equally variable. Around Australia the tidal range varies from 1.0 m at Little River, Victoria, to 12.2 m at eight estuaries in Western Australia (Royal Australian Navy Hydrographic Service 1987). The nature of fresh and salt water supply will affect the type and extent of wetlands associated with an estuary. While the tidal range of subtropical and tropical estuaries correlates positively with the mean wetland area per estuary (ie 571 out of 783), the relative proportions of mangroves and saltmarsh within estuaries show significant relationships with average annual rainfall (positive for mangroves and negative for saltmarshes) (Bucher & Saenger 1994) (Figure 1).

Other characteristics such as estuarine configuration and geology and low temperatures (particularly in the south) add further to the overall estuarine variability.

Human influence on Australian estuaries

Human activity has influenced some estuaries substantially but many remain almost unaffected. Clearance of estuarine catchments is widespread, particularly in central and southern coastal Queensland, New South Wales, Victoria and South Australia (see Table 4). However, more information on catchment clearance is needed especially for Queensland south of the Daintree River, Tasmania and south-western Australia.

Development within a catchment can have flow-on effects on water quality, either by discharge of chemical pollutants, by changes in the silt loads, or temperature of the water. Although adequate information for assessing water quality is available for few estuaries, the frequency of water quality categories in Australian estuaries, based on our current knowledge, is given in Table 5. Even so, these categories are a crude system of ranking as water quality consists of a number of parameters that cannot be combined in a single measure. This means that in most cases a subjective assessment was made and that no quantitative limits can be ascribed to each category. Some source of pollution has been identified in most of the estuaries listed in Table 5 under 'Insufficient information' and it is likely that further information about these estuaries

Table 4: Numbers of estuaries in categories of catchment clearance.

	Percentage of catchment cleared of natural vegetation					
State	**<25%**	**25–50%**	**50–75%**	**>75%**	**Insufficient data**	**Total**
Qld	170	17	4	3	113	307
NSW	20	31	25	5	-	81
Vic	8	6	6	15	-	35
Tas	17	-	-	-	46	63
SA	-	1	1	12	1	15
WA	125	2	3	1	14	145
NT	136	1	-	-	-	137
Total	476	58	39	36	174	783

will place them in the 'Fair' or 'Poor' categories rather than the 'Excellent' one.

To determine appropriate management, research and conservation priorities, some method of ranking estuaries in terms of their value is required. The values of estuaries are many but can be summarised under three categories: fishery, conservation and amenities. Briefly, fishery value is determined by the use of an estuary as a fishing ground or its value as fish habitat. The conservation value of an estuary may involve the presence of rare or endangered species or communities, use as a scientific study (or reference) area or educational resource, having representative examples of communities, and so on. Amenities value refers to the provision of shore access, boat ramps, jetties and port facilities. The national distribution of estuaries having fishery, conservation and amenities value are given in Table 6. Many of the value ratings are based on very inadequate information and further input may require some revision. However, estuaries are most likely to be under-rated due to lack of information than over-rated.

Real and potential threats to the fishery or conservation values of each estuary have also been identified (Table 7). Threats include such measures as pollution, catchment clearance, wetland reclamation, engineering works (eg dredging, training walls, marinas, wharves, flood mitigation, dams), boating activity, over-fishing, weed infestations and litter. Aside from boat channel silting and macroalgal blooms (which often pose a threat to fisheries and conservation values) no threats to amenities value have been identified.

Priority areas for protection would be those with high or moderate value that are, or may be, threatened. Table 7 shows the number of threatened estuaries with high conservation or fisheries value.

The overall conditions of Australian estuaries can be defined by several characteristics including the percentage of estuaries with (1) a low degree of catchment clearance, (2) high water quality, (3) high fisheries value, (4) high conservation value, and (5) a low threat to

Table 5: Occurrence of water quality classes in Australian estuaries.

	Water quality classes				
State	**Excellent**	**Fair**	**Poor**	**Insufficient data**	**Total**
Queensland	149	6	1	151	307
New South Wales	-	5	9	67	81
Victoria	5	2	2	26	35
Tasmania	16	1	1	45	63
South Australia	-	-	-	15	15
Western Australia	121	4	2	18	145
Northern Territory	133	-	-	3	137
Total	424	18	15	326	783

Table 6: Occurrence of fishery, conservation and amenities values.

	Value			
State	**High**	**Moderate**	**Low**	**Total**
Fishery value				
Qld	58	169	80	307
NSW	20	35	26	81
Vic	5	14	16	35
Tas	0	33	30	63
SA	1	12	2	15
WA	11	62	72	145
NT	24	87	26	137
Total	119	412	252	783
Conservation value				
Qld	67	164	76	307
NSW	13	41	27	81
Vic	17	11	7	35
Tas	4	21	38	63
SA	3	10	2	15
WA	11	58	76	145
NT	31	85	21	137
Total	146	390	247	783
Amenities value				
Qld	9	33	265	307
NSW	4	32	45	81
Vic	3	14	18	35
Tas	3	0	60	63
SA	3	3	9	15
WA	5	6	134	145
NT	2	5	130	137
Total	29	93	661	783

Table 7: Distribution of threats to fishery and conservation values.

	Threat to fishery value (no. of threatened estuaries with high fishery value)			
State	**Real**	**Potential**	**None identified**	**Total**
Qld	7(3)	79(29)	221	307
NSW	17(8)	21 (7)	43	81
Vic	8(4)	24 (1)	3	35
Tas	3(0)	5 (0)	55	63
SA	1(0)	5 (1)	9	15
WA	4(2)	9 (1)	132	145
NT	1(0)	4 (1)	132	137
Total	41	147	595	783

	Threat to conservation value (no. of threatened estuaries with high conservation value)			
State	**Real**	**Potential**	**None identified**	**Total**
Qld	7(5)	68(23)	232	307
NSW	17(8)	21 (5)	43	81
Vic	8(5)	24 (9)	3	35
Tas	2(1)	10 (1)	51	63
SA	1(1)	5 (2)	9	15
WA	3(0)	7 (1)	135	145
NT	1(0)	4 (3)	132	137
Total	39	139	605	783

Table 8: National overview of estuarine database.

	Percentage of estuaries having..............					
State	**Low degree of catchment clearance**	**High water quality**	**High fishery value**	**High conservation value**	**Low threat to conservation value**	**Reasonable literature coverage**
Qld	55.4	48.5	18.9	21.8	2.3	11.1
NSW	24.7	0.0	24.7	16.0	21.0	100.0
Vic	22.9	14.3	14.3	48.6	22.9	31.4
Tas	27.0	25.4	0.0	6.3	4.8	6.3
SA	0.0	0.0	6.7	20.0	6.7	13.3
WA	86.2	83.4	7.6	7.6	2.8	7.6
NT	99.3	97.1	17.5	22.6	0.7	5.1
Overall	60.8	54.2	15.2	18.6	5.2	19.2

conservation values. The percentage of estuaries falling into each of these categories for each State and Territory and for Australia as a whole is provided in Table 8. The last column in Table 8 shows the extent of scientific data available on each estuary and hence the possible accuracy of the assessments.

Most estuaries in northern Queensland, Western Australia and the Northern Territory have a lot of uncleared catchment area and high to excellent water quality characteristics. On the other hand, estuaries in southern Queensland, New South Wales and Victoria have lower water quality, greater catchment clearance and a high percentage of estuaries faced with real threats to their conservation values. Equally, the most extensive (but possibly still inadequate) literature available is on New South Wales and Victorian estuaries while the least studied are the estuaries of north Queensland, Tasmania, Western Australia and the Northern Territory.

Management

At present 62% of Australia's estuaries carry at least some form of administrative classification that restricts their use and 28.4% can be classified as having some formal conservation status (eg marine park, national park, game reserve, flora/fauna reserve). However, in only a few cases does this status apply to a significant proportion of the estuary and it rarely covers the entire estuary. More importantly, numerous classifications are used throughout Australia and there is little uniformity in nomenclature. As with the nomenclature, the reservation of estuaries in whole or in part, appears to reflect local interests and cannot be said to constitute a national system of representative estuarine reserves. Similarly, the level of surveillance and enforcement of existing estuarine reserves varies dramatically.

The difficulties associated with jurisdictional and administrative fragmentation of estuaries are similar to those of other tidal areas (Saenger 1987). However, given the detailed estuarine database (Bucher & Saenger 1991) available the start on a national network of representative and significant sites to cover the physical, hydrological and biotic variation in estuarine types merely requires coordination between Federal and State governments.

Major gaps in knowledge

Major gaps in our knowledge of estuarine habitats appear to be in seagrass distribution (except in Queensland, New South Wales and Victoria) and intertidal flats. Similarly, more information on catchment clearance is needed especially for Queensland south of the Daintree River, Tasmania and south-western Australia. Water quality data for most Australian estuaries are inadequate and need to be collected systematically as part of a national monitoring program. Inventories of estuarine biotas are adequate for southern Queensland, New South Wales, Victoria, South Australia and south-western Western Australia while the least studied are the estuaries of north Queensland, Tasmania, north-western Western Australia and the Northern Territory. However, with a few exceptions (eg Hodgkin 1978, Saenger, Stephenson & Moverley 1980, Staples 1980, Moverley, Saenger & Curtis 1986) the biotic dynamism (natural fluctuations, resilience, etc) of Australian estuaries is virtually unknown.

Conclusions

1. In general, Australian estuaries are moderately undisturbed, especially in the northern half of the country. Most estuaries in northern and western Australia have been largely unaffected by human activity but those same estuaries are poorly protected and are still very poorly known. In contrast, estuaries in eastern and southern Australia generally have lower water quality and greater catchment clearance, and a high percentage face real threats to their conservation values. The estuaries in New South Wales and Victoria have been studied most extensively.

2. Nationally, it is considered that 18.6% and 15.2% of estuaries respectively possess high conservation and fishery values which in turn, suggest that — given clearly defined management priorities — most representative and important features can be adequately conserved in a national system of estuarine reserves.

3. Like wetlands in general, establishment of a national network of estuarine reserves will require national coordination between State and Federal governments if management needs and conservation strategies are to be realised.

4. More research is needed on detailed individual catchment characteristics and water quality (especially where potential pollution sources have been identified) and there needs to be an on-going assessment of

fisheries and conservation values. More detailed mapping is needed, especially for seagrass beds and intertidal mud banks. Estuarine biotas and their dynamics generally require additional studies.

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Land-sea interactions and oceanographic processes affecting the nutrient dynamics and productivity of Australian marine ecosystems

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Introduction

Australia is an island continent (Figure 1). The coastal, continental shelf and oceanic ecosystems within Australian territorial waters and the Australian Exclusive Economic Zone span a 33-degree latitudinal range between the tropics (Cape York, c. 10°S) and temperate zone (Hobart, 43°S); a 58-degree span if Antarctica and the subantarctic islands are included. This large latitudinal range encompasses a diverse range of pelagic and coastal ecosystems. The biological productivity of these ecosystems is dependent upon nutrients delivered, removed and recycled by geochemical and biological processes which are, in turn, coupled to the physical oceanographic environment. This report briefly describes key processes relevant to understanding oceanographic controls on nutrient inputs, nutrient cycling and the productivity of Australian seas.

The mixing of seawater is determined by its density which is related to its temperature and salinity characteristics. In the absence of vertical mixing, water masses in the ocean form a layered structure. These layers are generally characterised by warm or low-salinity water of low density and usually low nutrient content at the surface overlying progressively cooler and/or saltier layers of increasing density. Away from freshwater sources, the specific gravity (density) of seawater falls within a fairly narrow range (c. 1.020–1.028 g/cm^3). Within this narrow range, small changes in seawater density control the amount and extent of vertical mixing. Figure 2 shows representative profiles of salinity, temperature and density measured in the upper 300 m of the Coral Sea (19°S) — where clear vertical changes in temperature and density are evident — and the Southern Ocean (54°S) — where the intensity of vertical mixing is much greater and there is little variation in density within the upper water column. Horizontal turbulent mixing is driven by wind stress and current shear as water masses move past each other. The extent of vertical turbulent mixing of water is governed by the magnitude of the vertical density gradient and fluxes of thermal and mechanical energy relative to the gravitational potential energy stored in the vertical density gradient. On a regional scale, rates of horizontal turbulent mixing are approximately one million (10^6) times greater than vertical turbulent mixing rates.

Much of Australia's continental shelf and coastal marine environment lies within the tropics. At low latitudes, intense, year-round solar heating produces a warm surface layer of sufficiently low density that turbulent mixing of water and nutrients across the *thermocline* is strongly inhibited. Below the thermocline, dissolved nutrient concentrations increase with depth and decreasing temperature. The productivity of surface waters is closely related to the rate at which water and its associated nutrients below the thermocline are transported or mixed into the *euphotic zone* (Eppley & Peterson 1979). In open ocean and continental shelf systems, regional wind stress (eg Clementson et al. 1989, Harris et al. 1991) and storm events (McGowan & Hayward 1978) directly affect vertical mixing of nutrients into the surface layer, regulating the level of primary production. Localised upwelling of nutrient enriched waters can occur at current divergences or in the vicinity of headlands, reefs and islands (eg Wolanski, Imberger & Heron 1984). At the same time, nutrient materials (eg nitrogen, phosphorus, silicon, iron) are continually removed from the surface layer by a rain of particulate material comprised of living plankton, detritus and faecal materials produced by pelagic grazers. Below the thermocline, organic materials are *mineralised* back into inorganic nutrients by microorganisms.

At higher latitudes, water temperature, levels of wind mixing and inputs of light necessary for photosynthesis go through pronounced seasonal cycles (Rochford 1984, Harris et al. 1987, Clementson et al. 1989, Harris et al. 1991). During

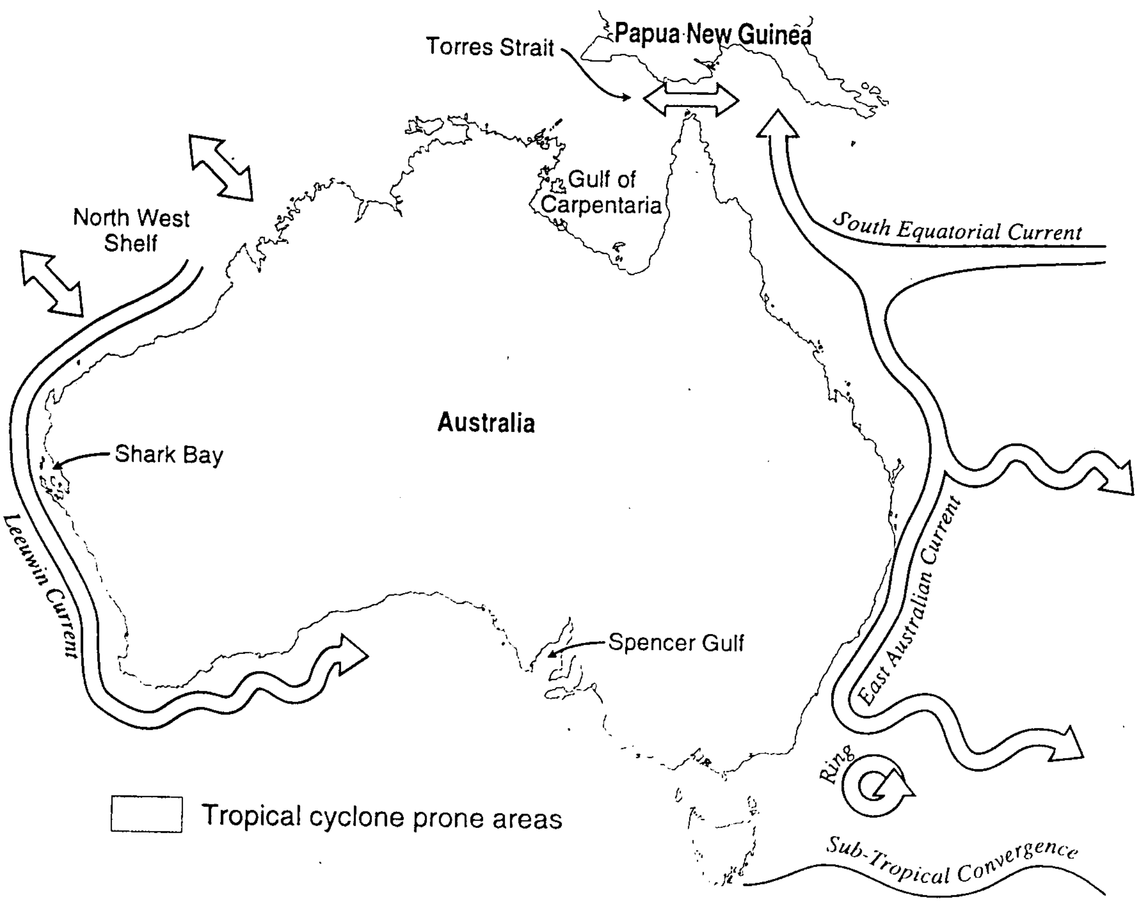

Figure 1: Regions in the Australian marine environment affected by major current systems, cyclones and oceanographic mixing regimes.

the winter, strong winds and cool surface water temperatures enhance vertical mixing processes, breaking down vertical density gradients and allowing nutrient-rich waters to mix into the surface layer. Winter production by planktonic algae is limited by seasonally low light levels and deep vertical mixing which keeps planktonic algal populations out of the euphotic zone for extended periods. In the spring, increasing solar energy inputs lead to a warming of surface waters, thus re-establishing a shallow thermocline. Phytoplankton above the thermocline remains exposed to higher light levels and, briefly, to high nutrient levels. These lead to the development of a spring bloom (eg Harris et al. 1987, Harris et al.1991). Temperature, per se, has relatively little effect on the initiation, magnitude and duration of spring blooms: the important factors are the thickness of the surface mixed layer, pre-existing nutrient concentrations and subsequent rates of nutrient input or recycling. The life cycles and population dynamics of many species inhabiting temperate and polar waters are directly or indirectly linked to the annual flowering of planktonic algae in the spring bloom (eg Thresher et al. 1989). Phytoplankton and organic matter which are not consumed during the spring bloom ultimately falls to the seabed to support benthic food chains and demersal fish stocks.

Because of the annual winter replenishment of nutrients, temperate oceanic and continental shelf regions tend to support larger seasonal blooms of algae and greater production at higher trophic levels than do (non-upwelling) tropical ocean and shelf systems. Lacking a well defined seasonal cycle, biological variability in tropical ecosystems is more closely related to disturbance events and oceanographic processes such as upwelling (Furnas & Mitchell 1986), floods (Brodie & Mitchell 1991), tidal mixing (Holloway et al. 1985) and cyclones (Furnas 1989). The overall productivity of temperate Australian continental shelf waters is restrained by the poleward transport of low-nutrient tropical waters along the continent's eastern and western margins by the East Australian Current (Nilsson & Cresswell 1980) and Leeuwin Current (Godfrey & Ridgeway 1985). There are no large seasonal blooms producing surpluses of organic matter. As a result, Australia lacks the large demersal fisheries that characterise northern hemisphere continental shelf systems.

The amount of biomass an ecosystem will support ultimately depends upon the stocks of nutrients available. There remains some debate as to whether nitrogen (N) or phosphorus (P) is the predominant limiting nutrient in marine systems (Smith 1984). Examination of nutrient stocks in many systems has generally led plankton biologists to favour N limitation, while geochemists favour P limitation as there is no significant atmospheric source of P. This is in contrast to N which can be fixed from atmospheric N_2 by specialised bacteria (Carpenter, Capone & Reuter 1992). An emerging

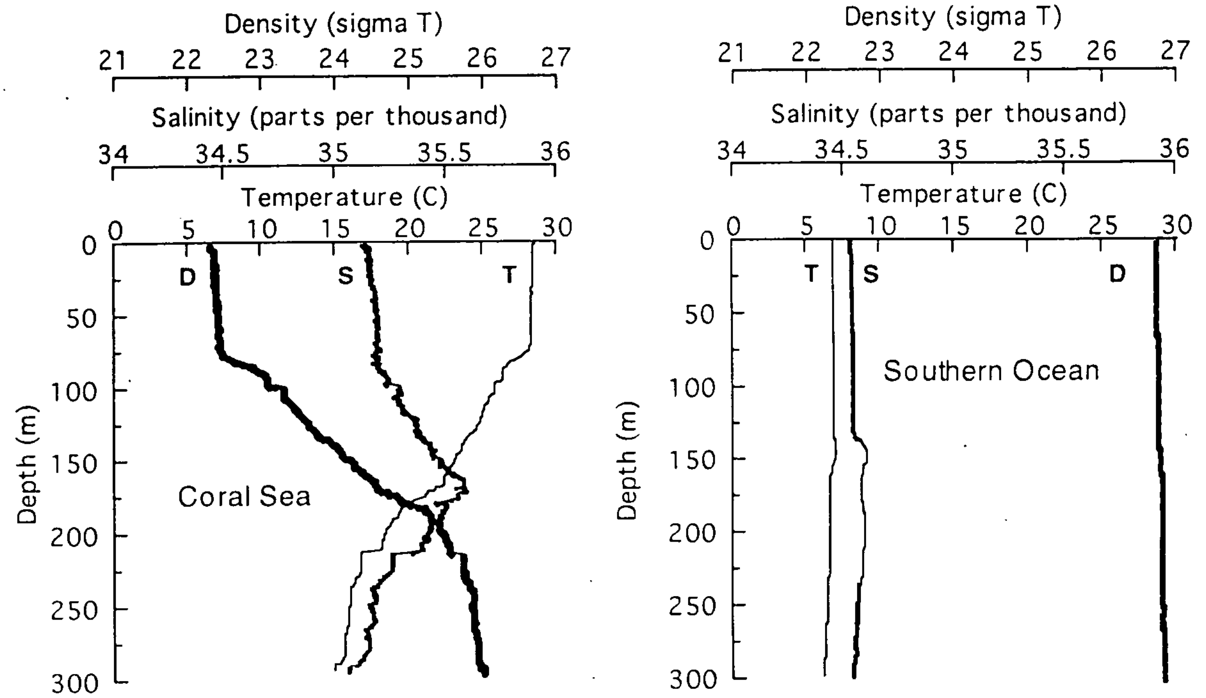

Figure 2: Representative vertical profiles of water temperature, salinity and density in the Coral Sea (19°S, left) and Southern Ocean (54°S, right).

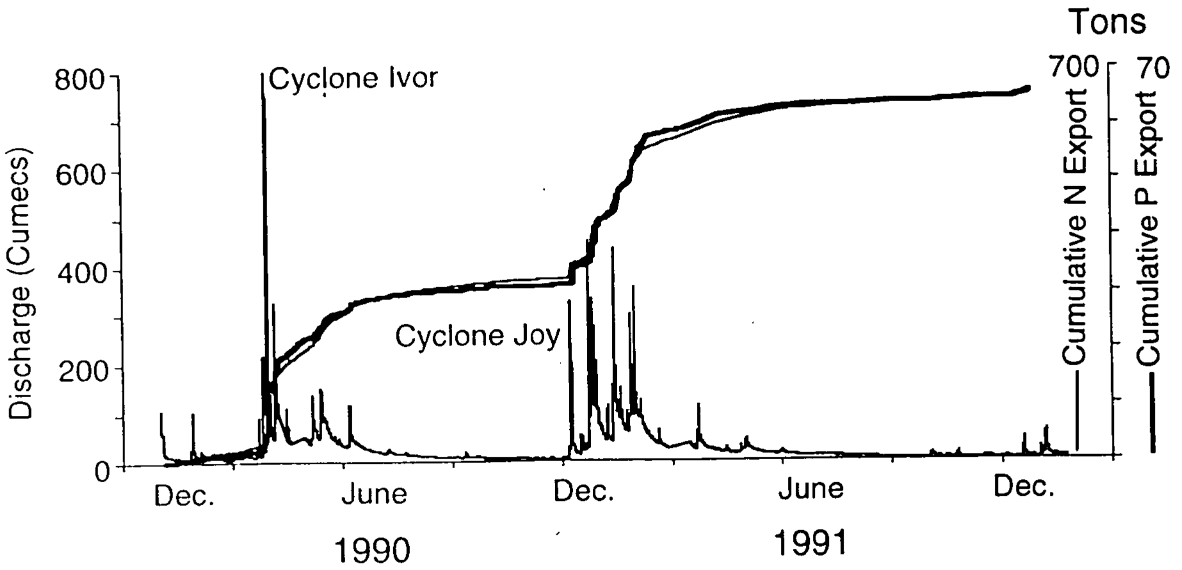

Figure 3: A two-year record of instantaneous discharge ($m^3 sec^{-1}$) in the South Johnstone River, north Queensland and the concurrent cumulative export of total water-borne nitrogen and phosphorus (tonnes).

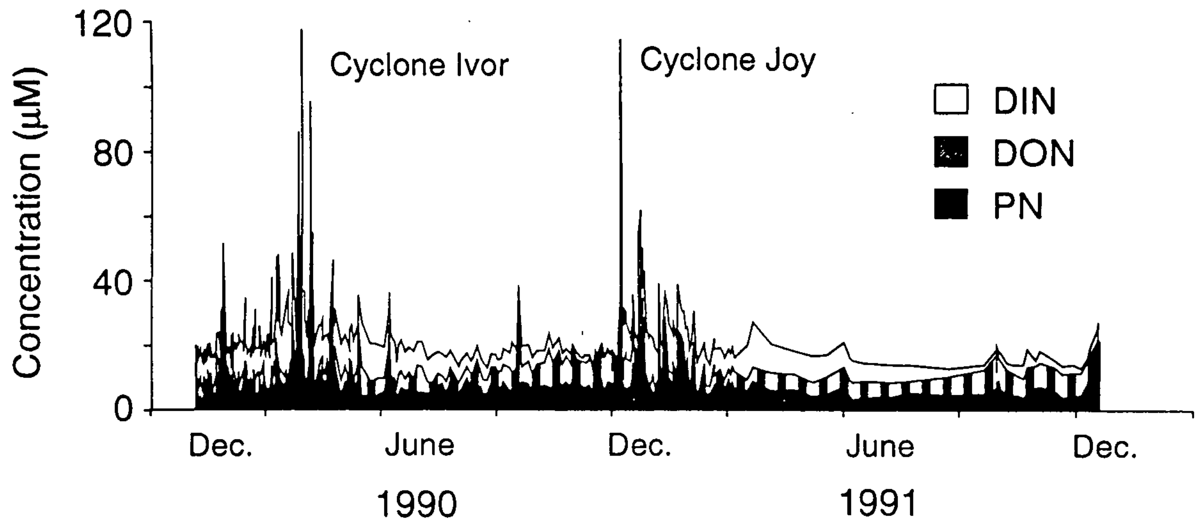

Figure 4: Short-term variability in concentrations of dissolved and particulate forms of nitrogen over a two-year period in the South Johnstone River, north Queensland. The instantaneous discharge pattern is shown in Figure 3.

view suggests that in some open ocean systems other elements such as iron (Fe) may the be limiting nutrient (Chisholm & Morel 1991), although this suggestion has never been tested in Australian waters. Even in systems with low nutrient levels, rapid rates of *in situ* N and P mineralisation (eg Harrison 1978, Smith, & Harris 1985) ensure that populations of planktonic algae and microorganisms have continual access to at least small amounts of these elements (and likely others) to sustain their growth.

River runoff and nutrient inputs

Most of the nutrients and sediments washed from the land into coastal waters are transported by rivers. Locally, submarine discharges of groundwater can also occasionally be significant (Johannes 1980). Reliable, year-round rainfall is largely restricted to narrow zones along the eastern and southern margins of the dry Australian continent. In terms of discharge, Australia has no world class rivers. With the exception of the Murray-Darling river system, Australian rivers are short. Because of the small size of most catchments, flood events are usually of short duration, often lasting only a few days (Figure 3). These high flow periods are closely coupled to local rainfall patterns. In the monsoonal climate of northern Australia, significant flow into and nutrient discharge from rivers is restricted to the summer wet season and is highly variable between and within years (Isdale 1984). Major flood events are related to the activity of monsoonal depressions or tropical cyclones. Relative to overall size of Australia's continental shelf environment, river runoff is a small and at most, a regional contributor to shelf nutrient processes and fluxes.

The high temporal variability of flow within most Australian river systems presents significant problems for sampling and the estimation of nutrient, pollutant and sediment delivery rates to coastal ecosystems. Fortunately, measurements of water levels and discharges are made in most significant river systems. Concentrations of suspended sediments and dissolved materials in river waters vary greatly in response to varying flow rates and flood events occurring in the annual cycle (Figure 4). Peaks in concentrations of suspended sediments, nutrient materials and other chemical compounds coincide with peaks in flow rates as soil and associated soluble materials are washed off the land and into the rivers. Dilution of specific nutrient or ion concentrations in river waters can occur when watershed stocks are limited or have been washed out in floods earlier in the wet season. The relative importance of dissolved and particulate-associated nutrient materials varies between individual nutrient elements, between rivers and between flow events within seasons. After prolonged dry periods, the first flood event of the season frequently contains high concentrations of nutrient materials which have built up within the watershed or are stored with sediments in upper reaches of individual streams. To resolve such variability, rivers must be sampled at frequent intervals (at least once and perhaps several times daily) during periods of high flow variability (eg Cosser 1989).

Estuaries

Estuaries are the zones where fresh and salt waters mix (Figure 5). Because nutrient delivery is spatially concentrated within the estuarine mixing zone, estuaries usually have a higher biological productivity than adjoining coastal and continental shelf waters (Nixon et al. 1986, Nixon 1988). Many regionally important fisheries are either based within estuaries or are dependent upon estuarine productivity at some time in the target species' life cycle (eg Rothlisberg, Hill & Staples 1985). As estuaries often provide convenient, sheltered locations to develop ports, virtually all of Australia's original settlements were located adjacent to estuaries. As these settlements have grown to towns and cities, the estuaries on which they were established have become the recipients of increasing volumes of nutrients and other human-derived materials and are now a focus for pollution.

Mixing between fresh and saline waters within estuaries is related to the relative magnitudes of wind stress, tidal energy and the density difference between freshwater and seawater. Where estuaries occur within enclosed coastal embayments or drowned river valleys, the freshwater floats upon a wedge of saltwater which intrudes landward along the bottom of the estuary. Current shear between the two layers and turbulence caused by winds and tides cause the layers to mix. The mixing in most cases produces a gradient of salinities at the surface which range from pure seawater outside of the estuary mouth to pure freshwater at the head of the estuary. The rapidity of the transition between fresh water and salt water depends on the freshwater discharge rate, the size and depth profile of the estuary, the tidal range and wind stress. Because river flow rates, tides and winds are variable, distributions of salinity, nutrients and plankton populations within estuaries are rarely stable. Benthic communities are located relative to longer term average distributions of salinity and tidal range and are more predictable. Under flood conditions, the discharge of freshwater may be so high that seawater cannot intrude into the estuary. On these occasions, mixing and other processes which normally occur inside the estuary take place within or on the boundaries of the flood plume on the adjacent continental shelf (eg Wolanski & van Senden 1983, Brodie & Mitchell 1991).

Because discharge from many Australian rivers is low for much of the year, the extent of vertical stratification is generally restricted in many estuaries. More commonly, tidal and wind mixing control the distribution of materials in the estuary, leading to the development of longitudinal gradients of salinity and other properties (eg Hodgkin et al. 1980) rather than layered saltwedge structures.

On arid coastlines, high evaporation rates from shallow embayments without significant freshwater inflows and restricted tidal exchange can create environments where salinities exceed those in normal seawater ('inverse estuaries'). These inverse estuaries may be permanent (eg Shark Bay: Smith & Atkinson 1983, Spencer Gulf: Nunes & Lennon 1986) or seasonal in monsoonal regions (eg Wolanski 1986). The productivity of

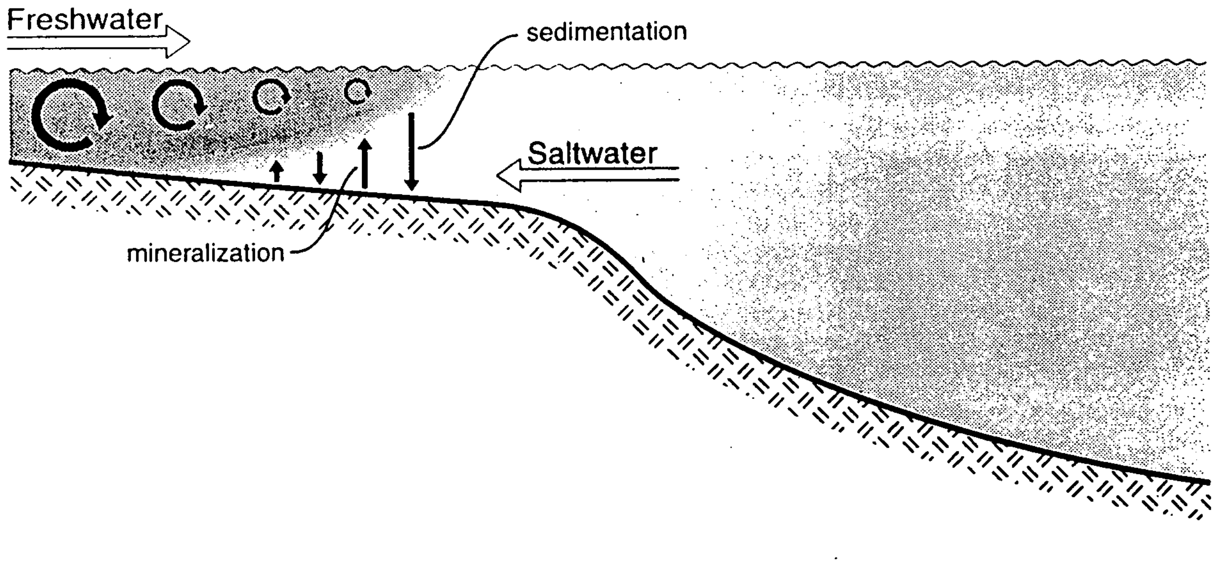

Figure 5: Time-averaged water flows, mixing patterns and major nutrient fluxes in a partially mixed salt-wedge estuarine system.

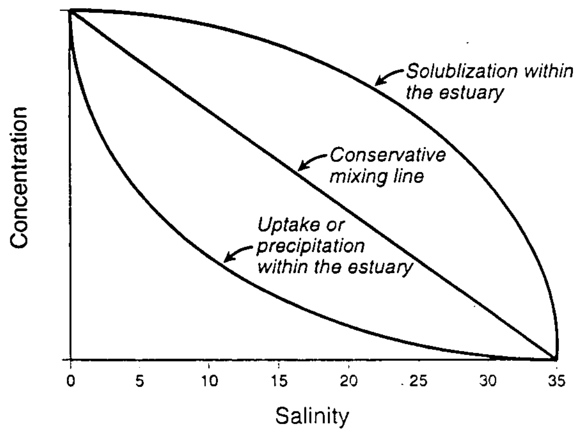

Figure 6: Schematic depiction of solute:salinity relationships in estuaries for materials not affected by estuarine processes, materials removed or solubilised within the estuary.

inverse estuaries is related to tide and wind-driven exchanges of materials with the neighbouring ocean, nitrogen fixation by benthic bacteria, sedimentation, and recycling within the ecosystem (Smith & Atkinson 1983; Smith & Veeh 1989).

Supra-tidal mud flats are found along a number of arid and dry-tropical coastlines, such as the southern Gulf of Carpentaria. These flats concentrate salt and nutrients for extended periods following tidal inundations, sheet runoff, rainfall and groundwater intrusions, then release salty, nutrient-laden water into the coastal zone following dissolution during spring tidal immersions (Ridd, Sandstrom & Wolanski 1988). The quantitative contribution of these release events to the coastal zone are not well known.

A wide variety of inter-connected geochemical and biological processes operate in the estuarine environment to alter the concentration and speciation of biologically important nutrient materials, metals and pollutants. Individual elements (eg sodium, chloride) may be unreactive within the estuary. Their concentration reflects the relative dilution of the fresh or saltwater end members (Kaul & Froelich 1984). Other materials are actively taken up or released into solution (Figure 6). The transition from freshwater (low ionic strength) to saltwater (high ionic strength) directly affects the concentration and speciation of many elements and ions carried in river waters. For example, iron, a biologically essential element common in terrestrial soils and rocks, is relatively soluble in freshwater but is highly insoluble in oxygenated seawater. As a result, most of the soluble iron transported by rivers is precipitated within or near estuaries (Boyle, Edmond & Sholkovitz 1977). In contrast, copper, which is also an essential trace element but highly toxic to marine life at only slightly higher concentrations, is less affected in the transition from fresh to seawater (Sholkovitz & Copland 1981). The solubility of copper and many other trace metals is strongly dependent upon the presence of organic matter in fresh and saltwaters to bind the metal ions and keep them in solution.

Much of the phosphorus transported by river systems is bound to soil particles (Meybeck 1982). Some, though not all of this phosphorus is solubilised to phosphate (PO_4) within the estuarine zone by equilibrium desorption processes (Fox, Sager & Wofsy 1986, Froelich 1988). Thereafter, it is available for uptake by planktonic algae.

Nitrogen undergoes a number of biologically mediated transformations in estuarine systems. Inorganic species such as ammonium (NH_4) and nitrate (NO_3) are both rapidly taken up and mineralised by plankton (McCarthy, Taylor & Taft 1977, McCarthy, Kaplan & Nevins 1984). Organic matter is actively mineralised in both the water column and benthos. Importantly, as much as half of the organic nitrogen which falls to the seabed in estuaries may be removed from the ecosystem by denitrifying bacteria (Seitzinger 1988). Rates of *nitrification* and *denitrification* are in turn directly coupled to the degree of nutrient loading (McCarthy, Kaplan & Nevins 1984, Seitzinger & Nixon 1985). All such processes need to be taken into account when estimating net fluxes of nutrients, pollutants or other compounds to coastal marine ecosystems.

The productivity of plankton communities in estuaries is dependent upon the rate of nutrient supply (usually through freshwater inputs: Nixon 1980), the amount of physical mixing within the estuary (Nixon 1988) and the residence time of water within the estuary (Ketchum 1954). Where nutrient input rates are high and water residence times are long relative to the generation times of algae, pronounced algal blooms can develop (eg the Peel-Harvey estuary in Western Australia: Hodgkin et al. 1980, Hillman, Lukatelich & McComb 1990). Residence times for water and plankton are determined by a number of factors, including rates of freshwater input, the shape of the estuary (eg long and narrow versus short and wide), the presence of adjoining wetlands (Wolanski & Ridd 1986a) and the volume of the estuary relative to the tidally exchanged volume. Where the estuary is relatively open to the adjoining shelf, wind-driven currents may be important in flushing it. In shallow estuaries and bays, exchanges of nutrients between the water column, the benthos and communities of benthic filter feeders exert a significant effect on water chemistry (Smith & Atkinson 1983) and system productivity (Nixon et al. 1986). However, for bays and continental shelf systems of even moderate water depths (c. 10 m), microbial mineralisation processes within the water column dominate short-term nutrient availability (Harrison et al. 1983, Furnas, Smayda & Deason 1986).

Once past the estuary, freshwaters and nutrient materials do not disperse haphazardly into the adjoining ocean. Buoyant plumes of river water generally flow along the coastline (eg Wolanski & van Senden 1983). As a result, terrestrial sediments, nutrient materials and contaminants are usually deposited close to the coast (eg Gagan, Sandstrom & Chivas 1987). Interactions between buoyancy-generated coastal currents, bottom friction and wind stress frequently lead to the formation of coastal boundary zones (King & Wolanski 1991). These dynamic features trap water and terrestrial materials near the coast enhancing nearshore nutrient concentrations and slowing dispersion into deeper shelf waters. In the Gulf of Carpentaria, annual prawn catches are directly correlated with the level of terrestrial runoff into the Gulf (Staples 1985). Much of this material can be trapped within the shallow nearshore zone (Wolanski & Ridd 1986b) for extended periods.

Cyclones

Tropical cyclones are common seasonal events in many parts of northern Australia (Figure 1). Cyclones, particularly those that cross the coast, exert pronounced effects on continental shelf and coastal marine ecosystems. In parts of northern Australia (eg the Kimberley region and north Queensland), the rainfall that accompanies cyclonic weather systems can be a major source of freshwater to the region, causing widespread, though episodic, flooding.

The high winds and associated waves generated in tropical cyclones induce mixing of surface waters in the deep ocean to a depth of approximately 100 m (Price 1981). This mixing often extends into the oceanic thermocline, bringing nutrient-enriched water to the surface along the storm track. As tropical cyclones move onto and over the continental shelf, cyclonic mixing (Figure 7) reaches to the bottom (Hearn & Holloway 1990) and is sufficiently powerful to re-suspend sediments over wide areas as the cyclone

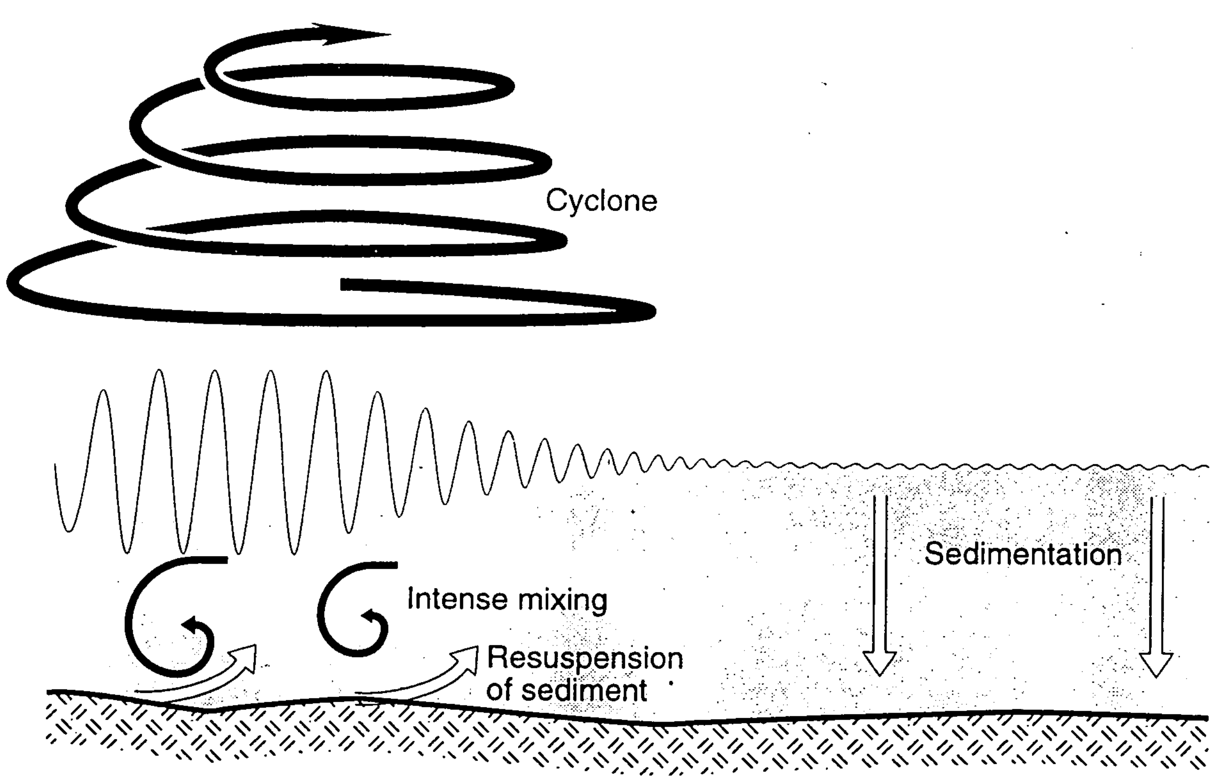

Figure 7: Mixing dynamics and significant nutrient fluxes associated with cyclonic storms moving over the continental shelf.

passes across the continental shelf (Gagan, Chivas & Johnson 1989). Dissolved and particulate nutrients within the sediments are re-suspended into the water column. There they are mineralised by bacteria, in turn triggering regional phytoplankton blooms. In conjunction with nutrients in rainfall and floodwaters near the coast, regional plankton biomass and primary productivity can increase 5–10-fold within a matter of days following a cyclone (Furnas 1989).

Upwelling

Ocean current and wind systems along the eastern and western coasts of Australia inhibit the development of the large, highly productive, *Ekman-forced* upwelling systems like those which occur along the western margins of North America, South America and Africa. This does not mean that upwelling processes are not important in Australia's marine environment. Rather, their spatial extent and trophic significance is more constrained. Locally significant upwelling of nutrient-enriched waters is known to occur along much of Australia's eastern seaboard (Andrews & Gentian 1982, Rochford 1984) and at sites along the southern coast of Victoria and South Australia (eg Schahinger 1987). At higher latitudes — near Tasmania — seasonal storm events accelerate the mixing of nutrients onto the shelf (Harris et al. 1991). On a local scale, upwelling frequently occurs in the lee of headlands and islands (eg Wolanski, Imberger & Heron 1984).

Along much of the eastern coast of Australia, the depth of the thermocline and associated *nutricline* is similar to the depth of the shelf break. Episodic oceanographic forcing leading to vertical excursions of the thermocline (eg Rochford 1984, Tranter, Leech & Airy 1986, Nof & Middleton 1989) results in the intrusion of cool, nutrient-enriched waters onto the continental shelf (Figure 8). These intrusion events lead to seasonal and episodic changes in the productivity of outer shelf waters (Holloway et al. 1985, Furnas & Mitchell 1986). For example, in the central Great Barrier Reef (GBR), one to several significant intrusion events may occur within a season (Andrews & Furnas 1986). Smaller mini-intrusions occur more frequently near the shelf break due to internal tides, shelf waves and local topographic effects (Wolanski & Pickard 1983, Griffin, Middleton & Bode 1987). The magnitude and duration of individual intrusion events is related to fluctuations in the direction and magnitude of both large scale and along-shore wind stress, current meanders, fluctuations in current strength and internal tides within the thermocline. Large intrusion events can transport substantial volumes of water onto the shelf. During large intrusion events in the central GBR, as much as 40% of the water volume over a section of the continental shelf can be intruded sub-thermocline water. In such cases, inputs of nutrients are proportional to the volume of water intruded. The importance of intrusive activity can only be resolved through the acquisition of long-term oceanographic and meteorological data sets (eg Rochford 1984, Harris et al. 1987, Harris et al. 1991) — particularly those with temporal resolution capable of resolving individual events and the environmental parameters accompanying them.

Tidally mixed systems (the Great Barrier Reef, Torres Strait, North West Shelf)

Tidally induced mixing is a major contributor to the nutrient dynamics of a number of Australian

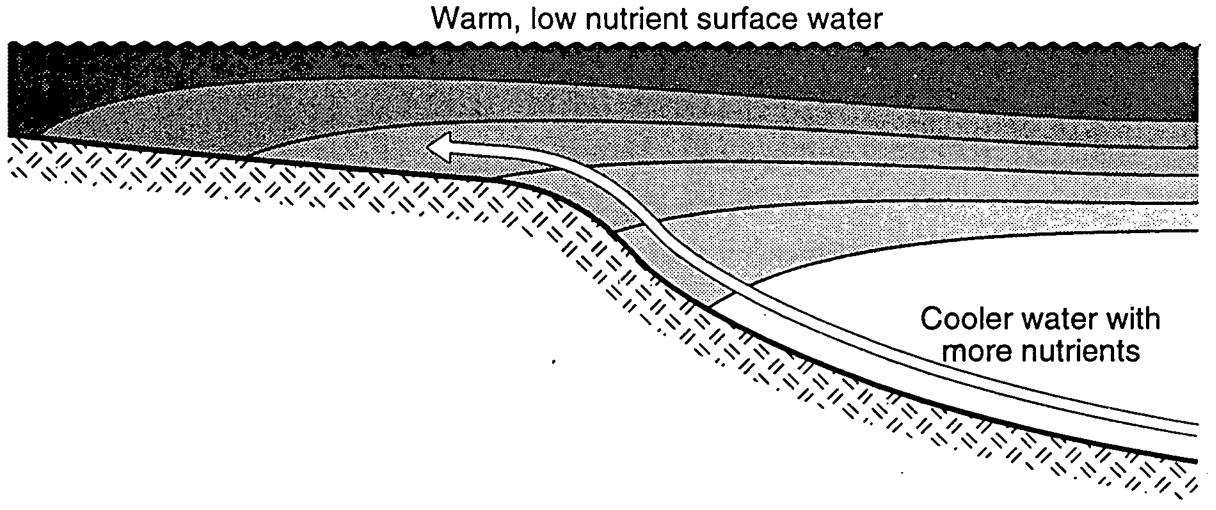

Figure 8: Temperature section through an intrusive upwelling event.

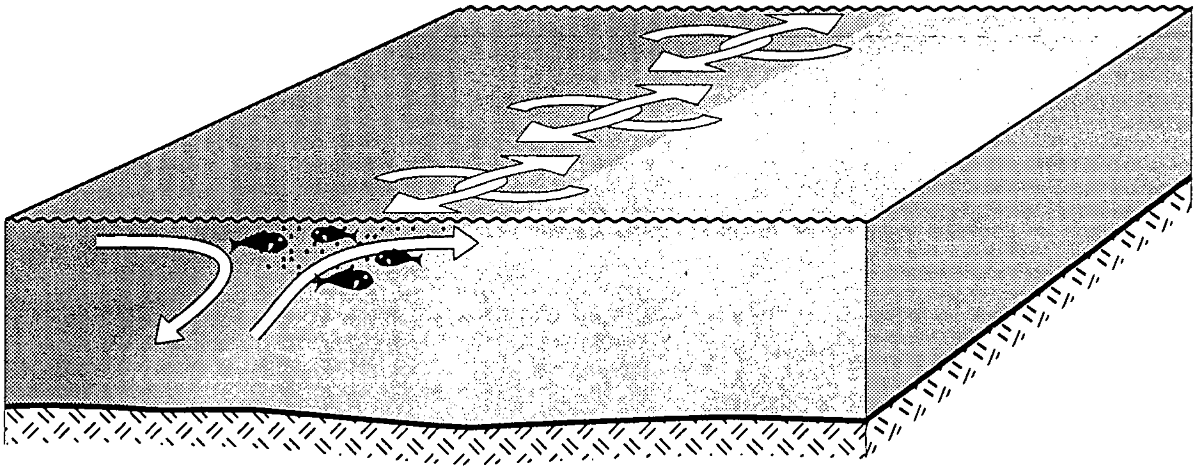

Figure 9: Water movements and mixing processes within a frontal zone.

marine ecosystems. In these systems, bottom friction acts in a manner analogous to wind stress on the surface to mix the water column.

In the complex reef matrix of the GBR, 2- and 3-dimensional wakes (Wolanski & Hamner 1988) are formed in the lee of individual reefs or behind gaps between reefs (Wolanski 1992, Liston et al. 1992). This constant mixing ensures that GBR shelf waters are largely homogenous in character, with nutrients being rapidly distributed throughout the euphotic zone. Topographically accentuated upwelling and mixing of tidal currents along the seaward margin of the GBR (eg Nof & Middleton 1989, Wolanski et al. 1988a) contributes to localised zones of high productivity along the seaward margin of the reef (Astley-Boden 1985). The influence of topographic stirring is greatest in the Torres Strait and far northern GBR where oscillating tidal currents through a dense reef and island matrix (Wolanski, Ridd & Inoue 1988) cause complete homogenisation of the water column and active resuspension of sediments. Phytoplankton populations in the Torres Strait and the far northern GBR exhibit high rates of productivity in surface waters (Furnas, unpub.). However, regional productivity is constrained by turbidity and the shallowness of the water column.

The continental shelf system of north-western Australia is also characterised by a highly energetic tidal regime. The tidal energy is dissipated through the breaking of internal tidal waves and the formation of benthic frictional boundary layers (Holloway 1983). Relatively little is known about the productivity and nutrient dynamics of this region. Holloway et al. (1985) proposed that a variety of mechanisms (shelf break upwelling, internal tidal activity and cyclonic disruptions) collectively contribute to nutrient inputs that support a productive shelf ecosystem. Tranter and Leech (1987) demonstrated that onshore propagation of the pycnocline and a subsurface chlorophyll maximum during the seasonal relaxation of the Leeuwin Current can bring phytoplankton biomass and nutrients onto the North West Shelf.

Frontal systems

In both the deep ocean (eg Andrews, Lawrence & Nilsson 1980) and shelf seas, frontal zones mark the horizontal boundary between discrete water masses or current systems (Figure 9). Water masses are usually defined by their temperature and salinity characteristics, though other features such as suspended sediment loads would also apply. Differences in the relative motion of two water masses at the frontal boundary creates a turbulent shear zone where extensive mixing can occur, both horizontally and vertically. Vertical water motions at frontal boundaries frequently transport nutrient rich water to the surface. The often sharp horizontal gradients at fronts can also concentrate organisms, either by passive physical processes or through modification of behaviour. Because of the enhanced mixing along the boundary, frontal zones tend to be more productive than the surrounding waters.

A number of significant frontal zones occur within or adjacent to Australia's marine Exclusive Economic Zone. In particular, the circumpolar Subtropical Convergence, which extends eastward from the southern corner of Tasmania, is a globally significant zone of enhanced marine primary production. It will be the focus for international scientific study over the next decade to determine the role of the Southern Ocean in ocean-atmosphere exchanges of CO_2 and climate variability. Latitudinal fluctuations in the location of the Subtropical Convergence have been shown to result in seasonal and inter-annual fluctuations in the presence and the persistence of high-nutrient subantarctic waters around Tasmania (Harris et al. 1987, Harris et al. 1991). By virtue of Australia's location, Australian oceanographers have the opportunity to observe this large and globally significant frontal system at close hand.

Ocean currents rarely move in a smooth straight path. When not constrained by physical boundaries, the current axis is constantly shifting, forming bends and meanders. Significant zones of eddy formation are associated with both the East Australian Current (Cresswell & Legeckis 1986) and the Leeuwin Current (eg Prata & Wells 1990). Where meanders of coastal currents detach from the coast, localised zones of upwelling can form in the pocket (Tranter, Carpenter & Leech 1986). After the East Australian Current has detached from the coastline south of Sydney, exaggerated meanders episodically pinch off and form free-drifting, rotating current rings (Figure 10). Depending upon the direction in which the meander bulges, the ring structures contain a central body of water warmer or cooler than the ring current itself or the surrounding ocean ('warm' or 'cold core' ring). As along frontal zones, shear between the current forming the eddies or rings and adjacent water masses can induce local upwelling, which brings nutrients to the surface and lead to the development of phytoplankton blooms (Tranter, Leech & Airy 1982).

Discussion

Although the nature and pattern of physical processes contributing to oceanic and continental shelf productivity around Australia are diverse and broadly identified, there is still very little quantitative information on the interaction between physical, chemical and biological processes in Australia's coastal, shelf and oceanic ecosystems. Longer-term climatic effects upon nutrient dynamics and ecosystem productivity are known, but as yet are only characterised for a few sites (eg Harris et al. 1987, Harris et al. 1991). Long term hydrographic data sets from a few sites (eg Rochford 1984) will prove useful in detecting such changes.

Few, if any, measurements of nutrient fluxes — as opposed to nutrient concentrations — have been made in Australian waters with the aim of developing a quantitative understanding of the productive processes at work in these systems. The lack stems from the immense area to be covered, the belated development of significant scientific programs aimed at understanding how large and medium-scale marine ecosystems operate, and a very real shortage of scientific personnel with the training and outlook to carry out such programs and demonstrate their importance.

Compared to a number of continental shelf regions in the northern hemisphere and global eastern boundary current upwelling systems, the primary productivity of Australian waters (eg Motoda, Kawamura & Taniguchi 1978, Furnas & Mitchell 1989) and associated fisheries production on a tonnage basis are not large. However, despite low to moderate primary productivity levels, a number of these regions support fisheries that are economically significant on a local or international basis due to their high monetary value (eg southern bluefin tuna, prawns, rock lobsters). The quality of water in Australia's estuaries and coastal zone and the productivity of the continent's fisheries are ultimately tied to the oceanographic processes responsible for delivering nutrients to the ecosystem(s) in question. Variability in biological processes such as primary productivity and recruitment are directly or indirectly linked to system-scale physical variability. Any effort to responsibly manage water quality in Australian estuaries or coastal waters and understand the processes controlling Australian fisheries must be based on an understanding of the limitations posed by the underlying environmental and oceanographic processes.

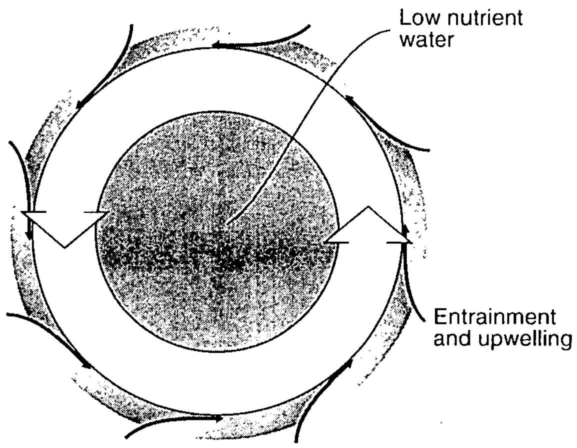

Figure 10: Water movements and mixing processes associated with current eddies or rings.

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Definitions

Euphotic zone – The upper layer of the ocean where sufficient sunlight penetrates to provide energy for photosynthesis by algae.

Thermocline – A vertical depth band where temperature changes rapidly. A thermocline defines the boundary between the well mixed surface layer and cooler, denser waters below. Thermoclines generally identify the boundary between vertically layered water masses in the ocean.

Sigma-t – Oceanographic shorthand for the specific gravity of water. Sigma-t equals 1000 times the specific gravity minus 1.000 (for s.g. = 1.0250, sigma-t = 25.0).

Nutricline – A vertical depth band where dissolved nutrient concentrations increase rapidly. In vertical profiles, the nitrate (NO_3) nutricline is usually embedded within the thermocline.

Front – A horizontal boundary between water masses.

Ekman forcing – When a layer of water is accelerated (eg by winds) or de-accelerated (eg by bottom friction), the Coriolis force causes the water to move at an angle to the direction of the acceleration. In the southern hemisphere, waters being accelerated (eg by wind stress) will move to the left. When winds blow parallel to the coastline, surface water will be either pushed toward the coast, leading to downwelling along the coastline, or offshore, resulting in upwelling as deeper water moves in to replace the surface water displaced offshore.

Mineralization – The biological conversion of organic matter to inorganic forms (CO_2, NH_4, PO_4) .

Nitrogen fixation – The conversion of atmospheric N_2 gas, which is normally biologically unreactive, to organic-N by specialised bacteria.

Denitrification – The conversion of NO_3-N to atmospheric N_2 gas by specialised bacteria, thereby removing the nitrogen from the ecosystem.

Nitrification – The oxidation of NH_4 to NO_3 by specialised bacteria.

The technical paper by Dr M. Furnas was reviewed by Dr J. Parslow, CSIRO Division of Oceanography, Hobart, and Professor J. Middleton, Centre of Marine Science, University of NSW.

Biogeography and diversity of Australia's marine biota

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The Australian continent's shallow-water marine biota, though less well known than our bizarre terrestrial flora and fauna, is notable for its high level of endemicity and for groups of animals and plants shared only with other southern hemisphere continents. The reasons for this are now emerging after nearly two centuries of taxonomic and biogeographic research.

Geological history

Although the present biota of Australia's coast and continental shelf can be explained in part with reference to modern conditions it is also the result of a long history in a changing environment. Since the mid-1960s theories of continental drift and plate tectonics have been applied to our interpretation of the Australian marine fauna — in particular, its position on the globe; its size, especially of its shelf; its connections to other land masses; the currents surrounding it; and the temperature and chemical nature of the sea surrounding it. The main events took place over the consecutive periods: Mesozoic, Tertiary and Quaternary.

Mesozoic

The history of Australia's modern marine biota began in the Mesozoic. Earlier and for most of this era Australia, South America, New Zealand, Antarctica, India and southern Africa were joined as a large southern continent, Gondwana. The fossil record suggests that most phyla had originated by the close of the earlier Cambrian times, perhaps 600 million years ago (600 mya) but new classes continued to appear until the Lower Carboniferous (300 mya). During the late Palaeozoic era the affinities of Gondwanan benthic marine faunas fluctuated between Palaeoaustral (cool-temperate) and Tethyan (warm-temperate). After the Permo-Triassic extinctions, rediversification of the marine fauna was mainly through evolution of new families, genera and species and the Gondwanan fauna became much like the fauna of today.

During the Triassic era (c. 200 mya) Gondwana to the south and Laurasia to the north were connected in the west but separated in the east by a major incursion from the surrounding Tethys Sea. The wide epicontinental shelf of Tethys included, in Australia, areas of present-day northern and southern Western Australia and south-eastern Queensland, then between 40°S and 65°S. Climates were warm and two faunal realms probably existed: a restricted cold- or cool-water Maorian or Austral Realm in the New Zealand–New Caledonia area; and a warm-water Tethyan Realm in what is now northern and western Australia.

Australia was positioned between 35°S and 65°S during the Jurassic period (200–129 mya) almost at right angles to its present orientation, and still connected to Antarctica (Figures 1A, 1B). A continental shelf surrounded only its present north-western and north-eastern coasts. Early Jurassic marine faunas were quasi-cosmopolitan but provinciality increased until the Australian (Tethyan) fauna became differentiated from the Boreal fauna of the northern hemisphere. The northern and southern coasts of Gondwana were at quite different latitudes but climatic differentiation was slight. The northern coast (incorporating north-western Australia) lay on the Tethyan Sea at about 30°S, while the southern coast (New Zealand, eastern Australia, West Antarctica, southern Africa and southern South America) lay on a southern arm of the 'Pacific Ocean' at 60°S.

The breakup of Gondwana commenced with the separation of Africa (c. 125 mya) and India (c. 118 mya) from Australia–Antarctica during the Cretaceous period by an arm of the Tethys becoming the proto-Indian Ocean by 100 mya. Several incursions of the sea from the west occurred along the line of the south coast of Australia. Much of the Australian land mass was submerged during Cretaceous incursions of the Tethyan Sea, especially from the north

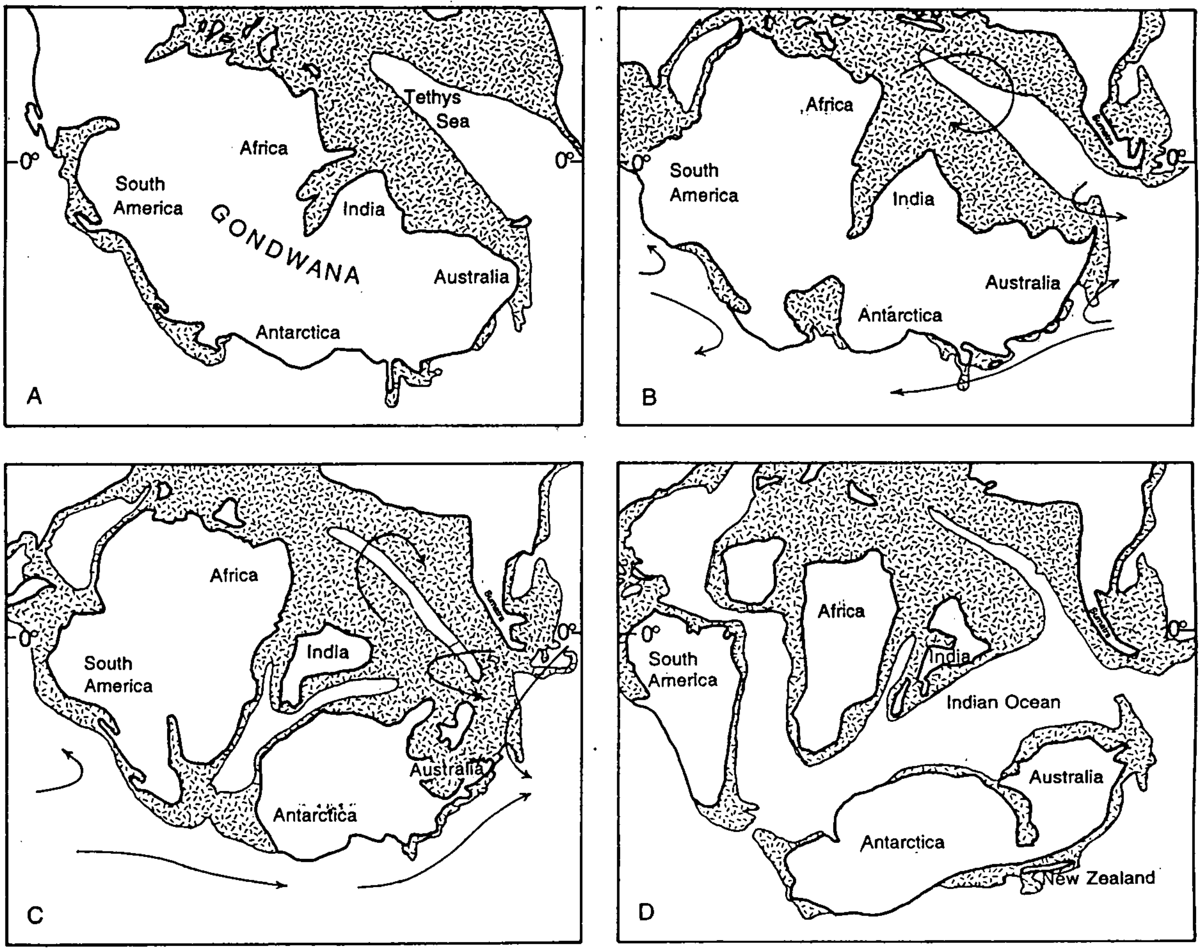

Figure 1: The relative position of Australian coasts and shelves among the southern continents (Gondwana) during the Mesozoic.
A. Early Jurassic (180 mya) — the Tethys Sea with its broad shelves separates Laurasia to the north from the fused continents of Gondwana. B. Late Jurassic (145 mya) — Africa and Antarctica begin to rift apart. C. Early Cretaceous (125 mya) — an ocean basin separates Africa and India from Antarctica; continental Australia is largely submerged by waters of the Tethys. D. Late Cretaceous (90 mya) — New Zealand has separated and Australia and Antarctica begin to rift from the north-west. The bold line indicates coastlines and shading shelf waters.

and north-west (Figure 1C). Temperatures warmed to about 10°C warmer than at present but cooled later. The Austral Province, which included Australia, New Caledonia, New Zealand, New Guinea, southern South America and eastern India, could be contrasted with the East African Province and accords with the tectonic isolation of Australia and western India–Africa at that time.

Later (82 mya), the Tasman Sea opened to separate Australia and New Zealand and later rifting between Australia and Antarctica began by intrusion from what is now the west. The colonising biotas of the Australian southern coast were therefore of tropical Tethyan origin (Figure 1D). The biota of what is now the eastern coast of Australia was part of a Weddellian Province which extended across Australia and northwards up the coast of Chile (Zinsmeister 1982).

Tertiary

By mid-Palaeocene (60 mya) the Coral Sea had formed and Australia had begun to rotate anti-clockwise towards its present orientation and to move northwards. Water temperatures were warm with a northerly current on the west coast and a south-directed offshoot of the Pacific Equatorial Current on the east coast. Although Australia moved northward during the Tertiary, the continent experienced successively cooler, rather than warmer, temperatures as a result of world-wide cooling. Sediments at this time were predominantly biogenic carbonates, much as they are now.

By the late Eocene–early Oligocene (40–35 mya) the sea was able to pass from the west between Antarctica and the south-eastern corner of Tasmania over the South Tasman

Rise at a latitude of about 65°S (Figure 2A). In the late Oligocene (30 mya) oceanic water moved freely between the two continents (Knox 1979) and water temperatures cooled quickly. For the first time biotas from the eastern coast of Australia and the western and southern coasts (mainly Tethyan) were able to mix along the south coast.

Although Australia was by then a separate continent with an isolated marine fauna two other tectonic events are relevant. The first event was the separation of West Antarctica and South America by the opening of Drake Passage during the Oligocene or Miocene (30–22 mya), thereby allowing the

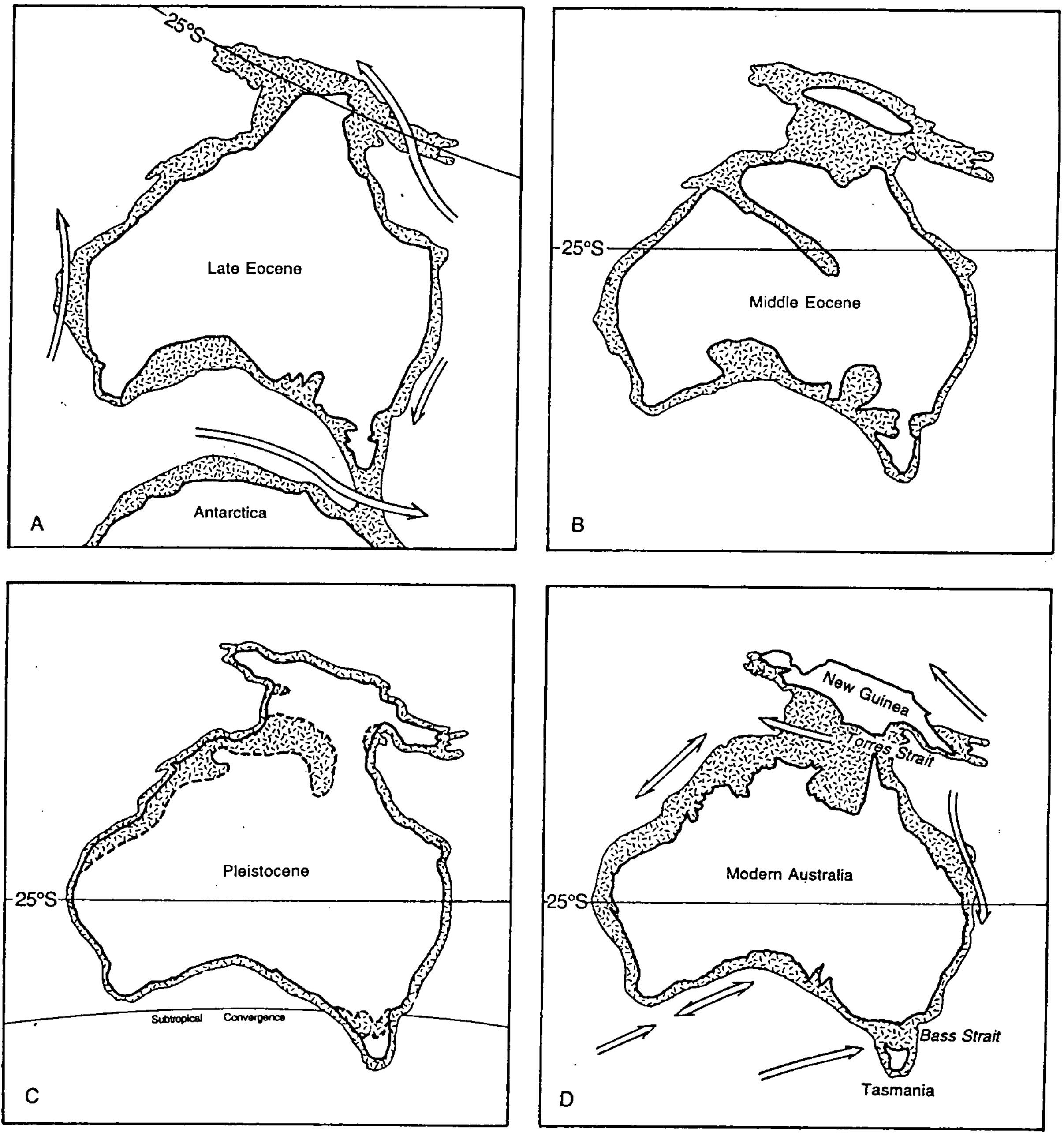

Figure 2: The changing shoreline and shelf of Australia during the Cenozoic.
A. Late Eocene (38.5 mya) — Australia and New Guinea still connected to Antarctica by the South Tasman Rise. B. Middle Miocene (11 mya) — extensive marine transgressions in the south and north. C. Pleistocene (18 000 ya, solid line, and 10 000 ya, dotted line) — a period of minimum shelf area with the 10°C–minimum Subtropical Convergence appearing across south-eastern Australia. D. Modern — both New Guinea and Tasmania separate.

Circum-Antarctic Current to form. Global cooling resulted in polar ice caps and a permanent steep water temperature gradient away from the poles. The Antarctic Convergence (a steep temperature gradient) formed during the early Miocene (22 mya) (Figure 2A) thus producing an important biogeographical boundary. This convergence and the Subtropical Convergence further north have persisted from those times but their latitudinal movements have had a profound effect on biogeographic regions of southern continents (Knox 1979).

At about the same time there was a notable invasion into southern Australian shelf faunas of warm-water elements. In a detailed analysis of the Tertiary Mollusca of the south-eastern coast, Darragh (1985) concluded that during the Tertiary the Tethyan Indo-Pacific element increased with time at the expense of the Australian and New Zealand elements. Cosmopolitan elements decreased to be replaced by greater endemism. It has been suggested that the development and decline of this warmer interval, which contrasts with the overall Cenozoic glacial cooling pattern, resulted from Australia's northward drift towards the equator overtaking the contraction of marine temperature zonation (Talent 1984). At the time (3–4 mya) the Subtropical Convergence crossed in the latitude of Tasmania and a cold water mass separated the eastern and southern coasts of Australia (Knox 1980).

The second tectonic event of significance to Australia's biogeography was its collision with South-east Asia in the north. This event began in the early Miocene (c. 20 mya) when the New Guinea margin of the Australian block came into contact with the Sunda Arc (eastern Indonesia). The two blocks continued to overlap and with the evolution of island arcs the Tethys Sea was no longer a barrier to marine shelf and coastal biota. This event contributed to a reduction in the east-to-west faunal differences across the tropical Pacific.

Quaternary

At the beginning of the Quaternary period (2 mya) the arrangement of land and sea in the south-west Pacific was essentially as it is today except that neither Torres Strait nor Bass Strait existed. However, the size and shape of the Australian coastline and shelf continued to change during the Pleistocene due to sea levels fluctuating over 200 m (Galloway & Kemp 1981; Figures 2C, 2D). Both Torres Strait and Bass Strait opened and closed repeatedly. The changing shelf size, coastline and latitudes of the passages between the east coast of the Australian mainland and the remainder of the continent would have affected the continuity of their biotas.

The coastal marine fauna of Australia did not respond to the breakup of Gondwana in the same way as did the terrestrial fauna because the separation of land masses created shallow shelves for marine invasion (barriers to terrestrial faunas) before these shelves were split by sea-floor rifting.

In summary, the most important tectonic events affecting the biota of Australia's coasts and shelf are:

- its separation from other Gondwanan continents;
- its separation from eastern Antarctica, allowing mixing of Tethyan and Austral faunas on the South Tasman Rise, and;
- the collision of Australia with South-east Asia, thereby providing pathways for the invasion of Laurasian Tethyan biota into northern Australia.

Distribution patterns

An understanding of the marine biogeography of Australia has been achieved through investigation of the geology and palaeontology of the continent and by examination of modern distribution of the biota. This distribution reflects past events and present-day environmental conditions.

Wilson and Allen (1987) compared the number of species and species composition of fishes, molluscs, echinoderms and corals throughout the continent. These groups were chosen because their taxonomy is relatively well-known, and they probably demonstrate general principles which can be applied to other groups.

Fishes

Some of the 3400 species of Australian marine fishes are pelagic or oceanic and wide-ranging in tropical or temperate seas. About three-quarters occur on the shelf and nearshore. The greatest number of species are in the tropics where approximately 120 families, 600 genera and more than half of the species (1900) are found; and most of these are common to the Indo-West Pacific region. Although most species have pelagic eggs and larvae a moderate level of endemicity (13%) is maintained with the help of southerly flowing currents on both the east and west Australian coasts.

In contrast, the fish fauna of southern temperate Australia comprises about 600 species of which 85% is endemic and 11% is shared with New Zealand. One contribution to this high level of endemicity is radiation in a few families with low dispersability: viviparous clinids, brooding syngnathids and nesting gobiescocids and gobiids. The pipefishes and seahorses (Syngnathidae), leatherjackets (Monacanthidae) and fishing frogfishes (Antennariidae) are especially rich in species in southern Australia.

Shallow-water reef fishes provide one of the best studies in zonation along the southern coast. Wilson and Allen (1987) recognised four ecological barriers which appear to inhibit dispersal: a sharp temperature gradient around Albany near the cessation of the Leeuwin Current; and the absence of nearshore rocky reefs in the centre of the Great Australian Bight, at the mouth of the Murray River, and in eastern Victoria. These barriers may act today to maintain allopatric eastern and western species pairs. Eighteen pairs of closely related fish species in 12 families were reported by Wilson and Allen (1987: Table 3.9).

Molluscs

The biogeography of molluscs is very much like that of fishes although there are many more species. Most families are represented in the Australian fauna and have more species in the north of the country than in the south. The level of endemicity in the north is low (about 10%) with most species distributed widely in the Indo-West Pacific region. In contrast, endemicity of the southern Australian fauna is about 95% and several endemic genera occur. Some endemic genera are relicts of the once widespread Tethyan fauna (as are the endemic families, Trigoniidae, Campanilidae and Diastomatidae, each with a sole living representative). Other endemic genera are relicts of the ancient Palaeoaustral fauna.

Echinoderms

Echinoderms, with only a few hundred Australian species, repeat the biogeographic patterns of the fishes and molluscs with 13% of species endemic in the tropical region and 90% in the temperate region. Many echinoderms have a long larval life which may explain why 22% of Tasmanian species occur also in New Zealand where they are transported by the West Wind Drift.

Corals

Reef-building scleractinian corals are essentially tropical and their distribution is substantially different from that of the three groups discussed above. The Australian fauna is a subset of Indo-West Pacific corals and is largely confined to the Great Barrier Reef and smaller reefs of the west coast. There is not a gradual reduction in the number of species with increasing latitude but rather an abrupt depletion of species at the termination of the Great Barrier Reef and at the Houtman Abrolhos. Coral reefs do occur further south —eg at Lord Howe Island and Solitary Islands — but the number of species is few. Few species of coral occur on the southern coast of Australia and reefs are not formed there.

Other taxa

Analyses of other groups in the manner of Wilson and Allen (1987) are more difficult because species composition is less well known. Similar patterns are expected but not all taxa are more diverse in tropical than in temperate environments. While there are more species of decapod crustaceans (crabs, hermit crabs, lobsters, shrimps, prawns) in warmer waters the same is not true for other groups such as peracarids (amphipods, isopods, and others). These biogeographic distributions have repercussions in functional ecology — for example, the taxonomic composition of crustacean scavenger guilds. Cypridinid ostracodes, cirolanid isopods and lysianassoid amphipods play this role but the last of these become relatively more important as one moves from tropical to temperate waters (J.K. Lowry pers. comm.).

East-west species pairs such as occur in the fishes occur in other groups. Dartnall (1974) figured pairs of brachyuran crabs, molluscs and asteroids whose distributions overlap or are contiguous in Bass Strait. Their distributions imply that this region is or has been a barrier stimulating speciation. Such an interpretation depends on establishing the sisterhood of the species pairs but in few cases have phylogenetic treatments been done.

Present Australian biogeographic provinces

In his 1953 classification of the world's marine environments, the biogeographer, S. Ekman, placed tropical and subtropical Australia within his Indo-West Pacific region. He recognised that southern Australia was separate and placed it within the warm-temperate fauna of the Southern Hemisphere. The Indo-West Pacific was recognised by Ekman as containing 'the greatest wealth of animal life' (1953:11). His view was not disputed by the most recent discussion of Australian marine biogeographic components (Wilson & Allen 1987).

Several marine provinces within Australia have been proposed and reviewed many times (Knox 1963). Suggestions of as many as three tropical provinces and three or four temperate provinces no longer have currency partly because their boundaries are doubtful and not defined quantitatively (for example, Figure 3A) and their definitions are intuitive. Edgar's (1984) presentation of a matrix of indices of similarity between sampling sites in Tasmania and the nearby mainland is one exception. Poore et al.'s (1994) analysis of the distribution of nearly 300 species along the Victorian coast found only weakly defined boundaries in this short section of the Australian coastline.

Nevertheless, division into tropical and temperate regions has never been seriously disputed. Wilson and Gillett (1971) and Wilson and Allen (1987) simplified the picture by recognising northern and southern Australian regions (Figure 3B) with transition zones between them — one on the east coast and one on the west coast. Wilson and Allen's (1987) conclusions on Australian biogeography derived more from inferred origins of the fauna than from its present classification based on species composition. The distribution patterns seen today are the result of contributions from two different early Tertiary biotas:

> (1) the pan-Pacific Tethyan biota and its successor, the Indo-West Pacific biota, have dominated the northern coasts of Australia since the beginning of the Tertiary and also contribute to temperate biotas, especially in the south-west. To the north, barriers to interchange of shelf and coastal biotas with South-east Asia are only slight. There are therefore many widespread tropical elements in the northern Australian biota and a low percentage of endemicity. At its southern limit the Tethyan element is limited by the latitudinal temperature gradient.
>
> (2) the temperate Palaeoaustral fauna has dominated south-eastern Australian coasts also from the early Tertiary and is now the major element of the biota of the entire southern coast. Its high level of endemicity results from its isolation by ocean basins from other southern continents and from a latitudinal temperature gradient to the north.

The boundary between the tropical and warm-temperate provinces coincides approximately with 18–20°C winter minimum

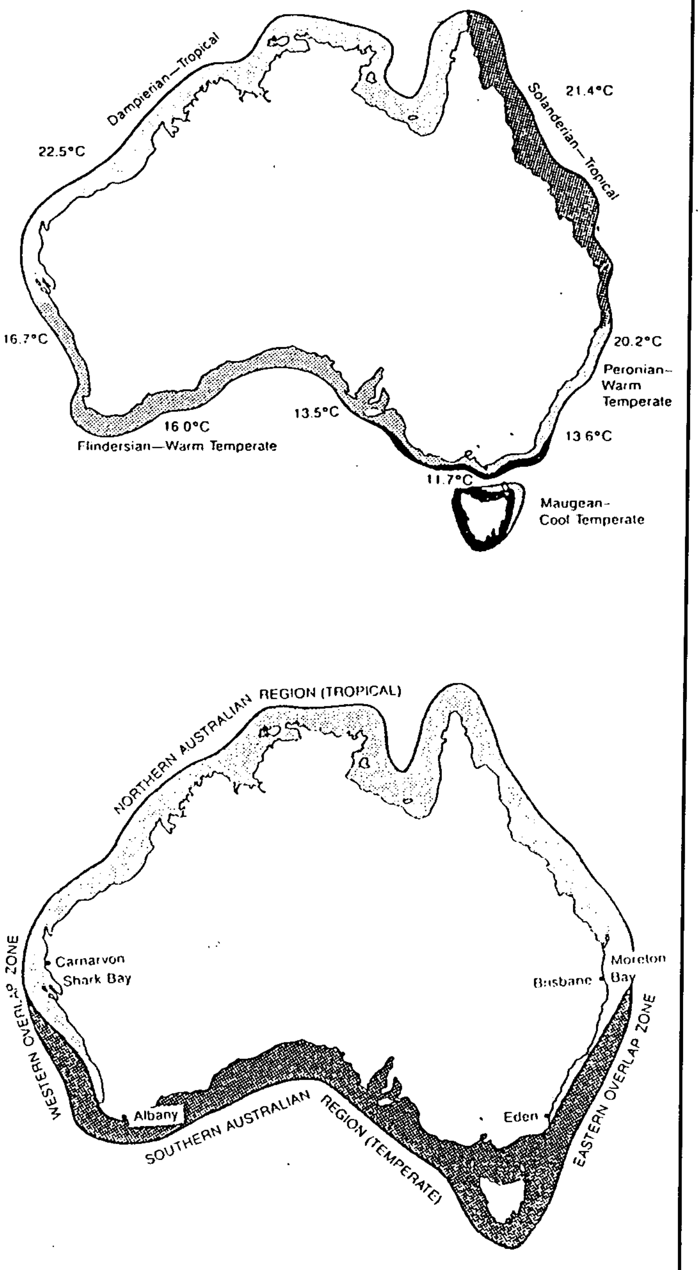

Figure 3: A. Biogeographic provinces of the Australian intertidal according to Bennett and Pope (Knox 1963).
B. Major faunal regions of the Australian coast (after Wilson & Gillett 1971).

surface temperature but this is variable and dependant on the influence of the East Australian Current and the Leeuwin Current.

This gross picture of Australia divided into a northern region with low endemicity and a southern region of high endemicity is superimposed on other patterns as yet poorly understood. Most obvious of these is the separation of the Great Barrier Reef community from that of the adjacent coast but it could be argued that this distinction is an ecological rather than a biogeographic division.

Similarly, George (1969) divided tropical coasts on the basis of water turbidity: northern Western Australia, Northern Territory and Queensland where high monsoonal summer rain and dry winters result in grey mud sediments inshore and well developed mangrove creeks; and north-western Western Australia where rainfall is low and irregular, with occasional cyclonic disturbances and flash flooding resulting in brown sediments. The two tropical regions contrast with the southern half of the continent where rainfall is more reliable — uniform throughout the year in the east and falling in the winter in the west. Again, this division may reflect modern ecological regimes rather than more ancient biogeographic events and is covered in the chapter on ecosystems.

Diversity

In an earlier section, numbers of fish, mollusc and echinoderm species in Australia were estimated. Also of interest is the number of species of all taxa in a circumscribed habitat: so-called 'diversity' or, in modern parlance, 'biodiversity'. Such data are known to correlate with environmental conditions and are a useful tool in understanding biogeographic trends.

The few quantitative attempts in Australia to obtain data on numbers of species are rarely comparable. The result depends on the habitat chosen, its size, methods, and on the effort and skills of the taxonomists involved. The most serious impediment to completion is the poor state of knowledge of the fauna: rarely are more than 40% of the true species complement known to science.

Less is known of the marine environments. Birtles and Arnold (1988) recorded 103 species of echinoderm and 196 species of mollusc from four sites on the Great Barrier Reef lagoon, and Ward and Rainer (1988) reported 308 species of decapod crustaceans from the North West Shelf. Both studies are taxonomically limited.

Attempts to identify all species (above a minimum size) in a defined small area have been made in estuaries. Poore (1982) reported that the number of species in the Gippsland Lakes (90) was 30% lower than in estuarine systems studied by others. Explanation might be sought in the relative sizes of the estuaries, degree of marine influence, and biogeographic history.

Less is known of marine environments. In Port Phillip Bay 713 macrobenthic species were taken from 430 samples of sandy and muddy habitats (total area = 43 m≤) (Poore et al. 1975). In Western Port fewer samples took 572 species (Coleman et al. 1988) but a total of 2000 has been estimated from all habitats.

A survey of 1.2 m≤ of sea-floor in eastern Bass Strait turned up 353 species (Parry, Campbell & Hobday 1990) but more detailed work at the same place has discovered about 800 species in 10 m≤ (unpubl.). The comparisons of Parry, Campbell and Hobday (1990) showed that even the lower figure was much greater than in other parts of the world.

Poore and Wilson (1993) and Poore, Just and Cohen (1994) summarised data on the number of species of isopod crustaceans on the south-eastern Australian slope where 359 species were identified. This is many more than found in similar studies in the northern hemisphere.

Less information has been published from macrobenthic communities in other parts of Australia and where data exists they are unlikely to be comparable. This points to the need for some basic protocols for quantification of 'biodiversity' before regions can be compared and latitudinal gradients measured.

Diversity has classically been viewed as being greatest at low latitudes and decreasing towards the poles. Recent evidence from terrestrial animals (Platnick 1991) and marine amphipod crustaceans (Barnard 1991) supports the view that this is not always the case, especially in the southern hemisphere. Australia has a very long coastline (both latitudinally and longitudinally) but trends in species diversity are poorly documented.

Gradients in both direction are to be expected but not all taxa will behave in the same way. While fishes, corals, molluscs, echinoderms and decapod crustaceans decrease in diversity from north to south in Australia the same trends are not apparent in other taxa. In amphipods (Barnard 1991) the reverse is true. In isopod crustaceans a diversity gradient overall is not yet apparent but relative dominance of families certainly changes with latitude as it does in many other taxa.

Conclusions: implications of biogeography for management

Biogeographic regions

One of the many criteria used in the selection of areas for management is 'biogeographic zone', the implication being that areas within zones have biotas more similar to each other than with areas in different zones. Thus it is argued that each area is representative of the zone from which it was selected. The division of Australia into a northern tropical region and a southern temperate region with broad transition zones is clearly not adequate for making such decisions.

However, I would argue that finer division of the coast (and continental shelf and slope) has not yet been achieved satisfactorily. The divisions of the early biogeographers were entirely intuitive and therefore hotly debated. Most were based on a single taxonomic group or habitat: molluscs, echinoderms, algae, fishes, ascidiaceans or intertidal habitats (see review by Knox 1963). But in reality, steep environmental gradients that might explain some of the provinciality do not exist and do not affect all taxa and habitats equally.

In 1986, the Australian Committee of the International Union for the Conservation of Nature and Natural Resources adopted in its policy for protection of marine and estuarine areas a classification of the Australian habitats and coastline prepared by the Australian Bureau of Flora and Fauna (Figure 4). Fourteen coastal (shallower than 200 m) geographic zones plus 18 oceanic zones and external territories are mapped. Boundaries between the 14 coastal zones do not coincide with suspected biogeographic boundaries and the geographic categories are inappropriate for environmental management. The absence of comparative quantitative data on the relative abundance of species in many taxa for most of the coast is a serious impediment to a classification which reflects distribution patterns of plants and animals.

A considerable body of data exists (mostly in museum collections) on the distribution of species around the coast. However, coverage of the coast and of taxa is certain to be uneven. These data could be collated and analysed objectively to determine where, if anywhere, biogeographic boundaries can be recognised. But success will be slight since the data were not collected for this purpose. Much better would be to gather new strategic quantitative data. Modern multivariate techniques such as hierarchical classification and ordination are appropriate analytical techniques worth investigating.

Ideally, the discovery of biogeographic boundaries — past and present — depends on the revelation of phylogenetic relationships within numerous taxa at the family and generic level. The congruence of their distribution patterns will indicate barriers as are hinted at in the study of species pairs on the southern coast and in Bass Strait. The costs of obtaining data and performing the required analyses will be high but until it is done biogeographic considerations can play little part in environmental management.

The concept of endemism as an important criterion in the selection of areas for management must be investigated. Such a criterion will shift emphasis from the north of Australia (low endemicity) to the south (high endemicity). It is in the south that Australian 'native' marine biota resides. Further, it is the southern temperate ecosystems which are the most threatened by the largest population centres.

Diversity

Management of marine environments must be geared to the management of communities of species rather than towards individual species. There are of course rare exceptions where a large and obvious species may warrant special attention.

Coral reefs in particular are said to be of special interest because of their high 'biodiversity' but this has not been quantified and the relative importance of different reefs from the point of view of diversity is unknown. By and large the species inhabiting tropical reefs are widespread through the Indo-West Pacific. Concentration of research effort and management on Australian coral

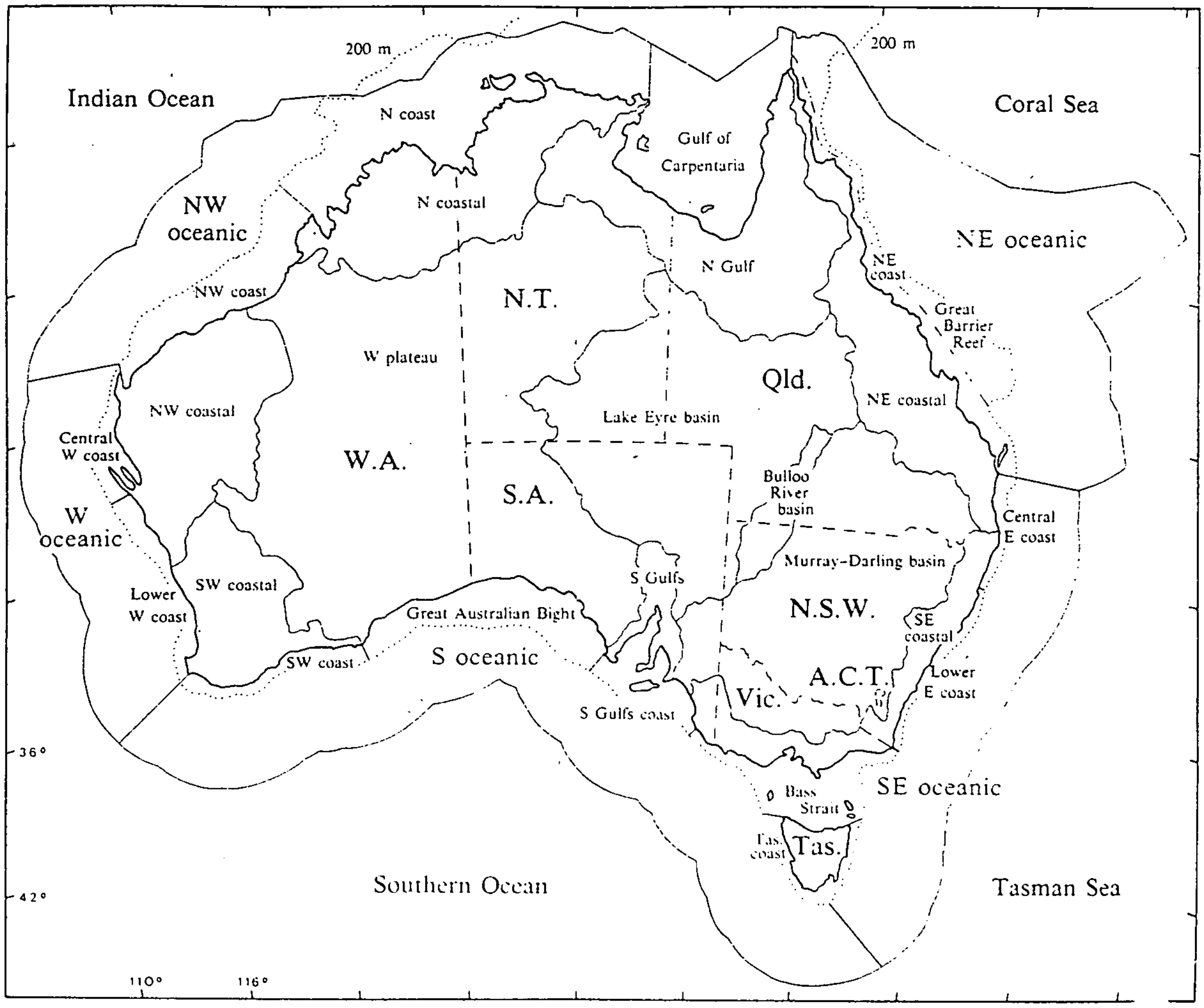

Figure 4: Coastal zones within the 200 m bathymetric contour and the 200 nautical mile Exclusive Economic Zone adopted by the Australian Committee of the I.U.C.N. in 1986.

reefs at the expense of more southern ecosystems with a much more endemic biota can only be justified on the grounds of international responsibility.

In fact, many temperate marine environments are inhabited by communities rich in species, with no species especially more abundant than others. Investigations in Bass Strait and the south-eastern slope have revealed soft-bottom benthic communities more diverse than elsewhere in the world; yet their geographic extent is unknown. It is certain that the relative taxonomic composition of communities does vary with latitude but details are sketchy. Communities of such complexity are worthy of attention and I argue that diversity should be one of the criteria on which areas are selected for management. Selection should not be on the basis of high diversity: communities with natural low diversity such as estuaries are equally valuable.

Although information on the diversity of coastal, shelf and slope communities in most of Australia is scattered attempts to obtain comparable data should be made. The greatest impediment to obtaining these data is the poor state of taxonomic knowledge of the biota. For many of the most numerous macrofaunal groups (crustaceans, polychaetes, micromolluscs) fewer than half and often less than a tenth of all species are named. For microfaunal interstitial species almost nothing is known taxonomically. This situation makes research into diversity and phylogenetic biogeography difficult.

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This technical paper by Dr Poore was reviewed by Dr. D. Alongi, Australian Institute of Marine Science, Townsville, and Dr. G. Ross, Australian Biological Resources Study, Canberra.

Marine phytoplankton communities in the Australian region: current status and future threats

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Introduction - value and usage

The phytoplankton algae that make up the floating pastures of Australia's marine environment are the food base that supports, either directly or indirectly, the entire production of the open sea. These microscopic plants range in size from 0.2 m to 200 m (a m is 1/1000 mm), with the exceptional few reaching 4 mm in length. The diversity of organisms involved is immense and encompasses representatives of 13 algal divisions. These classes include the well-known diatoms (5000 species), dinoflagellates (2000 species), golden-brown flagellates and green flagellates (several hundred species) and the coccoid picoplanktonic forms (cyanobacteria, prochlorophytes). Just like land plants, phytoplankton algae require carbon dioxide, water, sunlight and nutrients for growth and photosynthesis. Their depth distribution is limited by the extent to which photosynthetically available sunlight can penetrate, and this ranges from 200 m in the clearest oceanic water to several metres in turbid estuaries. For their nutritional requirements (nitrates, phosphates, silicates, trace elements) the phytoplankton are strongly dependent upon physical dynamic processes which move nutrient-rich deeper water into the upper lighted zone. 'Nutrient upwelling' can occur by many mechanisms, mostly driven by winds interacting with currents and continental shelf topography. While Australian waters have no major upwelling systems comparable to those off Peru, California or north-west Africa, enrichments of a lesser kind do occur regularly and provide the nutrients for the rich diatom blooms which support Australia's most productive fisheries grounds. Anthropogenic nutrient discharges via domestic and industrial wastes can also increase the algal biomass of coastal waters but, more seriously, this has the potential to dramatically alter the original phytoplankton species composition with far-reaching implications for the structure of entire marine food chains.

This review summarises the status of our knowledge on marine phytoplankton communities in the Australian region (excluding the Antarctic) and identifies the impact of human disturbances upon them.

Description and status

Phytoplankton species composition

Recognition of three distinct phytoplankton assemblages in Australian tropical and oceanic waters has come from light microscope and electron microscope studies of the extensive phytoplankton collections made by CSIRO scientists on the research vessels RV *Sprightly* and FRV *Soela* (1980–1985). A tropical oceanic community occurs in the offshore waters of both the Coral Sea and Indian Ocean, a tropical shelf community is confined to the Gulf of Carpentaria and the North West Shelf and a temperate neritic (inshore) community is found in coastal waters of New South Wales, Victoria and Tasmania (Figures 1, 2). The larger (30–200 m) diatom and dinoflagellate species in the assemblages are different, but the smaller nanoplankton (2–20 m) are remarkably similar in all environments studied. Sometimes typical warm-water species are found in subtropical or temperate waters. This phenomenon results from tropical waters penetrating southwards in the current or eddy formations and carrying their entrained plants and animals within them. Conversely, a fourth assemblage of subantarctic phytoplankton species has been observed episodically off southern Tasmania.

The phytoplankton flora of the Australian region has strong similarities with the warm- and cold-water phytoplankton floras of the northern hemisphere. In contrast to the macroalgal flora, there is virtually no endemism in the phytoplankton. The concept of threatened species is difficult to apply to phytoplankton communities, although there is growing evidence for immigrant species being

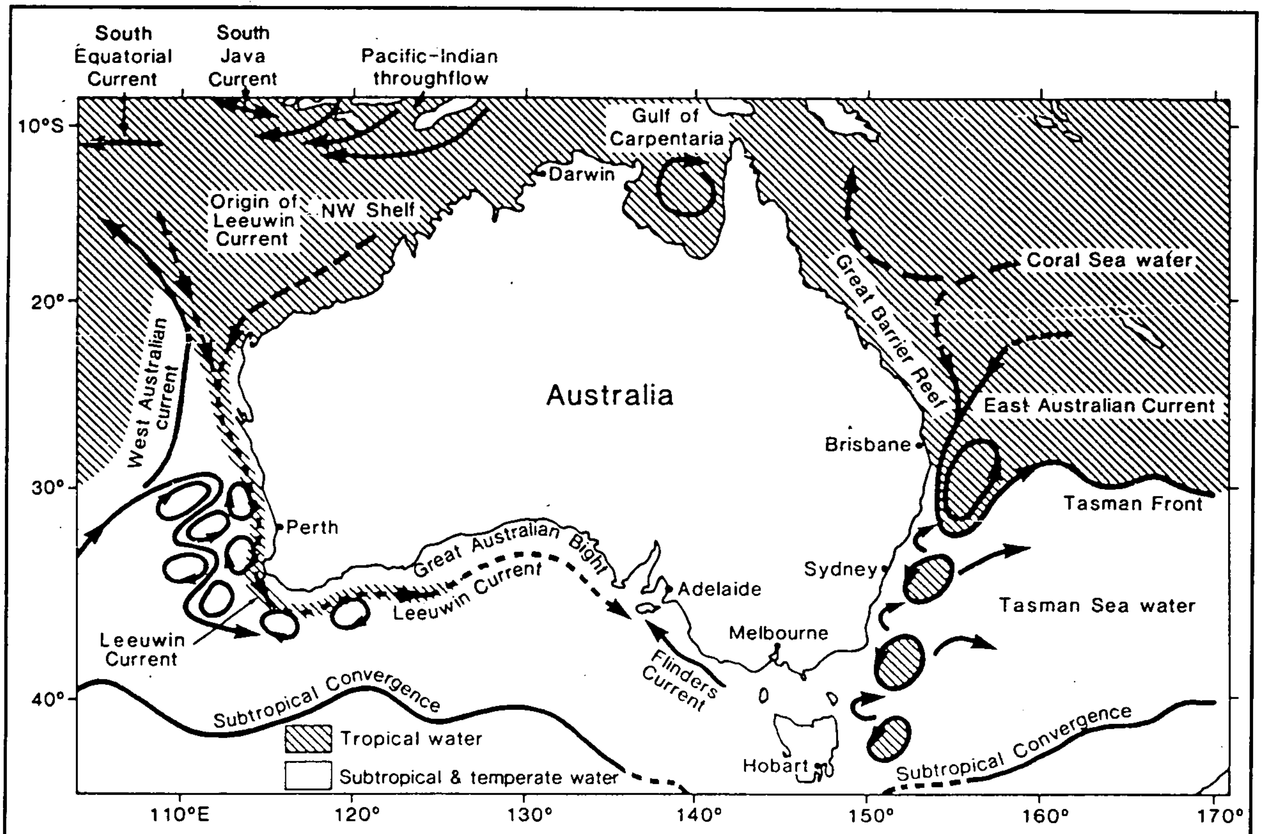

Figure 1: Map of the Australian region showing the major current systems (after Jeffrey & Hallegraeff 1990).

introduced into local waters of many countries via ships' ballast water (Hallegraeff & Bolch 1992). Well-documented examples are the diatom *Coscinodiscus wailesii* in the North Sea (which can clog fishermen's nets) and the dinoflagellate *Gymnodinium catenatum* in Tasmanian waters (which can contaminate shellfish with toxins harmful to the human consumer). Such introduced species or other human-made disturbances of the marine environment can only be identified from a thorough baseline knowledge of the 'normal' phytoplankton taxa and their seasonal changes in abundance and diversity in a region. In Australian coastal waters, phytoplankton checklists exist for the North West Shelf (Hallegraeff & Jeffrey 1984), the East Australian Current (Jeffrey & Hallegraeff 1987), the waters of the Great Barrier Reef (Revelante & Gilmartin 1982) and Port Hacking, New South Wales (Hallegraeff & Reid 1986). A diatom flora for the Swan River estuary in Western Australia has been prepared by John (1983) and a comprehensive phytoplankton flora for Bass Strait is also in preparation (D. Hill and co-workers). Such information is still lacking for many other parts of the Australian coastline.

Phytoplankton biomass and nutrients

Open ocean and coastal waters

The open ocean waters surrounding the Australian continent (Figure 1) are generally poor in nutrients (Jeffrey, Rochford & Cresswell 1990). The major factors responsible for this low nutrient status include: general nutrient impoverishment (particularly in phosphates) of coastal-drained soils; lack of a nutrient-rich eastern boundary current in the Indian Ocean originating in high latitudes; isolation by the Subtropical Convergence of Great Australian Bight waters from rich subantarctic waters to the south; and dominance of large areas by subtropical waters with limited nutrient reserves down to 100–200 m. Natural enrichment mechanisms in the Australian region include limited upwelling along the outer margin of the Great Barrier Reef, along the New South Wales coast between the Queensland border and Port Stephens, off the Gippsland coast of Victoria and off Port Macdonnell in South Australia; seabed coastal enrichments off New South Wales; deep convective overturn within warm-core eddies in the Tasman Sea providing nutrient enrichment to the surface; and some nutrient enrichment in

Table 1: Summary of the seasonal range of phytoplankton chlorophyll *a* biomass in Australian waters

	mg L^{-1}	mg m^{-2} (water column total)	Reference
Oceanic			
Indian Ocean	0.1–1.1	10–80	Humphrey 1966
Coral Sea	0.1–0.8	15–70	Furnas & Mitchell 1986
East Australian Current	0.05–0.9	20–80	Jeffrey & Hallegraeff 1987
Coastal			
Gulf of Carpentaria,	0.8–2.3	26–49	Hallegraeff & Jeffrey 1984
Arafura Sea	0.7–1.3	34–41	Hallegraeff & Jeffrey 1984
North West Shelf	0.2–1.1	20–55	Hallegraeff & Jeffrey 1984
Sydney coastal waters	0.1–15.0	20–280	Hallegraeff 1981
Bass Strait	0.1–1.6	–	Gibbs et al. 1986
Great Australian Bight	0.1–0.4	12	Motoda et al. 1978
Tasmanian coastal waters	0.1–6.0	–	Harris et al. 1991
Inshore			
Sydney Harbour	1.8–19.0	–	Revelante & Gilmartin 1978
Port Hacking estuary	0.1– 8.0	–	Scott 1978
Cleveland Bay, Townsville	0.1–10.0	–	Walker 1981
Port Phillip Bay, Melbourne	0.1–30.0	–	Axelrad et al. 1981

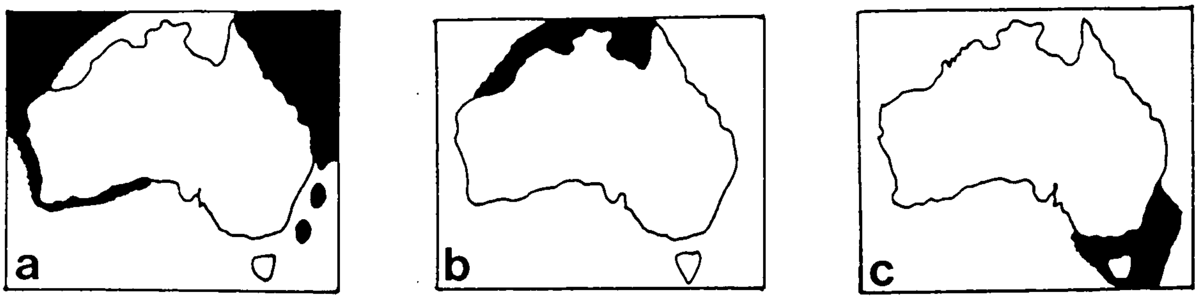

Figure 2: The distribution of three distinct marine phytoplankton assemblages in Australian waters: (a) tropical oceanic species; (b) tropical neritic species; and (c) temperate neritic species (after Jeffrey & Hallegraeff 1990). These assemblages support different marine food chains and are likely to have different sensitivities towards nutrient and pollutant stress.

the eastern Arafura Sea and northern Gulf of Carpentaria, possibly originating in the Aroe Island upwelling system. Table 1 summarises typical phytoplankton chlorophyll biomass values in various oceanic, coastal and estuarine Australian waters. Significant increases in chlorophyll biomass, exceeding these average values, are one of many signs of cultural eutrophication (see below). Well-studied phytoplankton communities in the Australian region are those of the North West Shelf and the Great Barrier Reef region, the New South Wales coast, and eddy systems of the East Australian Current.

North West Shelf and Gulf of Carpentaria: The phytoplankton chlorophyll biomass (10–55 mg m^2) of the warm continental shelf waters of north-west Australia, the Timor Sea, the Gulf of Carpentaria and Arafura Sea is significantly higher than that of other parts of the Eastern Indian Ocean. This is reflected in the fisheries resources of the region. The North West Shelf is the site of a productive demersal trawl fishery and the Gulf of Carpentaria supports a productive prawn fishery. In summer the North West Shelf region is invaded by nutrient-rich tropical waters, and a variety of processes such as tidal currents, internal waves and cyclone mixing carry these nutrients into the bottom waters of the shelf. Nutrient enrichment in the eastern Arafura Sea and northern Gulf of Carpentaria possibly originates in the Aroe Island upwelling system. The phytoplankton community of these waters is basically a diatom flora (*Bacteriastrum, Chaetoceros, Rhizosolenia, Thalassionema*), except on occasion of episodic cyanobacterial blooms of *Trichodesmium*

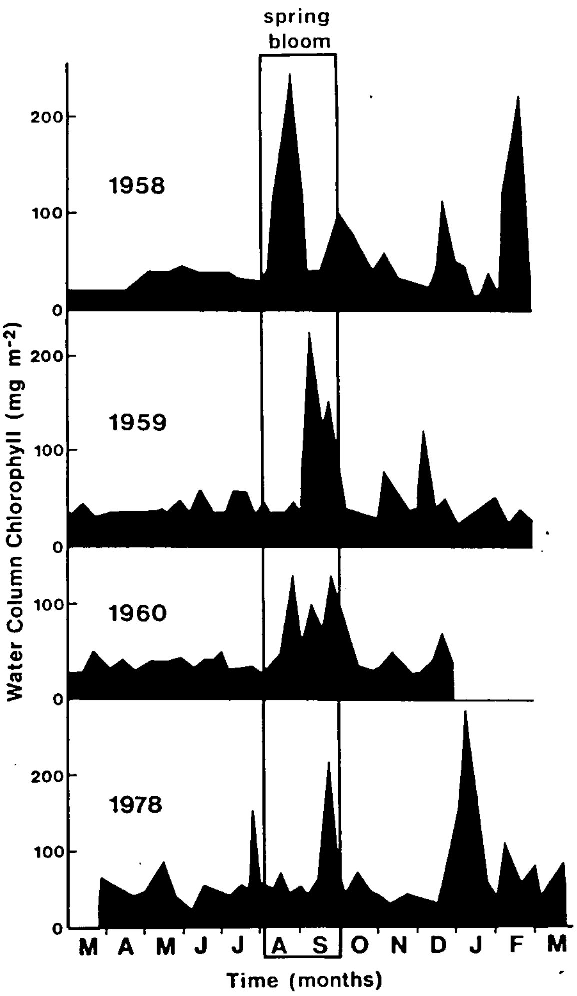

Figure 3: Annually recurrent phytoplankton blooms (as water column chlorophyll *a*) in Sydney coastal waters in the years 1958, 1959, 1960 and 1978 (after Hallegraeff & Jeffrey 1993). It would be of considerable interest to reexamine the phytoplankton species successions and nutrient status of these waters following the commissioning of the three new sewage outfalls in the area in 1990–91.

(Hallegraeff & Jeffrey 1984). Symbioses of algae within algae, or algae within animals are a feature of this tropical assemblage.

Great Barrier Reef region: The warm oligotrophic waters of the Coral Sea are characterised by low phytoplankton concentrations (15–20 mg m^2), with a dominant contribution (more than 50%) of picoplanktonic cyanobacteria and coccoid eukaryotes (Furnas & Mitchell 1986). Limited upwelling occurs along the outer margin of the Great Barrier Reef and the strong tides of the region periodically pulse deep nutrient-rich waters into the coral reef lagoons during the flood tide (Andrews & Gentien 1982). This results in diatom blooms (*Nitzschia, Pseudonitzschia)* close to the outer reef barrier (30–70 mg m^2). The Great Barrier Reef, the largest assemblage in the world of living coral-symbiotic dinoflagellate (zooxanthellae) communities, therefore represents an ecological response to tropical upwelling. Any change in the nutrient status or turbidity of these coral reef waters is likely to have significant impact on the diverse tropical benthic communities it supports. Episodic cyanobacterial blooms of *Trichodesmium* are also a prominent feature of the waters of the Great Barrier Reef (Revelante & Gilmartin 1982; see below).

New South Wales coastal waters: Coastal waters off Sydney are characterised by a series of sharp chlorophyll peaks (more than 10 times normal algal biomass) due to short-lived diatom blooms (*Lauderia, Pseudonitzschia, Rhizosolenia, Thalassiosira*) which usually occur in spring, early summer and autumn. This phenomenon, first recognised in the 1930s (Dakin & Colefax 1933), was documented in more detail in 1958–60 (Humphrey 1963) and 1978–79 (Hallegraeff 1981) (Figure 3). Surveys in 1981 and 1984 (Hallegraeff & Jeffrey 1993) demonstrated that these diatom blooms are a feature of the entire New South Wales coastline, from Cape Hawke in the north (32°S) where the East Australian Current separates from the coast, to Maria Island off Tasmania in the south (43°S). These blooms result from nutrients brought into surface waters by the shoreward transport of deep continental slope waters, induced by the action of the East Australian Current and its associated eddies. These phytoplankton peaks (100–280 mg chlorophyll m^2) must have profound significance for New South Wales coastal fisheries and coincide, for example, with the spring time spawning migrations of gemfish. It would be of considerable interest to re-examine the nutrient and phytoplankton status of Sydney coastal waters following the commissioning of three new submarine sewage outfalls in 1990–91. Extensive red tides by the dinoflagellate *Noctiluca scintillans* in Sydney coastal waters in early 1993 could well be an early sign of eutrophication.

East Australian Current eddies: Warm-core eddies of the East Australian Current are parcels of Coral Sea water which pinch off from meanders of the East Australian Current and drift southwards into the cooler Tasman Sea. As the surface waters cool and sink, the thermocline becomes eroded and convective overturn leads to a progressively deeper surface mixed layer of up to 300 m depth. Nutrients brought up by these deep mixing processes recirculate into the photic zone causing enrichments and diatom blooms (*Pseudonitzschia, Rhizosolenia*), especially at the eddy centre and sometimes also at the western boundaries where the eddies interact with the continental shelf (Jeffrey & Hallegraeff 1980, 1987). Valuable pelagic fish such as tuna tend to congregate at temperature discontinuities associated with eddy systems.

Estuaries

There is a growing public concern about the environmental quality of most major rivers, estuaries and coastal waters near Australia's large population centres where discharges of industrial, domestic and agricultural wastes are raising the nutrient levels in the water. Phytoplankton species that have always been present in low concentrations can respond to this increase by growing to bloom proportions (millions of cells per litre). Bloom-forming algae can become so densely concentrated that they generate anoxic conditions resulting in indiscriminate kills of both fish and invertebrates, especially in sheltered bays. The dinoflagellate *Scrippsiella trochoidea* has caused red-brown seawater discolourations and fish kills; for example, in the Hawkesbury River (New South Wales) and West Lakes (South Australia) (Hallegraeff 1991). This species (under the name *Glenodinium rubrum*) has also been implicated in causing fish kills in Sydney Harbour as early as 1890 (Whitelegge 1891). Other estuaries with annually recurrent algal bloom problems are Port Phillip Bay (Victoria), Huon and Derwent rivers (Tasmania), the Port River (South Australia) and the Peel-Harvey estuary and Cockburn Sound (Western Australia). As in many other parts of the world (Anderson 1989, Smayda 1990, Hallegraeff 1993), in the past two decades there has been a recognition of an apparent increase in the frequency, intensity and geographic distribution of such harmful algal blooms in the Australian region (Hallegraeff 1992).

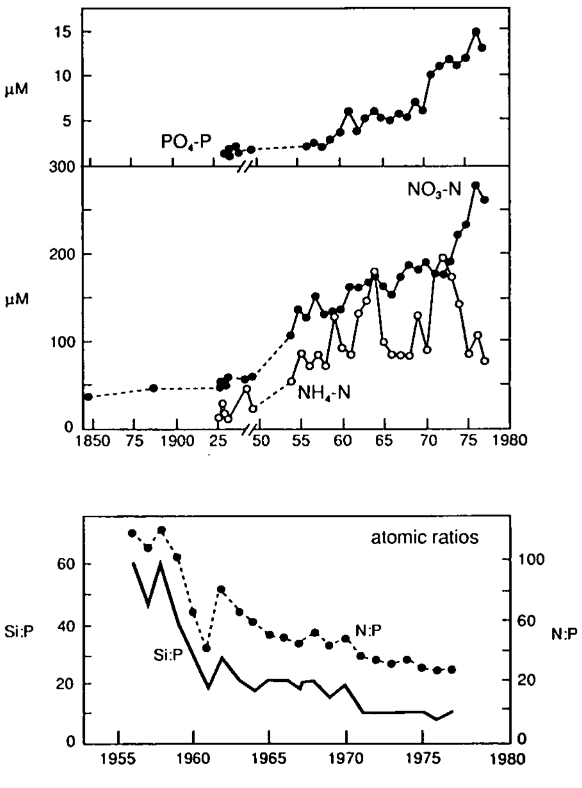

Figure 4: Long-term nutrient data for the River Rhine in Europe, showing evidence for a 7.5–fold increase in phosphate loading, a 3–fold increase in nitrate levels, but more seriously a significant decrease in Si:P and N:P nutrient ratios. Such altered nutrient ratios favour blooms of nuisance flagellate species (*Phaeocystis pouchetii, Chrysochromulina polylepis*) replacing the normal 'wholesome' spring and autumn blooms of siliceous diatoms (after Hallegraeff 1993).

Issues

Cultural eutrophication

Overseas experience from areas such as Hong Kong harbour, the Seto Inland Sea in Japan and northern European coastal waters indicates that 'cultural eutrophication' from domestic, industrial and agricultural wastes can stimulate harmful algal blooms. Figure 4 illustrates the pattern of long-term increase in nutrient loading of coastal waters of the North Sea. Since 1955 the phosphate loading of the River Rhine has increased 7.5–fold, while nitrate levels have increased 3–fold. This increase has resulted in a significant 6–fold decline in the silicate:phosphorus ratio, because long-term reactive silicate concentrations (a nutrient derived from natural land weathering) have remained constant. More recently, improved wastewater treatment has also been causing increases in the ammonia:nitrate ratio of River Rhine discharge (see review by Hallegraeff 1993). It is important to realise that the nutrient composition of treated wastewater is never the same as that of the waters in which it is being discharged, and indiscriminate reductions in nutrient discharges are therefore not addressing the problem of changing nutrient ratios of coastal waters. There is considerable concern (Smayda 1990) that such altered nutrient ratios may favour blooms of nuisance flagellate species which replace the normal spring and autumn blooms of 'wholesome' siliceous diatoms. Changed patterns of land use, such as deforestation, can also cause shifts in phytoplankton species composition by increasing the concentrations of humic substances in land run-off (Figure 5). Only three comparable long-term phytoplankton and nutrient data are available for Australian waters, viz. the CSIRO hydrological stations at Port Hacking (New South Wales), Maria Island (Tasmania) and Rottnest Island (Western Australia).

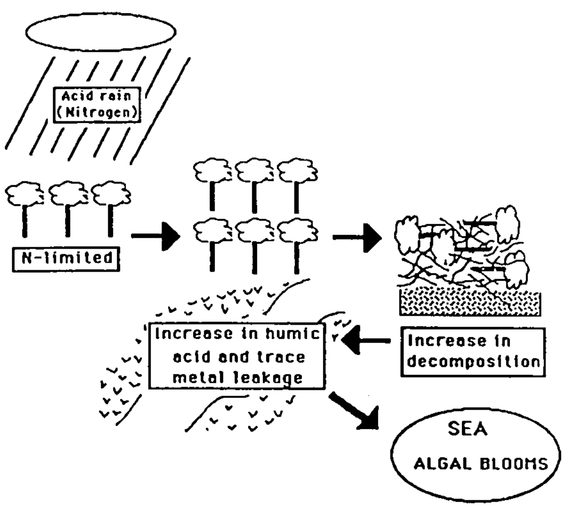

Figure 5: Changed patterns of land use, such as deforestation, increase the concentration of humic substances and trace metals in land runoff and can cause shifts in phytoplankton species composition (from Graneli et al. 1989).

Cyanobacterial blooms

Trichodesmium blooms in tropical waters

The filamentous cyanobacterium *Trichodesmium erythraeum* is the most common 'red tide' organism in tropical Australian coastal and oceanic waters. At the start of the bloom, the filaments usually appear throughout the water column, but during late-bloom stages the development of strong gas vacuoles causes a massive rise of the alga to the surface layers. This species produces seasonal (February–April) water blooms in the Java, Banda, Arafura and Coral seas, and from there the East Australian Current and Leeuwin Current transport the algal masses (covering up to 40 000 km^2) as far south as Sydney (Jervis Bay) and Perth (Albany) respectively. The alga is perceived as a nuisance to swimmers on Australian beaches and has significant impacts on recreation, but harmful effects on humans or marine life have seldom been reported. It is not yet known whether Australian strains of *Trichodesmium* produce neurotoxic compounds similar to those reported from populations from the Virgin Islands (Hawser et al. 1991). *Trichodesmium* red tides ('sea sawdust') were observed as early as 1770 during Captain Cook's voyage through the Coral Sea, and in a strict sense, they should be regarded as completely natural events. Differentiated cells in the centre of each minute *Trichodesmium* colony are capable of fixing atmospheric nitrogen, which allows the alga to

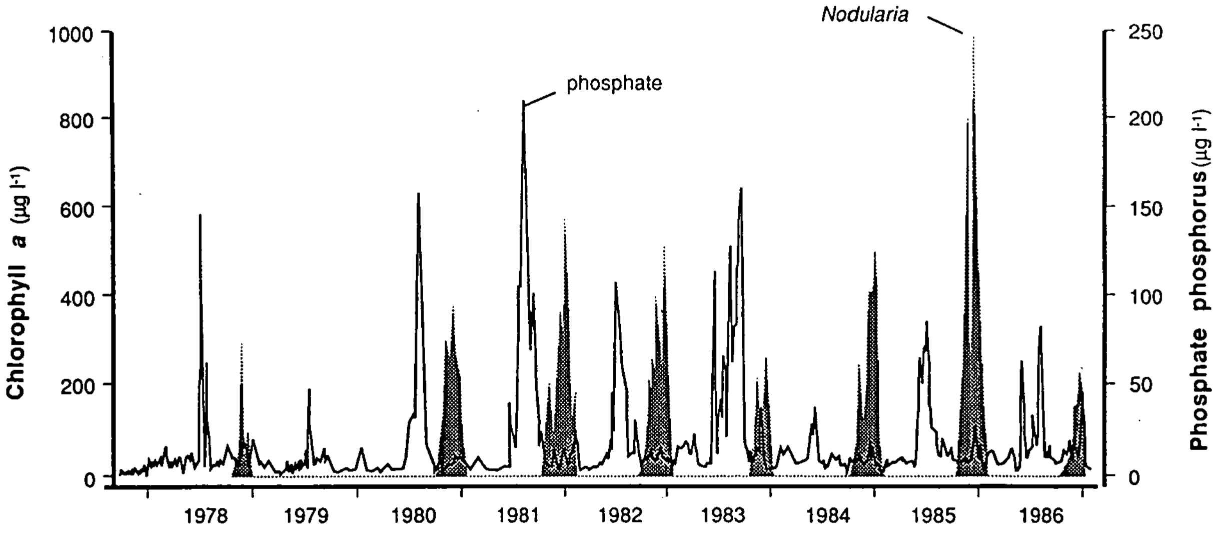

Figure 6: Long-term pattern of toxic *Nodularia spumigena* cyanobacterial blooms in the Peel-Harvey estuary, Western Australia, related to phosphate discharge from agricultural fertilisers in the previous winter period (after Hillman, Lukatelich & McComb 1990).

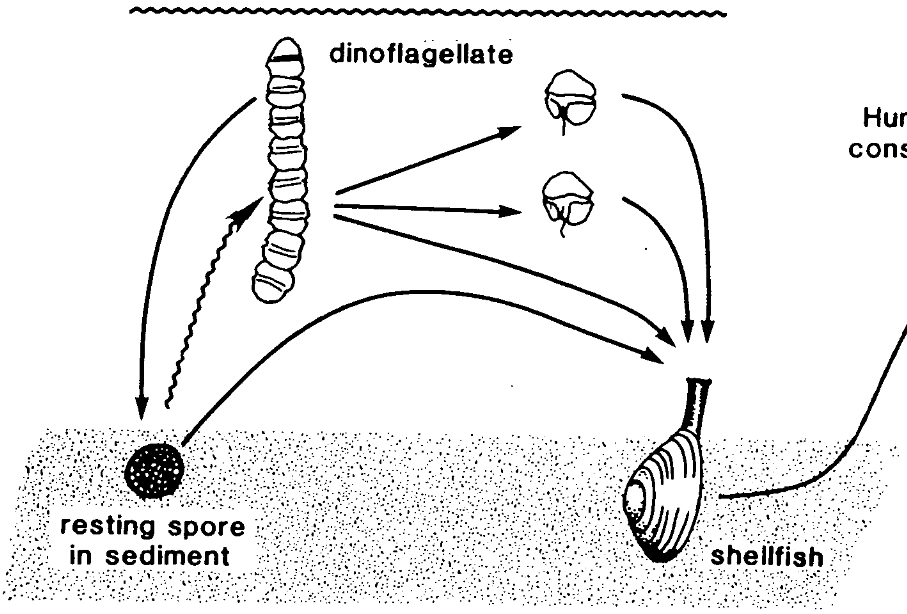

Figure 7: The production by plankton dinoflagellates of potent neurological toxins which accumulate in shellfish and thereby can poison human consumers. Until 1980, this phenomenon of paralytic shellfish poisoning (PSP) was unknown from the Australian region and there exist, for example, no historic Aboriginal reports of poisoning after shellfish consumption. There is strong circumstantial evidence that the PSP dinoflagellate *Gymnodinium catenatum* was introduced into Tasmanian waters via ships' ballast water originating from either Japan or Korea.

thrive under nutrient-impoverished oceanic conditions. It is possible, however, that coastal nutrients (especially phosphates) can stimulate or prolong the blooms once they are washed inshore. It is a contentious issue, at present, whether or not the apparent increase in *Trichodesmium* blooms in the Great Barrier Reef region is caused by coastal eutrophication (Kinsey 1991).

Toxic cyanobacterial blooms in freshwaters and estuaries

Toxic blooms of the brackish water cyanobacterium *Nodularia spumigena* were first recorded in Australia from Lake Alexandrina as early as 1878 (Francis 1878). However, their increasing frequency and distribution in the Gippsland Lakes system in eastern Victoria, in Orielton Lagoon in Tasmania, in the Darling and Murray rivers of South Australia and in the Peel–Harvey, Cockburn Sound and Vasse–Wonnerup estuaries of Western Australia, appears to be related to phosphorus from agricultural fertilisers and sewage being washed into the river systems (Hillman, Lukatelich & McComb 1990; Figure 6). This brackish water organism produces the hepatotoxic peptide, nodularin, which has killed domestic and wild animals that drink from the shores of eutrophic ponds, lakes and reservoirs (Main et al. 1977).

The species *Anabaena circinalis* (producer of the neurotoxic alkaloid, saxitoxin and derivatives) and *Microcystis aeruginosa* (producer of the hepatotoxic polypeptide, microcystin) cause similar problems, but tend to be confined to truly freshwater environments. In November–December 1991, a massive bloom of *Anabaena circinalis* throughout 1200 km of the Darling River system contaminated essential drinking water supplies for country towns, killing sheep, cattle and wildlife. This bloom has been attributed to a combination of sluggish river flow, high water temperatures and the build-up in sediments of nutrients from agricultural fertilisers, feedlot wastes and especially sewage from townships.

Impact of algal blooms on aquaculture

The considerable expansion of marine and freshwater aquaculture production in Australia (O'Sullivan, 1990) is focusing attention on phytoplankton species that can contaminate shellfish with human neurotoxins or damage the sensitive gill tissues of finfish, especially when held in intensive cage culture systems.

Algal blooms affecting shellfish aquaculture

Until the late 1980s, the phenomenon of paralytic shellfish poisoning (PSP; Figure 7) was unknown from the Australian region. Tasmania was the first State in Australia to suffer major problems with toxic dinoflagellates contaminating farmed shellfish. Blooms of the dinoflagellate *Gymnodinium catenatum* in 1986, 1987, 1991 and 1993 caused the temporary closure of up to 48 shellfish farms for periods of up to six months (Hallegraeff et al. 1989). The localised distribution of *G. catenatum* around the port of Hobart and the absence of plankton and benthic cyst records of this species prior to 1980 led to speculation that this organism had been introduced. Cyst stages of this species have been detected in ships' ballast water entering Australian ports from Japan and Korea (Hallegraeff & Bolch, 1992).

Red tides by the toxic dinoflagellate *Alexandrium minutum* were first recognised in the Port River area near metropolitan Adelaide in October 1986 (Hallegraeff, Steffensen & Wetherbee 1988). This species now produces annually recurrent red water blooms (up to 10^8 cells per l) in the period September–November (Cannon, 1990) and has also been detected in low concentrations in Western Australia and New South Wales. Wild mussels from the Port River area can be highly toxic to humans (Oshima et al. 1989), but fortunately no commercial shellfish farms are located in the affected area. Plankton and cyst surveys in Port River in 1983 failed to detect *A. minutum* in an area which now has recurrent blooms. This result has led to speculation that *A. minutum* could also be an introduced species, and genetic studies using ribosomal DNA sequencing have confirmed a close affinity between Australian and Spanish isolates of this species complex (Scholin, Hallegraeff & Anderson 1994). Finally, the toxic dinoflagellate *Alexandrium catenella* was first recognised in 1986 in Port Phillip Bay where it caused significant toxicity in wild mussels. Fortunately only minor effects on commercial shellfish farms have been reported (Hallegraeff et al. 1991). This species is also known from New South Wales (Port Jackson, Port Botany, Batemans Bay). Once an area has been infested with cyst-producing toxic dinoflagellates, there is little hope to eradicate the problem. The only solution is an avoidance strategy of regularly monitoring shellfish products for toxins and, on the basis of the results, imposing temporary closures of farms. Every attempt should be made not to spread the problem — for example, by resuspending cysts by dredging operations or by relaying shellfish stocks to non-infected areas.

Algal blooms affecting finfish aquaculture

A wide range of prymnesiophyte, diatom and dinoflagellate species can damage fishes' gills, either by purely physical mechanisms such as increased seawater viscosity due to secretion of algal mucilages, by chemical mechanisms such as the production of substances which affect cell permeability and/or cause a necrosis and sloughing of epithelial tissues of the gills and digestive system. While wild fish stocks have the freedom to swim away from problem areas, caged fish are extremely vulnerable to noxious algal blooms. In January 1989, a bloom of the raphidophyte flagellate *Heterosigma akashiwo* in Big Glory Bay, Stuart Island, New Zealand, killed NZ$ 12 million worth of cage-reared chinook salmon (Chang, Anderson & Boustead 1990). Within the Australian region, blooms of this species are known from Port Stephens (New South Wales), West Lakes (South Australia) and Cockburn Sound (Western Australia), but no major marine mortalities have been reported. The dinoflagellate *Gymnodinium mikimotoi* (related to the Norwegian fish-killer *Gyrodinium aureolum* Tangen 1977) has caused some mortality in fish farms in south-eastern Tasmania (Hallegraeff 1991) and a diatom bloom of *Chaetoceros criophilum* has been associated with irritation among caged fish in south-eastern Tasmanian waters. The diatom's spines (setae) can break off and penetrate the gill membranes of fish .

Sensitivity of phytoplankton to chemical pollutants

Chemical pollutants, including both organic and inorganic compounds, can cause selective inhibition of phytoplankton species, with wide ranging effects at higher trophic levels. Chlorinated organics, including DDT, dieldrin, chlordane, polychlorinated biphenyls (PCBs) and chlorophenols, are of particular concern because they readily absorb to particulates and sediments, are resistant to degradation and have the potential to bioaccumulate. PCB levels (1–10 mg per l) have been reported in Botany Bay and Port Phillip Bay, and may depress the growth of sensitive diatoms (Fisher & Wurster 1973). Pesticides in agricultural land run-off may inhibit zooplankton grazing, thereby stimulating algal blooms. The effects on phytoplankton of effluents from chlorine-bleaching pulp and paper mills, including chlorate, chlorophenols, resin acids and chlorinated lignin derivatives, are also receiving attention (J. Stauber, pers. comm.). Some of the chlorophenolic compounds, such as the more substituted chlorocatechols, are toxic to freshwater algae and marine diatoms (Kuivasniemi, Eloranta & Knuutinen 1985), with chlorate being particularly toxic to phytoplankton. Petroleum hydrocarbon contamination (1–23 mg per l) has also been documented in Australian waters — for example in Port Phillip Bay (Smith & Burns 1978). While occasional oil spills may lead to acute toxic effects (ie plankton mortality) chronic effects are less likely to occur because of degradation and removal processes.

The Australian coastline has a number of locations with high heavy metal concentrations derived from metal smelting, mining and other industrial processes. These include Townsville harbour (Queensland), Newcastle, Lake Macquarie, Sydney Harbour, Botany Bay, Lake Illawarra and Port Kembla (New South Wales), Corio Bay (Victoria), the north coast of Tasmania, the Derwent River estuary and Macquarie Harbour (Tasmania), Spencer Gulf (South Australia) and Cockburn Sound and Albany Harbour (Western Australia). The heavy metals of principal concern are zinc, copper, lead, mercury, cadmium, nickel, chromium, silver, arsenic and selenium. Maximum concentrations of cadmium (2 mg per l), lead (10 mg per l) and zinc (142 mg per l) have been reported in the Derwent River, while mercury concentrations up to 1 mg per litre have been found in Victorian coastal waters. However, it is the physico-chemical form of the metal and not its total concentration which determines its bioavailability and toxicity to phytoplankton. Metal speciation is influenced by temperature, pH, salinity, nutrients and the presence of other metals and organic chelators such as humic and fulvic acids (Stauber & Florence 1987, 1990). Dissolved metals, including the free metal ions, are considered to be the most toxic, but these readily absorb onto colloidal particles and/or complex with humic acids. In contrast, lipid soluble metal complexes produced in metal flotation processes can diffuse directly into the cell where both the metal and the ligand exert toxic effects. Low concentrations of metals can stimulate but higher concentrations inhibit algal growth. In general, the dinoflagellates, diatoms and cyanobacteria are the most sensitive, whereas the green flagellates are the most resistant. The sensitivity of different phytoplankton species varies by orders of magnitude, and the major impact of heavy metal stress would therefore be a shift in species composition (Thomas & Seibert 1977, Fisher 1981).

Global climate change

Global climate change and El Nino phenomena have the potential to modify existing current regimes in the Australian region and thereby alter natural nutrient enrichment patterns (Harris et al. 1988). Furthermore, ozone depletion has the potential to alter the species composition and depth distribution of phytoplankton organisms sensitive to ultraviolet radiation (Smith et al. 1992). Again, the most serious consequences would be for the structure of marine foodchains and possible alterations to nutritional quality of microalgae farmed in outside mariculture operations.

Management

Criteria for strict control of nutrient discharge must be developed for all Australian coastal waters and rivers if these waters are to be managed in a sustainable manner, and if conflict situations arising from new developments are to be solved on a sound scientific basis. Algal growth is controlled by both macronutrients (nitrogen, phosphorus and silicate) and micronutrients (vitamins, trace elements and chelators). In the past, simple phytoplankton biomass approaches to the eutrophication problem (eg chlorophyll analyses), often focusing on a single limiting nutrient (eg phosphate in freshwaters), have failed to achieve an adequate understanding of the eutrophication processes necessary to develop successful monitoring and management strategies. Wastewater treatment should not be confined to simply reducing nitrogen and phophorus, but silicate and other additions may be required to maintain a 'normal' phytoplankton species composition. Different geographic regions in Australia will have different carrying capacities for nutrient loading, dependent upon silicate concentrations and micronutrients in land runoff. Personnel responsible for management decisions on pollutant loadings of rivers and coastal waters (including decisions on agricultural and forestry practices in catchment areas) should be made fully aware that one probable outcome of increased nutrient loading will be an increase in harmful algal blooms. The discharge of ships' ballast waters should be strongly discouraged near marine parks or sensitive aquaculture areas when there is reason to believe they may contain harmful marine organisms non-endemic to the Australian region.

Conclusions

Adequate phytoplankton baseline studies are lacking for many parts of the Australian coastline and long-term nutrient data are virtually absent. This makes it extremely difficult to recognise the introduction of immigrant species (for example, those discharged via ships' ballast water) or to provide early warning of cultural eutrophication. Studies of algal microfossils in dated sediment depth cores can provide helpful indicators of environmental change. Overseas experience in North America, Europe and Japan has convincingly demonstrated that domestic and industrial nutrient discharges to inland and coastal waterways have the potential to increase phytoplankton biomass levels as well as dramatically alter phytoplankton species composition. While as yet there is no evidence that Australian offshore waters are affected, most rivers, estuaries and coastal waters near Australia's large population centres show signs of cultural eutrophication. Examples are Tuggerah Lakes and the Hawkesbury River (New South Wales), Gippsland Lakes and Port Phillip Bay (Victoria), Derwent and Huon rivers (Tasmania), Port River and West Lakes (South Australia) and the Peel-Harvey estuary and Cockburn Sound (Western Australia). Early signs of eutrophication include increased phytoplankton biomass and increased turbidity, novel phytoplankton blooms of species not previously recorded in bloom proportions, and in extreme cases, fish kills due to the generation of anoxic conditions or in some cases algal toxin production. The management of nutrient discharges to inland and coastal waterways is crucial to arrest the increasing impact of harmful algal blooms. Chemical pollution studies in Australian waters are still in their infancy, but the influence of growing mining and oil exploration ventures on marine phytoplankton communities should not be ignored in coastal monitoring programs. Global climate change has the potential to modify existing current regimes and natural nutrient enrichment patterns in the Australian region, as well as alter the species composition and depth distribution of phytoplankton organisms sensitive to ultraviolet radiation. Human-induced shifts in species composition of marine phytoplankton communities can have far-reaching consequences for the structure of the entire marine food chain leading to edible fish.

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The technical paper by Dr. V. Hallegraeff was reviewed by Dr. S. Jeffrey, CSIRO Division of Fisheries, Hobart, and Dr. L. Hammond, Director, Victoria Institute of Marine Sciences, Melbourne.

Saltmarsh

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Introduction

Definition

Coastal (intertidal) saltmarsh is defined as an intertidal plant community complex dominated by herbs and low shrubs. For the most part there is a clear structural distinction between saltmarsh and mangrove (which is defined as an intertidal community dominated by trees). However, at the southern limit of mangrove distribution, in Victoria, the structural distinction becomes blurred with stunted mangroves (*Avicennia marina*) being of lower stature than the tallest saltmarsh shrubs (*Sclerostegia arbuscula*). In these circumstances the separation of the two communities is floristic. Similarly, while at a few localities there is an intermingling of the lowest saltmarsh and the uppermost seagrasses(Figure 1), seagrass and saltmarsh are distinguished on the basis of floristics and physiognomy.

Composition and biogeographic variation

Saltmarsh vascular plant communities are characterised by being species poor and often clearly dominated by single species. Within a single marsh there is frequently a zonation from low to high elevations on the shore (e.g. Figure 2). For example, on the New South Wales coast the most frequent ordering of community dominants from low to high marsh (*Sarcocornia quinqueflora–Sporobolus virginicus–Juncus kraussii*) is subject to local modification along drainage lines and in depressions.

On the much larger scale of the continent itself, there is a striking trend of increased species richness and community complexity with increasing latitude (Saenger et al. 1977, Specht 1981, Adam 1990). Saltmarshes in northern Australia, although extensive, are very species poor, with frequently less then ten vascular plant species in their total flora. Compared to northern Australia, saltmarshes in Victoria and Tasmania which cover much smaller areas, frequently support at least three times as many species. Not only is the flora of individual sites much greater at higher latitudes, but the regional saltmarsh flora is also much larger.

In southern Australia a biogeographic distinction can be drawn between saltmarshes on arid or seasonally arid (Mediterranean climate) coasts and those on temperate coasts with relatively high rainfall (Bridgewater 1982, Adam 1990). On drier coasts saltmarsh vegetation is characterised by a diversity of succulent, shrubby chenopods with a tendency to more open vegetation towards the upper tidal limit, while on wetter shores vegetation is denser with more grassland and sedgeland communities. On the east coast there is a gradual transition from the more species poor, frequently *S. virginicus*– dominated, subtropical marshes to southern temperate marshes; but at Jervis Bay there is an important biogeographic nodal point where saltmarshes support a suite of species at, or close to, their northern distributional limit (Adam & Hutchings 1987). A similar trend is seen on the west coast of Australia but here the estuaries are more widely spaced, making the transition less apparent.

More permanently brackish sites in upper estuaries or on the shores of some coastal lagoons support floristically different communities although there is a continuum of variation between fully saline and brackish sites. Where topography permits, saltmarsh grades upwards into shrubland or forest characteristically dominated by *Casuarina* species (*C. glauca* in the south-east, *C. obesa* in the west), *Melaleuca* species or *Eucalyptus* species (for example *E. robusta* in New South Wales).

Although there is a high degree of endemism at the species level in the saltmarsh flora of Australia at generic and family level there is a strong similarity between Australian saltmarshes and those elsewhere in the southern hemisphere. In addition, many affinities to saltmarshes in the northern hemisphere are displayed (Adam 1990). The patterns of variation in structure and composition of Australian saltmarshes also are very similar to those exhibited on other continents. When viewed in a world context therefore, Australian saltmarshes are not as

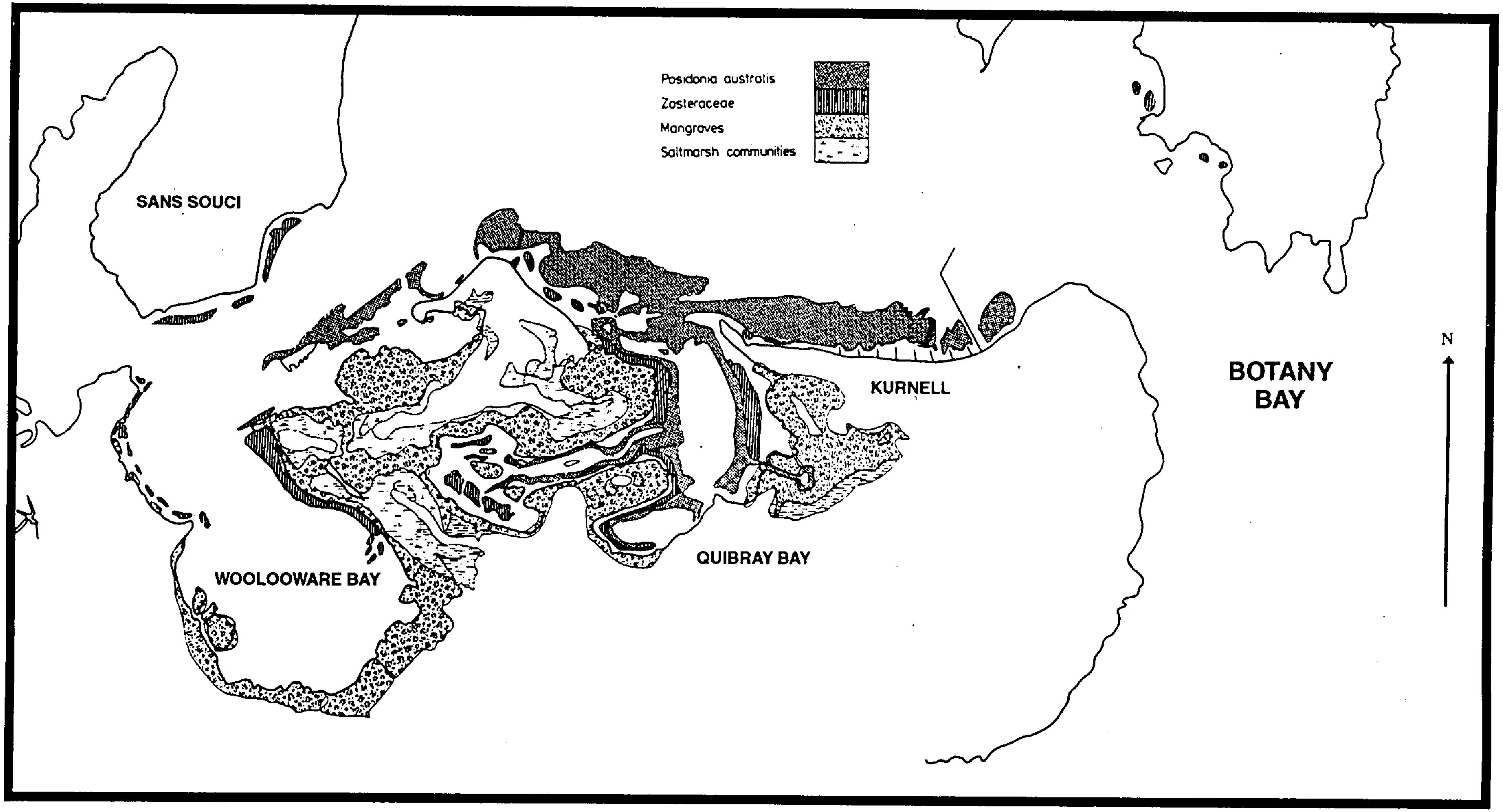

Figure 1: Distribution of seagrass beds, mangroves and saltmarsh in the southern part of Botany Bay, New South Wales (redrawn from West et al. 1985).

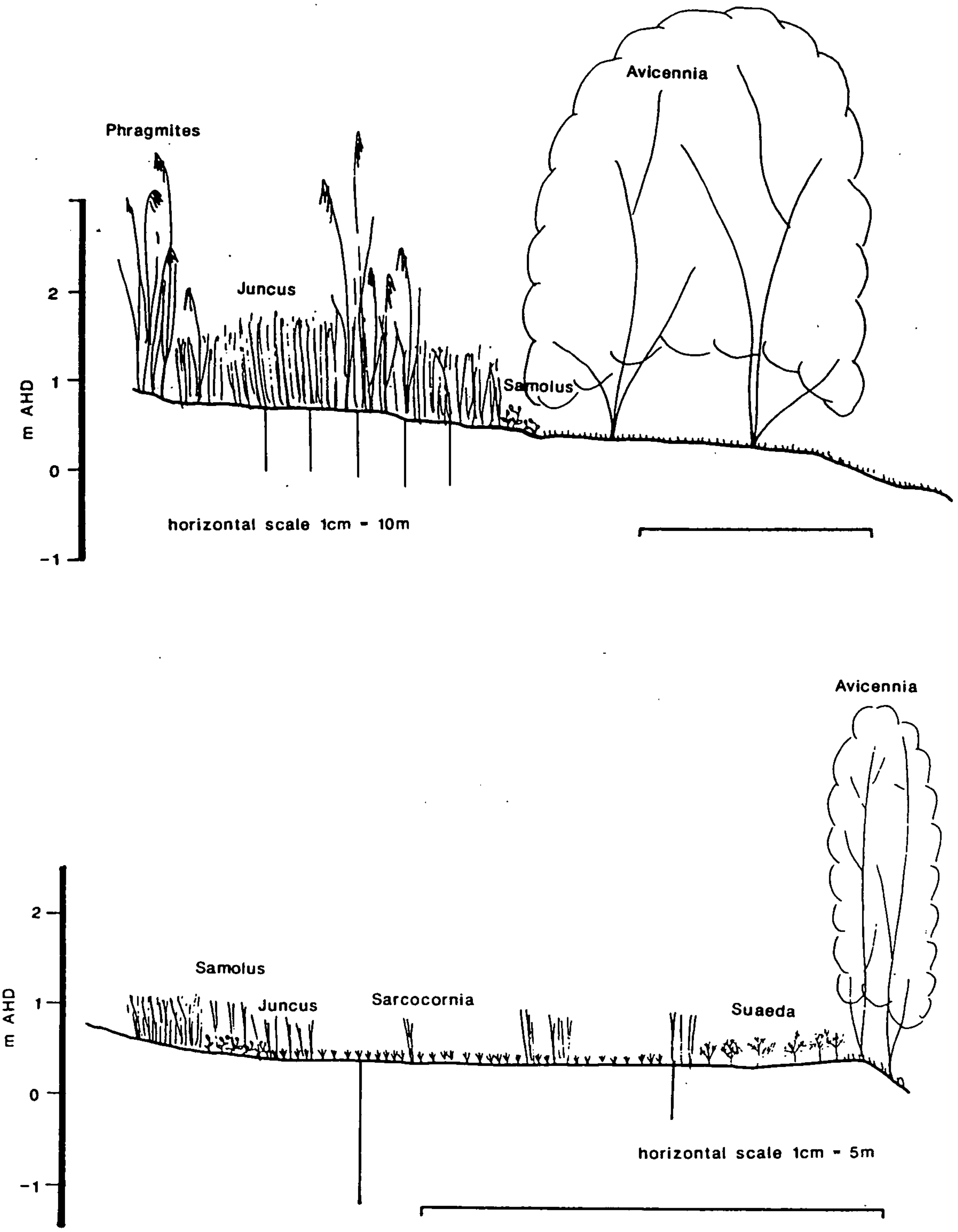

Figure 2: Relationship of saltmarsh and mangroves to elevation (expressed in m above Australian Height Datum) at various sites along the Georges River, New South Wales (from Mitchell & Adam 1989a).

distinctly 'Australian' as are the various terrestrial communities of the continent.

The vascular plants of saltmarshes are referred to as halophytes (salt-loving plants); but whereas a number of the species may show improved growth at moderate salinities there is no evidence that any are obligate halophytes in that it is possible for them to be grown experimentally under non-saline conditions. However, salt tolerance is achieved at the expense of growth (Adam 1990) so that in the field, under non-saline conditions, halophytes are out-competed by non-salt tolerant species.

Distribution

Saltmarsh is found on the coasts of all States. Where mangroves also occur, saltmarsh occupies higher elevations (except in Victoria where the introduced grass *Spartina anglica* extends seaward of the mangroves). This characteristic zonation has been interpreted as reflecting a successional relationship (Pidgeon 1940) but there is little evidence to support this view (Mitchell & Adam 1989a). In northern Australia both mangrove and saltmarsh may be restricted to the lower, more frequently flooded intertidal zone while the upper intertidal takes the form of extensive hypersaline flats with only very sparse and localised vegetation.

Extensive saline areas are found in the arid and semi-arid zones of Australia — both naturally and as a result of landuse practices. Physiognomically, and floristically at generic level, the flora of these inland saline habitats is similar to that of saltmarsh on climatically dry coasts. Indeed, it can be difficult to determine the upper limit of saltmarsh on some arid coasts as the intertidal saltmarsh merges with the fully terrestrial vegetation.

On cliffs and headlands exposed to very high inputs of salt spray, plant communities floristically identical to those of intertidal saltmarsh are present (Adam, Wilson & Huntley 1988). However, the total area of such vegetation is small.

The best available estimate of the area of estuarine saltmarsh in Australia has been provided by Bucher and Saenger (1991) (Table 1). This estimate, based mainly on remote sensing data, does not differentiate between closed saltmarsh vegetation and more open high level tidal flats because these two habitats are part of a continuum of variation, and separation on the basis of remote sensing data is difficult. In addition, saltmarsh on open shores was not covered by the survey (P. Saenger pers. comm.). Inclusion of open coast areas in the analysis would probably increase the estimate for northern Australia where there are extensive saltmarshes on open coasts (such as around Broome).

There is a tendency in the literature to regard saltmarsh and mangrove as being geographically mutually exclusive, with saltmarsh as the temperate equivalent of the tropical mangrove. However, the data from Bucher and Saenger (1991) demonstrate that the greatest extent of saltmarsh occurs in tropical Australia and that the total area is considerably greater than that of mangrove shores. Limited data are available on the distribution and extent of particular saltmarsh vegetation types in southern Australia (Kirkpatrick & Glasby 1981, Bridgewater 1982, Adam, Wilson & Huntley 1988) but none as yet for northern Australia.

Value and usage

Productivity

Spartina alterniflora – dominated saltmarshes on the east coast of the United States of America are amongst the most productive natural communities on earth. There has been a tendency in the popular literature to generalise from this observation and assume that all saltmarshes are equally productive, but this assumption is not supportable. The majority of data on saltmarsh productivity are from north America, with fewer reports from elsewhere in the northern hemisphere, and these data indicate a considerable range of productivities. Given the physiological costs to plants (Adam 1990) of living in a saline environment however, all saltmarsh productivity could be regarded as being high compared with that from terrestrial grasslands at similar latitudes.

An important difference between Australian and overseas marshes needs to be considered before extrapolating from overseas data to Australia. Saltmarshes around much of the Australian coast

Table 1: Areas of estuarine saltmarsh (km^2). Source: Bucher and Saenger (1991)

NSW	VIC	QLD	WA	SA	TAS	NT	TOTAL
57.16	124.88	5321.91	2965.02	83.67	36.99	5004.84	13 594.47

occupy the upper intertidal, and even at their lowest limit on the shore are not subject to daily flooding by tides. The zone around much of Australia equivalent to that of the most productive *S. alterniflora* marshes in North America is occupied by mangroves. In physiological terms, the habitat of Australian marshes is more stressful than that of *S. alterniflora*, with greater fluctuation in salinity and higher maximum salinities. This greater stress is likely to lower the maximum potential productivity.

Most estimates of saltmarsh biomass and productivity refer only to above ground material. The proportion of above ground productivity transferred to roots may be high, and the root:shoot weight ratio of plants may vary with salinity (Adam 1990). Productivity estimates also frequently exclude algal productivity. The soil surface in the wetter parts of saltmarshes may have a dense covering of algae (both microalgae such as diatoms and macroalgae such as *Chaetomorpha* and *Enteromorpha*) and these algal layers may be important contributors to local productivity. The significance of algal productivity to the ecosystem might be proportionately greater than its biomass contribution given that much higher-plant productivity becomes detritus whereas algae represent a high quality food source more directly available to grazers (Pomeroy et al. 1981).

Although claims have been made for high productivity in Australian saltmarshes there have been very few studies of either standing crop or productivity. The above ground productivity values reported for *Juncus kraussii* saltmarsh in the Blackwood River estuary (Western Australia) of 0.3–1.3 kg dry weight m^2 per year by Congdon and McComb (1980) are towards the lower end of the range from temperate northern hemisphere saltmarshes. In the absence of further studies generalisations regarding productivity in Australian saltmarshes are unwarranted. With the exception of saltmarshes deliberately grazed by livestock, consumption of biomass by herbivores appears to account for a small percentage of productivity in the north American situation (Pomeroy & Weigert 1981). Whether this generalisation holds in Australia is not known.

In the absence of a large energy and nutrient flow through direct grazing, and with — at least in most sediments — little accumulation of autochthonously-produced organic material, much of the primary production of saltmarshes will be utilised in detrital pathways. These pathways, which include microbially-mediated decomposition, take place either within saltmarshes or in adjacent waters.

In the popular literature the assumption of high productivity is accompanied by acceptance of the 'outwelling hypothesis', by which coastal wetlands act as a source of detritus and nutrients exported to offshore waters. This argument has been used to justify the conservation of intertidal wetlands but, as Nixon (1980) emphasised, is not well supported by data. Internationally the role of intertidal wetlands in the ecology of estuarine and near shore waters is the subject of considerable research. The current understanding (Jansson et al. 1988) is that there is substantial internal recycling of energy within saltmarshes as well as some fluxes to adjacent ecosystems (which vary in magnitude and direction between sites). Nutrient budgets also vary between sites. However, even where budgets are balanced overall, the transformation of the chemical form of nutrients within saltmarshes may be an important mechanism for increasing bioavailability. Data on the linkages of saltmarshes to adjacent waters in Australia are lacking.

Habitat

Saltmarsh provides habitat for numerous organisms of both terrestrial and marine origin.

A high proportion of the commercially important fish species in Australia are estuarine dependent (requiring estuarine habitat at some stage of their life cycle). Currently for example, about 60% by weight and 70% by value of the commercial catch in New South Wales is estuarine dependent (Leadbitter & Doohan 1991). Many of these estuarine dependent species utilise intertidal wetlands for part of their lives, particularly as juvenile nursery habitat. The most important habitats are seagrasses and mangroves: Australian saltmarshes, unlike those overseas, generally have few permanent creeks and pans and so provide few fish habitats. Nevertheless, studies in both Queensland and New South Wales indicate that at least some saltmarshes may provide habitats utilised by fish (Gibbs 1986, Morton, Pollock & Beumer 1987, Morton, Beumer & Pollock 1988). Further research on the importance of saltmarsh to fish is required.

Coastal wetlands are popularly identified as important habitats for birds. The number of species breeding in saltmarshes is small but upper marsh vegetation provides nest sites for some species (for example the white-fronted chat *Ephthianura albifrons*). A large part of the population of one of the rarest species in

Australia, the orange-bellied parrot (*Neophema chrysogaster*), overwinters on saltmarshes in Victoria where it feeds on the seeds of chenopods. Migratory waders feed largely on invertebrates in intertidal sand and mudflats but saltmarshes may provide secure high tide roosts. Conservation of waders is a matter for international concern and the Commonwealth is signatory to three agreements (the Ramsar Convention, the Japan–Australia Migratory Birds Agreement and the China–Australia Migratory Birds Agreement) which impose obligations to protect habitats utilised by migratory waders.

Little is known about the utilisation of saltmarsh by other faunal groups and this is a topic requiring further study.

Direct exploitation

There is limited direct exploitation of saltmarshes in Australia. In northern Australia many saltmarshes are accessible to livestock but the effects of grazing and trampling have not been studied. In southern Australia a number of saltmarshes are heavily grazed, but although this results in obvious changes to vegetation and soil structure the long term impacts are unknown.

Losses and threats

Reclamation

In settled areas saltmarshes have been reclaimed for port, industrial and housing development, road construction, parks and sports fields. In more recent decades saltmarshes at sites well removed from existing urban centres have been threatened by developments for recreation and tourism (marinas, resorts and canal estates). Construction of solar salt production ponds in Western Australia has also resulted in some loss of saltmarsh. Some reclamation for agriculture (mainly pasture) has occurred but this has probably involved much smaller areas than have been lost from freshwater wetlands on coastal floodplains. Although the pattern and extent of reclamation have been documented for particular locations there appear to be no inventories at State and national levels. Compared with the total extent of the habitat (Table 1) losses are likely to have been small, but they are concentrated in the south-east of the continent where the initial total area was small and where biodiversity is highest. The losses are therefore likely to be significant both nationally in terms of effects on biodiversity and regionally in terms of loss of habitat functions. Nevertheless, it is difficult to predict the specific impacts of losses in view of the paucity of quantitative information on ecosystem functions.

Degradation

A much larger area of saltmarsh than has been lost from reclamation has been damaged by various forms of habitat degradation.

Adjacent to settlements many saltmarshes are subject to illegal rubbish dumping and to disturbance through the construction and maintenance of easements for pipelines and powerlines. Vehicular use (4–wheel drive, trail bikes and 'BMX' bikes) alters the microtopography and drainage, leading to changes in vegetation. Even if access is prevented (an impossibility in most instances) recovery may take many years. Trampling — such as that associated with educational excursions — may cause long term damage to vegetation, in particular where succulent species predominate. Stormwater drains frequently discharge into saltmarshes. Apart from introducing gross pollutants, nutrients and weed propagules, freshwater discharge can cause local erosion and, through altering the salinity regime, promote the spread of fresh or brackish water species such as *Phragmites australis* and *Typha* species at the expense of more salt tolerant species (Pen 1983, Zedler, Paling & McComb 1990).

Saltmarshes are depositional sinks and pollutants from both terrestrial and marine sources may accumulate in them. Little is known about the nature and impact of such pollutants on saltmarshes in Australia. Sewage discharge and runoff from agricultural catchments may promote algal productivity in estuaries, and the accumulation of decaying masses of algae on saltmarsh may cause damage to the underlying vegetation (Hodgkin et al. 1985). Oil spills close to sea ports have affected some marshes (Anink et al. 1985) but these impacts have been inadequately studied. Many saltmarshes are potentially vulnerable in the event of a major oil spill in Australian waters.

Weed invasion

The harsh physico-chemical environment of saltmarshes could be assumed to provide protection against invasion by exotic species. Nevertheless a number of significant invasive weeds threaten the natural biodiversity and community structure of saltmarshes.

In the low marsh the only significant weed is the cord grass *Spartina anglica*, deliberately introduced at a number of sites in the early 20th century (Boston 1981). It has grown vigorously in Victoria and Tasmania, although populations in New South Wales and South Australia have remained small. In Victoria, cord grass has established to seaward of *Avicennia marina*, so changing the zonation pattern.

Cortaderia selloana (pampas grass) is a vigorous invader of disturbed bushland. It has considerable salt tolerance and has invaded a number of saltmarshes in southern Australia.

The rush, *Juncus acutus*, has invaded a range of wetlands in south-eastern Australia and in saltmarshes has displaced the native *J. kraussii* at a number of sites.

Groundsel bush, *Baccharis halimifolia*, is native to upper saltmarsh communities in the eastern United States of America. In Australia it is a major weed in coastal areas of southern Queensland and northern New South Wales and appears to be spreading southwards. Groundsel bush forms dense stands in disturbed saltmarshes and adjacent communities such as *Casuarina glauca* woodland.

Finally, there is a large number of small annual or short-lived, alien species found in upper saltmarshes in southern Australia (particularly on sandy soils) that do not appear to displace native species.

Insect control

Large populations of mosquitoes and sandflies associated with saltmarshes constitute a nuisance to humans. Through spread of diseases for which they are vectors, insects are a threat to human health in some circumstances. With an increasing population (both permanent and transient) living close to saltmarshes there are likely to be increased pressures from residents on local councils for the control of insects. Such control — through spraying of pesticides and/or hydrological modification — is already practised at many localities but the wider consequences of many of these control programs are unknown.

Sea level rise

As a consequence of global warming associated with the 'Greenhouse effect' sea level may rise. Intertidal wetlands have adjusted to previous sea level fluctuations but the consequences of a rise in the near future may be different from those of the past.

If the sea level were to rise, a regression of the seaward boundary of intertidal wetlands would occur. Where topography and other circumstances permit this regression would be accompanied by an extension landward (Bird 1988, Vanderzee 1988). For much of northern Australia there is no impediment to landward migration, so seaward losses would likely be matched by inland gains. However, in much of south-eastern and south-western Australia alienation of the hinterland for a variety of usages would mean severe limits on opportunities for landward movement and sea level rise would be accompanied by net loss of habitat. In the case of saltmarsh this loss would be exacerbated by the landward expansion of mangroves into former saltmarsh.

Increased temperatures accompanying the 'Greenhouse effect' may result in changes to the geographic distribution of individual species, while alteration to rainfall and storm regimes may also affect the composition of vegetation.

Invasion of saltmarsh by mangrove without a significant sea level rise has occurred at a number of sites in New South Wales over the last century (Mitchell & Adam 1989b). The factors responsible for this spread are unknown.

Conservation and management

A number of significant saltmarshes are included within national parks or nature reserves and, of those, several have been placed on the list of internationally significant wetlands under the Ramsar Convention (one such, the Towra Point Nature Reserve in New South Wales, owes its protected status to purchase by the Commonwealth to meet obligations under the Japan–Australia Migratory Birds Agreement).

However, the majority of saltmarsh is outside formal reserves. Protection from development can be conferred through planning, and increased public concern for wetland protection (particularly for coastal wetlands) has meant that planning authorities have, over the past decade, taken an increasingly sympathetic view towards saltmarsh protection. Although most planning decisions affecting saltmarsh are taken at the local council level, guidelines and more formal policies may be suggested at the State level. In New South Wales for example, the majority of saltmarshes outside the Sydney region are included in State Environmental Planning Policy 14 (Coastal Wetlands) which makes many proposals affecting coastal wetlands 'designated development'. This classification requires the production of an Environmental Impact Statement and concurrence from the Director of the Department of Planning to any consent to development. Although the policy does not prohibit development in saltmarshes it has been a major factor in slowing the rate of loss of saltmarsh habitat in New South Wales since its introduction in 1985.

The majority of the extensive tropical saltmarshes are not likely to be threatened by reclamation or development; however, there are few controls on access for grazing. While grazing does not destroy sites information is lacking on the impacts of grazing and associated activities such as burning to promote new growth.

In order to slow habitat degradation there is a need for implementation of catchment management regimes which address the input into estuaries of stormwater, nutrients and pollutants. It is also necessary to recognize the adverse impacts of uncontrolled access to saltmarshes and, even for sites which are not conservation reserves, to prevent the use of 'off road' vehicles.

Gaps in knowledge

There has been very little research on Australian saltmarshes and most of what there has been concentrates on describing the vegetation. While there is now a general overview of southern Australian saltmarsh vegetation (Kirkpatrick & Glasby 1981, Bridgewater 1982, Adam, Wilson & Huntley 1988) there are still many sites for which little has been recorded. In northern Australia there has been little detailed inventory of saltmarsh resources.

Data on the fauna of saltmarshes are particularly scarce — and the terrestrial invertebrate component (insects, spiders etc) is virtually unrecorded.

Ecosystem studies of Australian saltmarshes have been few. Topics such as productivity, energy and nutrient flows and linkages to other ecosystems, although the subject of speculation, remain largely unstudied.

There is an urgent need for research on:
- the effects of insect control measures on saltmarsh ecology;
- control of invasive weeds; and
- the effects of pollutants.

Following recognition of the extent of habitat degradation in south-eastern Australia there is increasing interest in rehabilitation of damaged saltmarshes. It will be important to monitor any rehabilitation programs and to carry out experimental work in order to develop appropriate methodologies.

Australia is fortunate in having extensive saltmarshes (Table 1), although many in the south of the continent have been severely degraded. In order to properly assess the status of saltmarshes and to develop measures for their conservation and management a much greater understanding of the functioning of saltmarsh ecosystems is required.

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The technical paper by Dr P. Adam was reviewed by Dr A. Robertson, the Australian Institute of Marine Science, Townsville.

Seagrasses

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Introduction

Seagrasses are rhizomateous marine angiosperms that cover extensive areas in the coastal waters of Australia. Australian seagrasses are characterised by high endemism and high speciation, especially in temperate regions. Australia has the largest number of seagrass species in the world with over 30 species (Walker & Prince 1987) which broadly can be categorised into tropical and temperate groups (Poiner & Roberts 1986). On the basis of species composition and distribution a transition zone occurs at approximately 30°S on the east coast and 25°S on the west coast (Larkum & den Hartog 1989).

Seagrass communities are of considerable importance in the processes of coastal ecosystems because of their high rates of primary production and their ability to trap sediments and organic nutrients. Their special importance to commercial and recreational fisheries is well documented. Additionally, seagrasses are important in substrate stabilisation, supply and fixation of biogenic calcium carbonate, detrital food chains, nutrient cycling and as substrate for epibiota and as critical habitats for many species (Larkum, McComb & Shepherd 1989).

Commercial and conservation value, and usage

Seagrass communities are important habitats for many commercially important species of fish and crustaceans, providing them shelter from predators and strong currents, and playing a vital role in the food cycle of coastal ecosystems. For example, seagrass communities are the major nursery grounds for juvenile tiger prawns (*Penaeus esculentus* and *P. semisulcatus*) and endeavour prawns (*Metapenaeus ensis* and *M. endeavouri*) (Poiner, Staples & Kenyon 1987, Staples, Vance & Heales 1985). Post-larvae of both tiger and endeavour prawns settle out from the water column into the shallow inshore seagrass beds and as they grow, they move into and through the deeper water beds. The productivity of the off-shore tiger and endeavour prawn fisheries of Australia are dependent on the presence of near-shore seagrass nursery grounds. Tiger and endeavour prawn fisheries are very important in Australia. They returned more than $80 million in export earnings in 1993 – approximately 7% of Australia's fisheries export revenue for that year (Australian Fisheries Statistics 1993).

Seagrasses are also critical in the life history of the western rock lobster (*Panulirus signus*) (Jernakoff 1987) and ornate rock lobster (*P. ornatus*) (R. Pitcher pers comm) as food and shelter for their peurulus and juvenile stages. The western rock lobster fishery is the most important in Australia returning $217 million in 1993 (Australian Fisheries Statistics 1993). The value of the tropical rock lobster fishery is small in comparison ($4.5 million) but it is extremely important to Torres Strait Islanders.

Seagrass is the principal food resource of the dugong (*Dugong dugon*). The dugong, the only herbivorous mammal that is strictly marine, is listed as vulnerable to extinction by the International Union for the Conservation of Nature. Seagrass is also a major food resource for the green turtle (*Chelonia midas*).

Status

Seagrasses of south-eastern Australia

Distribution and extent

South-eastern Australia is defined as the area between the Tweed River in northern New South Wales and Corner Inlet in south-eastern Victoria (Figure 1). Eight species of seagrass are present in this region. *Zostera* communities are the most common with three species of *Zostera* growing along this coastline. *Zostera capricorni* grows from the Tweed River to Mallacoota, where it is replaced by *Z. mulleri*. *Zostera* grows across a wide range of habitats and is the dominant genera in terms of occurrence and area. *Heterozostera tasmanica* occurs as far north as Port Stephens in New South Wales. Three species of *Halophila* occur in south-eastern Australia although their distribution and status is less well known than is *Zostera*'s. The range of *Halophila ovalis* and *H. decipiens* extends the length of New

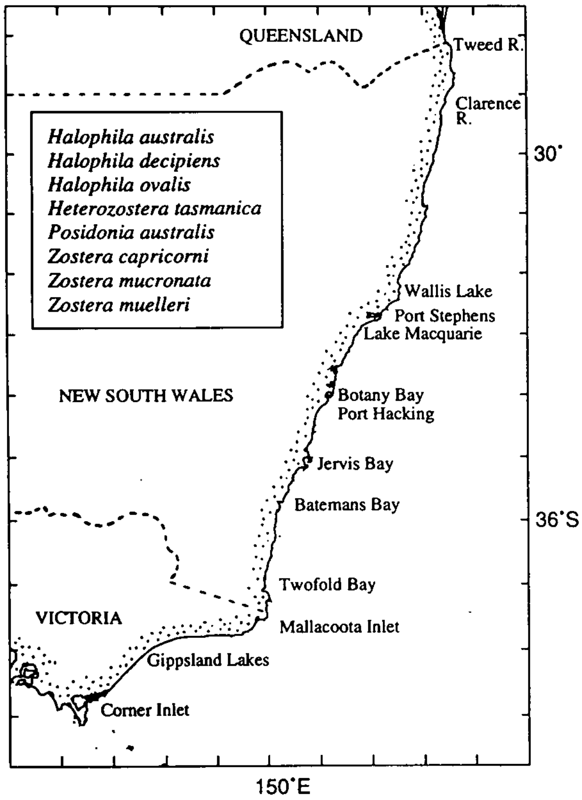

Figure 1: Seagrasses of south-eastern Australia

South Wales, while *H. australis* is present between central New South Wales and all of Victoria. *Halophila* has a wide habitat range but its growth appears to be very seasonal. It is present in low densities and occurs in small patches. *Posidonia australis* has a more limited range latitudinally, extending southwards from Wallis Lake, New South Wales. However, it is widely distributed in southern Australia. *P. australis* occupies fewer habitats than does *Zostera* or *Halophila* along the south-east coast, where it avoids hyposaline conditions and unstable sediments, but seems to be more tolerant of wave energy than are the other taxa (West, Larkum & King 1989).

The high energy coastline of south-eastern Australia means that seagrass growth is confined to estuaries and protected embayments. Although a generalised north-to-south distribution pattern can be given, the distribution and dominance of the three seagrass genera (*Zostera, Halophila* and *Posidonia*) is dictated by the occurrence and nature of those coastal features which offer a suitable growth habitat for them. The majority of the coastal features where seagrass growth occurs in New South Wales are estuarine. *Zostera* occurs in 93% of estuaries, *Halophila* in 66% and *P. australis* in 43% (West 1983).

Estuaries have been classified into three basic types according to their entrance characteristics: drowned river valleys, barrier estuaries and coastal lagoons (Roy 1984). Distribution and occurrence of seagrasses varies between type of estuary and is also affected by the age, catchment area and stability of each individual estuary. In general, there is a higher area of coverage by seagrass about 3 km back from the estuary entrance, although *Zostera* may be found in feeder creeks and cut-off bays further upstream. The species' distribution also depends on substrate type and its stability. All three seagrass genera occur predominantly on marine sand. At Port Hacking, *Zostera* and *Halophila* will readily recolonise seaward facing sands subjected to frequent disturbance by heavy seas and storms. *Posidonia australis* recolonises less readily: for example, at Jervis Bay, seismic testing 20 years ago blasted holes in the *P. australis* communities. These patches remain bare, even though they are surrounded by *P. australis* plants (West, Larkum & King 1989).

Coastal lagoons also harbour seagrass and floristic changes occur there over time. These in turn are affected by the frequency with which the lagoon entrance is broached by the sea; eg in Smiths Lake in New South Wales, the coverage by *Zostera* and *Halophila* seagrasses is greater during marine conditions than during less saline periods. Hyposaline conditions appear not to suit *P. australis* which is absent altogether from intermittently open lagoons. There are also several open embayments in New South Wales containing seagrasses. Those at the mouths of large estuaries may undergo periods of high turbidity and low salinities (eg Botany Bay and Batemans Bay). Bays which experience high wave energy are dominated by *P. australis* (eg Twofold Bay and Batemans Bay) (West, Larkum & King 1989).

Using aerial photographs, West, Thorogood, Walford and Williams (1985) reported that a total of 155 km^2 along the New South Wales coast was covered with seagrass. The largest area of seagrass occurred in Wallis Lake (30.785 km^2) where there is no *Halophila* – only *Zostera* and *P. australis*; followed by the Clarence River (19.072 km^2) of *Zostera* and *Halophila* and Lake Macquarie (13.391 km^2) of *Zostera, P. australis* and *Halophila*. There is very little information on eastern Victoria although large areas of seagrass are known to exist at Mallacoota Inlet, Gippsland Lakes and Corner Inlet. It was estimated in 1965 that there were 119 km^2 of *Posidonia australis* growing in Corner Inlet, as well as some *Zostera* and *Heterozostera* (Morgan 1986).

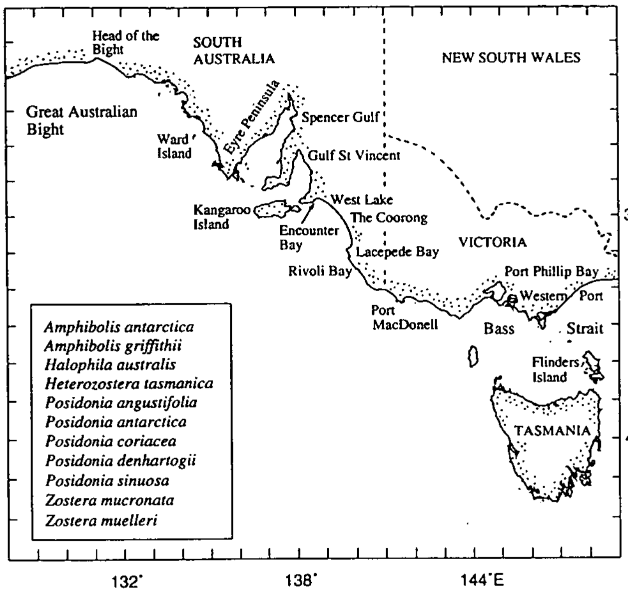

Figure 2: Seagrasses of southern Australia

Seagrass declines in south-eastern Australia

A number of seagrass communities in New South Wales have been studied in detail due to concerns that an apparent decline in the extent of seagrass beds has taken place. In 1953, the area of seagrasses in Lake Macquarie was estimated at 25.48 km^2 as part of an investigation into declining fish stocks. By 1985 the area of seagrass had declined by 11.31 km^2 to 14.17 km^2. The decline in seagrasses was probably due to increased turbidity in the lake from human activities (King & Hodgson 1986).

Botany Bay is a shallow sand embayment covering 46 km^2, with extensive areas of seagrass. Aerial photographs taken between 1942 and 1986 show a loss of 58% of the *P. australis* communities, an area of about 2.5 km^2. Reasons contributing to this loss may include: increased wave climate and erosion due to dredging; major storm events in 1974 and 1975; eutrophication due to sewage input; and grazing by the sea urchin *Heliocidaris erythrogramma* (Larkum & West 1990). Larkum (1976) suggested that once degeneration in *P. australis* begins it may be self-perpetuating: there has been no regrowth of *P. australis* over the 44-year period. *Zostera capricorni* has replaced much of the *P. australis* and covers an estimated of 3.09 km^2. However, this species has low productivity compared to *P. australis*. Both *Zostera* and *Posidonia* beds in Botany Bay house a diversity of juvenile fish species, and at least five species of commercially important fish utilise *Zostera* beds alone (Bell & Pollard 1989). The current development of the third runway for Sydney Airport in Botany Bay will probably cause further loss of seagrass from the Bay.

There have been unquantified reports of the decline of *Posidonia australis* in Corner Inlet (Victoria). No cause could be pinpointed and no man-induced changes are implicated. There is an estimated loss of 50% of *Zostera capricorni* communities from the estuaries of New South Wales. It is thought that the 60% losses at the Clarence River and the Tweed River over 30 years are due to increased turbidity associated with a general decline in water quality.

Although some of the figures concerning the decline of seagrasses in south-eastern Australia may be exaggerated due to a lack of conformity in survey and mapping methods, there has been an overall decline in the area of seagrasses along the south-east coast. Coastal development and other anthropogenic activities are probably responsible for much of the decline in seagrass area.

Seagrasses of southern Australia

Distribution and extent

This region extends from the Head of the Bight in the west to Bass Strait in the east (Figure 2). Species with warm temperate affinities – in the genera *Posidonia* and *Amphibolis* – decline in number from west to east, with a corresponding decrease in ocean temperatures. Encounter Bay is the easterly limit of *Amphibolis griffithi;* Lacepede Bay of *Posidonia sinuosa;* Rivoli Bay of *P. coriacea* and *P. denhartogi;* and Port MacDonnell of *P. angustifolia.* The cool temperate species *Halophila australis* is distributed throughout this region as are seagrasses in the genera *Heterozostera* and *Zostera.*

The distribution of seagrass is a function of the coastal topography and environment. The most extensive seagrass beds are found in Spencer Gulf and Gulf St Vincent in South Australia, with a total of over 5000 km^2 of seagrass. Spencer Gulf and Gulf St Vincent offer large expanses of sheltered water for seagrass growth. Both gulfs are dominated by *Posidonia* species. *Posidonia* beds in Spencer Gulf extend across 3700 km^2 while Gulf St Vincent boasts 1530 km^2 of *Posidonia* with *Amphibolis antarctica, A. griffithi* and *Heterozostera tasmanica* more sparsely distributed. *Halophila australis* is sparse but widespread and *Zostera mucronata* and *Z. mulleri* are found intertidally across the gulf (Figure 2) (Shepherd & Robertson 1989).

The clear waters of the Great Australian Bight allow chlorophyllous plants to live at considerable depths. *Posidonia angustifolia* and *P. coriacea* are found at depths of 25-30 m at the base of exposed cliffs of the western Eyre Peninsula where they are unaffected by swell. However, seagrass distribution along the exposed coast is patchy and most seagrass is found in the lee of reefs and islands. For example, seagrasses in the lee of Ward Island display a characteristic depth zonation and distribution, with *Amphibolis antarctica* occupying the shallows and *Posidonia coriacea* found at 30 m depth. *Halophila australis* is found on the more unstable sands (Figure 2).

Seagrasses are also distributed along the coastline in coastal lagoons. The larger ones in South Australia are West Lakes (Adelaide), with *Zostera mulleri, Heterozostera tasmanica* and *Halophila australis* at different depth zones, and the Coorong, an extensive lagoon system in the south-east of the State supporting *Zostera mulleri.* The distribution of seagrasses in Tasmania and much of Victoria requires further research. In sheltered waters of Bass Strait, patches of *Posidonia australis* and *Amphibolis antarctica* can be found. Tasmania is said to support abundances of *Heterozostera* and *Halophila australis* in its more sheltered waters, with large *Posidonia australis* meadows around its north coast and around Flinders Island (Figure 2) (Shepherd & Robertson 1989).

Seagrass declines in southern Australia

There has been a well documented decline of seagrass on the eastern side of Gulf St Vincent. In the northern section of the gulf, *Heterozostera tasmanica* dominates the intertidal flats and mixed *Posidonia sinuosa* and *Amphibolis antarctica* beds exist subtidally. By 1949, Outer Harbour (Adelaide) had lost over 9 km^2 of seagrass after the construction of retaining walls and groynes and subsequent sediment accretion (Sergeev, Clark & Shepherd 1988).

There have also been significant losses of seagrass due to sewage effluent. With the commencement of discharge from the Bolivar outfall in 1967, loss of *H. tasmanica* around Port Gawler was immediate. By 1976, there were *Posidonia* losses exhibited by a decline in leaf density, weight and length and an increase in epiphytism. Pre-1978, around the Port Adelaide outfall, 85% of the area was covered by seagrass, comprising 53% *Posidonia sinuosa,* 2% *Amphibolis antarctica* and 30% mixed beds. By 1981, the *P. sinuosa* was reduced by 50% and by 1982, 3.65 km^2 had gone. An even more extensive decline of *Amphibolis* took place, with 15 km^2 lost over this period. This suggests that *Amphibolis* may be particularly sensitive to the effects of effluent. In the southern section of the gulf, where *Posidonia angustifolia, P. sinuosa* and *Amphibolis antarctica* dominate, 8 km^2 of seagrass was lost around the Glenelg outfall between 1935 and 1987. Overall losses for the gulf amount to more than 60 km^2 of seagrass (Sergeev, Clark & Shepherd 1988, Neverauskas 1987).

In Victoria, Port Phillip Bay has large *Amphibolis antarctica* beds as well as *Heterozostera tasmanica* and *Zostera mulleri* beds. Further east, the 680 km^2 coastal inlet, Western Port supported extensive seagrass beds when it was first explored in 1899. In the early 1970s, 37% of the area of Westernport (250 km^2) supported seagrass and macroalgae. *H. tasmanica* dominated in terms of productivity (50%), with *Zostera mulleri* and *Amphibolis antarctica* also present. By 1984, only 72 km^2 of seagrass and macroalgae remained, with a 90% loss of *Heterozostera tasmanica.* Among the surviving seagrass there was a 50% decline in the above-ground biomass, giving an 85% decrease in the standing crop of Western Port. Such a decline appeared to be due to an increase in fine silt coming into the bay from river run-off, and this adhered to the seagrass leaf blades, blocking off light. Initial losses of plants caused further erosion of mud banks and the decline became self-perpetuating (Bulthius 1983).

Seagrasses of Western Australia

The coastline of Western Australia extends from 13°S to 35°S. Many different habitats and many communities occur over this latitudinal range (Figure 3). The diversity of seagrass species across this region (10 genera and 25 species) is unequalled elsewhere in the world. Tropical species have their southerly limits on this coastline: *Thalassia hemprichii* and *Thalassodendron ciliatum* extend to 22°S, a latitude which is also the northerly limit of the temperate species *Amphibolis antarctica. Cymodocea angustata* extends south to Shark Bay; *Halodule uninervis* and *Halophila spinulosa* extend south as far as 29°S; *Syringodium isoetifolium* reaches to 32°S – 5° further south than its range on the east Australian coast due to the warm Leeuwin Current. Eight *Posidonia* species are found on the south-western coast of Western Australia. The most widespread species is *Posidonia australis* with an uninterrupted temperature range from Shark Bay in the north to Lake Macquarie in New South Wales (Kirkman & Walker 1989, Walker 1989).

Despite the wide latitudinal range of the Western Australian coast, the sea temperature range is

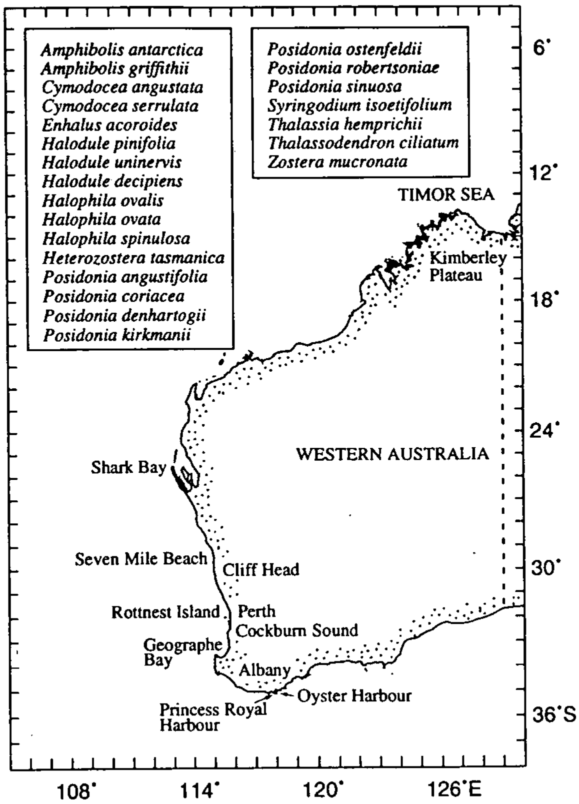

Figure 3: Seagrasses of Western Australia

very small and the distribution of seagrass species reflects the availability of a suitable environment. In the more southerly latitudes, two *Posidonia* seagrass complexes are recognised: "*ostenfeldii*", including *P. robertsoniae*, *P. kirkmanii*, *P. coriacea* and *P. denhartogii* and "*australis*", including *P. australis*, *P. angustifolia* and *P. sinuosa*. Each complex has distinct habitat requirements. *Posidonia* is very abundant along the south-west coast which is more exposed than coasts further north; and here the members of the "*ostenfeldii*" complex, with their deep roots and strong leaves, are most successful. Frenchman's Bay near Albany, Princess Royal and Oyster harbours and Geographe Bay have large areas of seagrasses. *P. australis* and *P. sinuosa* dominate these protected habitats, with some *Amphibolis* and other species in small areas. The distribution of species within these bays is a reflection of water movement gradients and depth zonation (Figure 3) (Kirkman & Walker 1989).

Around Perth, there are fringing reefs forming coastal lagoons. *Posidonia sinuosa* is the dominant species, covering many hundreds of ha in the lagoons. Other "*australis*" members occur within the lagoons and "*ostenfeldii*" complexes occupy more turbulent waters. Cockburn Sound is well protected by Garden Island. Here members of the "*australis*" complex are found together in the shallows with *Amphibolis antarctica* and *Heterozostera tasmanica*. The Swan-Canning estuary has 5 km^2 of seagrass mainly *Halophila ovalis* yet with some *Zostera mucronata*. Rottnest Island has some *Halophila ovalis* but in these clear waters overlying limestone rock, *Amphibolis* is the most important genus. In all, 9 species of seagrass are found in the Perth region (Figure 3) (Kirkman & Walker 1989).

Moving north, the sheltered Cliff Head lagoon has a 50 km^2 meadow of dense *Posidonia*. The reefs along this stretch of coast are the centre of the western rock lobster fishery. The seagrass beds are sites of settlement and nocturnal foraging for the juveniles of *Panilirus cygnus* (western rock lobster) with up to 13% of their diet consisting of seagrass and its associated macrofauna, particularly gastropods (Kirkman & Walker 1989).

Around Dongara further north, Seven Mile Beach is a highly disturbed habitat despite protective reefs. *Halophila ovalis*, *Halodule uninervis*, *Heterozostera tasmanica* and *Syringodium isoetifolium* form communities in the disturbed sands. There are natural seasonal fluctuations in the amount of cover of up to 80%. However, *Amphibolis antarctica* and *A. griffithii* form stable and extensive meadows, which did not fluctuate in area over a 6-year study period (Clarke & Kirkman 1989).

At 26°S, Shark Bay, with an area of 13 000 km^2 has some of the largest and most diverse seagrass meadows in the world, with seagrass the dominant organism in the bay. The high diversity is thought to result from a regime of intermediate disturbance and occasional cyclonic events. Shark Bay is also the site of an overlap between temperate and tropical seagrass species. The bay has restricted exchange with high evaporation leading to hypersalinity in its eastern parts. Through sediment accretion and slowing down of currents, this huge agglomeration of seagrass has further modified its own environment, with a subsequent build up of barrier banks. Behind these, hypersaline conditions exclude seagrasses, now favouring the growth of stromatolites. At the northern limit of its range, *Amphibolis antarctica*, is the most abundant species in Shark Bay, with monospecific stands accounting for 3676 km^2 or 85% of the total seagrass beds. Mixed *A. antarctica* and *Posidonia australis* stands are found in the eastern part of the bay: this is also the northern limit for *P. australis*. *Halodule uninervis* is the next most abundant plant. It may form an understorey to *A. amphibolis* and *P. australis* or grow on the

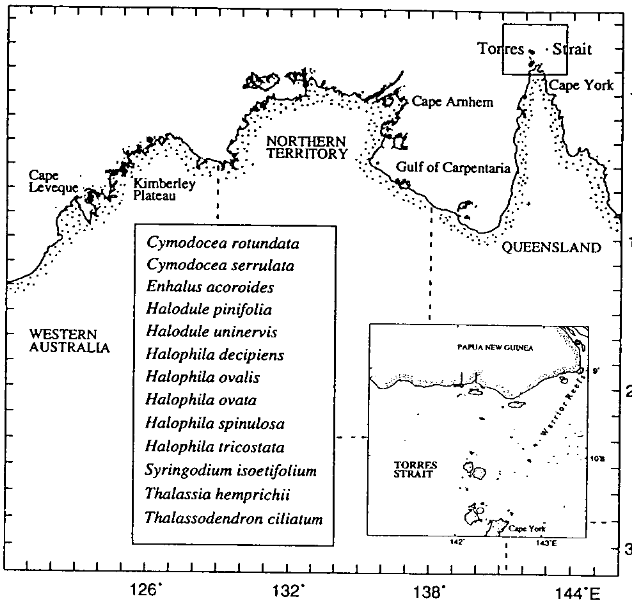

Figure 4: Seagrasses of northern Australia

intertidal flats in sparse mixed stands with *Halophila ovalis* and *H. ovata*. These stands are the preferred grazing area for the large number of dugongs that occur in the bay. North-east and east of the bay, monospecific stands of *H. uninervis* are the site of the dugong summer feeding grounds. Other seagrass species in Shark Bay include 'tropicals' such as *Syringodium isoetifolium* and *Cymodocea angustata* (which does not occur further south) (Figure 3) (Walker 1989).

Little is known about the extent of seagrasses north of Shark Bay, Nine genera with 14 species are found here. The coast can be divided into four main habitats. *Thalassia hemprichii* dominates atoll coral reefs, while large meadows of *Enhalus acoroides, Thalassodendron ciliatum* and *Cymodocea serrulata* occur on inshore reefs and banks. Offshore islands and sand cays support large populations of *T. ciliatum*. On extensive intertidal-subtidal flats, *Halodule* and *Halophila* dominate (Walker & Prince 1987).

The extensive and diverse seagrass communities along Western Australia's coastline may be attributed to the general suitability of the coast boasting a variety of habitats, together with a range of tropical and temperate species available for colonisation.

Decline in seagrass in Western Australia

Development of heavy industries and the discharge of industrial waste into Cockburn Sound commenced in the 1950s. By 1969 there were widespread losses of seagrasses and by 1978, 97% of the original 34 km^2 of seagrass had been lost (Cambridge & McComb 1984). Examination of the many possible causes showed that increased nutrient loading from effluent had given rise to an explosion in epiphytic growth. As a consequence, there was an overall 63% reduction in light reaching seagrasses in declining meadows. Controls on effluent input have helped to arrest the decline and there is evidence of slow regrowth (Cambridge, et al. 1986, Silberstein, Chiffings & McComb 1986). There have also been losses of seagrass in Princess Royal and Oyster harbours, probably associated with discharges of wastes. There is a 66% loss (over 7 km^2) of seagrass from Princess Royal Harbour and a 46% loss (over 8 km^2) from Oyster Harbour (Bastyan 1986, Walker & McComb 1990).

Seagrasses of northern Australia

Northern Australia is here defined as the area between Cape Leveque in Western Australia, and Torres Strait (Figure 4). This area encompasses three broad geographical regions: Torres Strait, the Gulf of Carpentaria and the north-western coasts of the Northern Territory and Western Australia.

Unlike temperate seagrass species, tropical seagrasses tend to occur in mixed-species stands. Thirteen species from seven genera are found across northern Australia. The region has a greater diversity of seagrass species and communities than elsewhere in the Indo-Pacific. Five *Halophila* species occur in northern Australia: *H. ovata, H. ovalis, H. spinulosa* and *H. decipiens* are ubiquitous, while *Halophila tricostata* is confined to Torres Strait. *Halodule uninervis* and *H. pinifolia* range throughout this region. *Cymodocea serrulata* is also found across the region, but *C. rotundata* is absent from Western Australia where it is replaced by *C. angustata*. Other wide-ranging species include *Thalassia hemprichii* and *Enhalus acoroides*. *Thalassodendron ciliatum* is found only in the north-western section, associated with hard substrates and corals (Figure 4) (Poiner, Coles & Walker 1989).

Torres Strait is a shallow (30-50 m deep in the east and 10-15 m in the west) body of water approximately 160 km long (north-south) and 220 km wide (east-west). The area has a complex bathymetry with large numbers of islands, shoals and reefs. Two physiographic regions characterise Torres Strait. The first is the western islands which represent the peaks of the drowned ridge extending from Cape York to Papua New

Guinea. Coral reefs fringe many of the islands and are well developed in the shallow water between them. The north-eastern portion of this region is an extensive shallow (less than 10 m) seabed with well developed sand waves. The second physiographic region encompasses the numerous platform reefs, atolls and barrier reefs in the eastern Torres Strait. The dominant feature of this region is a large, north-south oriented platform reef – the Warrior Reef complex that almost bisects Torres Strait. Winds are seasonal, with strong south-east trades in the austral winter (May-October) and north-western monsoons in the austral summer (December-February). Strong tidal currents (more than 1 m per sec.) flow alternatively east and west, but there is no evidence of net current flow through the strait.

Torres Strait supports one of the largest seagrass areas in Australia. A total of 17 500 km^2 of seagrass-supporting habitat associated with 295 km of coastline or reef has been identified and mapped. Twelve seagrass species have been recorded and the area supports a greater diversity of seagrass communities than does the rest of northern Australia. As well as the mixed-species reef-flat communities and depth-zoned open coastline communities similar to those found in the Gulf of Carpentaria, two other communities occur: sparsely distributed mixed-species open ocean communities; and subtidal *Halophila* communities. The open-ocean communities are similar to the diverse mixed species reef-flat communities but occur subtidally (to around 40 m depth) in the extensive shallow waters of the north-western and western Torres Strait with *Halophila ovalis, H. spinulosa*, *Syringodium isoetifolium, Halodule uninervis* and *Thalassia hemprichii* the most common species. *Halophila* (*H. ovalis* and *H. spinulosa*) communities are present subtidally off the large continental islands. The open-ocean communities are unique to the area and extensive. They especially occur in central and western Torres Strait and include very lush deep-water (more than 30 m) seagrass communities (*Halophila ovalis* and *H. spinulosa*) in south-western Torres Strait. Very little is known about the role of deep water, open-ocean seagrass communities.

The Gulf of Carpentaria, a shallow marine embayment with a variety of offshore and coastal features, supports 906 km^2 of seagrass habitat along 671 km of coastline. Seventy-four percent of the seagrass is found along open coastline with depth-zoned communities where *Halodule uninervis* and *Halophila ovalis* dominate intertidally, *Syringodium isoetifolium* and *Cymodocea serrulata* subtidally and *Halophila ovalis* and *H. spinulosa* offshore. Ten percent of communities occur on reef flats in mixed stands dominated by *Thalassia hemprichii*, 13% consists of a region of monospecific *Halodule uninervis* stands often associated with river mouths, and *Enhalus acoroides* dominates the remaining (Poiner, Staples & Kenyon 1987).

The north-western coast of Australia is the least well known of these northern regions. Eleven species, many with Indo-Pacific affinities, are reported for north-western Western Australia. North of Cape Leveque however, 1500 km of coastline remains unsurveyed for seagrass, as does 5000 km of the Northern Territory's coast west of Cape Arnhem. Some areas of the Kimberley coastline are known to be unsuitable for seagrasses but conditions along the rest of the Kimberley coast resemble Queensland seagrass habitat. Here, aerial photographs have revealed banks of *Enhalus acoroides, Thalassodendron ciliatum* and *Cymodocea serrulata* as well as intertidal *Halophila* and *Halodule* communities. More detailed studies are required to accurately map these communities (Poiner, Coles & Walker 1989).

Decline in seagrass in northern Australia

Sparse population and lack of development along this coast have meant that there has not been any major anthropogenic induced loss of seagrass. However, these areas are subject to tropical cyclones and floods. Since 1984 the Gulf of Carpentaria has been the site of a CSIRO study looking at the impact of cyclones on seagrasses. Only one of the four cyclones to have passed through since the study began (Cyclone Sandy, 1985), caused significant long term seagrass losses. This cyclone removed, undermined or smothered 70% of seagrass cover from the original 183 km^2 in the area. Scouring and smothering of the remaining beds caused further seagrass losses and by 1986, the entire seagrasses area was lost. This represented 20% of the total gulf seagrass area. Since 1986, recolonisation of the area by *Halodule uninervis, Halophila ovalis, Cymodocea serrulata, Syringodium isoetifolium* and *Enhalus acoroides* took place at relatively slow rates and much of the area had recovered to pre-cyclone Sandy conditions by 1994. Although these tropical seagrasses took approximately 10 years to recover, they contrast with the lack of recovery of disturbed temperate seagrasses.

In 1991-92 several hundred km^2 of seagrass in north-western Torres Strait disappeared. The actual cause of this loss is not known although it may have been due to high turbidities resulting from flooding of the Mai River in New Guinea.

The cause and extent of this loss is currently being investigated by CSIRO.

Seagrasses of north-eastern Australia

The region of north-eastern Australia encompasses the entire eastern coast of Queensland from Cape York to Moreton Bay (Figure 5). The majority of this coastline is sheltered and supports approximately 4300 km^2 of seagrass (Lee Long, Mellors & Coles 1993).

Seagrasses occur near estuaries, in coastal bays, and in areas sheltered from the predominant south-easterly trade winds. Moreton Bay (267 km^2), Hervey Bay (1026 km^2), and the area between Barrow Point and Lookout Point (1566 km^2) support extensive areas of seagrass. A total of eight genera and 14 species are found along this coastline. Two species of *Halophila* – *H. ovalis* and *H. spinulosa* – are found throughout north-eastern Australia and there are a further three *Halophila* species with more limited ranges. *Halodule uninervis* is perhaps the most widespread of all the seagrasses in this region, while *H. pinifolia* is restricted to the area between Cape York and Cairns. Whereas *Cymodocea serrulata* has also been found at most sites sampled along the Queensland coast, *C. rotundata* is not present south of Townsville. *Zostera capricorni* is only sampled south of 16°S. *Syringodium isoetifolium, Thalassia hemprichii, Thalassodendron ciliatum* and *Enhalus acoroides* are also found at intervals along this coastline (Lee Long, Mellors & Coles 1993).

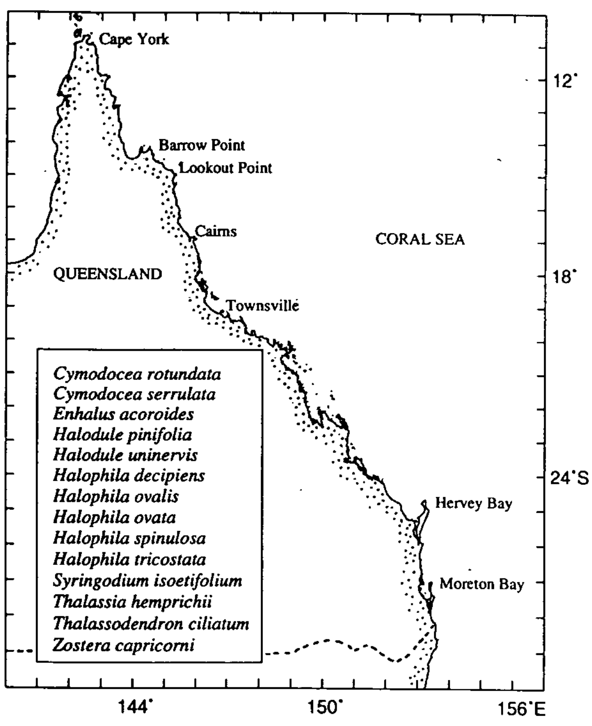

Figure 5: Seagrasses of north-eastern Australia

Declines in seagrass in north-eastern Australia

In the early 1970s some seagrass species in Moreton Bay experienced a decline or disappeared completely. Aerial photographs showed the loss of a stand of *Syringodium isoetifolium* and decline of a *Zostera capricorni* community at Toorbul Point. In neighbouring waters, there was a die-back of *Z. capricorni* stands, but by 1981 these had returned. Further south at Jumpinpin, a small bed of *Cymodocea serrulata* was lost in between 1972-73 and is not expected to return (Kirkman 1978). Increased sedimentation and a rise in substrate levels are thought to be responsible for these losses.

In 1992-93 an estimated 900 km^2 of seagrass in Hervey Bay disappeared. The cause of this loss is not known although it is thought that high turbidities, resulting from flooding of the Mary and Burrum rivers and run-off from cyclone Fran were responsible (Preen et al. 1993).

Summary of distribution of seagrass in Australia

Temperate species

1) *Cymodocea angustata* — restricted to the west coast; not reaching the Great Australian Bight, belonging to a genus with tropical affinities;
2) *Amphibolis antarctica, Posidonia sinuosa, P. angustifolia, P. coriacea, P. denhartogii, P. kirkmanii, P. ostenfeldii, P. robertsoniae, Thalassodendron pachyrhizum, Zostera mucronata* — restricted to the west coast, the Great Australian Bight and Victoria;
3) *Heterozostera tasmanica, Posidonia australis* — present in all temperate regions including the east coast;
4) *Halophila australis, Zostera muelleri* — very restricted distribution on the south coast;
5) *Zostera capricorni* — present only on the east coast as far south as Mallacoota (Victoria) and one record for Kangaroo Island (South Australia);

Tropical/subtropical species

1) *Halophila decipiens, H. ovalis, H. ovata, H. spinulosa, Cymodocea serrulata, Halodule uninervis* — broad distribution on the northern coastline;
2) *Cymodocea rotundata, Thalassia hemprichii, Thalassodendron ciliatum, Enhalus acoroides, Halophila tricostata, Halodule pinifolia* — found mainly off the Queensland coast and sometimes off the northern coasts but probably not extending far into Western Australia (some reports of *E. acoroides* and *Thalassodendron ciliatum* in the far north of Western Australia)

3) *Cymodocea angustata* and *Thalassodendron pachyrhizum* — both species endemic to Western Australia (also found in temperate waters);
4) *Syringodium isoetifolium* – extends from Moreton Bay on the east coast, throughout the northern coast and as far south as 32°S along the Western Australian coast.

Almost nothing is known about the distribution of seagrass communities of the north-west Australian coast between Cape Arnhem and Cape Leveque – a distance of around 6500 km.

Issues

Seagrass communities are extremely important parts of Australia's coastal ecosystems. To conserve and manage our seagrass resources we need to know the distribution and composition of the seagrass communities; how seagrasses respond to human induced changes to the coastal environment; how seagrass systems fluctuate both seasonally, interannually and in the longer term, and whether damaged seagrass systems can be repaired or replanted. Currently, we do not adequately know the distribution of seagrasses in Australia, nor do we know much about their temporal dynamics; and we do not have techniques to successfully replant them. There is information about the response of seagrasses to changes in the coastal environment but we cannot predict their response and we do not have an assay technique to measure the health of seagrass beds.

Australian seagrasses are being destroyed. This can occur through both natural and anthropogenic events. Human induced seagrass losses in recent years have been extensive with over 450 km^2 lost. There have also been extensive losses due to natural events such as floods and cyclones, with over 1000 km^2 lost. Most losses, both natural and anthropogenic, are attributed to reduced light intensity due to sedimentation and/or increased epiphytism from nutrient enrichment; but in some cases other factors such as sediment instability, dredging and poor catchment management interact to make the process more complex. Loss of seagrass results in loss of critical habitat for many species, declines in coastal productivity and increased sediment instability. For example, the loss of 183 km^2 of seagrass in the Gulf of Carpentaria has resulted in a loss of between 250 t/yr and 300 t/yr of tiger prawn production (Poiner, Loneragan & Conacher 1993). Recovery and recolonisation from such losses are rare for temperate species and long-term (more than 10 years) for tropical species. Attempts to replant seagrasses have not been successful in Australia. (Poiner & Conacher 1992).

We would recommend the following areas be investigated to improve the capacity for management of these important communities.

1. Methods. At all levels and scales there is a need to standardise procedures and definitions. It is almost impossible to compare and repeat many studies because of different methods used and/or a lack of description of procedures followed. This especially applies to production and distributional studies. Cost-effective, statistically robust methods of mapping and monitoring seagrass habitats need to be developed.

2. Distribution. The distribution of Australian seagrass communities should be mapped at appropriate spatial scales and, in key areas, monitored at appropriate temporal scales. Much of Australia's coastline has not been systematically mapped for seagrass.

3. Temporal information. This is an important area that has been neglected. In managing seagrasses it is very important to have an understanding of temporal variability.

4. Population dynamics. Little is known about the population biology of seagrasses both in Australia and in the rest of the world. Information is required on growth, reproduction and dispersal. With this information it may be possible to computer simulate the establishment, growth and spread of seagrasses over various time spans. Simulations offer considerable scope for predictive studies of the effects of varying the environment on seagrass communities and the impact of different management strategies.

5. Physiology. There is a need for an understanding of what physiological factors enable a seagrass to occupy and survive at a site. Initially this work should focus on the physiological and biochemical response of seagrass to reduced light conditions so that managers can set minimum water quality criteria to sustain seagrass growth. An understanding of the critical physiological processes offers the possibility of developing an assay technique to measure the health of seagrass beds.

6. Restoration. Once destroyed, seagrass systems do not readily recover. The plants require special conditions in the substrate and these are not present in disturbed or most sandy substrates. If an area does recover therefore, the time frames are long (years). The possibility of developing cost-effective methods of intervening in the seagrass recolonisation process to speed it up need to be investigated.

Conclusions and recommendations

Seagrass communities are critical for the long term sustainability of Australia's coastal zone. Despite the extensive area and species diversity of Australian seagrasses, there have been significant losses of seagrass communities. This loss has occurred through both natural events (eg cyclones, floods) and anthropogenic events (eg nutrient enrichment, coastal development). Most losses from these influences can be attributed to reduced light intensity. Once destroyed, seagrass systems do not readily recover.

There is a need to develop cost-effective, statistically-robust methods of mapping and monitoring seagrass habitats to complete mapping the seagrass communities of Australia. There is virtually no information about this important habitat along more than 6500 km of coastline. Furthermore in key regions and sites their condition needs to be monitored on a 1-5 year cycle because of the potential large scale changes that can occur in the extent of seagrass communities. The possibility of developing an assay technique to measure the health of seagrass beds needs to be investigated and cost-effective methods of intervening in the seagrass recolonisation process (to speed it up) should be investigated.

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The technical paper by Dr I. Poiner and C. Peterken was internally reviewed in CSIRO.

Mangrove systems in Australia: structure, function and status

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Mangrove and mangrove ecosystems: definitions

Mangroves are a diverse group of predominantly tropical trees, shrubs, palms and ground ferns growing above mean sea level in the marine intertidal zone. The term 'mangrove' is used to refer to the habitat although equally often the habitat is called a 'mangrove forest' or 'tidal forest'. It is also common for individual plants in this habitat to be referred to as mangroves.

As a group, mangrove plants share several highly specialised and well-known adaptations — notably exposed breathing roots, support roots and buttresses, salt excreting leaves and viviparous water dispersed propagules (Saenger 1982). Individual species do not possess all of these characteristics. The 69 recognised species of mangrove plants in the world belong to some 20 families (Duke 1992), and the species do not have shared ancestry. The term 'mangrove' is therefore an ecological term and not a genetic one.

The set of habitats including mangrove forests, their associated waterways (eg creeks) and estuarine regions or embayments with extensive mangrove forests (and their fauna) are collectively referred to in this paper as components of the mangrove ecosystem.

Habitat and area coverage

Mangrove forests show their greatest development on tropical shorelines with extensive intertidal zones composed of fine-grained sediment; for instance, as found on low gradient or macrotidal coasts (Woodroffe 1992). The forests are at their most luxuriant in areas of high rainfall or abundant freshwater runoff. Mangroves are thus generally associated with low energy, muddy shorelines, particularly tropical deltas. However, they can grow on a wide variety of substrates including sand, volcanic larva or carbonate sediments (eg Chapman 1976).

The most recent survey of the Australian coast showed that there were approximately 11 500 km^2 of mangrove forests in Australia (Galloway 1982). The largest forested areas occur in the humid tropics where there is abundant fine sediment and high rainfall and runoff from catchments. However, there are large local forests in the sub-tropics and as far south as Corner Inlet in Victoria at 38°S (Table 1).

Floristics

The mangrove flora of Australasia (the area including New Guinea, New Caledonia, Australia and New Zealand) is one of the richest in the world, having approximately five times the species richness of all other regions excepting the neighbouring region of Indo-Malesia (Duke 1992). Generally the greatest concentration of mangrove species in Australasia is found in southern New Guinea and north-eastern

Table 1: Area of mangrove forests in each of the Australian mainland States and their islands (after Galloway 1982).

State	Area (km^2)
Queensland	
tropics	4117
subtropics	485
Northern Territory	4119
Western Australia	
tropics	2507
subtropics-temperate	10
New South Wales	99
Victoria	12
South Australia	211

Australia, where 45 taxa of mangrove plants are shared.

The most recent review of mangrove species in Australia (Duke 1992) recognised 39 taxa of mangrove plants belonging to 21 genera and 19 families (Table 2). The taxa include at least four rare hybrids of more common species. Only one species, the newly discovered *Avicennia integra,* appears to be endemic to Australia (Duke 1988). All other species (Table 2) are widely distributed on the island of New Guinea or in South-east Asia.

The greatest species richness in mangrove communities of Australia is in the north-east humid tropics and there is a gradual decrease in the number of species with increase in latitude (Figure 1). Some estuarine systems on Cape York have up to 35 species of mangrove plants while to the south (Victoria and South Australia) only one species, *Avicennia marina,* occurs. Within the tropics of Australia, estuarine length, the size of the surrounding catchment, rainfall variation and the frequency of tropical cyclones have a significant effect on species richness on the east coast. These elements do not significantly affect the species richness of west coast mangrove forests however. Estuaries that are long and have large catchments tend to have more species than those with shorter and smaller catchments. Higher interannual rainfall variability and frequent cyclones tend to decrease species richness on the east coast tropics of Australia. The amount of freshwater runoff is inversely related to species richness in the western mangrove forests but not in eastern forests (Smith & Duke 1987).

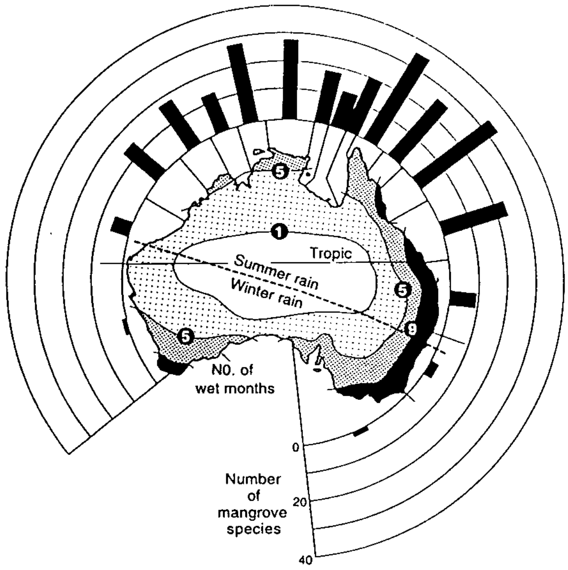


Figure 1: The pattern of mangrove plant species richness on the Australian coast. The number of species is generally much higher in the tropics, although the degree of aridity (shown here as the number of wet months per year) has a major control on species richness in the Pilbara region of Western Australia (based on Duke 1992).

Floral communities

Regional scale

Variations in factors such as air temperature, rainfall, river runoff, sediment type (and deposition rate), tidal amplitude and geomorphology along the Australian coast produce a variety of regional coastal settings for mangroves to grow. Interactions between such factors and the physiological tolerances of individual mangrove tree species control the growth forms and community structure of mangrove forests in different regions of the country (Clough 1992, Smith 1992). Thus in the high rainfall, humid tropics of north-eastern Queensland there are tall (up to 40 m), highly productive, closed canopy forests dominated by species of the Family Rhizophoraceae — in particular, species belonging to the genera *Rhizophora* and *Bruguiera.* This region also harbours the largest potential species pool of mangrove vegetation in Australia (see above); hence, a large variety of floral associations are observed in north Queensland estuaries (eg Bunt & Williams 1981). In this as in other regions of Australia, the observed floral communities on a local scale (ie within estuary) depends on position along both the estuarine salinity gradient and vertical height on the intertidal gradient (see below for details).

As conditions become more arid within the tropics, water and salinity stress on mangroves increase in the intertidal zone and open canopy woodlands or short (1–5 m), low productivity open shrublands may develop. Such shifts in forest types are well illustrated by decreases in average tree height, species richness (and hence forest composition) and greater development of mangrove shrublands from the more humid tropics of the Kimberley coast to the extreme arid zone tropics of the Pilbara coast in Western Australia (Semeniuk, Kenneally & Wilson 1978).

Open woodlands of the single species *Avicennia marina,* dominate mangrove habitats at latitudes greater than 30°S in Australia. Stem densities, basal areas and growth forms of *Avicennia* vary with region. For instance, on the east coast north of Sydney, the growing season is long enough for

Table 2: Australian mangrove plant species (based on Wells 1982 and Duke 1992). This list contains only obligate mangrove taxa. Putative hybrids are denoted by a 'X' before the specific epithet. * = present in that State.

	States and territories of Australia					
Family and species	**WA**	**NT**	**QLD**	**NSW**	**VIC**	**SA**
Acanthaceae						
Acanthus ebracteatus			*			
Acanthus ilicifolius	*	*	*			
Arecaceae						
Nypa fruticans		*	*			
Avicenniaceae						
Avicennia marina	*	*	*	*	*	*
Avicennia integra		*				
Bignoniaceae						
Dolichandrone spathacea			*			
Bombaceae						
Camptostemum schultzii	*	*	*			
Caesalpiniaceae						
Cynometra iripa			*			
Combretaceae						
Lumnitzera racemosa	*	*	*			
Lumnitzera X rosea			*			
Lumnitzera littorea		*	*			
Ebenaceae						
Diospyros ferrea			*			
Euphorbiaceae						
Excoecaria agallocha	*	*	*	*		
Lythraceae						
Pemphis acidula	*	*	*			
Meliaceae						
Xylocarpus granatum	*	*	*			
Xylocarpus mekongensis	*	*	*			
Myrsinaceae						
Aegiceras corniculatum	*	*	*	*		
Myrtaceae						
Osbornia octodonta	*	*	*			
Plumbaginaceae						
Aegialitis annulata	*	*	*			
Pteridaceae						
Acrostichum speciosum		*	*			
Rhizophoraceae						
Bruguiera gymnorrhiza		*	*	*		
Bruguiera sexangula		*	*			
Bruguiera exaristata	*	*	*			
Bruguiera parviflora	*	*	*			
Bruguiera cylindrica			*			
Ceriops australis	*	*	*			
Ceriops decandra		*	*			
Ceriops tagal		*	*			
Rhizophora apiculata		*	*			
Rhizophora X lamarckii		*	*			
Rhizophora stylosa	*	*	*	*		
Rhizophora mucronata			*			
Rubiaceae						
Scyphiphora hydrophyllacea	*	*	*			
Sonneratiaceae						
Sonneratia alba	*	*	*			
Sonneratia X gulngai			*			
Sonneratia caseolaris			*			
Sonneratia X 'merauke'		*				
Sonneratia lanceolata		*	*			
Sterculaceae						
Heritiera littoralis			*			

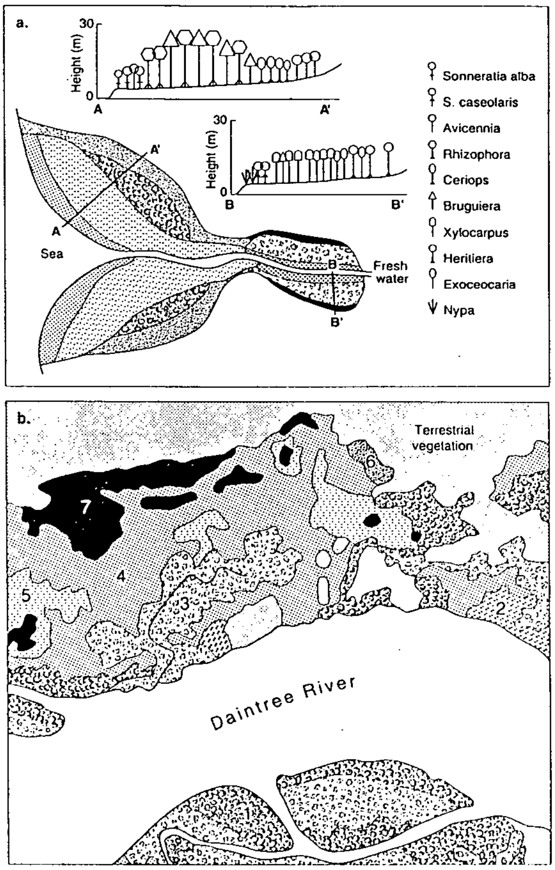
trees to reach 10 m in height. By contrast, *Avicennia* trees at Corner Inlet in Victoria (38°S) are stunted (less than 5 m in height) and have low stem densities.

Local scale: estuarine and intertidal gradients

Two key factors control forest community structure at specific mangrove localities within the (mainly) tropical regions of the Australian coast. These factors are the forest's position along the salinity gradient of an estuary, and its height in the intertidal zone. These factors in turn control the patterns of soil porewater salinities and soil wetness — key physiological constraints on mangrove species growth and competitive potentials (Ball 1988, Smith 1992). An idealised zonation pattern for mangrove plant communities in the humid tropics of north-eastern Australia is illustrated in Figure 2a. In the high salinity region of the estuary the forest composition changes from a *Sonneratia alba* or *Avicennia marina* fringe in the lowest section of the intertidal inhabited by mangroves (ie above mean sea level), through a mixed forest of *Rhizophora* (*R. apiculata, R. stylosa* and *R.* x *lamarkii*) and *Bruguiera* (usually *B. gymnorrhiza* and *B. parviflora*) to higher intertidal forests of *Ceriops* species and *Avicennia marina*. In low salinity regions of the idealised estuary, forest composition changes from a fringe of species such as *Sonneratia caseolaris* or *Nypa fruticans* through a zone dominated by *Xylocarpus granatum* to the high intertidal zone, whose characteristic species is *Heritiera littoralis* (Figure 2a).

Such an idealised zonation pattern is rarely, if ever, observed. The major reason for this is that the medium scale geomorphic features such as infilled creek beds, sand ridges and spits and areas of sedimentation and erosion occurring throughout the intertidal region provide a mosaic of microhabitat types available for colonisation and growth of mangroves (Thom 1982). This situation is illustrated by a surface map of mangrove forest community types in a section of the Daintree River estuary (Figure 2b). Local floral communities may, therefore, be highly complex. For instance, Bunt and Williams (1981) reported 29 species associations ('communities') in northern Australia based on a species pool of only 35 mangrove plants.

In regions where there is little rainfall, local patterns of mangrove floral communities are much simpler. On the Pilbara coast of Western Australia only seven mangrove species are present. Because there is little or no upstream–downstream salinity gradient, increases in the sediment porewater salinities with increasing intertidal height provide the strongest physiological gradient controlling patterns in local mangrove communities (Semeniuk 1983). In most localities on the Pilbara coast a seaward fringe of *Avicennia marina* (in areas of accreting sediment) gives way to a zone of *Rhizophora stylosa* followed by zones dominated by *Ceriops australis* and *Avicennia marina* in the mid- to high intertidal region.



1 = *Rhizophora* communities: total of 6 species; 2 = *Bruguiera gymnorrhiza/Osbornia* complex: total of 11 species; 3 = *Bruguiera gymnorrhiza* complex, with *Xylocarpus mekongensis, B. parviflora, Rhizophora* species and *Ceriops tagal*; 4 = *Ceriops tagal* complex with *Xylocarpus granatum, Bruguiera gymnorrhiza, Ceriops decandra* and other species; 5 = *Ceriops* open scrub; 6 = *Heritiera/Excoecaria/Bruguiera/Cynometra* terrestrial margin mangroves; 7 = saltpan.

Figure 2: a) Idealised intertidal and upstream (low salinity) — downstream (high salinity) distribution of mangrove forest community types for a high rainfall north Queensland estuary; and
b) a real situation for a small section of the Daintree River estuary (data taken from Le Cussan (1991).

Extensive bare salt pans (where soil porewater salinities are more than 90 parts per thousand) are the dominant feature of the higher intertidal areas of the region. In some locations, groundwater seepages from confined aquifers lower soil porewater salinities sufficiently to allow the growth of *Avicennia marina* on hinterland margins.

Physiological adaptations to physico-chemical gradients in the mangrove environment are not the only factor implicated in controls on mangrove forest structure at the local scale. Recent work in Australia has shown that seed predation by crabs and insects is also important in controlling the within-forest distribution patterns of mangrove tree species (reviewed by Smith 1992). For instance, almost 100% of the propagules (precocious seedlings) of *Avicennia marina* dispersed into the mid-intertidal region of north Queensland mangrove forests are consumed by crabs of the subfamily Sesarminae. Placing cages around *A. marina* propagules planted in the mid-intertidal causes a significant increase in the survival of propagules relative to controls subject to consumption by these crabs. Such predation helps to explain the bimodal distribution of *A. marina*, where the species occurs in low- and high-intertidal mangrove habitats but not the mid-intertidal habitat (Smith 1992).

Ecosystem structure and function

Interactions between physics, chemistry and biology

Hydrodynamic factors play a major role in the structure and function of mangrove ecosystems. Biogeochemical and trophodynamic processes are often driven by physical dynamics and forest structure and growth are intimately linked to tidal movement of waters.

In a recent review of mangrove hydrodynamics, Wolanski, Mazda and Ridd (1992) provided a succinct summary of the links between physical, chemical and biological processes in mangrove ecosystems (Figure 3). Water circulation is quite different in the two major constituents of mangrove ecosystems — the creeks and the forested portion of the system. Strong tidal flows occur in the long, often branching creek systems as a result of the tidal prism caused by the surrounding forested swamp. This causes a strong dispersion at the downstream end of creeks, which in turn helps flush material out to sea. In the upstream end of creeks there is weak dispersion and hence trapping of materials, often for up to several weeks. This can result in very low oxygen concentrations in the water column of such creeks.

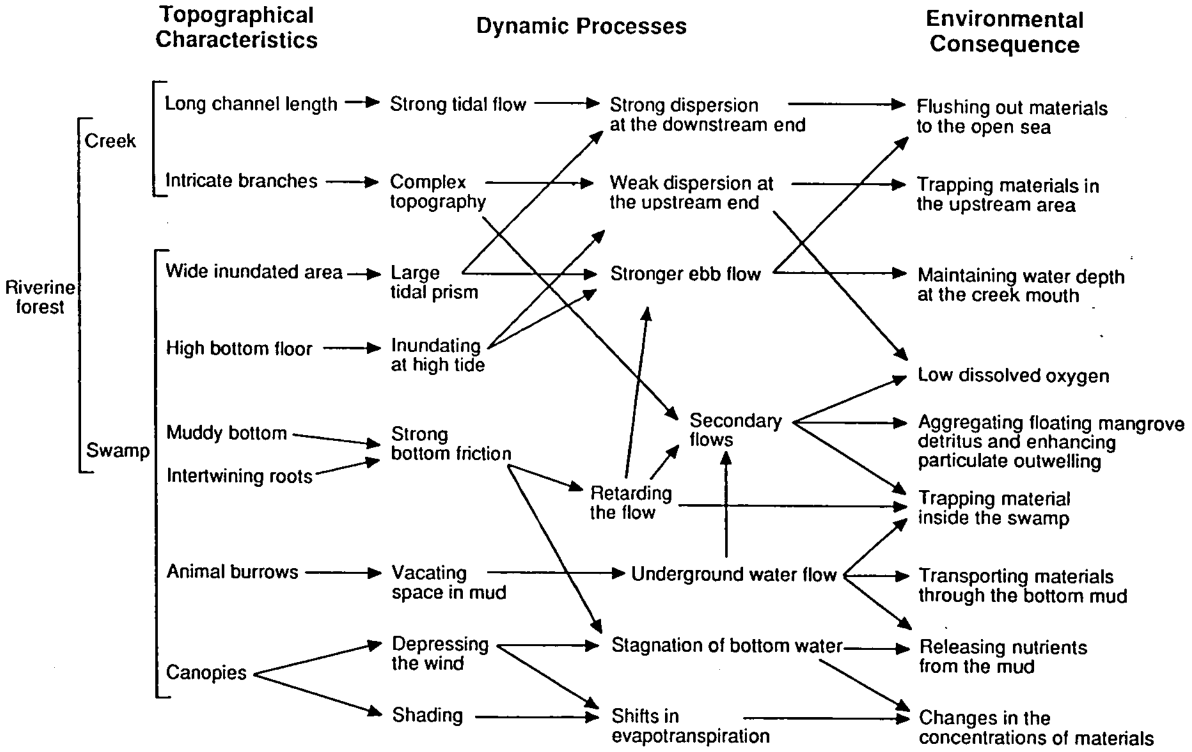


Figure 3: Links between physical dynamic processes and chemical and biological consequences in mangrove ecosystems. Note features special to creeks and forested areas of mangrove swamps (from Wolanski, Mazda & Ridd 1992).

Frictional forces and the presence of forested areas lead to an asymmetry of tidal currents, with ebb currents much greater than those in flood. This asymmetry maintains deep, self scouring, tidal channels. The high vegetation density in forests leads to high friction, retards flow, and results in trapping of water within forests. The result is that anoxic conditions can occur in waters near the sediment surface and the efflux of nutrients from sediments may increase.

The complex topography of the forested regions can also lead to secondary three-dimensional currents and small-scale topographically controlled fronts which aggregate floating mangrove detritus in long lines and enhance particulate export from forests. Biological structures such as crab burrows and decaying roots provide pathways for water (and salt and nutrient) filtration through mangrove sediments. This filtration prevents excessive accumulation of salt arising from evapotranspiration.

Finally, mangroves develop best on shores with low gradients. Mangrove swamps and nearshore waters often share a body of water called a 'coastal boundary layer', which mixes only slowly with offshore waters (Wolanski & Ridd 1990). Slow mixing means that there is often longshore transport of mangrove derived materials which are released to offshore waters only at distinct headlands (Wolanski & Ridd 1990).

Fauna

During the last decade there has been extensive work on the fauna associated with Australian mangrove forests and the role of heterotrophs in mangrove system dynamics (recent reviews by Alongi 1990, Robertson 1991, Alongi & Sasekumar 1992, Robertson & Blaber 1992).

Bacterial standing stocks in mangrove sediments and on decaying leaf litter range from 0.2 to 35.9 x 10^{10} cells per g, while the productivity of these bacterial populations are amongst the highest levels recorded for benthic bacteria and range from 0.6–5.1 g of carbon per m^2 per day (Alongi & Sasekumar 1992). The densities of benthic ciliates and flagellates are low in mangrove forests (range of mean densities; 6–260 cells per cm^2 of sediment) relative to other intertidal sedimentary habitats in Australia (Alongi & Sasekumar 1992). One possible reason for this is the low food quality of mangrove detritus.

The meiobenthos of Australian mangrove forests is dominated by nematodes and harpacticoid copepods, and their densities are generally low (less than 500 individuals per 10 cm^2) by comparison with other soft sediment habitats. Densities in the humid tropics of Australia are greatest during the wet season. In the dry tropics, the reverse is true as sediment temperatures can exceed 40°C in the summer. The soluble tannins derived from decaying mangrove tissues have a significant negative impact on the meiofauna of Australian mangrove forests and may be a major reason why their densities are so low. However, the very low nitrogen content of mangrove litter (the food of some nematodes) is also responsible for low densities (Alongi & Sasekumar 1992). Although it has been hypothesised that meiobenthic organisms play a major role in food chains and nutrient cycles in coastal sedimentary environments, recent work in Australia suggests that the meiofauna of tropical mangrove forests do not affect significantly the remineralisation of mangrove detritus because of their low densities although they may occasionally be important as a dietary component of nekton (Robertson 1988, Tietjen & Alongi 1990). Sediment bacteria are responsible for the bulk of detrital turnover in mangrove forests (Alongi 1990).

Studies of the complete macrofauna of mangrove forest sites in Australia are somewhat rare. Hutchings and Saenger (1987) provided an overview of mangrove macrofauna up to the mid-1980s. Detailed species lists of elements of the infauna and epifauna of mangrove forests are available in Hutchings and Recher (1982), Davie (1982), Wells (1983) and Hanley (1985). In this review it is possible only to provide general comments regarding mangrove macrofauna.

Relative to other intertidal sedimentary environments mangrove forested areas generally appear to have fewer species and lower densities of macrofauna. Bare intertidal mudflats associated with mangroves have a comparatively rich and diverse fauna. The epifauna in the creeks associated with mangroves has a low density when compared to adjacent subtidal habitats such as seagrass meadows (Daniel & Robertson 1990).

Alongi and Sasekumar (1992) suggested that there are several reasons for low infaunal and epifaunal densities and species richness in tropical mangrove forests. These include control by physical factors (monsoons, high temperatures and desiccation), poor quality of mangrove detritus as food (eg a very high carbon to nitrogen ratio in mangrove leaves), chemical defences by mangrove trees and predation by epifauna and nekton.

Decapod crustaceans, including the anomuran families Callianasidae, Thalassinidae and

Coenobitidae and the brachyuran families Grapsidae, Hymenosomatidae, Ocypodidae, Portunidae and Xanthidae, are usually the numerically dominant group within the mangrove macrobenthos. However, gastropods of the genus *Cerithium, Littorina, Littoraria, Nerita, Cassidula* and *Ellobium* are often locally dominant on the surface of mangrove sediments and on live and dead mangrove vegetation.

Macnae (1968) has given a general account of the behaviour of many macrofaunal species inhabiting Australian mangrove forests. However, recent work in the Australian tropics has shown that elements of the macrofauna play major roles in food chains and nutrient transformations in these intertidal forests. Grapsid crabs of the subfamily Sesarminae consume or bury up to 80% of the annual litter fall in tropical Australian mangrove forests (Robertson 1991). In so doing they have significant influences on the retention of litter nutrients (nitrogen and phosphorus) in forests that are flushed less regularly by tides, and hence on the total tidal export of materials from mangrove forests. Through their extensive burrowing activities the crabs also control aeration of mangrove sediments in some forests, thus ensuring the growth and survival of trees (Smith et al. 1991).

There is a diverse fauna associated with decaying wood in mangrove forests (Hutchings & Saenger 1987). In a 1989 unpublished study of wood-dwelling fauna in north Queensland, S.M. Cragg and A.I. Robertson found more than 120 species of animals inhabiting decaying wood. Surprisingly, insects were the most species rich taxa among the fauna with 17 species, while crabs and polychaetes were represented by 16 species. Teredinid bivalve molluscs (ship worms), represented by 12 species, were an abundant group occurring in more than half of the logs sampled.

Ship worms have a major role in the decomposition of trunk and branch wood in mangrove forests (Robertson & Daniel 1989). In the lower intertidal regions of mangrove forests, the trunks of fallen mangrove trees break down much more rapidly than do terrestrial trees and most of their weight loss during the first four years of decomposition is due to the consumption of wood by teredinids. Wood breakdown in the high intertidal is relatively slow and is caused by fungal decay.

Mangrove ecosystems are important habitats for nekton. Robertson and Blaber (1992) reviewed studies of fish communities in tropical Australian mangrove forests. Up to 197 fish species have been recorded from individual mangrove-dominated estuaries in tropical Australia, while in comparison, 65 and 46 species respectively have been taken during studies in Brisbane and Sydney. Fish densities vary widely depending on year, season, and period of the tide; mean densities in Australian mangroves range from 0.5–31 fish per m^2, while biomass ranges from 2.5–29.0 g fresh weight per m^2.

Fish species of the families Ambassidae, Clupeidae, Engraulididae, Gobiidae and Leiognathidae are numerically dominant in mangrove waters, while species of Sparidae, Haemulidae, Lutjanidae, Carcharhinidae, Centropomidae and Carangidae contribute significantly to the biomass of mangrove fish communities.

Fish and prawn abundances in mangrove systems can change over several orders of magnitude between years (eg Rothlisberg, Staples & Crocos 1985) depending on recruitment of postlarvae. Many of the dominant fish families occur in mangroves only as juveniles and recruitment from offshore waters is dependent on a number of factors, not least of which is the strength of the wet season and thus the prevailing inshore seawater salinities (Robertson & Duke 1990). Similarly, there are usually major differences in nekton abundance between locations within a region. Again recruitment factors are important, but variation in forest types and the availability of food and microhabitats are important factors in controlling spatial variation in fish densities and community structure (Robertson & Blaber 1992).

Mangrove waterways harbour an abundant zooplankton fauna, which differs in species composition and standing stocks from that in adjacent nearshore waters (Robertson, Dixon & Daniel 1988). Copepods of the genera *Oithona, Parvocalanus, Paracalanus, Acartia* and *Pseudodiaptomus* often dominate the more marine sections of mangrove waterways, while the larvae of macrobenthos — in particular, sesarmid, hymenosomatid and ocypodid crabs — are seasonally important in mangrove zooplankton communities. The degree of freshwater flushing in mangrove-dominated habitats and hence the interannual and seasonal variation in salinities, are the main factors controlling the species composition of mangrove zooplankton.

Food chains and nutrient cycles

With the exception of north Queensland, little work has been done on food chains in most

Australian mangrove forests. Measurements of primary production and the fate of carbon fixed by mangrove vegetation has been studied in north Queensland for the last decade (Clough 1992, Robertson, Alongi & Boto 1992). Because of these recent studies there has been some modification to the classic paradigm of mangrove food chains, ie that mangrove leaf detritus is the major source of energy in tropical estuaries and that microbial breakdown is the first step in the food chain leading to higher consumers.

A carbon budget for a mid-intertidal *Rhizophora*-dominated embayment in north Queensland (Figure 4) illustrates several important points about lower order trophic processes in such systems. Nearly 30% of the leaf litter fall in such systems is consumed by sesarmid crabs with the remainder exported by tidal action to adjacent creeks. Microparticulate export to creeks is much lower than leaf export. The annual turnover of wood detritus (due mainly to teredinid bivalves; see above) is of a similar order of magnitude in terms of carbon flux as the litter consumption by sesarmid crabs (Figure 4).

One of the most significant aspects of the carbon budget is the high sediment bacterial biomass and production. Indeed, bacterial production is greater than the sum of litter fall and dead wood turnover and depends on interstitial dissolved organic carbon for much of its carbon supply. These phenomena indicate that sediment bacteria are a significant 'sink' for carbon fixed by mangrove vegetation (Robertson, Alongi & Boto 1992).

There can be significant export of detritus from mangrove habitats to nearshore receiving waters (see Figure 5). The significance of such export can be gauged by calculating either the percentage of sediment bacterial production (in terms of carbon) or total sediment respiration that could be supported by exported mangrove detritus. In the north Queensland embayment referred to above these amounts were 13% and 48% respectively (Figure 5).

While these food chain models (Figures 4 and 5) incorporate significant new findings into the classical understanding of mangrove food webs (eg Odum & Heald 1975), they refer only to tidally driven (non-estuarine) mangroves, and to a particular area of the tropical Australian coast where there is a very large concentration of mangrove forests. These models are unlikely to

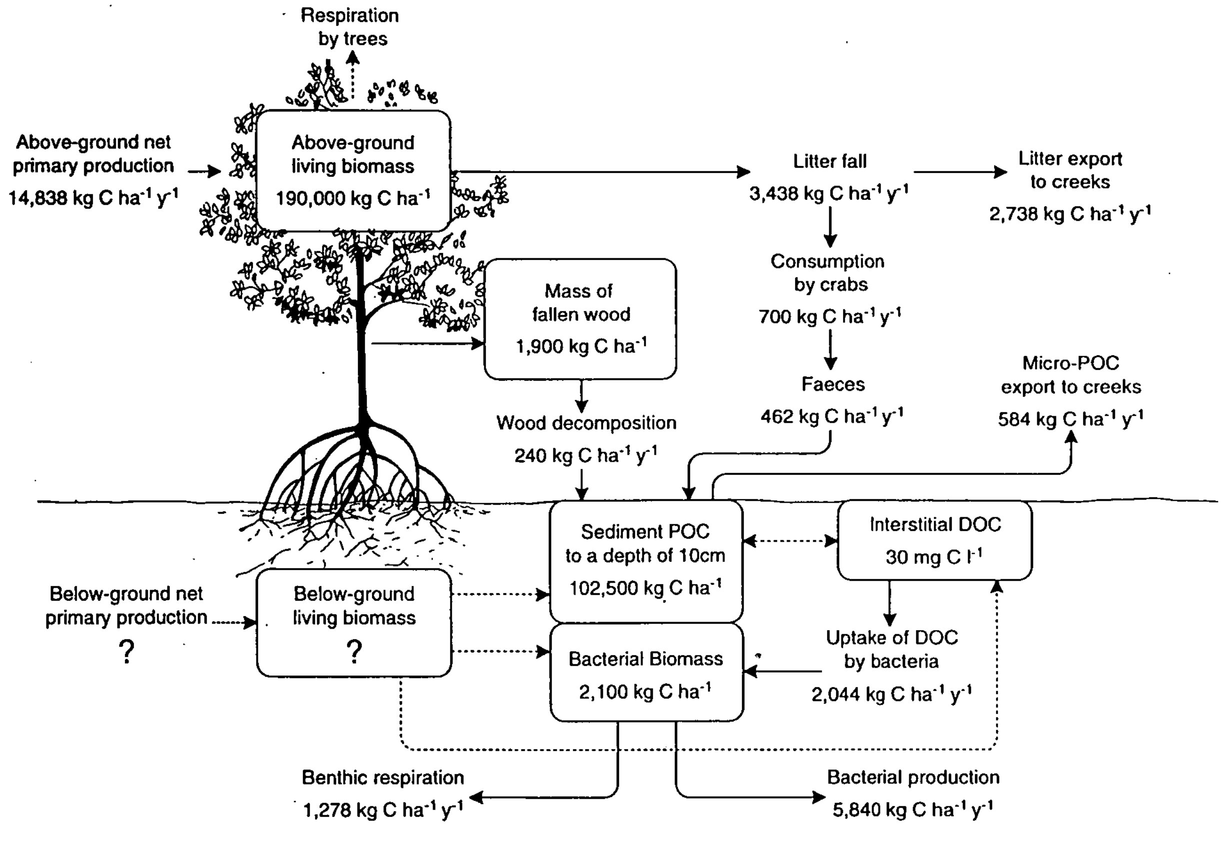


Figure 4: Fluxes of carbon within the *Rhizophora*–dominated mangrove forests of Missionary Bay in north Queensland (from Robertson, Alongi & Boto 1992).

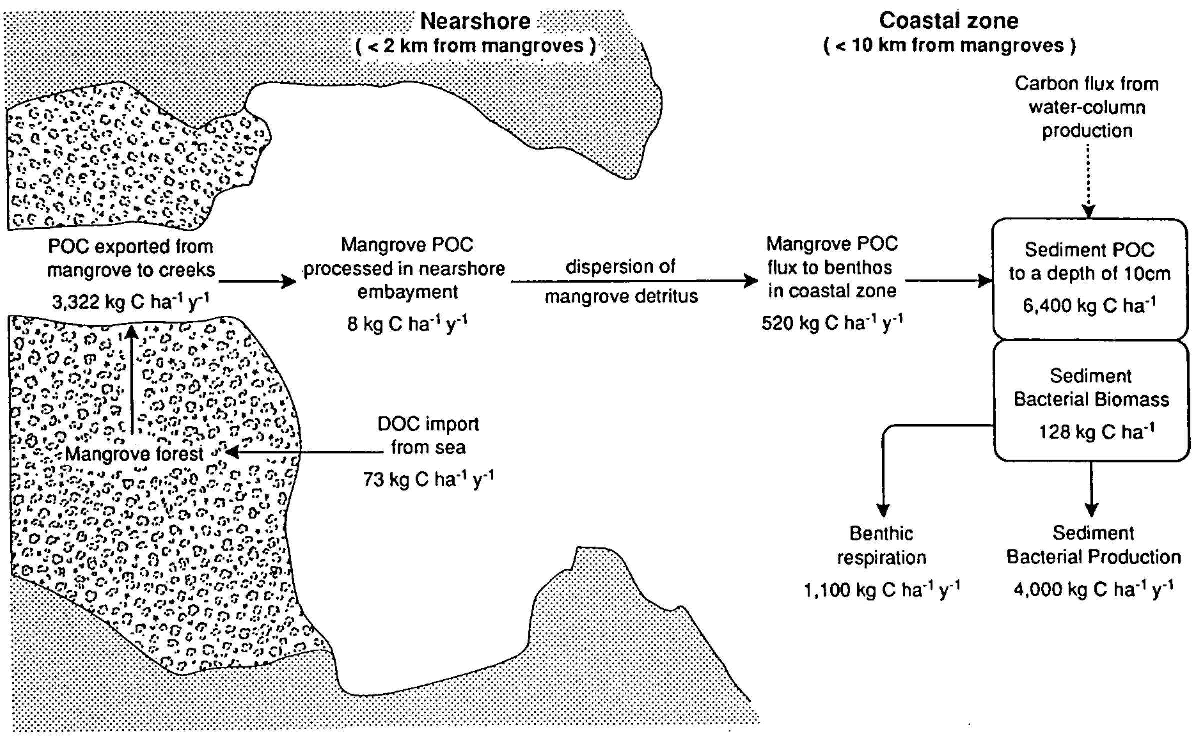


Figure 5: Export and fate of particulate carbon derived from the mangrove forests of Missionary Bay, on the north Queensland coast (from Robertson, Alongi & Boto 1992).

apply to the range of forest types in the different regions of Australia.

The important role of mangrove detritus in supporting higher consumers (such as fish and prawns) in estuarine and coastal areas of Australia has been clarified by recent studies. For instance, the zoea larvae of sesarmid and ocypodid crabs from mangrove forests are a major food source of juvenile fish in north Queensland estuaries (Robertson, Dixon & Daniel 1988) particularly during the fishes' early recruitment phase in the late dry and early wet seasons. The adults of both types of crabs are dependent on mangrove food chain links (Robertson, Alongi & Boto 1992). Thus there is a direct link between many juvenile fish and mangrove vegetation. However, during the remainder of the year juvenile fish feed on a variety of estuarine copepods whose links to mangrove primary production is unclear. It is quite possible that phytoplankton production in mangrove creeks might also be a significant carbon and energy source for zooplankton and their consumers.

The juveniles of some penaeid prawns feed on mangrove detritus or on meiofauna that is mangrove dependent. However, the importance of mangrove carbon to prawns decreases rapidly with distance from mangrove forests (Robertson, Alongi & Boto 1992).

Work on nitrogen and phosphorus budgets in north Queensland mangrove forests have shown that mangrove vegetation and soils are sinks for nutrients (Alongi, Robertson & Boto 1992). Plant uptake from sediments can account for annual rates of nitrogen (N) and phosphorus (P) removal in the order of 250 kg of nitrogen per ha and 20 kg of phosphorus per ha. Burial within sediments in rapidly accreting areas may also provide a large sink for nutrients, but this process has not been quantified (Boto 1992).

For Coral Creek, a tidal creek system in Missionary Bay in north Queensland, preliminary nutrient budgets (expressed as the percentage of N and P required to support forest primary production) (see Table 3) show that the major loss of nitrogen from the system is via export of particulate nitrogen (as leaves). This loss of nitrogen (approximately 8% of forest production) is balanced by inputs of dissolved nitrogen from the sea (Table 3) and *in situ* nitrogen fixation (Alongi, Robertson & Boto 1992). There is also a small but significant loss of phosphorus in the form of exported litter, but there is a substantial import of dissolved phosphorus from the sea (Table 3).

Importance to man

Fisheries and other faunal dependencies

Some of Australia's most important single species

Table 3: Nitrogen and phosphorus budgets for the Coral Creek mangrove system in Missionary Bay, north Queensland. Data are net annual fluxes of particulate and dissolved fractions via tidal transport. The units for net annual exchange are g of N or P per m^2 per year. Adapted from Boto (1992). A negative sign denotes export, a positive value denotes import.

Component	Net annual exchange	Proportion of forest primary production requirement
Particulate matter		
Particulate organic nitrogen	-3.7	-13.4
Particulate organic phosphorus	-0.25	-12.2
Dissolved materials		
Dissolved organic nitrogen	1.3	4.7
Dissolved organic phosphorus	0.37	17.9
Ammonia	0.15	0.6
Nitrate and nitrite	-0.03	-0.1
Phosphate	0.13	6.3
Total dissolved nitrogen	1.45	5.4
Total dissolved phosphorus	0.50	24.2

commercial fisheries are either directly or indirectly linked to mangroves. For example, the banana prawn, *Penaeus merguiensis*, has a life cycle that includes a post-larval and juvenile phase which is restricted to mangrove-lined estuaries (Rothlisberg, Staples & Crocos 1985). During 3–6 months spanning the late dry season and early wet season in the Gulf of Carpentaria and the north-east coast of Queensland, juvenile banana prawns are captured only in mangrove-lined creeks or within mangrove forests, and do not inhabit adjacent shallow water habitats (Staples, Vance & Heales 1985, Robertson & Duke 1987, Robertson, 1988, Vance, Haywood & Staples 1990). The link between mangroves and banana prawns is such that those areas in the Gulf of Carpentaria having the highest concentrations of mangroves also produce the greatest catches of adult banana prawns (Figure 6). Other locally important commercial fisheries in Australia directly dependent on mangroves include those for mud crab (*Scylla serrata*), prawns used for bait (*Metapenaeus* species) and barramundi (*Lates calcarifer*).

Other commercial fisheries have more indirect but equally important connections to mangroves. The seagrass meadows on which juvenile tiger prawns, *Penaeus esculentus*, depend (Staples, Vance & Heales 1985, Coles & Lee Long 1985) often occur immediately seaward of mangrove forests in the wet tropics of north-eastern Australia. Removal or damage to estuarine mangroves is likely to result in high sediment loads smothering adjacent seagrass meadows and then lead to decreased catches of tiger prawns (Robertson & Lee Long 1991).

Mangroves in Australia are also important nursery grounds for a variety of small non-commercial fish species (Figure 7; and see Bell et al. 1984, Robertson & Blaber 1992). These fish are an important food source for carnivorous fish in estuaries such as barramundi and trevallies and jacks (Carangidae) (Davis 1985, Robertson & Duke 1990). When mangrove-associated baitfish (Clupeidae, Engraulididae) mature and move offshore, they also support billfish and mackerel (Williams & Cappo 1990).

Many of the inshore fish species targeted by recreational fishermen in the tropics and subtropics of Australia are directly dependent on mangrove habitats. For instance, bream (*Acanthopagrus australis* and *A. berda*), grunter (*Pomadasys kakaan*), barramundi and mangrove jack (*Lutjanus argentimaculatus*) are all mangrove dependent during at least part of their life cycles (Bell et al. 1984, Pollock 1984, Morton 1990, Robertson & Duke 1990, Sheaves 1992).

Other fauna of high conservation value are also dependent on mangrove forests. Mangrove forests in the arid and semi-arid tropics often form the only closed canopy forest communities available for birds. A recent study in north-western Australia has shown clearly that 22 species of bird either were confined to mangroves, or limited to them for at least part of their range in Western Australia (see Table 4).

Coastal protection

In the Australian tropics mangroves line much of the soft-sediment dominated coast. Galloway

(1982) estimated that greater than 30% of the linear dimensions of the coasts of the Northern Territory and Queensland are lined with mangrove forests. Experience elsewhere in the world has shown that removal of mangrove forests has allowed cyclone-driven waves and storm surges to reach further inland, to be more destructive and to have greater erosive capability than was previously the case (Fosberg 1971, Lu & Lin 1987). This indicates that mangroves in northern Australia are likely to play a major role in ameliorating the effects of cyclones and storm surges on nearby urban centres and low lying agricultural lands.

Mangrove forests are also major sites of trapping and stabilisation of sediment derived from river catchments. The barrier to water flow provided by the dense tangles of prop roots, pneumatophores and trunks in mangrove forests reduces current speeds (Wolanski, Mazda & Ridd 1992) and thus aids sedimentation. Accreting sediment is bound by the dense root growth of mangrove trees. Removal of mangroves is likely to result in increased sedimentation in adjacent marine habitats (see above).

In addition, because mangroves trap water and cause ebb currents to be more rapid than flood tide currents, drainage creeks in mangrove systems are deep and self scouring (see earlier). Removal of mangrove forests leads to rapid siltation in tidal creeks.

Nutrient filtering

Because of their position in estuaries between river systems and marine waters, mangrove forests are likely to serve as efficient filters of nutrients (and other human derived contaminants) brought down from river catchments. The growth of most mangrove forests is limited by the supply of nitrogen and phosphorus and mangrove trees are known to be efficient in the uptake of high inputs of nitrogen (eg Nedwell 1975, Clough, Boto & Attiwill 1983). Additions of nitrogen and phosphorus at the level of 400 kg per ha per year for one year to a *Rhizophora*-dominated mangrove forest in north Queensland resulted in increased growth of trees and had no detrimental effect on forests. Calculations based on known rates of tree growth indicated that 63% of the nitrogen and 5% of the phosphorus were taken up by trees (Boto & Wellington 1983). Adsorption and burial within the sediments was the likely fate of the remaining phosphorus. These data indicate that, like other wetland systems, mangrove forests (and their sediments) are likely to be efficient barriers to nutrient runoff from land to sea. This role of

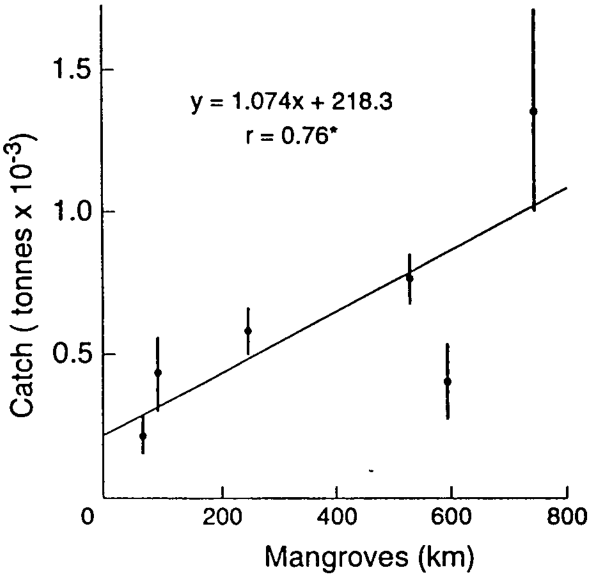


Figure 6: The relationship between the linear dimensions of the coast with mangrove forests and the commercial catch of banana prawns, *Penaeus merguiensis*, in different regions of the Gulf of Carpentaria, Australia (based on Staples, Vance & Heales 1985).

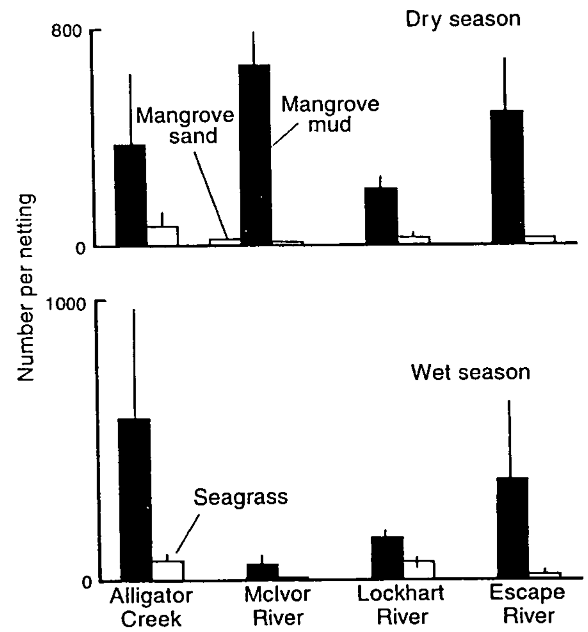


Figure 7: Relative abundance of small, juvenile fish (captured with a 3 mL mesh seine net) in mangrove and adjacent seagrass or mudflat habitats in four estuaries in north Queensland. Data are for the dry (July) and wet (January) seasons (from Robertson & Duke 1987).

Table 4: Species of birds dependent on mangrove forests in different regions of north-western Australia (based on data in Johnstone 1990).

			Region		
Bird species	**Cambridge Gulf**	**NW Kimberley**	**SW Kimberley**	**Pilbara**	**Carnarvon-Shark Bay**
Great-billed heron	*	*	*		
Mangrove heron*	*	*	*	*	
Grey goshawk	*	*			
Chestnut rail	*	*			
Bar-shouldered dove	*	*	*	*	
Little bronze cuckoo	*	*	*		
Mangrove kingfisher	*	*	*	*	
Lemon-breasted flycatcher	*	*	*		
Mangrove robin	*	*		*	
Mangrove golden whistler	*	*	*	*	
White-breasted whistler	*	*	*	*	*
Wood fantail	*	*			
Mangrove grey fantail	*	*	*	*	*
Broad-billed flycatcher	*	*	*		
Shining flycatcher	*	*	*	*	
Mangrove flycatcher	*	*	*		
Dusky flycatcher		*	*	*	*
Large-billed flycatcher	*	*			
Yellow white-eye	*	*	*	*	*
Red-headed honeyeater	*	*	*		
White-breasted woodswallow	*	*	*	*	*
Black butcherbird	*				
Number of bird species	21	21	16	11	6

mangrove ecosystems may be particularly important in regions such as north Queensland, where high rainfall and agricultural industries in catchments of the wet tropics are likely to result in the delivery of large quantities of dissolved and particulate nutrients to the naturally oligotrophic waters of the Great Barrier Reef (see Yellowlees 1991).

Conservation and management

The latest available estimate indicates that approximately 8% of Australia's mangrove forests are included in various types of conservation reserve (Table 5). Almost two-thirds of the total reserved mangroves are in Queensland which has the most extensive reserve system. Relative to most other nations with extensive mangrove habitats, Australia has a good record in the conservation of intertidal forests, mainly owing to the remoteness of our mangrove resources and the lack of population and development pressure on them (Saenger, Hegerl & Davie 1983, McComb & Lake 1988). However, large, unique areas of mangrove forest — such as those that abut the World Heritage areas of the Great Barrier Reef and the wet tropics rainforest habitats (eg Daintree River) — lack adequate protection. Similarly, large, undisturbed areas of arid zone mangroves in the Pilbara region of Western Australia are not protected under any special conservation reserve system, although the Western Australian Environmental Protection Authority has a policy of 'no net loss of mangroves'. These are rather special areas as they provide the only closed canopy forest in this arid region of Australia (see earlier: their significance as bird habitat) and are some of the best representative areas of undisturbed arid zone mangroves in the world.

There are three major problems facing the future conservation and management of mangrove areas in Australia

> First is the coordination of the number of State and Federal government departments and bodies involved in managing human activities in mangrove forests.
>
> Second, until mangroves and other estuarine areas are seen as part of a continuum between catchments and the sea, alteration to the hydrology of catchments will inevitably affect

Table 5: Australian mangrove forest areas in conservation reserves of some type. Based on data in Arthington and Hegerl (1988) and McComb and Lake (1988).

State	Number of sites reserved	Area in reserves (km^2)	% of total area of mangroves
Queensland	54	600.2	13
Northern Territory	2	281.5	7
Western Australia	-	-	0
South Australia	7+	112.9	56
New South Wales	8	4.5	5
Victoria	2	2.8	23
TOTAL	73+	1001.9	8

mangrove forests (McComb & Lake 1988, Hatcher, Johannes & Robertson 1989).

Finally, apart from intensive work in a few localities and except for vegetation species lists, little is known about most mangrove forests in Australia. Further management-oriented research is required — on dependent fauna and the effects of different types of human disturbances on mangroves — before the full conservation value of different mangrove areas is recognised.

Status in Australia and overseas

Of the 80 nations with mangrove resources, only Brazil and Indonesia have larger areas of mangroves than does Australia. While there has been localised destruction of mangroves near coastal cities such as Cairns, Brisbane, Newcastle and Sydney, there also has been regrowth of mangroves in some of these areas (eg Hyland & Butler 1988, McComb & Lake 1988). By international standards, the losses of mangroves in Australia are negligible. For instance, a number of developing countries such as the Philippines, Thailand, Malaysia and Ecuador have current rates of loss of mangrove habitat close to 1% of forest area per year (eg Saenger, Hegerl & Davie 1983, Umali et al. 1987). If this trend continues, Australia will be left with the least disturbed mangrove forests in most of the Asia-Pacific region and possibly the world (Arthington & Hegerl 1988).

Nevertheless, the continuing rapid increases in the population of Queensland in particular, has and will continue to put pressure on mangrove habitats. Large areas of mangroves in Darwin, near Cairns, near Gladstone and in south-east Queensland are currently under direct threat from a variety of development projects. Rational planning for these areas will require better understanding of the variation in conservation value of the different types of mangrove forests in each region.

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The technical paper by Dr. A.I. Robertson and Dr. D,M. Alongi was reviewed by Dr. Paul Adam, University of NSW, and Professor A.J. McComb, Murdoch University, Perth.

Coral reefs - an overview

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Introduction

The coral reef phenomenon

Coral reefs are geological structures, yet they are made by living organisms. As such, they are the biggest and most conspicuous of all non-anthropogenic structures made by living organisms and have been a major physiographic feature of the earth's surface for the past 150 million years.

The key to this 'phenomenon' is a symbiotic relationship between plant and animal which allows the practically unlimited resources of sunlight and inorganic carbon to be used for the building of structures so large that they are the creators and controllers of their own macro-environment. Both energy (in the form of organic nutrients) and building materials (in the form of calcium carbonate or limestone) are products of the same process which is the nutrient base of all major terrestrial ecosystems: photosynthesis. In this sense, coral reefs are the marine equivalent of terrestrial forests.

This often-made analogy between reefs and forests readily extends to the diversity of organisms they house, for reefs and forests each provide the food and the environment, for the earth's greatest number and variety of species.

The world-wide distribution of coral reefs (and reef-building corals) is controlled primarily by bathymetry and temperature, for reefs can only develop in shallow, sun-lit waters where the temperature seldom falls below 18°C. Secondary constraints are related to Cenozoic geological history (eg the complete absence of reef building corals in the Mediterranean Ocean), surface circulation patterns (eg their paucity in the far eastern Pacific) and regional environments of which salinity (the absence of reefs in areas influenced by major rivers), substrate type (the paucity of reefs in extensive areas of soft terrigenous substrates) and nutrients are the most important. Clearly, surface circulation patterns and temperature are interlinked, as is bathymetry and the existence of extensive river deltas, soft terrigenous substrates and light availability (Veron 1986). The relative lack of nutrients is critical to long-term stability in reef development: reefs thrive around islands in the 'nutrient deserts' of the remote oceans; they also thrive around the more nutrient-rich waters of continental margins, but they appear to do so there only where natural ecological checks and balances do not permit corals to be outgrown by other organisms.

> Like a patchwork of miniature forests, the coral reef is a microcosm of different communities, each separate, but linked to the next by a complex web of ecological interactions. These communities are distinctive because, on a single reef, they form a series of narrow bands or zones, each having a particular place in an array of rapidly changing environmental gradients.

Because coral reefs are constrained to shallow (less than 100 m depth) oceans, they are profoundly influenced by global climatic changes that affect not only ocean temperature and circulation patterns, but, much more importantly, ocean depth. Sea-level change has caused reefs to be repeatedly emerged and submerged throughout most of their geological history: thus, for example, only 20 000 years ago, all currently identifiable coral reefs were completely emerged as the sea level decreased to approximately –120 m. The coral reefs of today are living veneers on older formations, which are themselves based on successively older foundations and so on, back in geological time.

Australian coral reefs in global perspective

At least 50% of the world's coral reefs, covering an area of more than 300 000 km^2, occur in the central Indo-Pacific. This region is the world's centre of reef coral diversity (Figure 1). The reasons involve a complex mixture of geological history, oceanography and biology but the principal reason concerns past and present sea

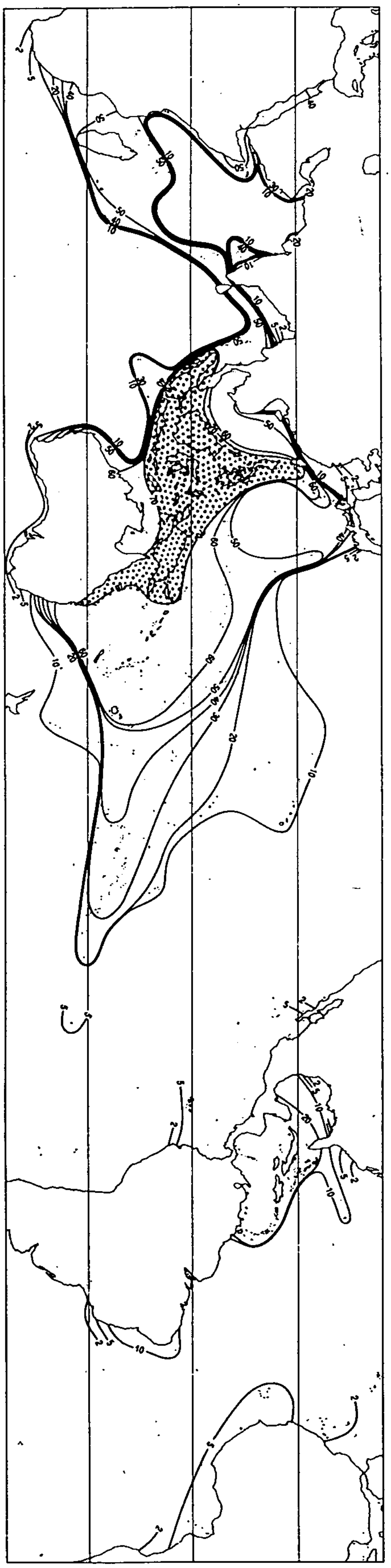


Figure 1: Isopangeneric contours of hermatypic (reef-building) coral diversity of the world (from Veron 1992). The Indo-Pacific centre of diversity is highlighted.

Millions of hectares of coral reef dominate much of the world's tropical coastline. These massive structures result from the accumulation and cementation of skeletons of innumerable corals and algae over thousands of years. Yet, imposing as they are, the existence of modern coral reefs is the result of a most intricate and subtle relationship between the coral polyp and the minute single-celled algae which live symbiotically within the cells of the polyp. These algae, commonly called zooxanthellae, belong to a group of unicellular brown plants known as dinoflagellates.

surface circulation patterns, for it is they which provide the means of long-distance dispersal for all groups of organisms capable of maintaining a planktonic existence.

Australia, because of its geographic position within the world's centre of marine diversity, is critical to the conservation of corals. Coral reefs mostly occur around the developing, over-populated countries or the world's tropics. Within major regions of the central Indo-Pacific 'Centre of Diversity', only western Micronesia, northern Papua New Guinea, Australia and (perhaps) Japan, currently have a low population pressure and/or the capacity to permanently regulate human impacts. These impacts are, as yet, poorly known: my own estimate is that 70% of all central Indo-Pacific coral reefs have been significantly degraded. This degradation has occurred primarily through over-fishing (which has effectively removed the top of the food pyramid in most South-east Asian and Japanese areas of reef), eutrophication and increased sedimentation (from urban outfall, deforestation, agricultural run-off and coastal zone development) and direct intrusive activities (principally through subsistence food gathering, particular mining practices, shell collecting and unregulated tourism).

The now often-made distinctionbetween acute and chronic impacts on reefs is intuitively useful. Acute impacts, whether anthropogenic or not, are generally limited in area (the starfish *Acanthaster* and, to a lesser extent, the gastropod *Drupella* outbreaks being the dramatic exceptions) and are often associated with widespread local death of corals. Chronic impacts are generally sub-lethal, long-term and environmental. It is the latter which are of principal importance to coral reefs and which present the main challenge for scientific study. Environmental deterioration of the type that has so widely affected European forests opens a Pandora's box of present and future possibilities for coral reefs, among which is their capacity to cope with the synergistic effects of multiple chronic influences (such as eutrophication together with over-fishing) and their capacity to recover from acute impacts while under the influence of chronic ones.

The result of anthropogenic influences are perhaps best seen by international comparisons. Truly pristine reefs, such as those of the remote outer northern Great Barrier Reef (GBR), some parts of the Coral Sea, and the reefs off the North West Shelf are inhabited by sharks and other big predators, turtles, whale-sharks and marine mammals in numbers that are seldom seen in the central and southern GBR (some specifics excepted) and which are rare anywhere in South-east Asia. Similar comparisons are valid for most collectable objects of value or interest, notably the big and/or valuable molluscs. Putting Australian reefs in a broader context is a subjective undertaking, but it appears clear that in the next few generations, they will play a critical role in the conservation of a significant proportion of the species of the world's centre of diversity.

Australian coral reefs (Figures 2, 3)

Australia's coral reefs form seven distinctive groups:

High-latitude reefs of eastern Australia

The Solitary Islands (30°S) (Clayton 1989) adjacent to the central New South Wales coast, are a group of rugged islands that do not have coral reefs as such, but do have a combination of reefal and non-reefal biota that is not found elsewhere in the world. This biota includes 53 species of reef corals and 280 species of fish of which 80% are tropical. North Solitary Island has very large populations of giant anemones and attendant clown fishes. The fauna of the Solitary Island has long been largely unprotected but recently was made a marine reserve.

Lord Howe Island, a spectacularly mountainous national park, is situated on a submerged volcanic seamount of the Lord Howe Island Rise. The reef which extends for approximately 6 km along the island's western side is the world's southern-most coral reef. The outer slope, broken by three passages, rises from a sandy sea floor at 15–20 m depth. The reef is dominated by algae with tropical affinities. The biota includes 65 species of coral (some in temporary populations) and 427 species of fish. The fish species also have primarily tropical affinities (Francis in press). Elizabeth and Middleton reefs (Hutchings 1992) are large platform reefs, also on seamounts of the Lord Howe Island Rise and only 95 km north of Lord Howe Island. In all essential characteristics, these coral reefs closely resemble those of the tropics yet they exist far to the south in very marginal conditions for reef development. They are much less accessible than the Lord Howe Island reef and have not been as well studied. Their intrinsic interest is nevertheless enormous, because of their environment and isolation. One hundred and twenty-two species of reef coral, which include most species found at Lord Howe Island, have been recorded. Elizabeth Reef was one of the first eastern Australian reefs to have a major *Acanthaster* outbreak in the 1980s. Since then both reefs have been extensively damaged with the result that abundant coral is now restricted to the reef lagoons.

There are no other limestone reefs south of the GBR although reef fauna and flora may occur in great abundance at some coastal localities, notably the little-studied 'Flinders Reef' off Brisbane. Flinders 'reef' is actually a sandstone outcrop, but has a diversity of corals which rivals that of Elizabeth and Middleton reefs. The same is likely to be true of other benthic groups on the outcrop, although most await study.

The Great Barrier Reef

The GBR is the largest single coral reef in the world. It is not the most diverse in terms of species (Indonesian and Philippine reefs have greater numbers of coral species) but it is extremely diverse in terms of reef types, habitats and environmental regimes. Why this is so, is because the GBR is large enough to extend from the low latitude tropics to subtropical zones, to have regions with very different climates (wind patterns and rainfall), tidal regimes, water qualities, bathymetry, island types, substrata and even geological histories. To some extent the GBR faunas have regional identities, but in general there is more variation across the GBR, than there is down its length. This variation is because the western (inshore) edge is dominated by shallow seas with terrigenous substrates and is exposed to periodic river run-off and consequent low salinity and high turbidity. Also, high (continental) islands occur only in inshore regions of the GBR, and it is these islands which provide much of the habitat diversity.

The Capricorn and Bunker reefs are the southernmost reefs of the GBR and are among the best known. The region as a whole is characterised by well-defined, distinctly elevated platform reefs with entire, steeply sloping sides. Inter-reefal water is moderately deep. Many reefs

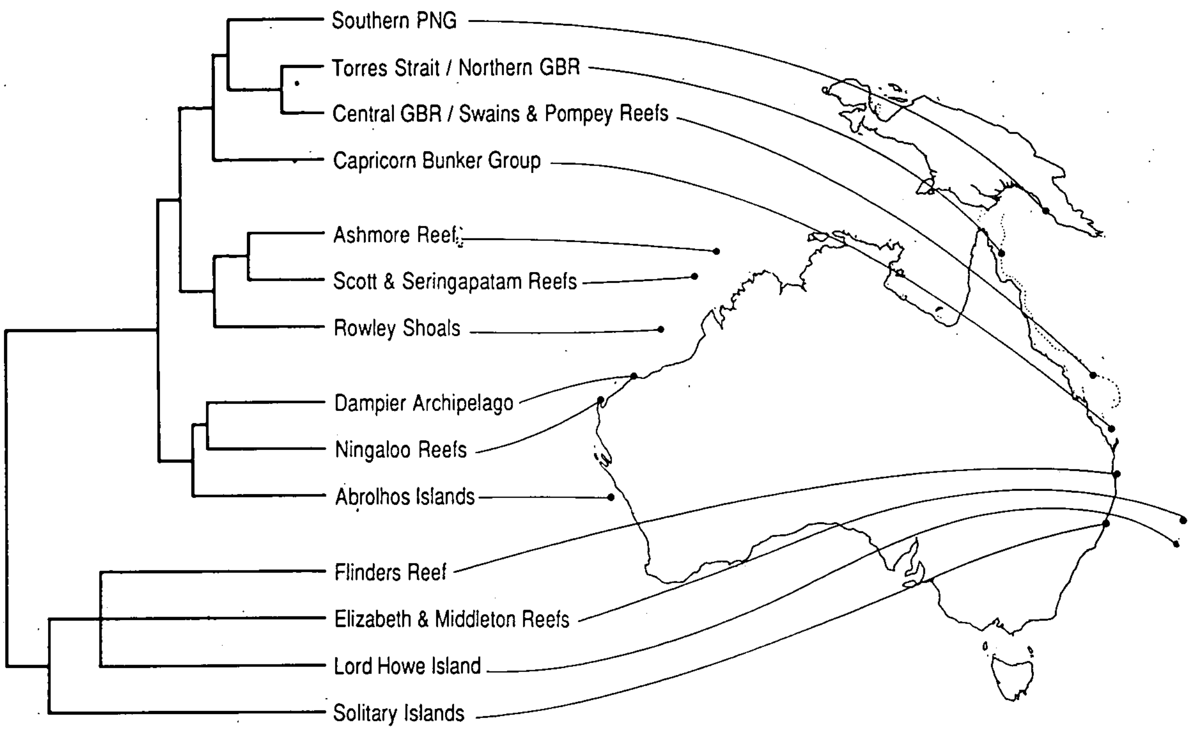


Figure 2: The relationship between the coral faunas of Australia based on presence/absence data. Links (nodes) in the dendrogram are measures of dissimilarity, the more remote the node the greater the dissimilarity. The diagram shows that there is a high degree of uniformity in the tropics and that tropical species become attenuated towards higher latitudes. This broad pattern is repeated by reef species of most major groups. However, unlike corals, most major groups show replacement of tropical species by temperate ones, rather than attenuation.

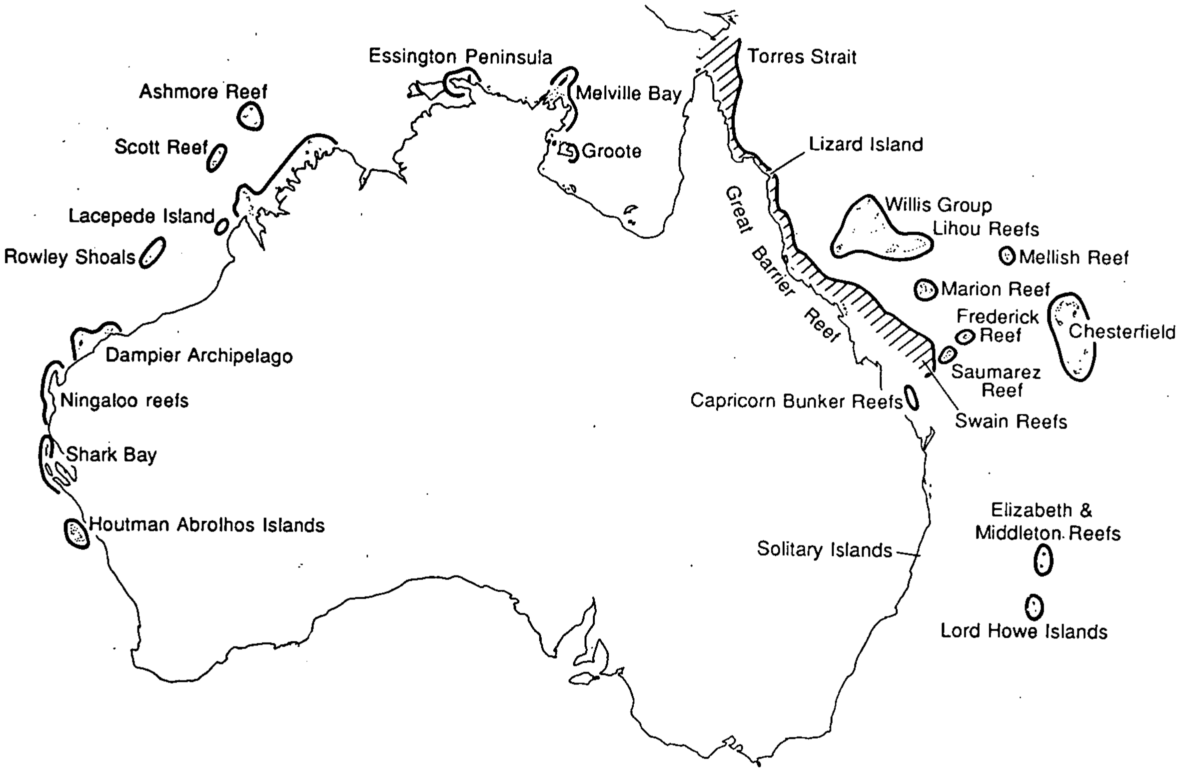


Figure 3: The distribution of coral reefs around the Australian coastline.

have vegetated cays which are much sought after by visitors. Faunistically, the reefs are very uniform with the same zones or community types being repeated from one reef to the next. Because of this uniformity, the overall diversity of corals is low (and so probably is the diversity of most other faunal and floral groups) compared with other major regions of the GBR.

The Swain Reefs and Pompey Complex extend further from the coast than any other part of the GBR and have, until recent times, been known only from a brief description in Maxwell's (1968) Atlas of the Great Barrier Reef. Seen from the air, the Pompey Complex is a spectacular panorama of interlocking reefs, channels, sandbars and lagoons, all set in the highest tidal range of the GBR and forming immense barriers to tidal water movement. So-called 'deltaic' reefs of the outer 'hard line' of the Pompeys resemble river deltas in reverse, the deltas being solid limestone and the tributaries being 'U'–shaped channels carrying extremely strong, reversing, tidal currents. The Swain complex forms a southward pointing wedge, the reefs on both sides of this wedge having exposed outer faces and protected inner margins. The two sides are ecologically dissimilar: the eastern side has several sparsely vegetated cays. The increased habitat diversity displayed in the Swain Reefs and Pompey Complex is reflected in their having a higher diversity of corals compared with the diversity in the Capricorn/Bunker Reefs.

The central GBR is a vast area, primarily characterised by lower reefs than those further south or further north, and by the absence of cays and well-defined outer barrier reefs. Perhaps the best-studied aspect of it is the change in fauna that takes place across the continental shelf from inshore to offshore in response to major environmental gradients. The relatively shallow, turbid, terrigenous coastal waters that are protected from strong wave action and subject to seasonal river flooding (and attendant pulses of silt and organic nutrients), support a reefal and inter-reefal fauna and flora of a very different character to that found offshore. The complex of high islands of the Whitsunday and Lindeman groups have a very high diversity of benthic fauna — perhaps the highest diversity of the GBR — and certainly one of the most varied.

The continental shelf is narrowest in the northern section of the GBR. It is here that the Queensland Trough forms, deepening as it extends northwards. 'Ribbon reefs' occur where the trough and GBR shelf meet. The ribbon reefs follow the shelf-edge break all the way to Torres Strait (720 km) and form the most conspicuous physiographic feature of the whole GBR. On the eastern side where they are very exposed to ocean swells, the ribbon reefs plunge steeply into the abyssal depths of the Queensland Trough. Although the water is very clear, the lower slopes are too deep for SCUBA divers to explore and almost nothing is known about them.

Inside the ribbon reefs is a band of open water mostly devoid of reefs where, particularly in the vicinity of passes between the reefs, the substrate consists of enormous mounds of *Halimeda* (Roberts & Macintyre 1988). The mid-shelf is occupied by extensive areas of reefs with roughly parallel east-west margins, cut in the past by rivers at low sea levels. The inner shelf contains a wealth of reef types, high islands and coral cays, many of them heavily vegetated.

A research station on Lizard Island, a high island on the mid-shelf, provides the only land-based access to the northern and far northern GBR. Inaccessibility of this enormous region has thus truncated even the most basic faunistic and descriptive studies.

The continental shelf widens in the far northern GBR, but the extensive shelf-edge reefs remain. Raine Island is a special place by any standards, having the largest green turtle rookery in the world and some of the largest sea bird rookeries as well. The outer barrier in the far north abounds with life in dramatic abundance no longer seen in the south. Numerous large nearshore reefs are found only in this region.

In Torres Strait, the outer barrier reefs become broken up into a series of 'deltaic' formations similar to reefs of the Pompey Complex, and then an almost impenetrable line of 'dissected' reefs. Inside this barrier line is an aggregation of reef complexes, high islands and cays of great variety. The sea becomes progressively more shallow and turbid towards the west, finally forming the Warrior Reefs which are essentially vast mud flats fringed in the east by coral. In terms of interest and variety, both above and below water, Torres Strait and the far northern outer barrier has, in my view, no equal anywhere on the GBR.

Reefs of the Coral Sea

The Western Coral Sea essentially can be divided latitudinally into three parts. To the north, and not far removed from the GBR, are Ashmore Reef, Portlock Reefs and Eastern Fields. Each of these groups is very different from the others, Ashmore Reef being atoll-like. Very deep, 'empty' ocean to the south separates these groups from the central

Coral Sea which contains the widely dispersed reefs of the Queensland plateau. Some of these reefs have cays. Further to the south are even more isolated reefs including Marion, Kenn, Frederick, Cato and Wreck reefs. Of all of these reefs, Flinders Reefs nearest to Townsville are the best studied, but even these have only been the subject of expeditionary cruises.

Reefs of northern Australia

There is a scattering of little-known fringing reefs along most of the complex coastline of the Northern Territory. The shallow, turbid waters of the eastern Arafura Sea are not conducive to coral reef growth and what reefs there are, are mostly shallow. Reef development increases to the west and reaches modest diversity in the vicinity of Essington Peninsula. Strong tidal currents dominate the environment in this region and increase towards the west. The sponge-dominated soft bottom communities of western Northern Territory have, in general, attracted more interest than the sparsely developed reefs there.

Cocos (Keeling) Atoll and Christmas Island

Cocos (Keeling) Atoll in the eastern Indian Ocean is Australia's only true atoll. Most scientific interest is in its geomorphology — especially as influenced by sea-level changes — and its isolation, which is of particular interest in faunistic studies. Due to its long occupation and recent expeditions, the atoll's fauna is moderately well known. Christmas Island to the east is a high mountainous island with a plunging shoreline. Its reefal fauna is similar to that of Cocos (Keeling) Atoll.

Reefs of the North-west Shelf

Ashmore Reef (Australian National Parks and Wildlife Service 1989), 350 km off the Kimberley coast on the outer edge of the Sahul Shelf, is basically a large sedimentary accumulation with reef patches. Nevertheless, it has the highest diversity in Western Australia of corals, sea snakes and probably most other major reefal taxa. Scott Reef, Seringapatam Reef and Rowley Shoals are all 'shelf-edge atolls', a reef type not found in the east of Australia. They are visually spectacular due to clear oceanic water with a high tidal range, and each has its own distinctive characteristics. Again, knowledge of them is mostly limited to faunistic studies (Berry 1986).

Coral reef fishes have complex life cycles involving two or more developmental stages that live in very different habitats. With rare exceptions, reef fishes do not protect offspring through their full development so that most species spend some part of their early life history in a planktonic refuge where they may be dispersed to new locations. Recruitment of juveniles from larval sources to reefs is highly variable and has given rise to debate about whether the size of adult populations is primarily determined by the success of juvenile settlement or post-settlement history (Doherty 1991).

Reefs of coastal Western Australia

Compared with the reefs of the east coast, these have been much neglected scientifically and remain little-known. Western Australian reefs are interestingly distributed down the coast in a series of 'stepping-stones', each connected to the other by the southward flowing Leeuwin Current. The influence of this current results is a chain of geographically and environmentally discrete localities forming a natural setting for long-distance dispersion of reefal fauna from Indonesia.

The reefs of the Kimberley coast are only superficially explored. They exist in turbid waters constantly mixed by large tidal fluctuations. Of all coastal reefs of north-western Australia, those of the Dampier Archipelago (Simpson 1988) off the Pilbara coast are the best known and probably the most diverse. With an inshore muddy environment and offshore clear waters, all mixed by strong tidal currents, the archipelago has a range of marine environments probably unmatched by any other area of similar size in Australia.

The 230 km-long Ningaloo Reefs (May et al. 1988), by far Australia's biggest fringing reef, is situated at the barren far western extremity of the country where the continental slope is closest to the Western Australian coast. The Ningaloo Reefs are readily accessible to visitors from Perth and, until recently, were heavily fished.

The Houtman Abrolhos, situated approximately 400 km north of Perth, are the most southerly reefs of the Indian Ocean (29°S) and are some of the most interesting coral reefs of Australia. Although they form the southern distribution limit of most Western Australian coral species, the corals show few signs of environmental stress and in some areas they form the most luxuriant communities to be found on any Australian reef. Curiously, *Acanthaster* has never dispersed to these islands although on the east coast it has reached the Solitary Island and Lord Howe Island. In other areas, corals grow with kelp and

The complex distribution of fishes among the 3000 reefs of the GBR is driven by cross-shelf gradients in water quality. The major determinant is the extent of intrusion of oceanic water onto the continental shelf. This intrusion is influenced by the width of the shelf, the proximity of reefs to the shelf-edge and the extent of an outer barrier. It varies greatly from region to region (Williams 1991).

Sargassum seaweed in an extraordinary mixture of the tropical and temperate biota.

Value and usage

The Great Barrier Reef

The GBR is arguably the most valued part of Australia's natural inheritance. Its importance to life on this planet and its intrinsic value to future Australians is beyond measure. Unlike most of the other great natural wonders of this earth, the GBR has nothing of the robustness we naturally associate with vast and apparently pristine regions. It is, as aforementioned, only a veneer of life on a limestone foundation and, in my opinion, that veneer is fragile and as sensitive to environmental degradation as is any other ecosystem on earth. It is the challenge of the present and future to preserve that veneer for all time, and do so in the face of human usage that appears likely to undergo an exponential increase.

Some would place the value of the GBR to individual Australians second only to its value to World Heritage. Most Australians, and indeed most educated people from any country who take an active interest in global issues, would place a high value on the conservation of the GBR. This value, I believe, is part of our national and international culture and is thus difficult to describe and impossible to define.

It is clear that Australian coral reefs in general are only at the dawn of international tourism. The GBR, as no other reef region in the world, offers true wilderness areas of vast proportions. As yet, the remoteness of these regions has preserved them almost completely from the tourist industry. How long this will last is guesswork, for future projections of tourist numbers and activities are inevitably prone to error because of the difficulty of predicting technological advances in transport and accommodation (such as high-speed aluminium catamarans and floating hotels), not to mention international economics.
At this point in time, most of the reef tourist industry is catering for a combination of speed and ease of access. In these respects there are many other places in the Indo-Pacific which effectively compete with the GBR. In perhaps a decade or less, this situation will almost certainly change as better informed visitors demand more personal experience, more 'adventure' and better access to remoteness and the unknown. In a decade or so beyond that, the place Australia will have in the global conservation of coral reefs will create management issues that can only be imagined at present.

In October 1981, the GBR was inscribed on the World Heritage list as it satisfies all criteria set out in Article 2 of the World Heritage Convention: an example of a major stage in the earth's evolutionary history; an outstanding example of geological processes, biological evolution and human-environment interactions; a place with unique, rare and superlative natural phenomena; and a place which provides habitats for rare and endangered species of plants and animals (Great Barrier Reef Marine Park Authority in prep.). At present the GBR region supports directly economic activity estimated to be worth at least $1 billion annually to the Australian economy.

Issues

The conservation of our natural heritage in the face of threatened impacts of the next technological and economic generation is the overwhelming issue of today. Assuming a continuation of the present level of management, key future impacts are to be from the same sources as at present: extractive industries, coastal zone development and, in the west, the oil industry.

Extractive industries

Considerable difference of opinion exists concerning the impact of fishing on the GBR. While some studies have claimed significant effects of fishing on some reef fishes, there has not always been agreement on interpretation of the limited data available. I confine my own view, derived from my comparisons with other countries, to a simple statement of what I believe is obvious: there is no permanent protection of fish of any sort throughout almost all of the Australia and South-east Asian Nations (ASEAN) region; that long-term conservation within the Indo-Pacific centre of diversity is rapidly becoming dependent on Australian legislation; and that the central and southern GBR is nowhere near as 'pristine' as is sometimes claimed.

Coastal zone development

The two principal aspects of environmental impact resulting from coastal zone development, which are now profoundly affecting reefs worldwide, are the sedimentary environment and the nutrient environment. Corals, being immobile and sensitive to environmental deterioration, are affected by sediment directly through physical contact and the energetic cost to the organism of removal, and indirectly through the effect of turbidity on light attenuation. Increase in sedimentation on coastal reefs comes primarily from soil erosion after deforestation and from agriculture, coastal construction and dredging. The nutrient environment is perhaps more complex, nutrient increase coming from sewage and agricultural outfall as well from the nutrient content of sediment resulting from increased erosion, all of which may affect ecological checks and balances.

The oil industry

This is a consideration for the future, especially in relation to the North West Shelf in general and Scott Reef in particular.

The long-term issues

These are global in nature and largely beyond the scope of this article. They vary greatly from one region or situation to another, but in the long-term the major ones are likely to be:

the status of our knowledge of what actually exists,

the relative merits of strategic versus applied research,

the interface between management and science,

the relative merits of long-term vision as opposed to 'quick fixes',

the consequences of change,

how to measure slow, long-term change,

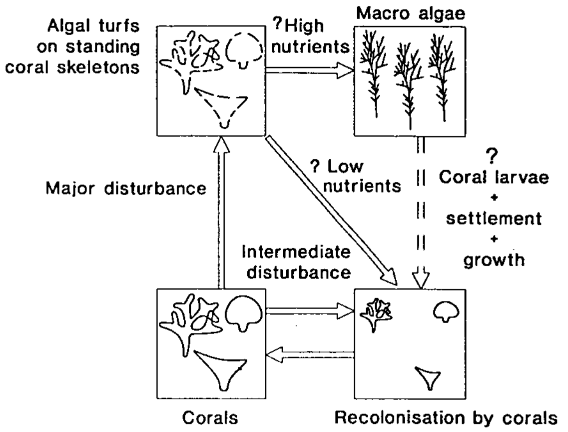
how to manage change,

public education and awareness.

Status of knowledge

Two decades ago, the GBR was one of the most poorly known of the world's major reef systems and all publications on it would fit onto a small bookshelf. Very little had been published about any of the other coral reefs of eastern Australia and virtually nothing known or published about the reefs of Western Australia other than Dakin's (1917) book on the Houtman Abrolhos and the geological studies of Fairbridge (1948). Almost no research had been undertaken using scuba diving. Since the 1970s, the GBR has become one of the best studied of the world's major coral reef regions, making Australia in general (and Townsville in particular) globally pre-eminent in reef research.

Postulated effects of episodic and chronic disturbance on the benthic community of a coral reef.



Boxes indicate stages the community passes through when subject to disturbance and/or stress. Arrows indicate that reversion of disturbed areas to coral dominance does not necessarily take place. Question marks indicate location-specific factors influencing community trajectory (after Done, in press).

Expeditionary work has been undertaken throughout the GBR and, at least superficially, on most of the reefs of the Coral Sea, the Cocos (Keeling) Atoll, all the reefs of coastal Western Australia and some of the reefal areas of northern Australia. The nature of research undertaken so far varies greatly from 'natural history' to state-of-the-art field and laboratory studies. As Australians have become more recognised in reef research, the direction of international information flow has reversed and there is now constant demand for Australian expertise in other countries, especially in South-east Asia and the tropical Pacific. Australia has moved from an era of 'youthful vigour' in reef research to one of long-term consolidation and maturity. A new generation of research has emerged through the activities of highly trained and dedicated doctoral and post-doctoral students.

The weakening of government support for marine science is being felt across the research spectrum. At personal levels it is diminishing career prospects and the nature and levels of achievement for many professionals. At institutional levels there is greatly increased emphasis on strategic research with direct conservation or commercial application. This emphasis has created a dichotomy in the nature of research, one direction being a return to low-cost basic biology, the other being to applied, or otherwise high-profile research, which remains relatively well funded. Past spending on research, combined with high achievement but now diminished funding, has left most research institutions moderately well capitalised but with restricted opportunity to put those investments to best use. Some results of this situation are positive: increased competition for resources has lifted research performance; institutions have become more cost-effective and research has generally become strategically 'useful'. However, most results from this situation are negative and will become more so in the longer term.

The studies carried out across the GBR shelf have resulted in a good understanding of environmental and faunistic differences and relationships, at least for the dominant biota accessible to SCUBA divers. However, some of the more underlying ecological processes — especially the role of micro-organisms, autotrophic/heterotrophic balances and other aspects of trophodynamics — still require a lot more study before it can be claimed that a true 'picture' has emerged.

All institutions concerned with reef research have been involved in studies of *Acanthaster*. These have been conducted along the length of the GBR as well as in other countries. Other phenomena affecting, or reflecting, the health of reefs, including coral bleaching and the effects of cyclones, are similarly under study and the reefs where the studies are being performed are extensively monitored. As with other aspects of reef research, the use of molecular techniques has opened up a new horizon, the possibilities of which are only just now being felt.

Some of the more interesting recent research developments are in:

- studies of the geological history of the GBR, a history known to affect everything from the structure and morphology of reefs to temporal, spatial and genetic patterns;
- studies of coral reproduction following the discovery of a spectacular annual mass spawning of corals on the GBR by James Cook University students in the late 1980s and subsequently in Western Australia;
- studies of the ecology of reef fish, largely initiated at Sydney University in the 1970s and continued primarily at the Australian Institute of Marine Science and James Cook University. These not only elucidate essential aspects of reef fish ecology: they have become the key component to a broad understanding of the principles of reef population and community ecology;
- multi-disciplinary research, which provides a potential for major advances that would remain closed to more categorised endeavours. Examples are in physical oceanography and genetics in relation to faunistics; chemical oceanography, geochemistry and reef biology in relation to upwelling, primary productivity and zonation patterns; coral biology, climatology and genetics in relation to past environments;
- development of predictive and explanatory computer models, which provide enhanced means of generating hypotheses and, with appropriate data (especially from long-term monitoring) to rigorously test them; and
- the application of research and attendant technology stemming from some aspect of the faunal diversity contained within the GBR to other uses. Examples at AIMS are the linking of fluorescent bands in corals with weather records of the past, the discovery of commercially valuable ultra-violet light blocking compounds in corals, and the potential for finding biologically active compounds for medical applications.

Conclusions

The most general comments that can be made about our present knowledge of Australian coral reefs are that, compared with terrestrial counterparts, it:

is full of gaps. Major advances have been made in particular areas by individual scientists, institutions, or groups, but these are widely separated;

has largely excluded inter-reefal regions. These comprise more than 90% of the area of the GBR and are heavily trawled for fish and prawns;

has inadequately considered biology and processes at a total-systems level;

has largely excluded the far northern GBR, the reefs of the Northern Territory and, most importantly, the reefs of Western Australia;

has failed to consider microbial processes;

has not yet adequately addressed the particular needs of long-term conservation. Foremost among these are the effects of sediments and nutrients on reefs, the root cause of *Acanthaster* outbreaks, and the long-term effects of commercial and recreational fishing; and

has not yet utilised the wide spectrum of opportunities afforded by molecular techniques to ascertain the distribution, characteristics and taxonomic relationships of populations and species.

Finally, because mainstream Australian research now predominates in a global sense, it is out of balance with knowledge in most other countries. The extent of Australia's responsibility to contribute to global knowledge can be debated, but Australia's role in long-term conservation in the Indo-Pacific realm is clear.

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The technical paper by Dr. Veron was internally reviewed within the Australian Institute of Marine Science and by Dr. D.W. Kinsey, Chairman, SOMER.

Australia's marine ecosystems: the continental shelf and slope

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The long coastline of the Australian continent is surrounded by a gradually sloping submarine shelf of 2.5 million km^2. About half of this area is less than 50 m deep. Beyond a depth of 150–200 m the gradient increases quickly away from the shore and the sea floor becomes the continental slope, another 1.5 million km^2 in area.

Knowledge of the environment and biology of the continental shelf and slope can best be described as patchy. Much of the coast is remote from human habitation and its large size and climatic range make generalisations difficult. The geology of surface sediments, which is an important determinant of the biological communities, has been studied superficially and the literature reviewed briefly by Bunt (1987). His contribution is summarised here with some additions.

The biology of the benthos of the continental shelf and slope has not been systematically studied. Only three areas have been investigated intensively: the North West Shelf and slope, the Great Barrier Reef lagoon and Coral Sea and Bass Strait and the south-eastern Australian slope. The methods and objectives of these studies are very different and I treat them as case studies almost impossible to compare. Almost nothing is known of other areas and, surprisingly, there is little information on the biology of the continental shelf off Sydney, Australia's largest city, apart from a superficial report edited by Jones (1977).

Information on off-shore commercial fisheries is reviewed elsewhere in this volume.

Geology

The continental shelf is a continuous feature ranging from 15 km wide off the south-eastern coast to 400 km wide in the Timor Sea. It joins Australia to New Guinea in the Arafura Sea and Torres Strait. The shelf has a gradual slope to its greatest depth at 150–200 m and its generally featureless surface is broken by occasional reefs, dunes and ridges. Its topography is more complex in the north-east (where the Great Barrier Reef occupies the outer shelf) and north-west.

The Australian slope is continuous with that of Indonesia and New Guinea. In parts it is much steeper than continental slopes elsewhere in the world. Gradients of up to 40° are common and extend to the abyssal plain at about 4000 m depth. The slope is intersected by canyon systems, especially in eastern and western Bass Strait, where very steep gradients and rocky outcropping occur. It is further interrupted by several terraces, major plateaus and troughs, best developed on north-eastern and western margins of the mainland and on southern Tasmanian coasts. Each plateau has its shallowest depth at between 900 m and 2000 m. Minor terraces also occur in the Great Australian Bight. The south-eastern slope is steeper, yet without major troughs and terraces (Figure 1).

Surface sediments, which play an important part in the nature of infaunal and epifaunal communities, vary geographically and with depth. Bunt (1987) reviewed current knowledge and referred to the major recent works.

The sediments of the Arafura Shelf and most of the west coast are mostly coarse and calcareous with finer sediments further offshore. Bryozoans are predominant but foraminiferans and calcareous algae contribute at greater depth. There is little sedimentation from terrigenous sources except in estuaries.

Terrigenous muds and quartz sands dominate the inner shelf sediments of the Great Barrier Reef lagoon. Carbonates, with a fair contribution from corals, increase in concentration towards the outer lagoon and are predominant on the outer continental shelf. In the Gulf of Carpentaria sediments are much finer than further south and are rich in marine faunal remains.

The shelf of New South Wales has mostly relict sandy sediments — terrigenous close to the coast

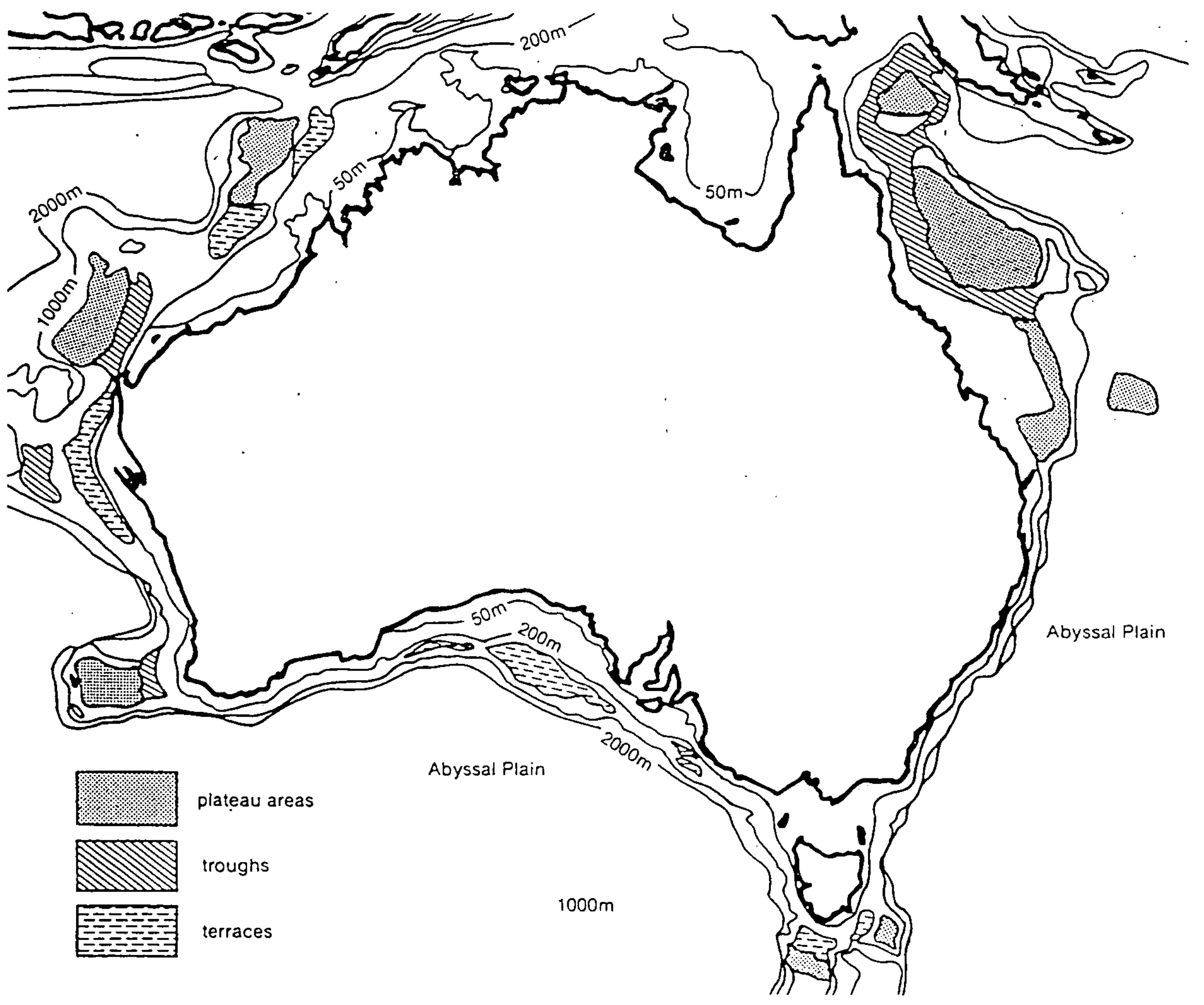

Figure 1: Major features of benthic environments of the Australian shelf and slope (from Bunt 1987).

and carbonates with increasing depth below 60 m. Muds are uncommon. It is thought that sandy sediments are transported north and feed the shelf of southern Queensland.

The southern continental shelf is covered with course calcareous and mainly relict sands, the remains of bryozoans, molluscs and foraminifera. This predominantly carbonate province which extends from south-eastern to south-western Australia is the largest of its kind in mid-latitudes and differs from that in the tropics where coral is the major contributor (James et al. 1992). Terrigenous sediment input is negligible except near the only major river, the Murray. The dominance of carbonates prevails in Bass Strait, the only anomaly being the presence of quartz sands in varying percentages off eastern Victoria.

The information needed for an appraisal of the relationships between fauna and grain size, sorting and composition is only at a gross level (Bunt 1987). Nevertheless, it can be said that the Australian shelf environment is peculiar, especially in the south, in the dominance of coarse particles and the virtual absence of terrigenous material. Combined, these two attributes contribute to complex microhabitats which may be responsible for high macrofaunal densities and species diversity.

Biological communities

The physical environment of the continental shelf and slope is almost entirely soft sediments and is inhabited by infauna (burrowing into the top few centimetres), epifauna (attached to or walking on the sediment) and demersal species (swimming over the bottom). Offshore reefs of hard rocky substrate are a small part of this environment and are little known except in shallow water where diving makes sampling possible. Exceptions are where the biology of abalone on reefs has been investigated. Information on this is covered elsewhere in this report.

North-western Australia

In 1982 CSIRO began a multidisciplinary study of the demersal fish and invertebrate communities of the North West Shelf with the objectives of investigating and managing finfish and crustacean fisheries. The epibenthos of this region is composed of sponges, gorgonians, soft corals and sea-pens and it is scattered over areas of rippled bare sand. The demersal crustaceans taken by Ward and Rainer (1988) from 47 trawl and sled samples at 40 m and 80 m depth numbered 308 species. Most were crabs but penaeid prawns and carid shrimps were also diverse. Depth was the only environmental parameter correlated with the distribution of these species. The number of taxa from the North West Shelf is higher than that caught in more intense surveys in the western Atlantic. The infauna of the North West Shelf has an extremely rich species complement and is dominated by polychaete worms (Rainer 1991). Rainer suggested that the high diversity is consistent with the combined effects of disturbances from tidal and storm-generated water movement, predation by fishes and crustaceans and low productivity in an oligotrophic environment. Small individual body size, another peculiarity of this North West Shelf fauna, may enable rapid turnover in an area of low nutrient levels.

Great Barrier Reef shelf and slope

Information on the lagoon of the Great Barrier Reef comes from several studies with very different objectives.

Cannon, Goeden and Campbell (1987) recorded about 700 species of fishes and macroinvertebrates from trawls on the northern and southern continental shelf and slope. Their data were uneven and they were able to detect only slight depth-related distributional patterns. Rainer and Munro (1982) and Rainer (1984), working in the nearby but very different Gulf of Carpentaria, found that depth played a major part in explaining the distribution of fishes and cephalopods.

Near Townsville, Birtles and Arnold (1983, 1988) found that the most diverse epifaunal taxa (echinoderms with 103 species and molluscs with 196 species) were divided into an inshore community on muddy sediments at 22 m depth and an offshore community on sandy sediments at greater depths. Calcareous rubble acts as a point of attachment for solitary and colonial animals and these 'multispecies isolates' are important in contributing to the diversity of shelf epifauna. The relative importance of different feeding strategies (filter-feeding, browsing, carnivory and deposit-feeding) varied haphazardly across the shelf. Tietjen (1991) reported high diversity of nematode assemblages across the Great Barrier Reef lagoon resulting from sedimentary heterogeneity (caused by cyclones and trawling), a rich bacterial flora, continuous warm temperatures and low macrofaunal abundance.

Alongi and colleagues have examined shelf communities from the point of view of structure and function without the taxonomic resolution of other studies (Alongi 1989, Alongi & Christoffersen 1992). They found that benthic standing crops and processes on the central shelf appear to be regulated by low and intermittent inputs of detritus, warm temperatures and physical disturbance, both natural and anthropogenic. These factors and oligotrophic conditions perpetuate the dominance of a pioneering regime.

Cruises on the north-eastern slope led to the discovery of new macrofaunal taxa but the published work on communities is devoted to microbial and meiofaunal composition (Alongi 1987, 1992; Alongi & Pichon 1988). Alongi and Pichon suspected that densities of metazoan meiobenthos in the western Coral Sea was lower than could be expected due either to low rates of detrital input or to rapid transport of turbidites. The slope in this region seems more oligotrophic than in other parts of the world.

Bass Strait and the slope off south-eastern Australia

Several scientific exploratory expeditions have passed through Bass Strait since a French expedition in 1802. Yet none has resulted in more than very local reports on the biological communities or taxonomic publications (Poore 1979). The Museum of Victoria's Bass Strait survey (1979–1984) aimed to provide material for systematic study — specimens on which to base descriptions of new species and from there to work towards an understanding of the area's evolutionary history. In the survey, about 300 samples were taken from a wide area of the shelf using a variety of grabs, sleds and trawls. Dozens of papers describing new taxa in many taxonomic groups of animals have resulted from this survey and many more will appear over the next decades. For the first time, there may be sufficient distributional information to present a synthesis of the Bass Strait fauna — at least for some taxa.

During the survey, the only quantitative study in Bass Strait was on a very small area in eastern Victoria. Here, grab samples from 1.2 m^2 of benthos enabled 353 species of invertebrates to be discriminated, about half of them crustaceans, the rest polychaetes and molluscs (Parry, Campbell & Hobday 1990). Compared with data from similar areas in other parts of the world, species diversity for this eastern Victorian area was exceptionally high (Parry, Campbell & Hobday 1990). None of the four sites sampled had more than 45% of the total number of species collected, suggesting that greater effort would reveal more species. More recent sampling from 10 m^2 in the same area turned up over 900 species (unpublished data). Interestingly, the fauna 100 km west at similar depth and sampled in the same manner is also rich but many species recorded in the eastern samples are replaced by others there.

Qualitative sampling on the continental slope of south-eastern Australia at depths between 200 m and 3000 m have also discovered a rich fauna. For example, the aplacophoran molluscs (a group of Tethyan origin) are concentrated on the slope and there are numerous undescribed species of them in this region (Scheltema 1990). Three species-pairs, with one of each pair occurring on the slope and the other on the shelf, indicate local speciation and a slope origin for some shelf species. Such a pattern is expected in other taxa.

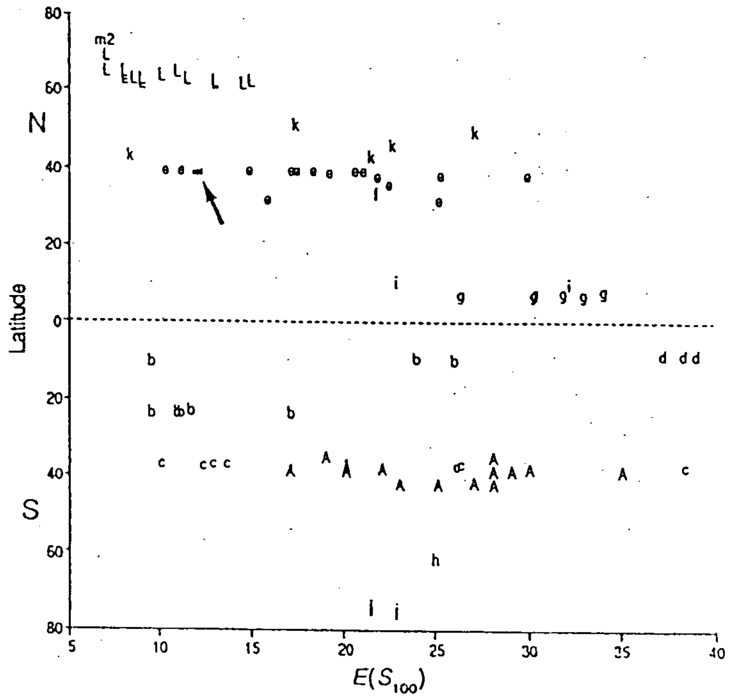

Figure 2: Expected number of species of benthic isopod crustaceans in 100 individuals calculated from Hurlbert rarefaction curves ($E(S_{100})$) versus latitude. Data labelled: **A** - South-eastern Australia; **b** - Angola Basin, **c** - Argentine Basin, **d** - Brazil Basin, **e** - Gay Head/Bermuda Transect, **f** - Mediterranean Sea, **g** - Guiana Basin, **h** - Scotia Basin; **i** - Sierra Leone Basin, **j** - Weddell Sea, **k** - West European Basin; **L** - Norwegian Basin, **m** - Greenland Basin[3]; ***** - off New Jersey and Delaware, USA, arrowed (from Poore & Wilson 1993).

The isopod crustaceans are the best studied taxon in the area so far, and 359 species belonging to 36 families are represented. Ten per cent can be identified to known species — a much smaller fraction than in comparable surveys conducted elsewhere on the slope. Poore, Just and Cohen (1994) found that depth is the main environmental factor correlated with species composition. The expected numbers of species calculated by Poore and Wilson (1993) varied between samples and matched the highest recorded in other oceans (Figure 2). The results were similar for ostracode crustaceans where more than 90 mostly new species were discovered (Kornicker & Poore in press).

Conclusions: unexplored seas

Vast areas of the continental shelf and all except two tiny areas of the slope have never been surveyed by biologists.

The results of the three case studies discussed here reflect differences in study objectives. Hence it is not possible to compare them in terms of species diversity, community composition – environment relationships or functional ecology.

Latitudinal gradients in species diversity are hinted at and, as would be expected, these differ with taxonomic group. Benthic sediments are dominated by coarse calcareous biogenic particles and this may be responsible in part for a diverse fauna throughout all of Australia's continental seas, even in its colder waters. Fine muddy terrigenous sediments dominate only near the major rivers of the northern half of Australia and may be the only places where species diversity is depressed.

Recent research in north-eastern Australia has concentrated on benthic processes and there is now a fair taxonomic knowledge of the largest epifauna. However, taxonomy of the infauna is poorly studied. To some extent this emphasis reflects the alternative interest of biologists in the effects of large rivers on the Great Barrier Reef and its lagoon. In contrast to our knowledge of north-eastern Australian benthic processes, almost nothing is known of processes in south-eastern Australia where recent terrigenous inputs are negligible. However, the infauna in south-eastern Australia — especially of crustaceans, polychaetes and molluscs — is taxonomically better understood.

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The technical paper by Dr Poore was reviewed by Dr. G. Ross, Australian Biological Resources Study, Canberra; and Dr. D. Brunckhorst, Australian Nature Conservation Agency, Canberra.

The reptiles and mammals in Australian seas: their status and management

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Introduction

There is considerable concern for the status of many marine mammals and reptiles. The giant Steller's sea cow, *Hydrodamalis stelleri*, was exterminated from the north Pacific by hunters in the 18th century. Of the seven extant species of sea turtles, five are listed by the International Union for the Conservation of Nature (1990) as endangered and one as vulnerable to extinction. Although humans have not caused the extinction of any species of whale or dolphin, some are now severely threatened by human activity (Cooke 1991).

All major groups of marine mammals and reptiles are represented in Australia. There are over 30 species of sea snakes, six species of sea turtles, one species of sirenian or sea cow, three species of fur-seals and sea-lions, and more than 40 species of whales and dolphins. This paper, which is adapted from Marsh, Corkeron, Limpus, Shaughnessy and Ward (1994), reviews the biology and management of the marine mammals and reptiles of Australia as a basis for an assessment of their status.

The saltwater crocodile, *Crocodylus porosus*, is not included in this review. It is found in estuaries, large streams, lakes, and swamps as well as marine environments. We regard it more appropriate to consider the status of this species in the context of a review of the terrestrial environment.

Marine snakes

Species distribution

Most sea snakes are venomous, benthic piscivores. All species are viviparous (ie bear live young) and display numerous adaptations to the marine environment (eg sub-lingual salt gland, paddle-tail, nasal valves). However, hydrophids, aipysurids and *P. platurus* also display significant differences in morphology (eg ventral scales), ecology (eg habitat) and biogeography, and may not be as closely related as was assumed by Smith (1926).

Sea snakes (Hydrophiidae) and kraits (Laticaudidae) are restricted to the Indian and Pacific oceans (Heatwole 1987). Most species occur in the warm, shallow seas of the Indo-Malaysian Archipelago, northern Australia and Oceania. Three groups within the Hydrophiidae (hydrophiids, aipysurids and *Pelamis platurus*) are commonly found on the northern Australian continental shelf (McDowell 1972, Marsh et al. 1994). No breeding population of laticaudids (McDowell 1967) has been found in Australian waters, but these snakes occasionally strand on the north-east Queensland coast.

Five genera of hydrophiids (*Acalyptophis, Astrotia, Disteira, Enhydrina, Hydrophis*) inhabit the northern Australian continental shelf. Hydrophiids have narrow ventral scales and typically use soft bottom (inter-reefal) habitats. Most species are specialist piscivores, feeding on eels (eg *Gymnothorax* species) and slender fishes such as gobies.

The two genera of aipysurids, *Aipysurus* and *Emydocephalus*, are characterised by their wide ventral scales. They are typically associated with coral reefs (Cogger 1989). The seven known species of *Aipysurus* are distributed between the Sahul Shelf and the Gulf of Carpentaria, and five of them are endemic to the northern Australian continental shelf. Most aipysurids are generalist piscivores. However, the olive sea snake, *A. laevis*, also eats cephalopods, crustaceans and fish-eggs; and the teeth and venom apparatus of *A. eydouxii* and the endemic *Emydocephalus annulatus* are substantially reduced as these species feed exclusively on fish eggs (McCarthy 1987).

The yellow-bellied sea snake, *P. platurus*, differs from other sea snakes in not being part of a benthic community. It is a pelagic species most commonly found in groups associated with `drift lines', where it feeds on surface fish such as mullet, trevallies and anchovies (Kropach 1975). *Pelamis platurus* inhabits tropical and subtropical seas from the western Indian Ocean to the eastern Pacific Ocean. Individuals are occasionally found stranded on beaches throughout Australia (see Cogger 1975, Guinea 1981).

Life history

Sea snakes are smaller than other marine reptiles. The heaviest sea snake is *Astrotia stokesii* which can weigh more than 2 kg. The longest sea snake inhabiting Australian waters is *Hydrophis elegans* which can reach almost 3 m.

Olive sea snakes (*A. laevis*) from the Keppel Islands and Swain Reefs are estimated to reach sexual maturity at 3 years of age (males) and 4-5 years (females) and live for at least 10 years (Burns 1984 1985). Recent data indicate that the hydrophiid sea snake, *Lapemis hardwickii*, may have a much longer lifespan.

Sea snakes tend to have smaller broods and larger offspring than have terrestrial snakes of a similar size (Lemen & Voris 1981, Shine 1988). The mean number of offspring in 15 broods of *A. laevis* from the southern Great Barrier Reef was 2.6 (Burns 1984). Brood sizes for *P. platurus* range from about two to six (see Visser 1967, Pickwell 1972, Kropach 1975).

Population sizes and trends

Inter-reefal sea snakes (hydrophiids) are probably carried large distances by currents. Populations of inter-reefal sea snakes are not amenable to mark-recapture studies because of the low rates of recapture (see Heatwole & Burns 1987). Demersal trawls provide estimates of relative abundance, but difficulties in estimating depth:time budgets and rates of net avoidance/escape complicate the calculations of population density and size.

The reefal (site attached) habits of aipysurids make them amenable to mark-recapture studies. Marking techniques include freeze branding and removal of subcaudal scales (Burns 1984). Because of logistical difficulties associated with sampling the enormous expanse of ocean through which *P. platurus* travels, the development of reliable estimates of the population sizes of this species is restricted.

Movements

There are few records of seasonal movements by hydrophiid sea snakes. The possibility of an inshore migration during the summer (wet season) was suggested by Shuntov (1971). Seasonal variation in the species composition of the sea snake by-catch of prawn trawlers was documented by Heatwole and Burns (1987). *Lapemis hardwickii* in the Gulf of Carpentaria were caught more frequently by prawn trawlers in shallow, inshore waters during spring, and in deeper waters, further offshore, during autumn (Wassenburg et al. 1994).

Mark-recapture, sonic tracking and visual mapping studies of olive sea snakes (*A. laevis*) from the Swain Reefs indicated that these snakes generally confine their movements to particular sections of reef (home ranges of about 0.18 ha for females and 0.15 ha for males).

Surface currents carry individuals of *P. platurus* over large distances. These probably account for the wide distribution of the species (Kropach 1975).

Impacts

The major impact on the habitats and populations of inter-reefal (hydrophiid) sea snakes on the northern Australian continental shelf is prawn trawling. Between 10% and 42% of sea snakes caught in prawn trawls die. The proportion killed is correlated with duration of the trawl (Heatwole & Burns 1987). Products made from sea snakes obtained from prawn trawlers (such as wallets, belts and bikinis) have been sold in markets and souvenir shops throughout northern Australia for at least 20 years.

The Queensland Department of Primary Industries has issued three licences for the collection of 10 000 sea snakes per annum from prawn trawlers on the east coast of Queensland (two licences) and the Gulf of Carpentaria (one). This harvest includes several species and its sustainability has not been assessed.

Stocks

No information is available.

Management

Sea snake products have been traded internationally since the 1930s but there are no data suggesting that any species of sea snake is endangered or threatened and no species is currently listed by CITES (the Convention on International Trade in Endangered Species of Wild Fauna and Flora). The Commonwealth's *Wildlife Protection (Regulation of Exports and Imports) Act 1982* was invoked to provide legislative support for CITES. This Act restricts the export of sea snake products from Australia (eg by tourists).

Reefal habitats in northern Australia are generally well protected and there are few impacts on Australian populations of aipysurids. Some olive sea snakes have been illegally collected from reefs in the Great Barrier Reef Marine Park for use in the aquarium trade (unpublished data). The low population densities and fecundity of aipysurid (reefal) species make them unsuitable for commercial harvest. Conservation of reef species should continue to be conducted in the context of large-scale habitat management (eg Great Barrier Reef Marine Park).

Except for the pelagic *P. platurus*, the shallow-water habitat (less than 100 m depth) of sea snakes impedes their crossing of international boundaries. Thus they can be managed within national conservation strategies. Responsibility for managing the sea snakes of the northern Australian continental shelf is shared between six State and two Commonwealth fisheries management and wildlife conservation departments. The development of national management plans for the commercial use of Australian hydrophiids caught in prawn trawls should be a high priority.

Sea turtles

Species distribution

Of the world's seven species of sea turtle, six occur in Australia. They represent all extant families and genera:

Family **Dermochelyidae**
    Dermochelys coriacea, leatherback turtle
Family **Cheloniidae**
    Caretta caretta, loggerhead turtle
    Chelonia mydas, green turtle
    Eretmochelys imbricata, hawksbill turtle
    Lepidochelys olivacea, olive ridley turtle
    Natator depressus, flatback turtle

Leatherback, loggerhead, green and hawksbill turtles have pan-tropical distributions. The olive ridley is widely distributed in the tropical and subtropical Indo-Pacific. Unlike the other species, the flatback turtle has a limited distribution and is effectively endemic to Australia (Walker & Parmenter 1990). It has not been recorded nesting outside Australian territorial waters, and its feeding grounds extend only to the Indonesian Archipelago and the Papuan coast.

Life history

Sea turtles are amongst the largest extant reptiles. Breeding adults range in size from approximately 40 kg for olive ridley turtles to over 300 kg for leatherback turtles. Sea turtles are exceptionally long-lived and breed only occasionally. For example, it is estimated that, on average, female loggerhead turtles migrate to breed on about five occasions at intervals of several years (Limpus 1992).

All sea turtles have a similar life history (Carr 1968, Bustard 1972). They spend most of their life at sea but return to land to lay eggs on traditional nesting beaches. Fecundity varies with species and ranges between about 50 eggs per clutch in flatback turtles to 130 per clutch in hawksbill turtles. There is no parental care of the eggs or young and sex is determined by the temperature at which the eggs are incubated (Limpus, Reed & Miller 1985).

The hatchlings of all species have a pelagic dispersal phase mediated through ocean currents and lasting for several years. Leatherback turtles remain pelagic; they feed on jellyfish. The other species of sea turtles recruit to shallow benthic feeding areas after they reach 35 cm or more in curved carapace length (more than 6 kg). They spend the rest of their life in that habitat except for intermittent breeding migrations. Their diet varies with species (Bjorndal 1982): green turtles eat algae and seagrasses; loggerhead turtles molluscs and crabs; and hawksbill turtles sponges and algae.

Population sizes and trends

The absolute size of some sea turtle breeding units is in the tens of thousands to hundreds of thousands. For example, the olive ridley turtle forms massive breeding aggregations of up to 150 000 individuals (Harris 1994). An estimated 11 500 green turtles were recorded nesting at Raine Island on a single night in 1984 (C.J. Limpus, unpublished data).

Population models (Crouse, Crowder & Caswell 1987) have shown that sea turtles will not withstand significant increases in mortality above natural levels. Crouse, Crowder and Caswell found that the most sensitive life history parameter is the survivorship of large pre-reproductive and adult females, the very size/sex classes targeted for harvest.

The status of most sea turtle breeding units is difficult to assess for the following reasons:

(1) the paucity of census data;
(2) the difficulties of estimating abundance and determining trends in localised feeding grounds (see Marsh & Saalfeld 1989a, Taylor & Gerrodette 1993) and in fluctuating breeding ground populations (Limpus & Nicholls 1988);
(3) the mixed stocks in feeding areas;
(4) the dispersed feeding and breeding behaviour and associated migration of members of a breeding unit (Limpus et al. 1992); and
(5) the long age to maturity, long breeding life but generally unquantified life history parameters.

Despite their apparent abundance, declining populations of green, loggerhead, hawksbill and leatherback turtles (Figures 1 and 2) have been reported from Australia (Pritchard 1969, Hirth 1971, Marquez, Villanueva & Penaflores 1976, Witzell 1983, Dodd Jr 1988, Groombridge & Luxmoore 1989, Limpus & Reimer 1991).

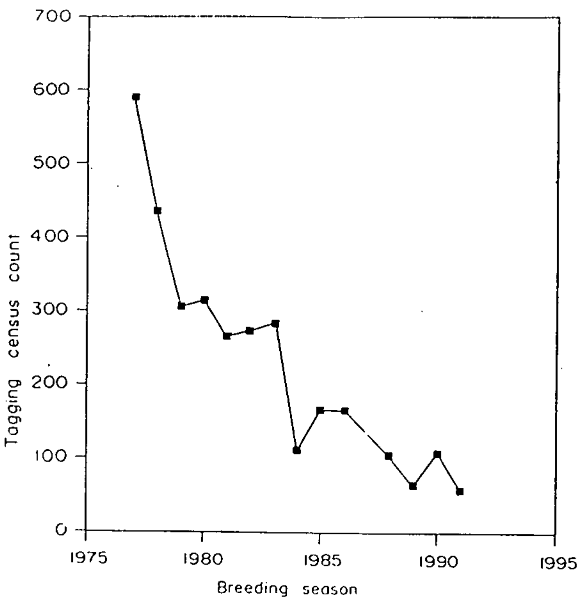

Figure 1: Population decline as evidenced by the trend in the number of nesting loggerhead turtles recorded during the annual tagging census conducted during the peak of the nesting season (last two weeks of December) at Wreck Island in the southern Great Barrier Reef since 1977. Wreck Island is the most significant breeding site for this species in the south-west pacific.

Movements

Breeding migrations are typically for hundreds of kilometres (km) and may exceed 3000 km (Bjorndal 1982). Tag re-sightings and recoveries of 118 loggerhead and 273 green turtles from eastern Queensland rookeries have documented migrations between nesting beaches and feeding areas of between 8 km (green turtle) and more than 2600 km (green and loggerhead turtles). Both of these species are capable of high-speed sustained migration in excess of 30 km per day for several weeks. They demonstrate fidelity in their choice of both feeding and nesting areas (Limpus et al. 1992). Recent genetic studies support the hypothesis that turtles home to their natal beach at least at the regional level (Gyuris & Limpus 1988, Meylan, Bowen & Avise 1990). Hatchlings from rookery regions within eastern Australia colonise feeding areas that encompass regions spanning 10-30o in latitude and 30-37o in longitude. Thus turtles from widely separated rookeries co-habit the same feeding area (Limpus et al. 1992).

Impacts

Impacts on the sea turtle populations which feed or breed in Australian waters are widespread and varied but generally unquantified (Bjorndal 1982, Groombridge & Luxmoore 1989, Limpus et al. 1989). The eggs of all species are harvested at nesting beaches. The green turtle is favoured for its meat and is the species generally targeted by traditional and commercial hunters, although loggerhead (Limpus et al. 1992) and hawksbill turtles are also eaten in some areas. Vitelogenic females (those preparing to breed) are favoured because they are fattest. Kwan (1991) monitored the harvest of turtles between 1984 and 1987 by the inhabitants of Daru Island off the Papuan coast. He found that adult female green turtles comprised up to 83% of the catch. Most turtles were caught coincident with the migration of green turtles through Torres Strait to rookeries in eastern Australia. The greatest threat to the hawksbill turtle is the tortoise-shell trade. The present rate of harvest of green and hawksbill turtles throughout the Indo-Pacific region appears unsustainable.

Some sea turtles drown when they are caught in trawl gear. The flatback turtle is the main species caught in prawn trawls in northern Australia (43%), followed by loggerhead (19%), olive ridley (15) and green (4%) turtles (Poiner, Buckworth & Harris 1990). The rate of mortality varies with the duration and depth of the trawl.

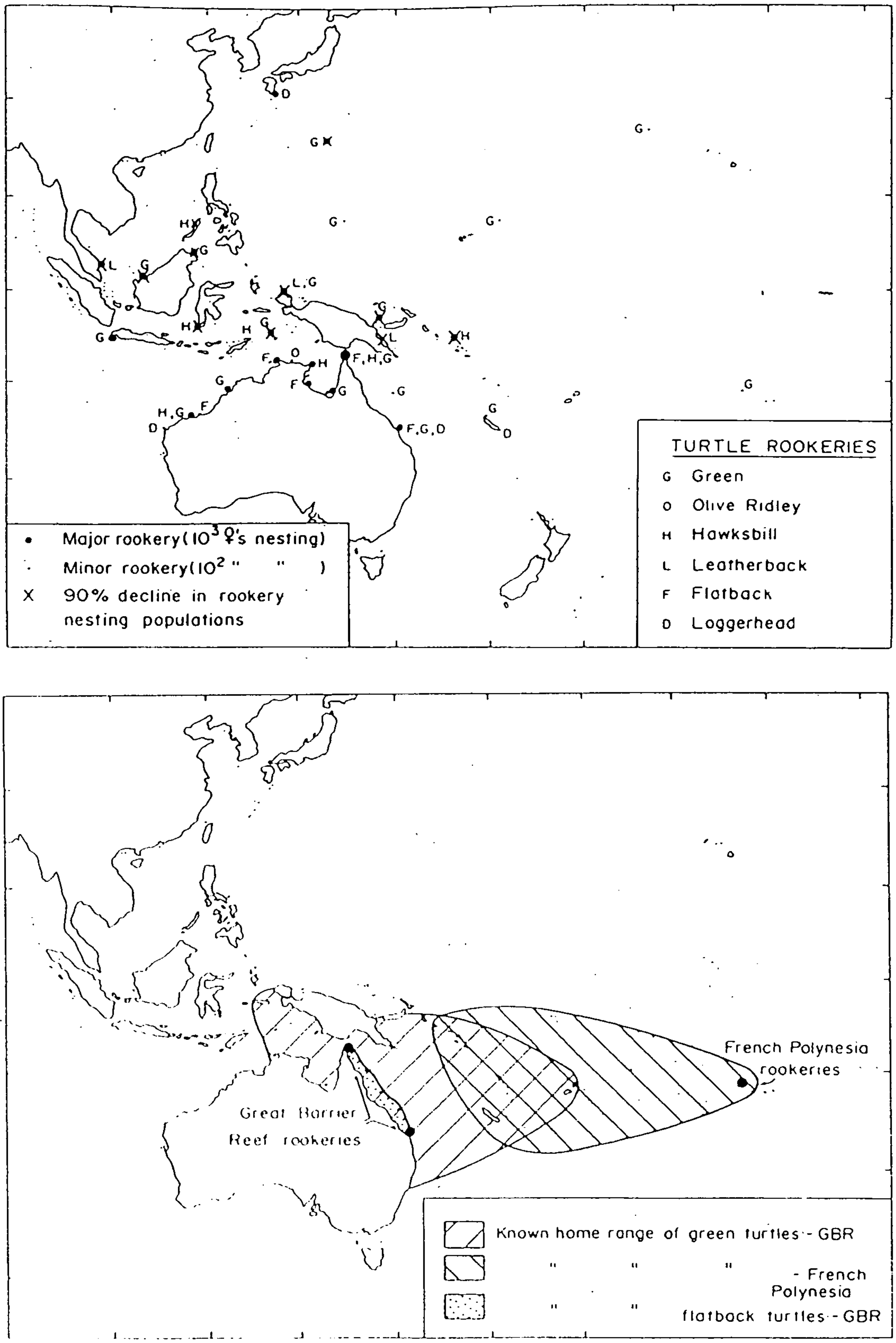

Figure 2: *Top:* Known rookeries of each species of sea turtle in Australia and Oceania at the beginning of western colonisation. Note the large number of rookeries (x) at which significant declines in the number of nesting females have occurred and/or where major harvests of eggs and adults are continuing. *Bottom:* Known home ranges, based on recoveries of tagged females while nesting; of green turtles that breed at the rookeries at the northern and southern ends of the Great Barrier Reef region; of green turtles that breed in French Polynesia; and of flatback turtles that breed in the southern Great Barrier Reef region.

An estimated 4114 ± 1369 turtles are estimated to have been caught in the Australian northern prawn fishery in 1988, of which an estimated 247 ± 90 drowned (Poiner, Buckworth & Harris 1990). Although Poiner, Buckworth and Harris concluded that the impact of trawling on sea turtle populations in the northern prawn fishery is not of immediate concern, their view is challenged by some other scientists such as Parmenter (1994), who considered that the mortality rate from fishing has been underestimated. Turtles are also drowned in gillnets and shark nets set for bather protection (Paterson 1990), but the extent of the mortality in gillnets is largely unquantified. Other causes of indirect and direct anthropogenic mortality include: feral fox predation on sea turtles nests on mainland beaches in northern Australia; destruction of nesting beaches by urban developments; and turtles being killed as a result of collisions with boats. The unintentional killing of turtles and their eggs as a result of human activities in the Australian region is threatening the survival of regional populations of loggerhead and olive ridley turtles.

Stocks

Genetic studies by Bowen, Meylan & Avise (1989) and Norman et al. (1994) demonstrated that the major sea turtle breeding aggregations are essentially genetically discrete and therefore should be regarded as separate breeding units (stocks). For example, green turtles that breed in the Gulf of Carpentaria, northern Great Barrier Reef and southern Great Barrier Reef are separate stocks. Not all the members of an individual breeding unit follow the same migratory paths. Turtles from different breeding units can be mixed in the one feeding area; eg green turtles breeding at rookeries in Java, Western Australia or the northern Great Barrier Reef have been caught in the one feeding area in north-western Arnhem land (Limpus et al. 1992).

Management

All species of sea turtle are listed on Appendices I and II of CITES. The International Union for the Conservation of Nature (1990) lists the leatherback, green, hawksbill and olive ridley turtles as endangered and the loggerhead as vulnerable. Kennedy (1990) listed flatback turtles as potentially vulnerable. All species of sea turtles occurring in Australian waters - except the flatback turtle - are listed under the Commonwealth's *Endangered Species Protection Act*: the loggerhead turtle as endangered the others as vulnerable.

All sea turtles are protected in all States of Australia, although State and Commonwealth laws allow Aborigines and Torres Strait Islanders to hunt them subject to various restrictions. Management practices in neighbouring countries frequented by sea turtles are more limited. Indonesia protects selected species (loggerhead, olive ridley and leatherback turtles); Papua New Guinea has some protected areas and bans export; the Solomon Islands protect certain size classes; and Fiji has a closed season. As a shared international resource, sea turtles are particularly relevant for inclusion in international agreements addressing the conservation of migratory species. Such an agreement between the countries bordering the Arafura and Coral seas (Australia, Indonesia, Papua New Guinea, Solomon Islands, Vanuatu and New Caledonia) is urgently required.

Sea cows (Sirenia)

Species distribution

The dugong, *Dugong dugon*, the only strictly marine herbivorous mammal, is the only member of the mammalian Order Sirenia (sea cows) present in Australia. Its range (Figure 3) extends throughout tropical and sub-tropical coastal and island waters from east Africa to the Solomon Islands and Vanuatu, and between about 26o and 27o north and south of the equator (Nishiwaki & Marsh 1985). Over much of this range, dugongs are believed to have been

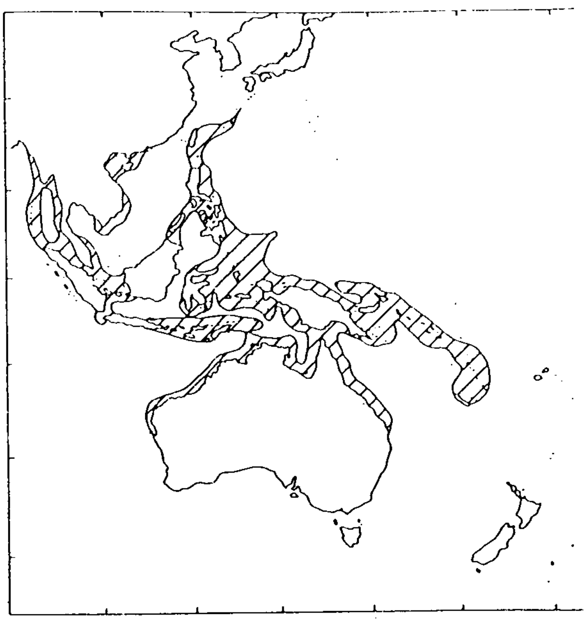

Figure 3: The distribution of dugong in south-east Asia, Australia and Oceania.

reduced to relict populations. These populations are separated by large areas where dugongs are close to extinction or are extinct. This assessment is, however, almost entirely based on anecdotal information and the actual extent to which their range has contracted is unknown. Australia appears to be the dugong's stronghold, as a significant proportion of the dugong stock is believed to be present in northern Australian waters between Moreton Bay (Queensland) and Shark Bay (Western Australia).

Life history

Adult dugongs reach a length of 3 m and weigh up to about 420 kg (Spain & Heinsohn, 1975). Individuals may live for 70 years or more, but a female does not have her first calf until she is at least 10 years old, and then only bears a single calf every three to five years after a gestation period of about 13 months (Marsh 1980, Marsh, Heinsohn & Channells 1984, Marsh, Heinsohn & Glover 1984, Marsh, Heinson & Marsh 1984, Marsh 1986). Population simulations (Marsh 1986) indicate that dugong numbers are unlikely to increase at more than about 5% each year, even if all the females in a population are breeding to the maximum.

Population sizes and trends

The only quantitative information on dugong population size comes from dedicated aerial surveys. Such surveys conducted in Australia indicate that dugongs are the most abundant marine mammal in the inshore waters of northern Australia (Marsh, unpublished data). Even though not all areas of suitable habitat have been surveyed, the population estimates sum to more than 80,000 (Bayliss 1986, Marsh & Saalfeld 1989b, 1990, Bayliss & Freeland 1989, Marsh, Saalfeld & Preem 1990, Marsh, Saalfeld & Prince 1991, Marsh et al. 1994, Marsh & Lawler 1992).

The major problem in determining the status of the dugong is the difficulty of detecting trends in abundance. Marsh and Saalfeld (1989b) considered a hypothetical situation in which the estimated 8000 dugongs in the northern Great Barrier Reef region were declining at 5% a year. Using power analysis, they calculated that it would take at least 10 years of annual surveys before it could be determined within the usual limits of statistical error whether the population was declining or not. By that stage, numbers could have declined to about two thirds of their value at the time of the first survey. The difficulties of detecting such a trend at a more localised level are much greater than this. Using present techniques, it is impossible to detect trends at spatial and temporal scales which are useful to management.

Movements

Marsh and Rathbun (1990) and Preen (1993) studied dugong movements and habitat preferences using satellite radio-tracking techniques. They found that most movements of the 18 dugongs fitted with satellite transmitters were local and in the vicinity of seagrass beds. However, one individual moved between two localities a straight line distance of 140 km apart three times in six weeks, indicating that the management of dugongs is an international matter in areas such as Torres Strait.

Impacts

Humans have been hunting or netting dugongs for thousands of years, mainly for their meat and/or oil. Dugongs are still killed for food by Aborigines in many parts of northern Australia. Contemporary hunting statistics are almost non-existent. Harris et al. (1994) estimated that the 1991 dugong catch in the Australian parts of Torres Strait was approximately 1000 dugongs. It is not known whether this harvest is sustainable (Marsh & Lawler 1992). In contrast to the estimated Torres Strait harvest, the estimated annual catch of dugongs by members of the two main Aboriginal communities on the east coast of Cape York was less than 100 (Smith and Marsh 1990) - much less than the estimated sustainable yield.

An unknown number of dugongs are killed incidentally in commercial gillnets in northern Australia (Marsh 1988). A total of 576 dugongs were killed in shark nets set for bather protection in Queensland between 1964 and 1988 (Paterson 1990).

Most of the dugong's habitat in Australia is remote from human settlements. Loss of seagrass beds because of coastal residential, industrial and tourist developments is a potential threat along the eastern Queensland coast south of Cooktown (Morissette 1991). In 1992, more than 1000 km2 of seagrass was lost from Hervey Bay - approximately 25% of the total known area of seagrass along the eastern Queensland coast - as a result of two floods and a cyclone (Preen, Lee Long & Coles, unpublished). Consequently, the regional dugong population dropped from an estimated 2206 (±420) in August 1988 (before the die off) to 600 (±126) in November 1993 (21 months after the floods; Preen & Marsh, submitted). At least 99 dugong carcasses were recovered from Hervey Bay and surrounding areas following the seagrass loss, the peak of mortality occurring 6-8 months after the floods (Preen and Marsh, unpublished data). The subsequent decline in dugong population was probably due to emigration. Some dugongs moved at least 900 km from Hervey Bay (Preen & Marsh, unpublished data).

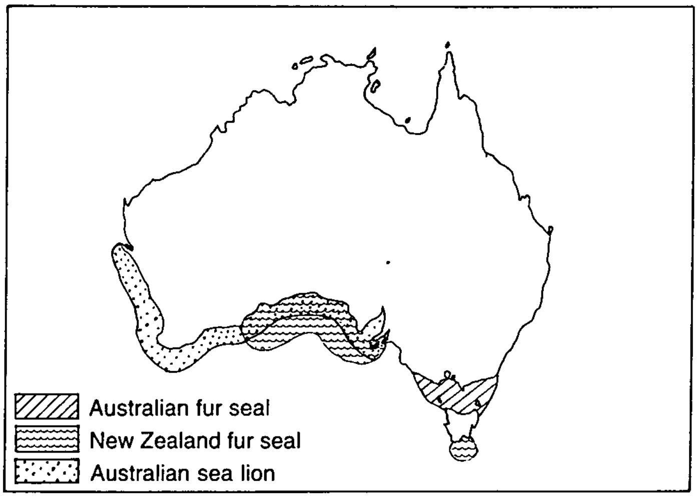

Figure 4: The breeding range of fur seals and sea lions in Australia.

Stocks

The only information available on dugong stock identity in Australia is a morphometric study by Spain and Marsh (1981). They demonstrated that the skulls of dugongs from Mornington Island in the Gulf of Carpentaria can be statistically distinguished from those from Townsville on the east coast, approximately 1850 km away by sea.

Management

The dugong is listed as vulnerable to extinction in the *IUCN red list of threatened animals* (International Union for the Conservation of Nature 1990). Trade in dugong products is either regulated or banned (depending on the dugong population involved) by CITES.

In Australia, relevant State and Commonwealth laws allow Aborigines and Torres Strait Islanders to hunt dugongs subject to various restrictions. The management of dugongs in Torres Strait is an international matter for Australia and Papua New Guinea and dugong hunting is a nominated fishery under the Torres Strait Treaty for bilateral management.

Sea lions and fur seals (Pinnipedia)

Species distribution

Three pinnipeds breed in Australia (Figure 4): Australian sea-lion, *Neophoca cinerea*, New Zealand fur-seal, *Arctocephalus forsteri* and Australian fur-seal, *Arctocephalus pusillus doriferus.* All belong to the fur-seal and sea-lion family Otariidae. The Australian sea-lion is endemic; the New Zealand fur-seal is common to both Australia and New Zealand and the Australian fur-seal also breeds in South Africa. Five species of Phocidae, or true seals, occur in the Australian Antarctic Territory and its subantarctic islands.

Life history

Males are considerably larger than females. Australian sea-lions may reach 400 kg, Australian fur-seals 360 kg and New Zealand fur-seals 185 kg. All three species breed ashore (generally on remote islands) and feed at sea, mostly on fish and cephalopods. They are territorial and highly polygamous. Females give birth to a single pup. Pupping is synchronised in the fur-seals and occurs during early summer. The Australian sea-lion has a 17-18 month pupping cycle and there is asynchrony in the pupping season between colonies. The reason for this asynchrony is not known.

The rate of increase in seal populations is limited by their members exhibiting delayed sexual maturity (3-5 years in females, older in males), and bearing single offspring each year. The maximum reported for an Australian population is 15% per annum for New Zealand fur-seals on Kangaroo Island, South Australia (Shaughnessy 1990).

Population sizes and trends

Determining the distribution and abundance of these seal populations has not been straightforward for two reasons: (1) the proportion of the whole population ashore or on the sea surface at any one time is not often known, and (2) they occur in remote locations. Shaughnessy and Gales (1990) attempted to determine the breeding distribution of the Australian sea-lion and New Zealand fur-seal in South Australia and Western Australia in the 1989-90 summer by visiting more than 200 islands. They found 13 breeding colonies of fur-seals (only five of which were known previously) in Western Australia and four new breeding colonies in South Australia. Shaughnessy and Gales (1990) concluded that the population of New Zealand fur-seals is increasing at the three locations for which previous records are available. They also discovered several new breeding locations for sea-lions along the southern Australian coast, but were unable to assess whether sea-lion numbers were decreasing or increasing because of the lack of longitudinal data.

Behavioural traits of these seals useful for determining abundance ashore are their tight aggregations and - with the exception of Australian sea-lions - their brief pupping season. Abundance estimates are usually directed at pups, since this is the only age class recognisable and ashore together. However, this approach raises the problem of converting pup numbers into an estimate of overall abundance. Good demographic data on which to do this accurately are usually not available.

Movements

None of the seal species undertakes long migrations. The number of seals ashore in breeding colonies varies throughout the year and is greatest during the pupping season. Adult females suckle their pups ashore for about 10 months, and intersperse periods of several days ashore nursing pups with several days at sea feeding. The need to return to shore to feed their young means that the movements of some pinnipeds tend to be more restricted than those of other mammals such as cetaceans and sirenians which nurse their young at sea.

Impacts

Important impacts on seal populations are interactions with fisheries, oiling, entanglement in man-made debris and disturbances ashore (tourism). Both *A. forsteri* and *A. p. doriferus* have been shot at fish farms in south-eastern Tasmania in recent years, although this practice should decrease as fish farmers install predator-proof fencing (Pemberton 1989). An oil spill affected pups of *A. forsteri* in Western Australia in February 1990 as a result of the wreck of the *Sanko Harvest* (Gales 1991). These incidents are likely to increase with the expansion of the transport and mining of petroleum products in the waters of the region. The rate of entanglement of *A. p. doriferus* and *N. cinerea* in plastic straps and fragments of netting in Tasmanian waters (Gales 1990, Pemberton, Brothers & Kirkwood 1991) is high by world standards. Disturbances to breeding colonies are also increasing as people become more mobile and more interested in wildlife.

Stocks

Shaughnessy (1970) detected genetic differences between populations of *A. forsteri* in New Zealand and Australia using electrophoretic techniques. Little else is known of stock identity among these species, but research into this aspect is underway using isozyme electrophoresis and DNA finger printing.

Management

None of these species is listed in the IUCN Red Data Book (IUCN 1990). The Australian sea-lion is considered by the South Australian and Western Australian governments as rare and is listed as endangered by Kennedy (1990). *A. fosteri* and *A. pusillus doriferus* are considered to be secure.

In Australia, seals are managed by a variety of State conservation and fisheries agencies and, beyond the three-mile territorial limit, by the Australian Nature Conservation Agency.

Whales and dolphins (Cetaceans)

Species distribution

The mammalian Order Cetacea includes two modern suborders, the Mysticeti or whale-bone whales and the Odonotceti or toothed whales, porpoises and dolphins. Forty four species of cetaceans are present in the waters around Australia, including eight species of baleen whales and at least 38 species of toothed whales and dolphins. However, knowledge of cetacean taxonomy is such that this list may be incomplete. For example, minke whales (*Balaenoptera acutorostrata*) may comprise more than one species (Hoelzel & Dover 1991) as may Bryde's whales (*B. borealis*: Report on the workshop on the genetic analysis of cetacean populations 1991).

No cetacean species is endemic to Australian waters. Australian cetaceans can be broadly grouped on the basis of their distributions as follows: cosmopolitan species which occur throughout the world's oceans; species which occur in temperate and polar waters worldwide; species with a pantropical -generally warm temperate distribution; species with a southern hemisphere (generally circumpolar) distribution; and species with an Indo-Pacific (generally tropical and warm temperate) distribution. The resident species in Australian inshore waters include the bottlenose dolphin

(*Tursiops truncatus*), the Irrawaddy River dolphin (*Orcaella brevirostris*) and the Indo-Pacific humpbacked dolphin (*Sousa chinensis*). The latter two are restricted to tropical and warm temperate waters. Humpback whales (*Megaptera novaeangliae*) pass close to the eastern and western coasts of Australia on their annual breeding migrations (Figure 5, top) and southern right whales (*Eubalaena australis*) breed in southern coastal waters (Figure 5, bottom).

Life history

Cetaceans are all large mammals, and those in the waters of Australia range in size from dolphins less than 2 m long to the blue whale (*Balaenoptera musculus*), which can be up to 33.5 m long, the largest animal that has ever existed. Most of the largest cetaceans are baleen whales. However, there are some smaller baleen whales including the 10 m minke whale and the 7 m pygmy right whale(*Caperea marginata*). Toothed whales range in size from the sperm whale (*Physeter macrocephalus*), the males of which may be up to 18 m long, to the small dolphins.

Cetaceans spend all their lives in the water. They typically bear one young at a time at intervals of several years after a pregnancy lasting from 10 months in small dolphins to 17 months in beaked whales (Perrin & Reilly 1984). The calves of some toothed whales and dolphins may suckle for several years; lactose has been detected in the stomachs of 13-year-old immature male sperm whales (Best, Canham & Macleod 1984). The pre-reproductive period varies from three years in some dolphins to 20 years or more in male sperm whales (Perrin & Reilly 1984, Best, Canham & Macleod 1984, Olesiuk, Bigg & Ellis 1990). The female reproductive cycle in baleen whales is usually two years, with a gestation period of about a year. Lactation lasts about half a year (Lockyer 1984). Normally one calf is born in warm temperate or subtropical waters. Most baleen whales undertake extensive migrations for feeding and the reproductive cycle appears to be geared to this annual cycle of migration and feeding.

Population sizes and trends

Population estimates for many species can be determined only by carrying out aerial or boat based surveys over large areas of ocean. Such surveys present huge logistic and financial difficulties. Even species occurring sufficiently close to land masses and therefore being suitable for shore-based census work pose serious difficulties in population estimation. The decay of an observer's capacity to sight animals with increasing distance from shore is confounded by the influence of weather conditions. Further difficulties may arise from the pattern of distribution of groups and individual whales. Because cetacean populations increase slowly, surveys must span several years before population trajectories can be ascertained, while imprecise population estimates reduce the power of a survey series to determine the existence of a trend.

Two separate series of land-based surveys of humpback whales migrating off the east Australian coastline have produced similar estimates of population size and rate of increase. These are: 790 animals in 1987 (with 95% confidence intervals of 732-884, although total numbers are probably underestimated by around 15%), with the population increasing at 14.4% per annum (no confidence intervals given) between 1981 and 1987 (Bryden, Kirkwood & Slade 1990); and 1107 animals (no confidence intervals given) in 1987, having

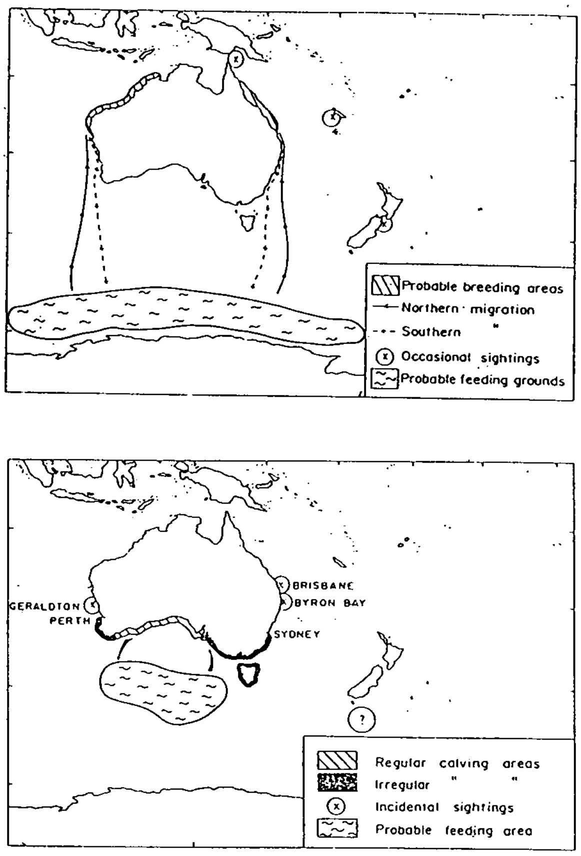

Figure 5: The breeding and feeding areas and migratory routes of humpback (top) and right whales (bottom) in Australian and New Zealand waters.

increased at 9.7% per year (95% confidence intervals 6-13%) since 1981 (Paterson & Paterson 1989). Regular aerial surveys since 1976 of the humpback whales migrating along the coast of Western Australia indicate that this population has increased at 8.8% per annum (95% confidence intervals 3-14.6%) since whaling ceased in 1963 (Bannister, Kirkwood & Wayte 1991). The results of annual aerial surveys along the south coast of Western Australia since 1977 suggest that right whale numbers are increasing at 11.7% per year (95% confidence intervals 4.5-18.9%) (Bannister 1990).

Movements

There is considerable specific variation in the movement patterns of cetaceans. Some large mysticetes undertake annual migrations over 40o of latitude from the Antarctic to tropical waters. In contrast, some inshore delphinids maintain home ranges of perhaps tens of square km. The ranging patterns of many pelagic species are unknown, as are the pelagic migration paths of some mysticetes.

Only the migration patterns of those mysticetes which undergo coastal migration are even vaguely understood. Although the movements of humpback whales along parts of the eastern coast of Australia are known in some detail (Paterson & Paterson 1989, Bryden, Kirkwood & Slade 1990, Bryden & Corkeron 1990), the northern termini of the migrations, and the breeding areas (if they can be defined at all) along both coasts are unknown (Figure 5) as are the Antarctic feeding grounds. The areas along the southern coast of Australia where southern right whale females and calves spend the winter (Figure 5) are reasonably well known (Bannister 1990). However, the patterns of movement of other age and sex classes of right whales are not described, nor are their Antarctic or subantarctic feeding areas and their migration routes.

Impacts

Until recently, the major human impact on cetaceans was hunting. The exploitation of minke whales continues, and the resumption of limited commercial hunting of this species in the northern hemisphere is a management issue in the International Whaling Commission (IWC); the IWC is under pressure to lift the moratorium on commercial whaling and to implement catch limits under a revised management procedure. However, in the southern hemisphere the IWC's 1994 decision to implement the Southern Ocean Sanctuary in which catch limits will be set at zero for all commercial whaling precludes the resumption of operations by large-scale factory vessels in this region.

Many scientists consider that small cetaceans such as dolphins are under greater threat than the large whales. Human activities which are having a deleterious impact on cetaceans now include fisheries by-catches, pollution and habitat degradation.

Gillnets set by Taiwanese vessels in offshore waters within the Australian Fishing Zone in the Arafura Sea caused significant mortalities of a variety of delphinid species in the 1980s (estimated at 4700 animals between June 1981 and March 1983; Harwood et al. 1984). Gear restrictions placed on the fishing fleet in December 1986 rendered fishing within the Australian Fishing Zone uneconomic, and removed the problem from Australian waters.

Organochlorine pollutants, and particularly polychlorinated biphenyls (PCBs) pose a significant threat to the viability of cetacean stocks worldwide (Tanabe 1988) and could threaten mammals with extinction (Cummins 1988). Mammals are particularly sensitive to reproductive failure due to bioaccumulation of PCBs (Subramanian et al. 1987; Cummins 1988). There are few data on the organochlorine loads present in cetaceans in the waters off Australia. Levels of PCBs measured in the tissues of fish which form the basis of the diet of inshore dolphins are such that these pollutants could affect the dolphins' reproductive viability (Preen, Thompson & Corkeron 1992).

Coastal development, fisheries and offshore industrial development affect cetaceans although quantification of such impacts is exceptionally difficult. Reduction in prey numbers by destruction of suitable habitat, and by commercial and amateur fishing must cause an increase in the time devoted to foraging by local cetacean populations. The increase in boat traffic and noise pollution associated with offshore oil development could have impacts on coastal migrators such as humpback and right whales, and local dolphin populations.

Tourism, particularly vessel based whale-watching is increasing in popularity, and will represent a significant cause of harassment to coastal migrating whales. Humpback whales off the east Australian coast are observed by whale-watching vessels at points along almost their entire migration, and the presence of these vessels results in changes in the whales' behaviour (Bryden & Corkeron 1990). Tourist activities designed to allow people to interact with resident bottlenose dolphins occur in several areas. There will undoubtedly be further pressure to establish similar areas. While these interactions with free-ranging dolphins may prove of conservation value by making the general public more aware of their local cetacean fauna, care is needed to ensure that these animals are not harassed.

Stocks

There is no good information on stock differentiation for most of the cetacean species in Australian waters. Recent molecular genetic work has verified the results obtained from marking during whaling, demonstrating that humpback whales migrating along the east and west coasts of Australia are two distinct stocks (Chittleborough 1965, Baker et al. in press). No significant genetic differences have been found between humpback whales migrating off eastern Australia and those off Tonga, although this may be due to small sample size (Baker et al. in press). There has been a suggestion that the right whales off south-eastern Australia may form a separate stock from those off south-western Australia (Report of the workshop on the status of right whales 1986). However, as there is evidence of intermingling between right whale populations across the south Atlantic Ocean (Best et al. 1993), it is likely that the right whales occurring off southern Australia form one breeding aggregation. Photo-identification and molecular genetic work both under way, should resolve this question.

Management

Most species of cetacean are classified by the International Union for the Conservation of Nature as insufficiently known, reflecting the paucity of knowledge of the order generally (Klinowska 1991). Blue whales are classed as endangered, and southern right, humpback, sei and fin whales as vulnerable. Longman's beaked whale (*Mesoplodon pacificus*) is considered to be the rarest whale in the world; a live specimen has never been positively identified. The species is known from only two specimens (both skulls and jaws), one of which was found near Mackay on the east coast of Queensland.

Responsibility for the management of the stocks of large cetaceans in international waters is vested in the International Whaling Commission (IWC). With over-exploitation of stocks, the IWC placed a moratorium on all commercial whaling from 1986. Some hunting of minke whales continues for `scientific purposes' but this is strongly opposed by Australia which was at the forefront of the decision to place a moratorium on commercial whaling. With the Seychelles, Australia initiated the 1979 IWC declaration of the Indian Ocean whale sanctuary.

Australia passed the *Whale Protection Act* in 1980 which prohibits the killing, capturing, injuring, harassing, chasing and herding of whales, dolphins and porpoises in the Australian Fishing Zone. Cetaceans are protected under complimentary State legislation within 3 nautical miles of land. The maintenance of dolphins in captivity was tightly controlled after a Senate Select Committee investigation in 1983-85. Cetacean management and administration under the Whale Protection Act is undertaken by the Australian Nature Conservation Agency.

To date, no international body oversees the management of small cetaceans in international waters off Australia. While national and local bodies (eg national parks services, fisheries services) oversee the management of cetaceans within their jurisdiction, national and international co-operation and continuity of regulation is lacking.

A strategy for conserving marine reptiles and mammals in Australia

The listing of sea turtles, dugongs, and some of the cetaceans occurring in Australia as endangered or vulnerable to extinction is an expression of concern for their status and not a definitive diagnosis (nor a cure!). The status of most of the marine mammals and reptiles of this region is unknown and likely to remain so for the foreseeable future without a major long-term research effort.

It is not appropriate to infer the status of these species solely from their abundance. For example, a conservative estimate of the abundance of large sea turtles in Torres Strait is of the order of 65 000 (Marsh & Lawler 1992). However, because of the long pre-reproductive period of sea turtles, it will be decades before we can assess the impact of current levels of hunting and egg harvest on these animals.

It is almost impossible to determine trends in abundance for most marine mammals and reptiles at temporal or spatial scales that are useful to managers (see Taylor & Gerrodette, 1993). Most species are large and long-lived with a slow rate of natural population increase; many species are wide-ranging. All are incredibly difficult to census. It may be more fruitful to concentrate on species demography in assessing anthropogenic impacts rather than attempt to monitor trends. For a demographic approach to work, information is required on the species' life history, the size of populations and the levels of anthropogenic mortality.

In the recent past, the major impact on sea turtles and marine mammals has been from commercial and traditional hunting. As values change, the major impacts are becoming more indirect. The effects of commercial fishing on marine mammals and reptiles are generally unquantified but are of major concern for some species. More insidious impacts are increasing: loss of habitat from coastal developments, pollution and collisions with boats to name a few.

The **most effective approach** to the conservation of marine reptiles and mammals is to give high protection to these species and their habitats as part of large, multiple-use management areas such as the Great Barrier Reef Marine Park. Surveys are required to identify priority areas for protection. Dugongs and green turtles - as the focus of traditional hunting cultures - are a management challenge in some areas. However, it must be acknowledged that the hunting culture is dependent on the survival of the prey. Species which are caught as by-catch in lucrative commercial fisheries are also difficult to manage. Some reductions in by-catch can be effected by gear modifications but this could meet with resistance from the fishing industry. Would the community be prepared to meet the cost of modifing fishing procedures to save an endangered sea snake?

It is appropriate to manage sea snakes, flatback turtles, dugongs, seals, and coastal dolphins at a national level except in international regions like Torres Strait. Wide-ranging species (such as the other sea turtles and many of the oceanic cetaceans) cannot be managed effectively without international co-operation. Sea turtles with mixed stocks on both feeding grounds and nesting beaches (Limpus et al. 1992), present a difficult challenge.

In some parts of Australia, departments of environment have responsibility for marine reptiles and mammals. In other areas they are the responsibility of fisheries departments. Fisheries agencies usually do not fund research on species which do not have commercial value and most departments of the environment do not have the funds nor expertise to undertake marine research. Many potential graduate students yearn to study marine reptiles and mammals; however, these animals are generally unsuitable for graduate student projects. They are long-lived, difficult and expensive to study, and not amenable to experimental work. Unless the problem of conducting research on Australian marine reptiles and mammals is addressed strategically, a significant proportion of mammal work will be conducted by visiting North Americans. This pattern is already evident in cetacean research. Meanwhile competent local cetologists struggle on limited and short-term funding. The conservation of marine reptiles and mammals in Australia and Oceania needs to be based on the results of studies by local biologists. Otherwise much of the information and expertise will be exported and unavailable to managers and a major rationale for the research will be lost.

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The technical paper by Professor Marsh and colleagues was based on a paper in press in Conservation Biology in Australia and Oceania. It was also reviewed by Dr G. Ross ABRS and Dr. F. Michaelis, Department of Environment, Sport and Territories, Canberra.

The status of Australia's seabirds

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Introduction

The seabird fauna of Australia and its external territories is diverse and comprises 110 species representing 12 families (Serventy, Serventy & Warham 1971, Marchant & Higgins 1990, Selkirk, Seppelt & Selkirk 1990, Woehler 1991, Woehler & Johnstone 1991, Woehler, Hodges & Watts 1991). Of these, 76 species (69%) breed, and many spend their whole lives in one or more parts of the region. A further 34 species are regular or occasional visitors to the region where they feed in the non-breeding season. The species diversity of the Australian region is intermediate between that of the North Pacific Ocean (62% breeding, total of 154 species in 13 families) and the North Atlantic Ocean (68% breeding, 81 species in 12 families) (Harrison 1983).

Although the biology of many Australian seabird species is better known than that of many terrestrial birds and other marine organisms, information on population levels and status for most species is poor. Non-sustainable exploitation by man of breeding seabirds largely has ceased, but a variety of other anthropogenic effects is putting pressure on some breeding populations around Australia.

Population estimates for each breeding species assessed against various human impacts, provide a measure of the status of Australian seabirds. The value of seabird populations for monitoring purposes, and the research needed to implement effective monitoring, are discussed in this report.

Value and usage

Australian seabird breeding colonies have provided a valuable and fairly predictable food source for indigenous communities, mariners, explorers and entrepreneurs for centuries.

Seabirds breeding on islands close to shore were probably an important food source to coastal Aboriginal communities. The impact of their harvesting on seabird populations is unknown as few records remain. Shearwater chicks from Mutton Bird Island, New South Wales were eaten in initiation ceremonies (Swanson 1976). Serventy, Serventy and Warham (1971) suggested that some Western Australian islands suited to seabird habitation yet presently unoccupied by them, may have been overexploited by Aborigines in the past. South-west Tasmanian Aborigines valued seabirds highly and travelled at least 15 km on paperbark rafts to Maatsuyker Island to hunt seals and nesting short-tailed shearwaters (Flood 1989).

In Torres Strait, islanders from Murray, Darnley and Stephens islands continue a traditional harvest of booby chicks and eggs on Bramble Cay and several smaller islands, their harvest facilitated by modern small craft (Elvish & Walker 1991). Such harvests occurred previously on Raine Island (Serventy, Serventy & Warham 1971). Torres Strait Islanders also harvest a large proportion of eggs from Kusamet Island every year. Adults of several bird species are collected for feathers to be used in ceremonies (Johannes 1991).

The impact of intermittent harvesting of seabirds is impossible to estimate. Harvesting was initiated by South-east Asian trepang fishers and followed by European mariners, pearling crews and more recently Indonesian and Japanese fishers. However, the impact of harvesting is likely to be less than that of sealers on Australian and subantarctic islands, or resident labourers on islands during commercial guano collection. Until recently, the practice of killing seabirds for crayfish bait was prevalent (Serventy, Serventy & Warham 1971), although less so in Western Australia. Little penguins were preferred, and black-faced shags, short-tailed shearwaters and Australasian gannets were also used widely. The effects of such collection were seen most strikingly in the decline of the Cat Island gannetry in Bass Strait from 2500–5000 pairs to fewer than seven pairs between 1908 and 1977 (Warham & Serventy 1978).

Up to 150 000 king and royal penguins a year were boiled down for oil on Macquarie Island between 1890 and 1920 (Cumpston 1968). The penguins reputedly provided a meagre return of about £20 per year.

Muttonbirding encompasses the traditional and sustainable commercial harvest of short-tailed shearwaters. It is conducted on seven Bass Strait islands largely by Tasmanian Aborigines, under management of the Tasmanian Government. The current harvest of 400 000 chicks a year is valued at $330 000 to the harvesters and retails at $ 3.2 million (Beaton 1990). The impact of illegal muttonbirding on populations on several Tasmanian islands is uncertain, or limited on Victorian islands.

Providence petrels saved the fledgling Norfolk Island settlement from starvation in 1790, when some 180 000 petrels were eaten over a three-month period (Hicks 1988). Norfolk Islanders have a traditional harvest of sooty tern eggs from nearby Phillip Island (Jurd 1989). Cocos Island Malays have harvested large numbers of red-footed boobies from North Keeling Island, an annual take estimated at 3000–10 000 birds (J. Hicks, pers. comm.). Following a cyclone, a moratorium was placed on harvesting in 1989 to allow the colony to recover. In contravention of Australian legislation enacted in 1992 however, illegal harvesting of red-footed boobies and frigatebirds around Cocos (Keeling) Atoll continues.

Tourism potential of seabird colonies is limited by the problem of constraining human disturbance and access to most colonies on offshore islands. The successful 'Penguin Parade', established on Phillip Island (Victoria) in 1928, presently draws nearly 500 000 visitors annually. Although measures to reduce disturbance have been successful, the colony on the island is in decline. The decline is apparently the result of increased adult mortality at sea and on land, where penguins are killed primarily by foxes, dogs and motor vehicles (Dann 1992). In New South Wales, penguins form the basis of a new tourist initiative in Jervis Bay and on Montague Island. Carefully-managed tours by charter boat to the Houtman Abrolhos (Western Australia) are increasing, and an industry developing around Michaelmas Cay, off Cairns (Queensland), is being encouraged to regulate itself to ensure protection of the seabird colonies. Numerous islands — including several seabird islands in the Capricorn/Bunker group of the southern Great Barrier Reef — form the focus of tourist activities.

Description and status

Australia

In broad terms, the tropical and southern ocean elements of the seabird fauna reflect Australia's geographical location (Table 1). A few tropical species have adapted to a broader range of environmental conditions. Almost all tropical species extend south to about Fraser Island (25°S) on the east coast. The influence of the warm Leeuwin Current enables tropical roseate terns, bridled terns and red-tailed tropicbirds to breed as far south as 32–34°S on the west coast (Dunlop & Wooller 1990).

Estimates of the **number of breeding pairs** of seabirds along the Australian coast (Table 1) have been compiled from published population data for 294 islands surveyed and a few coastal sites as well as unpublished data from several sources. Excluded were gull-billed terns, white-winged terns and whiskered terns, great cormorants and little black cormorants, because they breed inland and generally feed infrequently in marine environments.

The Seabird Islands Report (SIR) series (Numbers 1–215 published in *The Australian Bird Bander*

Table 1: Estimates of numbers of breeding pairs of Australian seabirds along sections of the Australian coast and Coral Sea

Region	1	2	3	4	5	6	7	8	9	10	11	Total	
												Min	Max
Little Penguin [C]	800-1200					13.3-23.5e3	23.4-29.5e3	2489-2748	243-447	85.6-151.8e3	23.3-40.6e3	149130	249900
Shy Albatross [C]										2700-3300	4200-5200	6900	8500
Great-winged Petrel [C]									33.1-84.1e3			33050	84100
Herald Petrel [T]					3							3	3
Black-winged Petrel [T]						3						3	3
Gould's Petrel [T]						250-500						250	500
Fairy Prion [C]							1359-1639			71.8-118.7e3	98.2-156.2e4	1055060	1682000
Wedge-tailed Shearwater [T]	10.7-10.9e5	30.1-60.3e3			15.1-15.3e4	43.3-53.3e3						1301150	1344400
Flesh-footed Shearwater [C]								br	10.5-31.1e4			104540	310600
Sooty Shearwater [C]						296-510					3-700	300	1210
Short-tailed Shearwater [C]						26.6-27.1e3	1.96e6	495300	3330-3200	7.8-10.1e6	2.5-3.4e6	12787070	16059700
Little Shearwater [C]	24.0-54.4e3								3060-7140			27060	61600
White-faced Storm-petrel [C]	8399-19370					10.6-12.2e3	32600	143770	9620-21020	160600	5000-8000	370180	396600
Common Diving-petrel [C]							20+			83.2-116.0e3	46.0-68.0e3	127220	184000
Australasian Pelican [I]		428-988	<200		140-220		177		30-70	47		1030	1680
Australasian Gannet [C]							3140+			1000-1500	1406-1500	5540	6140
Masked Booby [T]		590-610		399-614	2763-3046							3750	4270
Red-footed Booby [T]		13		1290-4896	78							1380	4990
Brown Booby [T]		3.4-3.8e4	16.2-22.7e3	722	9001-12418							59940	73900
Pied Cormorant [T]	3400-4000	6554-12594	130		90	500	138	1465-1645				13080	19120
Little Pied Cormorant [I]	10				7	100-150		25-30				140	200
Black-faced Cormorant [C]	21					50-120	90+	4280-4420	46	2700	700	7740	8110
Great Frigatebird [T]		3-7	1	1581	22							1610	1610
Lesser Frigatebird [T]		9530-10100	4919	2253-2265	2160-2347							18680	19430
Red-tailed Tropicbird [T]	1	1-4		267	22-102				3-7			290	380
White-tailed Tropicbird [T]		0-2										2	2
Silver Gull [I]	6480-14967	1150-2190	129		398-504	61.1-66.6e3	41.8-51.5e3	12.6-13.0e3	3400-7550	4200	2100	133890	163620
Pacific Gull [C]	74-84	14					437	80-121	7	1100	185	1900	1950
Kelp Gull [C]						11	3				300	315	315
Caspian Tern [I]	141	767-927			7-9		89-94	83-201	14	41	13	1160	1410
Roseate Tern [T]	4310-5060	300-760	990		1618-6560							7220	13370
White-fronted Tern [C]										44		44	44
Black-naped Tern [T]					1706-2076							1710	2080
Sooty Tern [T]	28.3-33.8e4	10.0-50.0e3			35.8-76.5e3							328760	383750
Bridled Tern [T]	10.1-44.0e3	3080-4080			6882-9738				1			20040	57870
Little Tern [T]			11		56	80-87	400	6		10		560	570
Fairy Tern [I]	1512-1962	375-454					200+	100-130	135	109		2420	2990
Crested Tern [I]	1380-1780	5870-13770	13.0-15.0e3		24.1-29.2e3	11.7-12.7e3	2705-3355	4750-5830	3630-8430	6900	400	74350	89940
Lesser Crested Tern [T]		2-500	35		4676-7634							4710	8170
Common Noddy [T]	116500	13.6-35.1e3			44.4-62.5e3							174480	214130
Lesser Noddy [T]	79500											79500	79500
Black Noddy [T]		50-100			11.9-13.1e4							119340	130840
											TOTAL	17025500	21673500

Key: 1, Cape Leewin to Shark Bay; 2, Shark Bay to Northern Territory border; 3, Northern Territory & Gulf of Carpentaria; 4, Coral Sea; 5, Cape York to New South Wales border; 6, New South Wales; 7, Victoria; 8, South Australia; 9, South Australian border to Cape Leeuwin; 10, Bass Strait; 11, Southern Tasmania.; [C], [T] and [I] refer to Southern Ocean species, tropical species and those derived from tropical species, respectively; e3, e4 and e6 = x 1000, 10 000 and 1 000 000; br = breeding, but numbers not estimated

Volumes 11–14, 1973–1977 and its successor, *Corella* Volumes 1–16, 1977–1992) formed a primary data source. Though incomplete, this series provided useful regional comparisons in this report. Other data sources are Serventy (1952), Green and Mollison (1961), Serventy and Whittell (1962), Warham (1962), Stirling, Stirling and Shaughnessy (1970), Lavery and Grimes (1971), Bush and Lodge (1977), Kolichis (1977), Burbidge and George (1978), Cox (1978), Abbott (1979), Storr (1980), Harris and Norman (1981), Whinray (1982), Storr (1984), Berry (1986), Storr, Johnstone and Griffin (1986), Burbidge et al. (1987), Australian National Parks and Wildlife Service (1989), Burbidge and Fuller (1989), Hill and Rosier (1989), Marchant and Higgins (1990), King, Hicks and Cornelius (1992), Walker (1992), King (1993) and Menkhorst (1993).

Population estimates for Western Australian islands are based wholly on unpublished data from the Department of Conservation and Land Management database, developed and maintained by A.A. Burbidge and P.J. Fuller. Similarly, the unpublished seabird island database established by the Tasmanian Department of Parks, Wildlife and Heritage is used here. Estimates for South Australia are based largely on unpublished data held by the Department of Environment and Land Management (Robinson, Canty, Mooney & Rudduck, unpublished manuscript). The SIR series provides limited spatial and temporal coverage for these areas, as well as for Torres Strait, the Coral Sea and parts of the Great Barrier Reef.

To improve their robustness, estimates have been expressed as maximum and minimum values where possible. Estimates derived from non-numerical comments by several observers were expressed as maximum and minimum numbers of breeding pairs in the following manner: 'few pairs' or 'several tens of thousands' became 3–7 pairs or 30 000–70 000, on the basis that 3×10^n and 7×10^n are appropriate maximum and minimum levels around a central midpoint of 5×10^n. Where estimates were available within the same year or 2–3 year period, they were expressed as maximum and minimum numbers.

Several factors increase the likelihood of error in these estimates.

Population estimates based on nesting burrow densities are almost certainly too high, because burrow occupancy by breeders may range from 43% to 86% in sooty, short-tailed or wedge-tailed shearwaters and little penguins (Harris & Norman 1981, Marchant & Higgins 1990, Dyer & Hill 1992). Estimates for South Australia and Western Australia were based on burrow occupancy rates. It is possible also that colonies of some surface-nesting species which breed almost continuously (such as some tropical terns and boobies) may comprise sub-populations with different breeding seasons. This factor, coupled with interannual fluctuations in the size and timing of breeding, suggest that the single counts made for many of the island seabird populations are probably too low (Serventy, Serventy & Warham 1971, King, Hicks & Cornelius 1992).

Conversely, addition of records for different years from several islands in the same region will overestimate numbers for species which frequently change breeding sites (for example, pied cormorants, and crested, roseate or fairy terns). Less information is available for remote islands or those difficult to land on and survey data are often spread over the last two decades. Publication dates of the SIR series indicate that 14% of islands were surveyed before 1975, 59% from 1975–1984 and 26% after 1984.

The total estimated number for each Australian seabird species (Table 1) ranges across seven orders of magnitude. Four species are known from fewer than 100 breeding pairs. Breeding of the white-fronted tern was first proven in 1979 (Whinray 1982). Less certain is the status of the miniscule, incipient colonies of white-tailed tropicbirds, herald petrels and black-winged petrels. The status of Gould's petrel and the little tern, two of five species with fewer than 1000 breeding pairs, are discussed below under 'Threatened species'. Although low in numbers, the widespread, endemic Pacific gull seems secure, and numbers of the burrowing common diving-petrel are more likely to be underestimated than those for most species. The kelp gull is considered a recent arrival in Australian waters (Serventy, Serventy & Warham 1971).

At the other end of the scale, eleven species exceed an estimated 100 000 pairs (little penguin, fairy prion, three shearwater species, white-faced storm-petrel, common diving-petrel, silver gull and three tern species). Most remarkable of these is the short-tailed shearwater, which constitutes some 16 million pairs or 74% by number of all 42 Australian breeding seabird species. Wedge-tailed and flesh-footed shearwaters form a further 7.7% and 6.2% respectively.

The estimated **biomass of breeding seabirds** in each 3° block around Australia provides a simple basis for distributional and ecological comparisons between areas. For present purposes, species were grouped according to dietary preferences as feeding predominantly on plankton, small or medium-sized fish (more than approximately 100 mm long), cephalopods and/or crustaceans, and scavenged material. No distinction was made between inshore and offshore feeders. Diet and weight data are from Serventy, Serventy and Warham (1971) and Marchant and Higgins (1990) (see Table 2).

Seabirds feeding on plankton

The southerly distribution of breeding planktivorous species is particularly evident, especially in and around Bass Strait, and to some extent in South Australia. An estimated 13 690 t (tonnes) of these birds occur in Bass Strait — primarily short-tailed shearwaters — and are equal to 67% of the estimated biomass of all seabirds breeding around Australia. Until their nestlings require food, breeding short-tailed shearwaters feed largely on krill, *Nyctiphanes australis*. Although little is known of its feeding distribution in this period, they probably feed in the more productive waters over the continental shelf of southern Tasmania (Skira 1991). While nutrient levels — and therefore productivity — are comparatively high around Tasmania, nutrient levels and productivity in Bass Strait are generally low (Bunt 1987). Despite this, the total biomass of breeding seabirds in Bass Strait far exceeds that of birds in the southern Benguela region off Africa (1157 t: Duffy, Siegfried & Jackson 1987), an area renowned for its high productivity.

Seabirds feeding on small fish

The highest concentration of breeding colonies of species which feed on small fish occurs in the Houtman Abrolhos, one of the most important breeding locations in Australia in terms of biomass and species diversity. High concentrations of birds at this trophic level breed on islands off south-western Western Australia, especially flesh-footed shearwaters breeding on Sandy Island, near Windy Harbour. The southern Great Barrier Reef also supports a large biomass of several tern species and wedge-tailed shearwaters. In the northern Great Barrier Reef and islands in Torres Strait, terns and least frigatebirds are the most important birds in this feeding group. North Bountiful Island (Gulf of Carpentaria) supports perhaps the largest known colony of crested terns in the world (Walker 1992). Little penguins comprise 98% by mass of birds feeding on small fish in Bass Strait.

Seabirds feeding on medium prey

The few large concentrations of birds feeding on medium-sized prey consist almost entirely of masked boobies and brown boobies and some Australian pelicans. These feeding concentrations are found off north-western Australia, in Torres Strait, islands west of Cape York and the Wellesley Islands (Walker 1992). Walker linked the last of these with high productivity in the Gulf of Carpentaria during the breeding season, although trawl discards may influence numbers of brown boobies.

The 'snapshot' nature of the data limits the detection of population trends in almost all species. A notable exception is the silver gull, which is increasing in some areas at an exponential rate of 10–13% annually in response to increased edible domestic garbage and the development of pastoral areas and more reservoirs. Such increases in gulls are especially evident around Sydney, Melbourne, Adelaide and Perth (see Figure 1a; scavenger biomass, Table 2) and are correlated with human population increases (Meathrel, Mills & Wooller 1991).

Numerous **anthropogenic activities** have an effect on Australian island seabird colonies. About 47% of 215 islands in the SIR series are subject to one or more indirect and direct human threats. The lowest (23%, n = 39) and highest (88%, n = 8) regional variations in such disturbance occur off the southern and western coasts of Western Australia respectively, although the significance of such variation is difficult to evaluate. Islands which are difficult to land on (30%, n=64) are far less prone to human impacts than those which are moderately difficult (22%, n=47) or easy (48%, n=104) to access. This trend remains, even though difficult islands also occur closer to shore (mean distance offshore, 12.7 km) than those with moderately difficult (23.2 km) or easy access (28.5 km) and are therefore potentially more open to visitation.

Studies on little penguins on Phillip Island illustrate the complexity of anthropogenic effects on seabirds, where loss of habitat, introduced predators and mortality from vehicles coupled with the possible adverse effects of commercial fisheries, have led to the decline of populations (Dann 1992).

As described under 'Value and Usage', direct exploitation — including illegal poaching of adults, chicks or eggs — has obvious impact unless it is conducted under careful management.

Table 2: Weight of breeding seabirds (t) per 3° block, based on the most recent figures available. Data sources and descriptions of categories are given in the text. Horizontal and vertical headings indicate longitude and latitude of the top left corner of each square respectively.

	112	115	118	121	124	127	130	133	136	139	142	145	148	151
9														
12														
15														
18														
21														
24														
27	1.38													
30	0.008	0.78						366						7.1
33				11.1				108	101	0.40			10.6	0.17
36										0.36	96.4	1000	24.6	
39 Planktivores											4051	2376	5486	
42 Total weight: 17646 tonnes												3789	216.0	
9										0.01	26.6			
12				37.9							13.1	1.2		
15				3.06	11.4					20.7		34.9	2.0	0.79
18		25.2	6.07									0.07	9.1	
21	21.7	6.36												139
24	0.86													7.3
27	1037													9.8
30	2.44	9.89												35.9
33		368	9.65	10.0				0.39	19.7	0.2			29.2	1.5
36									0.68	0.79	7.9	26.6	48.9	
39 Small fish, cephalopods and/or crustaceans											88.1	96.1	8.9	
42 Total weight :2259 tonnes												65.6	14.0	
9											17.5			
12				0.13	0.25	0.71	0.46				20.7	0.001		
15			0.002	41.7	36.1				33.6	19.3			4.72	1.4
18		3.37	23.4										3.7	0.13
21	0.10	2.40												4.9
24	35.6													0.006
27	4.76													
30	1.43	7.86												0.39
33		80.2	8.04	7.99				0.03	2.2				0.13	0.13
36									3.7	13.7	1.67	0.58	0.17	
39 Medium-sized fish, cephalopods, and/or crustaceans										26.6	0.02	0.46		
42 Total weight: 451 tonnes												40.9	0.11	
9											0.02			
12											0.00			
15				0.04						0.08				
18		0.65											0.05	
21		0.44												0.22
24	0.26													0.02
27	0.28													0.00
30	0.03	9.18						0.01						0.68
33		4.44	0.18	0.10				0.68	8.00	0.04			33.9	0.62
36									0.01		31.4	1.09	6.23	
39 Scavengers											12.40	2.19	0.89	
42 Total weight: 116 tonnes												1.30	0.06	
9										0.012	44.1			
12				38.1	0.25	0.71	0.46				33.8	1.2		
15			0.002	44.8	47.5				33.6	40.1		34.9	6.7	2.2
18		29.2	29.5									0.07	12.9	0.13
21	21.8	9.19												144.4
24	36.7													7.4
27	1043													9.8
30	3.90	27.7						366						44.1
33		453	17.9	29.3				109	131	0.64			73.9	2.4
36									4.39	14.82	137	1028	79.9	
39 All categories										4178	2474	5497		
42 Total Weight: 20 472 tonnes											3897	230.00		

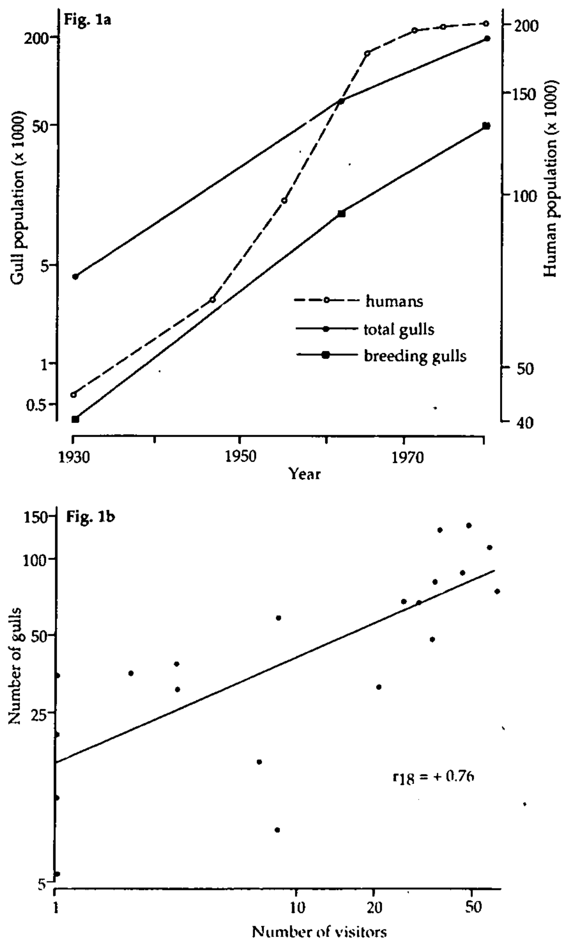

Figure 1: a) Growth in silver gull populations. Changes in the resident human population in the Wollongong and Port Kembla region (open o), the total gull population (closed o) and the gull population breeding on the Five Islands group (closed square), 1930–1980. b) The relationship between daily numbers of non-breeding gulls and human visitors on Heron Island, Great Barrier Reef (after Meathrel, Mills & Wooller 1991).

Several additional direct and indirect factors which are less easily assessed also affect Australian seabird populations. They are noted below.

Bushfires, caused by accident, lightning, or lit deliberately, have destroyed thousands of seabirds or their nests on at least 16 islands in four States.

Birds in Australian pelican and cormorant colonies are used occasionally as shooters' targets.

Recreational visits to islands have most effect on surface-nesting species, although nests in burrows may be damaged or destroyed through inadvertent trampling and subsequent collapse. Disturbance of adult breeding birds on islands increases the potential exposure of eggs and chicks to predation and the sun. Hill and Rosier (1989) claimed that breeding wedge-tailed shearwaters and black noddies can co-exist successfully with tourists on Heron Island. However, Walker (1991) maintained that these authors' approach was too simplistic. He stated that far more detailed biological and ecological data were required to substantiate their claims and he emphasised the need for caution in further development. Some species, such as bridled terns, crested terns and frigatebirds, may abandon their breeding colonies when approached too closely. Pelicans and both species of giant petrel are readily disturbed when breeding (J. Warham, pers. comm). The minimum approach distance suggested by Erwin's (1989) studies may not be attainable on very small breeding islands.

The effect of aircraft noise on seabirds is being investigated (Brown 1990).

Species nesting on beaches are prone to disturbance by vehicles, walkers and dogs.

Feral cats were recorded at eight islands in the SIR series. They are often associated with lighthouse staff and their impact on seabirds on these islands is unknown. Gulls are breeding again on North-West Island, Great Barrier Reef, after cats were eradicated (Walker 1987). Some damage to eggs and chicks has been attributed to black rats, known from eight islands, all close to shore. However, these predators have affected islands not in the SIR series. in Western Australia, cats, rats and guano mining together extirpated 'millions' of seabirds from Rat Island, Houtman Abrolhos (Storr, Johnston & Griffin 1986) and rats probably eliminated common noddies from Bedout Island. Domestic dogs contributed to the decline in shearwaters on Sawtell Island (New South Wales) and little penguins at Twofold Bay (New South Wales). Foxes and dogs also prey on penguins on Phillip Island (Dann 1992). In Victoria, foxes are known to cross the 1.8 km of water to Benison Island, and were eliminated on Griffiths Island. Erosion caused by rabbits and goats on islands may result in burrow collapse. However, sheep had little effect on populations of short-tailed shearwaters on Big Green Island, Bass Strait (Norman 1970). Birds may become impaled or

> entangled in exotic weeds such as African boxthorn or kikuyu grass, which grow thickly enough to restrict access to nests (Twyford 1992).

Discarded by-catch from fishing vessels may provide an important food source for seabirds. For example, pied cormorants, little pied cormorants and crested terns in Moreton Bay depend largely on discarded by-catch from prawn trawlers (Blaber & Wassenberg 1989). In Torres Strait, more scavenging of by-catch by seabirds, dolphins and fish occurs in trawled areas compared with non-trawled areas — perhaps reflecting learned behaviour by scavengers (Hill & Wassenberg 1990). Such scavenging could increase the numbers of these species artificially. Conversely, fishing gear kills large numbers of seabirds each year. In the North Pacific, an estimated 131 000 – 281 000 shearwaters are killed annually in Japanese gillnets. Short-tailed shearwaters formed most of this catch (Ogi 1984, in Skira 1991). The serious decline in albatross populations of several species is attributed to mortality in Japanese longline fisheries in the Southern Ocean of approximately 44 000 birds annually (Brothers 1991, de la Mare & Kerry in press). Few data are available on the effects of low-level oil spillage and plastic pollution on Australian seabirds.

Few **quantitative studies of seabirds at sea** have been made around Australia and fewer still are related to oceanographic parameters. Very little work has been done along the north-east coast, where the oceanography is complex and timing of breeding might be expected to reflect short, localised pulses of higher productivity (for example, Smith 1993). Brandis, Chafer and Smith (1992) summarised data for 46 species from monthly observations between 1984 through 1990 off Wollongong, New South Wales, and referred to publications based on these data. Seasonal species composition changes rapidly during April–May and September–October, and may reflect seasonal changes in the southerly extension of the Tasman Front. The temperature change across this front appears to be an important distributional barrier for several Southern Ocean species, as shown by the abundance of four species of albatrosses off New South Wales (Figure 2) (Wood 1992). Norman (1992) related seabird distribution and biomass in Port Phillip Bay and off Phillip Island to diet and seasonal movements. Five seabird assemblages identified off western and north-western Australia were linked to oceanographic conditions (Dunlop, Wooler & Cheshire 1988). Seabird biomass and species diversity were highest over the South Equatorial Current and, overall, seabird distribution patterns were linked to levels of marine productivity.

Oceanic islands and Australian Antarctic Territory

The **numbers of seabirds breeding** on the tropical Cocos (Keeling) and Christmas islands and the more temperate Norfolk and Lord Howe islands are provided in Table 3. Humans have had extensive impact on the seabirds of all four island groups.

The main atoll of Cocos (Keeling) has lost most of its bird fauna as a result of long-term human interference. This situation is in stark contrast to the significant populations of seabirds on North Keeling Island 24 km to the north, where Stokes, Shiels and Dunn (1984) reported an increasing amount of hunting of red-footed boobies, least frigatebirds, great frigatebirds and other species by local inhabitants. Least frigatebirds have declined from an estimated 4400 pairs in 1985 (J. Hicks, pers. comm.) to perhaps 200 pairs in 1993 (J. Tranter, pers. comm.) on North Keeling Island.

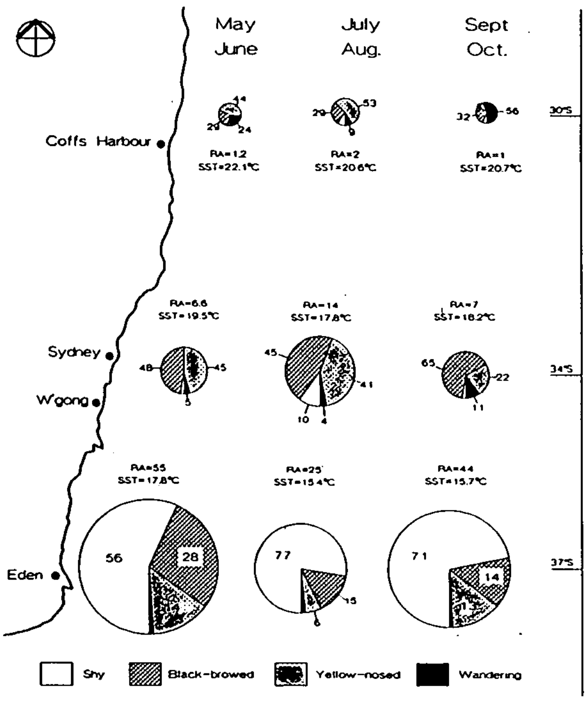

Figure 2: The abundance of shy (no shading), black-browed (hashed), yellow-nosed (stipple) and wandering (black) albatrosses over the continental shelf off New South Wales, from May to October. Relative abundance off Coffs Harbour in September–October is set arbitrarily at RA = 1, and species abundance is a percentage; SST = mean sea surface temperature (after Wood 1992).

Table 3: Numbers of breeding pairs of seabirds for Australian tropical and sub-tropical oceanic islands. Data sources are given in the text.

Species	Cocos (Keeling) Islands	Christmas Island	Lord Howe Island	Norfolk Island
Providence Petrel			27000	
Kermadec Petrel			few pairs	
Black-winged Petrel			100-1000	50-100
Wedge-tailed Shearwater	<100			sev 00 000
Flesh-footed Shearwater			20000-40000	
Little Shearwateter			4000	100-1000+
White-bellied Storm-petrel			>1000	
Australasian Gannet				<5
Masked Booby	75		300	300
Red-footed Booby	22800	12050		
Brown Booby	40	4910		
Abbott's Booby		3000		
Great Frigatebird	c. 200	3250		
Least Frigatebird	c. 200			
Christmas Frigatebird		1280		
Red-tailed Tropicbird	1	1380	200	230-300
White-tailed Tropicbird	40	6000		
Sooty Tern	40		50000+	40000-70000
Common Noddy	4800	5390	1000	100s -1000s
Black Noddy				1000-10000
Grey Ternlet			<1000	1000-10000
White Tern	abundant		<10	abundant

Since the closure of the Christmas Island Phosphate Mining Corporation, hunting on this island no longer affects populations of red-footed boobies, brown boobies and great frigatebirds. Loss of forest breeding habitat through clearing for mining has had most effect on the endemic Christmas frigatebird and the endangered Abbott's booby. Reville, Tranter and Yorkston (1990) suggested that breeding success of the latter species may be 10% lower than before clearing, through the effect of increased wind close to clearings and displacement of breeding pairs. However, based on a comprehensive nest survey, an estimate of nearly 3000 pairs suggests that the population was underestimated previously (Yorkston & Green 1992). Feral cats and rats have little impact away from settlements (Stokes 1988). Declaration of the Christmas Island National Park has protected 63% of the forest habitat.

Feral cats and rats also have an impact on breeding by little shearwaters, white terns and red-tailed tropicbirds on Norfolk Island (Schodde, Fullagar & Hermes 1983). Clearing of trees on Norfolk Island and the complete destruction of forest on nearby Phillip Island by pigs, goats and rabbits has reduced nesting habitat for black noddies and white terns, but this deforestation has probably assisted the ground-nesting masked booby. The harvesting of sooty tern eggs from Phillip Island is managed on a seasonal basis. The extirpation of providence petrels from Norfolk Island in the 1790s by colonists for food was hastened by foraging pigs (Schodde, Fullagar & Hermes 1983), although a few pairs were rediscovered nesting on Phillip Island in 1985 (Hicks 1988).

On Lord Howe Island, breeding populations of Kermadec petrels and white-bellied storm-petrels may have been eliminated by feral cats and black rat predation, although populations of these seabirds survive on adjacent islets (Garnett 1992).

The **seabird fauna of Macquarie Island** is subantarctic and has several species in common with the southern Australian fauna, itself derived largely from subantarctic groups (see Table 4). Although Heard and McDonald islands are south of the Antarctic Convergence, their seabird faunas are subantarctic in nature (Watson 1971) and share faunal elements with Macquarie Island. Numbers of breeding pairs for species in the Australian Antarctic Territory (AAT) and on subantarctic islands are given in Table 4.

Table 4: Numbers of seabirds breeding on Macquarie, Heard and McDonald islands, and in the Australian Antarctic Territories.

Seabird	Macquarie Island	Heard and McDonald Islands	Australian Antarctic Territory
King Penguin	218000-250000	5700	
Emperor Penguin			59335
Gentoo Penguin	5000	16574	
Adelie Penguin			661965
Rockhopper Penguin	500000	10000	
Macaroni Penguin		2000000	
Royal Penguin	810000-960000		
Wandering Albatross	7	(1)	
Black-browed Albatross	70	680-790	
Grey-headed Albatross	80-100		
Light-mantled Sooty Albatross	500-700	200-700	
Southern Giant-Petrel	4000	4400-4600	163
Northern Giant-Petrel	1000		
Southern Fulmar			29065
Antarctic Petrel			164521
Cape Petrel		br	4033
Snow Petrel			8599
White-headed Petrel	7850		
Blue Petrel	500-600		
Antarctic Prion	48900	>10000	2
Fairy Prion	40		
Fulmar Prion		>1000	
Sooty Shearwater	1770		
Wilson's Storm-Petrel		br	41624
South Georgian Diving-Petrel	10000-20000		
Common Diving-Petrel	20	>1000	
Macquarie Island Shag	760		
Heard Island Shag		89	
Subantarctic Skua	br	550	>100
South Polar Skua			357
Kelp Gull	br	50-100	
Antarctic Tern	40	1348	

br = breeding occurs, but no estimate of numbers; ( ) = breeding doubtful. Data after Woehler (1991), Woehler & Johnstone (1991) and Selkirk, Seppelt & Selkirk (1990).

Royal penguins and probably king penguins on Macquarie Island had recovered fully from exploitation before 1930. The decline in the wandering albatross populations from 29 pairs to seven between 1967 and 1984 is attributed primarily to incidental capture by longline fishing vessels (de la Mare & Kerry in press). Rabbits threaten burrowing species by inducing removal of vegetation with subsequent slips, soil and burrow loss; also they assist in maintaining populations of cats and skuas, which prey on seabirds. All feral wekas or Stewart Island rails — which were introduced last century — were eradicated in the 1980s. Wekas, cats and black rats prey directly on seabirds, their eggs or chicks, and were probably responsible for the extermination of grey petrels on the island.

Pesticide and mercury residues in penguins, giant petrels and a skua sampled in 1978 ranged from 0.02–2.9 parts per million (ppm) and 0.16–2.6 ppm respectively; and no polychlorinated biphenyls (PCBs) were detected (Rounsevell & Brothers 1984).

No anthropogenic impacts of significance have been identified at Heard or McDonald islands (Woehler 1991).

Estimates of **minimum breeding populations** in the Australian Antarctic Territory (AAT) (Table 4) range between 0.3% (Antarctic fulmar) and 43% (Antarctic petrel) of total breeding populations for these species (Woehler & Johnstone 1991). One third of all emperor penguins and Adelie penguins breed in the AAT. For unknown reasons, giant petrel populations have declined by 55% in the AAT since 1956, while populations of most other species are stable or have increased, at least locally. Threats to seabirds are minimal and strictly controlled. Levels of pesticides, PCBs and mercury are very low in these species (Luke, Johnstone & Woehler 1989).

Seabird distributions at sea between Tasmania and the Antarctic between 1980 and 1986 have been described by Johnstone and Kerry (1976), Kerry, Horne and Dorward (1983) and Bretagnolle and Thomas (1990). Woehler, Hodges and Watts (1991) have mapped all records for the same area taken between 1981 and 1990 and these records are to be analysed further.

Status of threatened species

Fourteen species or subspecies of seabirds in Australia and its Territories under discussion have been identified as threatened. Objectives for recovery and management actions required or already initiated have been identified (Garnett 1992).

The wandering albatross population at Macquarie Island, Abbott's booby on Christmas Island, and the New Zealand subspecies of Antarctic tern *Sterna vittata bethunei* on Macquarie Island are classified as 'Endangered' using IUCN criteria. The Antarctic tern population comprises 20–50 pairs. Comments on the first two species have been made above.

Vulnerable species include the Kermadec petrel and the white-bellied storm-petrel at Lord Howe Islands, and the Christmas frigatebird, which breeds in a single colony susceptible to cyclones. Several species comprise small populations at risk from possible capture by longlining fishing vessels, or predation. On Macquarie Island they include the grey-headed albatross and the Macquarie Island shag *Phalacrocorax purpurascens*; on Heard Island they include the Antarctic tern, *Sterna v. vittata*, the little tern, *Sterna albifrons sinensis*, the fairy prion and the Heard Island shag, *P. nivalis* — which comprises less than 500 individuals confined to one island.

In Australia, the little tern, which nests between the high tide mark and shore vegetation, has declined to less than 500 pairs, as a result of excessive disturbance by humans, dogs and off-road vehicles. Driscoll (1993) reviewed the biology and banding data for this species. The nominate subspecies of Gould's petrel, *Pterodroma l. leucoptera*, breeds only on Cabbage Tree Island (New South Wales) and may not exceed 150 pairs. The northern subspecies of the soft-plumaged petrel, *Pterodroma mollis deceptionis*, is included in this review since it is believed that a colony of less than 100 pairs breeds on Maatsuyker Island, Tasmania (Garnett 1992). The only other population of this subspecies breeds on Amsterdam Island, southern Indian Ocean, where it is under threat from feral cats.

Management

Almost all Australian seabird islands within three nautical miles of the coast are managed by the relevant State or Territory wildlife authority. Many of these islands are declared nature or faunal reserves, or form part of a larger national park. Parts of islands with lighthouses may be leased by the Australian Department of Transport. A few islands are under local district or borough councils.

Between three nautical miles and 200 nautical miles from the coast, seabirds are protected by the National Parks and Wildlife Regulations in force under the *National Parks and Wildlife Conservation (NPWC) Act 1975*, administered by the Australian Nature Conservation Agency (ANCA). Within the Great Barrier Reef, the *Great Barrier Reef Park Act 1975* protects seabirds seawards of the low water mark and on Commonwealth-owned islands. Ningaloo National Park is managed jointly by Conservation and Land Management (CALM), Western Australia and ANCA under the *National Parks and Wildlife Conservation (NPWC) Act 1975*. Offshore, the Houtman Abrolhos and Barrow Island are managed by Western Australia. The Lord Howe Islands, including Balls Pyramid, are a declared reserve under New South Wales legislation and are managed by the National Parks and Wildlife Service of New South Wales.

Seabirds within the Coral Sea islands, North Keeling and parts of southern Cocos (Keeling) Islands territories are protected under National Parks and Wildlife Regulations. Lihou and Coringa reefs in the Coral Sea are included in national nature reserves declared under the NPWC Act. The Cocos (Keeling) Islands are largely privately owned. Two ANCA officers stationed in the Territory as Government conservators advise on environmental issues, including environmental education. Most of Christmas Island is administered as a national park declared under the NPWC Act through the Christmas Island conservator. The Norfolk Island Government permits continued harvesting of sooty tern eggs on a seasonal basis. Phillip Island is reserved under Norfolk Island legislation, and an area on Norfolk Island is protected under the NPWC Act. Following removal of rabbits by ANCA, vegetation is regenerating on Phillip Island.

Ashmore Reef National Nature Reserve, in the Territory of Ashmore and Cartier islands, was declared under the NPWC Act, and is managed by the ANCA. Traditional harvesting is now prohibited, and the activities of Indonesian fishers are controlled under Memoranda of Understanding (1974, 1989) between the governments of Australia and Indonesia.

The Japan-Australia Migratory Birds Agreement (1974) and the China-Australia Migratory Birds Agreement (1988) impose obligations on Australia to protect birds that migrate between Australia and Japan or China. The agreements are implemented by a combination of Commonwealth, State and Territory legislation in the various jurisdictions within Australia. Similar agreements are being considered with Russia, Thailand, Indonesia and Papua New Guinea under the Convention on the Conservation of Migratory Species of Wild Animals (the Bonn Convention). An agreement on the Conservation of Migratory Waterbirds of the Asia-Pacific Region is nearing completion. APNA contains a management plan and a principal action plan. Australia is coordinating efforts within the region to identify priority groups of species for which specific action plans can be developed.

The South Pacific Region Environmental Program (SPREP) and the Convention on the Conservation of Nature in the South Pacific (the Apia Convention) promote conservation of seabird species with restricted breeding distributions in the South Pacific region.

Australia is party to the Convention on International Trade in Endangered Species (CITES). Controls on the import and export of endangered wildlife required by CITES are implemented through the *Wildlife Protection (Regulation of Exports and Imports) Act 1982*. The Act also controls exports of all national wildlife.

Macquarie Island, which comes under the jurisdiction of Tasmania, was declared a wildlife reserve in 1933 and became the Macquarie Island Nature Reserve in 1971 under the *Tasmanian National Parks and Wildlife Act 1970*.

Heard and McDonald islands were listed on the Register of the National Estate in 1983, in terms of the *Australian Heritage Commission Act* of 1975. Seabirds and other fauna are protected under the *Territory of Heard Island and McDonald Islands Environment Protection and Management Ordinance 1987*, administered by the Antarctic Division of the Department of the Environment, Sport and Territories.

The protocol on environmental protection to the Antarctic, known as the *Antarctic Treaty (Environmental Protection) Act 1980*, incorporates the Agreed Measures for the Conservation of Antarctic Fauna and Flora (1964). It implements the provisions of the Madrid Protocol under which treaty parties agree to manage Antarctica as a national reserve devoted to peace and science. The Agreed Measures protect all seabirds, and additional protection is provided within Special Protected Areas and Sites of Special Scientific Interest, although these areas and sites protect only a small proportion of bird sites (see Woehler & Johnstone 1991). The *Antarctic Marine Living Resources Conservation Act 1981* applies to the territorial waters of the AAT. Plans of management for each station there are being drawn up.

Status of knowledge

The basic biological information on seabird species within the Australian region ranges from inadequate to excellent (Marchant & Higgins 1990). The short-tailed shearwater, which has been studied continuously since 1947, is particularly well known (Skira 1991). For most species, activities at sea (especially regarding diet and energetics), feeding areas and the effects of fishing on seabirds, are poorly understood, if at all. With some exceptions (for example, King, Hicks & Cornelius 1992), available data on breeding population sizes are too imprecise to show trends in population or species numbers, even though they were collected over a long period. Very few species have been monitored or thoroughly investigated experimentally and general breeding biology is poorly understood.

In terms of biomass, Bass Strait and its environs support 75% of breeding seabirds in Australia. Most of these are planktivorous and feed largely on *Nyctiphanes australis* — although probably not in Bass Strait itself. This species of krill is confined to the inshore continental shelf and is an important prey of several commercial fish species.

Databases of seabird breeding populations are maintained by the Department of Conservation and Land Management, Western Australia, and by the Victorian Department of Conservation and

Natural Resources (*Atlas of Victorian wildlife*). Queensland, Tasmanian and South Australian State wildlife authorities are assembling databases from data on file. Some seabird data are included in the *RAOU atlas of Australian birds* (1977–1981) (Blaber, Davies & Reilly 1984) and data for all banded seabirds are maintained by the bird and bat banding unit of ANCA. However, there is no national database dealing with Australian seabird populations. The Australian Antarctic Division holds census details for breeding seabirds in the AAT and approximately 40 000 sighting records of birds at sea over ten years between Tasmania and Antarctica (Woehler, Hodges & Watts 1991). The forthcoming RAOU seabird atlas will describe the distribution and abundance of pelagic birds at sea off south-eastern Australia.

Environmental reporting

Seabirds have a variety of biological attributes which suit them to environmental monitoring programs. Most breed in colonies in a fixed location well suited to ready observation and counting and in numbers appropriate for statistical analysis, although access to colonies may be expensive or difficult logistically. As seabirds are long-lived and many species take several years before breeding, variation in diet and reproductive parameters generally reflect environmental changes rather than variation in average experience of a seabird population. Seabirds can be used as indicators of such environmental perturbations.

Seabirds are useful indicators of regional prey abundance (for example, Baird 1990) and selected responses to changes in food abundance, such as body condition, age at first breeding, or mortality (for example, major changes associated with ENSO events: Schreiber & Schreiber 1989) can provide integrated views of their environment at different time scales. Seabird diet can be monitored readily by regurgitation or pellet analysis. Eggs are ideal for assessing bio-accumulation of persistent chlorinated hydrocarbons, and heavy metals may be measured from tissues (Brothers & Brown 1987) or feathers. Such parameters are easier to obtain and more economical to monitor in seabirds than in other marine organisms, especially on a continental scale. The Commission for the Conservation of Antarctic Marine Living Resources' ecosystem monitoring program (CEMP) (Commission for the Conservation of Antarctic Marine Living Resources 1991) would provide a useful model in establishing such a system. Projects could be developed as part of a national environmental awareness program involving regional interest groups supported by modest Federal or State funding.

Conclusions

Overall, the status of most Australian seabirds appears to be satisfactory. However, it should be noted that 13 of the 110 species found in the Australian region are endangered or vulnerable and there have been marked declines in populations of several other species, particularly albatrosses. Information useful for determining changes in status is less than satisfactory and for some species, such as the fairy prion and common diving petrel, distribution and population size are uncertain. In most instances, populations harvested previously have recovered well from past exploitation, although oceanic islands such as Norfolk and Lord Howe appear more susceptible to irreversible loss of species. Few continental islands are afflicted with feral predators such as rats, cats and foxes. However, disturbance by human visitors may be an increasing problem, although some research suggests that management solutions are feasible. National guidelines for human visitation to seabird islands will be an important contribution to seabird conservation.

To improve management, it is essential that a national database on seabird breeding populations be assembled: a network of State and Territory databases would be most effective. The model devised by the Canadian Wildlife Service may prove useful (Nettleship & Glenn 1990). The number of seabird islands prohibits regular censuses of all seabird populations, but it is important to establish the distributions and sizes of breeding populations before effective monitoring can proceed. Regular, frequent surveys of specific breeding populations should be conducted to establish their potential as long-term environmental monitoring stations for identified environmental parameters. In 1993, a national workshop was convened by the wildlife monitoring section of ANCA to establish procedures for assembling a national seabird database and to identify the most appropriate parameters, species and populations for such long-term monitoring. The workshop considered also the development of archives of frozen biological materials (such as eggs) for long-term contaminant studies.

A better understanding of the population dynamics of krill (*Nyctiphanes australis*) is important for effective management of the seabird resources of southern Australia, especially in relation to interannual variation and potential long-term changes in oceanographic conditions.

Carefully designed, continuing studies of seabirds at sea at specific locations may reveal long-term trends in oceanographic conditions. Appropriate areas for such studies should also be identified. The seabird counts conducted off Wollongong through most of the 1980s by interest groups illustrate the usefulness and potential of data gathered close to an oceanographically dynamic area such as the Tasman Front. Further data collection should be actively encouraged as they provide a continuous assessment of the relative abundance of seabirds at sea. Complementary work in northern and southern New South Wales would be especially valuable in assessing the effects of variation in the position and strength of the Tasman Front as determined from satellite imagery.

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Conservation of rare, threatened and endemic marine species in Australia

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Introduction

The primary goal of the conservation legislation of most countries is to prevent the extinction of rare and endangered species, whether this is by providing programs to rehabilitate species on the edge of extinction or by protecting the habitats or ecosystems on which they depend (Franklin 1993). With the exclusion of high profile marine organisms (mammals, reptiles, birds) most of the concerns and conservation practices undertaken have been based on terrestrial or freshwater species. The effects of habitat loss and fragmentation, the introduction of exotic organisms, exploitation and pollution are well documented (Soule 1991, Allan & Flecker 1993). There are long lists of recent extinctions and species threatened with extinction that can be directly related to these factors. The principles of establishing and maintaining viable populations of endangered species in terrestrial environments are being formulated and applied with some success (Soule & Wilcox 1980, Soule 1986, 1987, Simberloff 1988).

Recognition of the threats to marine biodiversity is recent (Ray & Grassle 1991, Thorne-Miller & Catena 1991). There is widespread public concern over the world-wide decline of coral reefs (Bunley-Williams & Williams 1990, Veron 1993), changes to temperate kelp bed communities (Estes, Duggins & Rathbun 1989), decline in seagrass beds (Thayer, Wolfe & Williams 1975) and loss of saltmarshes and mangroves (Hatcher, Johannes & Robertson 1989). These all appear to relate directly or indirectly to either coastal development and pollution (Veron 1993) or exploitation by humans (Aronson 1990, Hughes 1993), but the processes are complex and any predictions are made with great uncertainty. Although there are obvious examples of marine mammals and birds that have either become extinct (great auk, Steller's sea cow) or are considered endangered (eg blue, fin, humpback, sei and southern right whales), little is known of this problem for the vast majority of marine animals — including fish and invertebrates (Vermeij 1989). Although there has been only one recorded post-Pleistocene extinction of a marine invertebrate (the limpet *Lottia alveus* was last recorded in the eelgrass beds off the eastern coast of North America in the 1920s: Carlton et al. 1991), we have no idea whether there are many extinctions that have gone unnoticed, whether such extinctions are likely to become a major problem, and whether or not the current approaches to dealing with the conservation of fauna and flora will be appropriate for marine organisms.

In focusing attention on potentially threatened marine species, we address a number of important questions in this article. They are:

> What is the extent of species endangerment and extinction in the marine environment?
>
> To what extent can the basic concepts of conservation biology developed for terrestrial organisms be applied to marine species, particularly to those with a dispersive larval phase?
>
> What are the characteristics of species most likely to become threatened?
>
> What are currently viable conservation strategies for potentially threatened marine species?

The Australian perspective: no systematic approach

In Australia, most public attention and research into conservation has focused on the unique bird and mammal fauna (Common & Norton 1992). With 18 or so species becoming extinct over the last century and approximately 120 thought to be threatened (Kennedy 1990) there is considerable cause for alarm. The lack of recorded marine extinctions should not fuel complacency about our present and future impact on marine ecosystems however. There is evidence of overfishing and collection of a wide variety of marine organisms, including offshore pelagic fishes (eg southern bluefin tuna, *Thunnus maccoyii*: Caton, McLoughlin & Williams 1990), deepwater fishes (eg gemfish, *Rexea solandri* in New South Wales: Bureau of Rural Resources 1992), shallow water reef fishes (eg three species of coral trout, *Plectropomus*: Goeden 1979, Craik 1979, Rigney 1990) and giant clams, *Tridacna* (International Union for the Conservation of Nature 1983, Braley 1987). Also potentially important is the over-collecting of black corals (Antipathidae), cowries (*Cyraea* species), cone shells (*Conus* species) and tritons (Cymatiidae). Coastal development and industrial pollution (Cambridge & McComb 1984) and destructive fishing methods (Hutchings 1990) are having significant impacts on shallow marine habitats in some areas. The long-term prognosis for individual species — particularly those of great ecological and social significance — needs to be assessed.

There has been no systematic species-level approach to the conservation of Australian marine organisms. Apart from the *Whale Protection Act 1980,* which protects all whales, dolphins, seals, sea lions and dugong, individual species protection legislation does not generally consider marine organisms as animals, and only a handful of species have been given 'endangered' species status (Bates 1992). The threat of extinction of tridacnid clams was recognised by the International Union for the Conservation of Nature (IUCN) in 1983. All seven species of tridacnid clam, including the giant clam *Tridacna gigas,* are endangered. Local extinctions of giant clams have been reported throughout Micronesia (Crawford, Lucas & Munro 1987) and the species has been overfished on the Great Barrier Reef (Braley 1987). Three of the largest grouper species in the Great Barrier Reef complex (*Epinephelus tukula,* potato cod; *E. tauvina,* estuary grouper; *E. lanceolatus,* giant grouper) are so few in number that now they are protected completely during their adult phase. The Queensland National Parks and Wildlife Service also lists several other fish species as being endangered.

In a recent review of Australian endangered species, Kennedy (1990) listed only five marine fish, all of which have temperate and subtropical distributions. They are the great white shark (*Carcharodon carcharias*), the grey nurse shark (*Carcharias taurus*), the sand tiger shark (*Odontaspis ferox),* the black cod (*Epinephelus damelii*) and the southern bluefin tuna. Although Kennedy (1990) considered these species 'vulnerable' rather than 'endangered', he did not define the classifications. Fry and Robinson (1986) listed over 230 mollusc taxa that are potentially vulnerable and in need of monitoring to determine their status. Their list included 44 tropical cowries, 23 cone shells, six tritons and few representatives of numerous other tropical families. Fry and Robinson considered that only the Queensland cowrie, *Cypraea queenslandica,* is endangered.

As can be seen, the approach to marine species conservation in Australia so far has been inconsistent. The approach highlights deficiences in both the scientific theory and information being used to manage marine species.

Rare and threatened species: the marine perspective

It is important to draw a distinction between 'rare' and 'threatened' species (Edgar et al. 1991). The vast majority of marine organisms are, for whatever reason, naturally **rare** in that they have exceedingly low local abundance (*sensu* Rabinowitz, Cairns & Dillon 1986). However, they may have wide geographical distributions and overall numbers of individuals in the species may be high. For example, most of the 1500 or so fish species on the Great Barrier Reef have a broad Indo-Pacific distribution (Randall, Allen & Steene 1990), yet a large fraction of them would be considered rare on the basis of average density. Although naturally rare species may be more vulnerable to extinction than abundant ones (particularly if they have narrow geographic ranges or are highly specialised), not all rare species are (necessarily) threatened (Mace & Lande 1991). Recent studies are showing that — at least for terrestrial species — rarity can be associated with particular life history traits: for example, low reproductive effort, asexual reproduction and poor dispersal (Kunin & Gaston 1993).

In contrast to rarity, **endangered** or **threatened** species are those for which 'survival of the species' is unlikely if the causal factors (threats) continue operating (International Union for the Conservation of Nature 1988). Just as rare species are not necessarily at risk, species at risk are not necessarily rare (Mace & Lande 1991). In a first attempt to cope with this variability in status, the IUCN recognised a range of categories including Extinct, Endangered, Vulnerable, Rare and Indeterminate.

Unfortunately, the classification of species is highly subjective and no standards can be applied across taxa or habitats. This lack of standardisation is particularly a problem if the categories of threat are to be used to set priorities for research and management. While there are many clear examples of endangered terrestrial insects, amphibians, reptiles, birds and mammals, the same is not true for fauna occupying the marine environment. Nevertheless, whether that statement applies because endangered marine species do not exist or because we have not detected them, remains to be established. It is possible that at this stage, human impact on marine species has not been as devastating as it has been in the terrestrial environment — perhaps because we are not as efficient species exploiters or habitat destroyers when it comes to the sea.

The majority of marine organisms have life histories and population characteristics that make global extinction unlikely on ecological time scales. The following series of events illustrates this point. A dispersive larval stage in the majority of teleost fish and 70% of marine invertebrates is associated with generally broad geographic ranges and low historic extinction rates (Jablonski 1986). Larval dispersal links isolated subpopulations of adults and can replenish local areas where extinction occurs (Fairweather 1991). The magnitude of recruitment varies in relation to planktonic processes and can be to a large extent independent of a falling adult stock size or processes impacting on the adult population.

On the other hand, marine populations have characteristics that confound any attempt to detect dangerous levels of depletion. Fluctuation in recruitment and breeding population size can obscure any long-term trends and an imminent population crash may be impossible to forecast (the collapse of fisheries for Peruvian anchovy and south-east Australian gemfish are good examples). Patchy distributions can make reliable estimates of density or population size difficult to obtain, even for very common species: only quantum changes in numbers may be detected (Schroeter et al. 1993). Added to these, the methodologies for detecting and determining trends in the abundance of rare species are themselves scarce (but see Gerrodett 1987, Green & Young 1993) and are needed urgently.

Although the global extinction of marine organisms appears to have been rare in the last 200 years, there are other forms of extinction which can have a major impact on the structure of marine assemblages and the functioning of marine ecosystems. Estes, Duggins and Rathbun (1989) recognised the importance in certain species of detecting 'local' extinctions (the disappearance of a species from part of it range). To illustrate their point, they cited observations on the Californian sea otter, *Enhydra lutris.* Where the sea otter has become locally extinct, benthic communities are dominated by sea urchins and the abundance of kelp is reduced by grazing. This situation is reversed where populations of sea otters have re-established.

Such shifts in community structure can occur without local extinction. 'Ecological' extinction arises when a species is reduced to such low abundance that, although still present, it no longer plays the ecological role it used to (Estes, Duggins & Rathburn 1989). For example, in North America the overfishing of lobsters — once important predators in shallow kelp forest systems — has led to alternative benthic community structures even though lobsters are still present though in low numbers. Overfishing of triggerfish on Kenyan reefs also has not led to local extinction. However, because these fish had a natural predatory role, the reduction in their number had a major impact on coral reef communities (McClanahan & Muthiga 1988). It is our view that, in the marine environment, we must be more concerned about the local or ecological extinction of species playing 'keystone' ecological roles (*sensu* Mills, Soule & Doak 1993) than about the unlikely global extinction of the vast number of essentially 'redundant' species (*sensu* Walker 1992).

The theory for conserving terrestrial species is inappropriate

It is clear that much of the current theory developed in conservation biology cannot be uncritically applied to marine species and habitats. There are four areas in which we see the theory as deficient.

Minimum viable population size

The minimum viable effective population size for terrestrial animals is usually based on the '50–500' rule, ie an effective population size (Ne) of greater than 50 being the critical limit to avoid inbreeding problems, and an Ne of greater than 500 being the critical limit to avoiding loss of genetic variation through genetic drift. Both processes can lead rapidly to extinction in 'closed' populations. An Ne of 500 may equate with an actual population size of up to 2500 individuals depending on the species, because it depends on the number of mature, breeding individuals in the population (Shaffer 1981, Nunney & Campbell 1993). The 50–500 rule has a chequered history and is not universally accepted even for terrestrial organisms (Simberloff 1988). One thing for certain is that such genetic factors are not problems for small 'open' populations of marine organisms with high levels of gene flow among subpopulations (Slatkin 1981).

Threatened species categories

Despite the chequered history, minimum viable population estimates lie at the heart of the most recently published threatened species categories (Mace & Lande 1991). These are:

CRITICAL: effective population size (Ne) <50; (or actual population size (N) <250)
ENDANGERED: Ne <500; N <2500
VULNERABLE: Ne <2000; N <10 000

We know of no marine fish or invertebrate in this range. Nor are there any for which the data could be obtained, except perhaps for endemic species with exceedingly restricted geographic ranges. Simple thresholds in population size are unlikely to apply to the majority of marine organisms which characteristically exhibit extremely high juvenile mortality, fluctuating recruitment and a poor relationship between recruitment and adult numbers. Marine mammals appear to be classified arbitrarily as 'endangered' when numbers are reduced to four digit estimates (Kennedy 1990).

Population viability analysis and habitat fragmentation

The more general approach to the factors affecting the persistence of small populations is called Population Viability Analysis (Gilpin & Soule 1986). Preliminary models are available to assess population persistence under different extinction pressures which include demographic variation, stochastic environmental factors, metapopulation structure and increasing fragmentation (see Mace & Lande 1991). While this approach is promising in the long-term, there are few models that realistically can be applied to open marine metapopulations. Increasing habitat fragmentation in terrestrial environments is seen as a major threat to population persistence, but a high level of fragmentation is the natural condition for marine species and may not represent an extinction threat at all. Large numbers of subpopulations in combination with high levels of dispersal theoretically make the extinction of the metapopulation unlikely (Hanski 1989). On the other hand, if large-scale catastrophic processes lead to correlated population trends among subpopulations, then extinctions are more likely and may occur even if metapopulations are extremely large. No doubt increasing habitat fragmentation will occur as seagrass beds and coral reefs decline. However, until we have realistic measurements of the numbers of subpopulations, the degree to which trends in different subpopulations are correlated, the level of larval and adult exchange and knowledge of any source-sink relationships, it will not be feasible to base management strategies using Population Viability Analysis *per se*.

Reserve design and shape

Terrestrial ecologists have also been pre-occupied with the community and ecosystem responses to habitat fragmentation (see Saunders, Hobbs & Margules 1991). This pre-occupation has also led to discussions on reserve size, design and shape which culminated in the so-called SLOSS ('single large or several small') debate (see Simberloff 1988 for review). Reserved habitat patches are treated as 'islands' and island biogeography theory is applied to determine the optimal size and arrangement of reserves for maximising colonisation and minimising extinction. The basic assumption that reserves of a particular configuration have an equilibrium number of species has not been supported by long-term studies of community composition on natural reef isolates (eg Sale 1991). Rates for colonisation and extinction appear to be largely independent of variation in the number of resident species, patch size and shape, and proximity to the source because of the unpredictable nature of larval supply and adult mortality. While most marine reserves have been set up on an *ad hoc* basis, even very small, isolated reserves have had measurable effects on the abundance of highly mobile exploited species (Jones, Cole & Battershill 1993).

The pre-occupation of terrestrial conservation biologists with 'corridors' to connect habitat fragments or reserves (see review by Hobbs 1992) has no strictly marine analogue.

Potentially threatened marine species: their characteristics

At this stage in our knowledge, any attempt to grade marine species according to the degree of threat (critical, endangered and vulnerable) is likely to prove futile. An alternative yet still subjective approach is to recognise the characteristics of species that are at least 'potentially threatened' by extinction, and develop management strategies as a precautionary measure.

1. *Species with unusually restricted geographic ranges.* Examples include starfish *Marginaster littoralis* (known range 1 ha of the intertidal zone in the Derwent River estuary, Tasmania), the triplefin blenny *Forsterygion gymnotum* (confined to the Derwent River estuary), the Maugean skate *Raja* sp.L of Last and Stevens (1994) (confined to the Bathurst Harbour estuary, Tasmania) and four hermatypic coral species endemic to the Houtman Abrolhos, Western Australia (Edgar et al. 1991, Veron 1993). These sorts of species may suffer global extinction from relatively small-scale impacts.

2. *Species with unusually restricted breeding sites.* Many highly mobile marine organisms converge on specific breeding grounds that represent only a small part of their geographic range. Southern bluefin tuna is one example.

3. *Species which are very large, long-lived and/or have low fecundity.* Typically such species are also naturally rare and aggregated, slow to mature, and have consistently low recruitment. These characteristics make them easy to overexploit and slow to recover from overexploitation. Australian examples include giant clams, the great white shark and black cod. Large, live-bearing fish such as the elasmobranchs (sharks and rays) are particularly susceptible (Manire & Gruber 1990) as are most marine reptiles and mammals.

4. *Species which are subject to large-scale mass mortality.* A variety of marine organisms appear to exhibit catastrophic declines in abundance 'overnight'. For example, the mass mortality of sea urchins in the Caribbean Sea (Lesson 1988), mass mortality of seagrass beds in North America due to wasting disease (Den Hartog 1987) and mass mortality of marine mammals due to parasites or toxins (Harwood & Hall 1990). Such species may be threatened, although never classified as rare.

5. *Species which are subject to prolonged periods of recruitment failure.* While most marine organisms exhibit variable recruitment and 'year-class phenomena', there are those for which the good years are rarely recorded. One example is the potato cod.

6. *Species which are highly susceptible to environmental stress.* These may be the first species to become extinct locally, and may be the first to succumb to global threats such as ocean warming. They may represent 'early warning' indicators of impacts which may progress to other components of the community (Jones & Kaly in press). An example of this kind of species is reef-building corals which are susceptible to changes in water temperature or sedimentation (Veron 1993).

7. *Species which are extreme habitat specialists,* particularly those associated with one species of habitat forming organism; eg fish and invertebrates associated with a single coral or anemone species.

8. *Obligate supra-tidal, intertidal, estuarine and coastal embayment species,* particularly those restricted to the most populated latitudes. Most of their habitat may be anthropogenically degraded by coastal enrichment or pollution, particularly in the most populated areas.

9. *Species which are or have been subject to over-exploitation.* A number of commercially exploited fish species in Australia are now being rigidly controlled due to catastrophic declines in their abundance or catch per unit of effort. For example, the catch of eastern gemfish declined from over 4000 tonnes in 1987 to 400 tonnes in 1991 and the recommended yield is currently at nil catch.

Many species exhibit combinations of these characteristics and those species may be most at risk. We stress however, that species exhibiting these characteristics are not necessarily endangered. For the majority of species we simply do not have information on such matters as distribution, recruitment and population variability, susceptibility to stress and degree of overfishing sufficient to make judgements about the likelihood of local, ecological or global extinction. Arbitrary thresholds could be used to create a list of 'potentially vulnerable' species (eg geographic range less than 'x' km, organisms growing to 'x' m, or living to more than 'x' years). However, such arbitrary thresholds should not be used to establish a hierarchy of threatened species categories nor establish research priorities.

Conservation strategies for rare, threatened and endemic marine species

One of the oldest debates in ecology — whether research effort should favour the species by species approach or focus on communities/ ecosystems — is being revisited by conservation biologists and managers (Franklin 1993). It is perhaps a lesson that the debate was never resolved (we still have population and ecosystem ecologists!), yet it is the diversity of approaches that will in the long term maximise our chance of protecting species within natural systems. The sheer number of marine species (many undescribed), our inability to recognise the endangered among them and the potentially high level of ecological redundancy, favour the 'ecosystem' perspective (Walker 1992). The 'species' approach has never served small, low profile but ecologically important trophic groups such as detritivores and decomposers.

However, Noss (1990) identified five types of species deserving special conservation status because of their relevance to the ecological communities of which they are members. They are:

1. **ecological indicators** — species which may provide an early warning of detrimental impacts on the community;
2. **keystone species** — pivotal species upon which the diversity of a large component of the community depends;
3. **umbrella species** — those with large area requirements which, when given sufficient protected area, will bring many other species under protection;
4. **flagship species** — popular species that serve as rallying points for major conservation initiatives; and lastly
5. **vulnerable species** — those which are actually prone to extinction. For marine environments, it is the latter which are most difficult to identify.

The best way to ensure some degree of protection of widely distributed but rare marine organisms is to establish a network of marine reserves encompassing a proportion of all marine habitats and biogeographical regions (Pressley et al. 1993). There is no doubt that marine reserves have had considerable beneficial effects on excessively exploited species in tropical and temperate regions (Roberts & Polunin 1991) and can shift communities back toward a more 'natural' community structure (Jones, Cole & Battershill 1993). However, their effectiveness as a management tool for rare or endangered marine species has yet to be evaluated and may be a problem for highly mobile organisms. Protecting communities and ecosystems does not guarantee species survival, and a 'safety net' of species-orientated approaches will also be necessary (Walker 1992).

As an additional precautionary measure, marine protected areas should also be chosen to contain populations or breeding areas of 'potentially vulnerable' species, particularly those endemics with small geographic ranges, localised breeding sites and unusual aggregations of long-lived, large species. Rare habitats supporting endemic species will be high priority. An example is the Shark Bay region in Western Australia. This 2.1 million ha area was nominated for world heritage listing in 1990, and includes the famous stromatolites at Harmelin Pool, the extensive Wooramel seagrass banks and the habitats of several rare species (Rigney 1990). Another area, the Solitary Islands Marine Reserve in northern New South Wales, was gazetted in 1990. This area is a unique mixture of coral and kelp-dominated reefs and contains unusual aggregations of subtropical endemic species with restricted ranges. The Solitary Islands Marine Reserve is similar only to the Houtman Abrolhos in the west, where another suite of subtropical endemics are found (Veron 1993).

Other conservation tools that could be applied to protect particular species include complete closure to collecting or fishing and strict pollution or development controls in areas where sensitive marine organisms are concentrated (eg coral reefs). The possibility of enhancing excessively exploited marine species also needs to be explored. The technology for captive breeding and outplanting of juveniles has been developed for some invertebrates (eg giant clams: Crawford, Lucas & Munro 1987). For those species which cannot be reared through the larval stage, the opportunities for recruitment enhancement should be evaluated. Many reef fish species, for example, have very specific habitat requirements at settlement. Once these are known, artificial substrata may be developed and deployed on a scale that will enhance recruitment. The 'last resort' approach for terrestrial species is to transfer remaining individuals to island refuges containing suitable habitat that is free of exotic competitors or predators. This practice may not work for marine species with dispersive larvae, although the evidence is there: introduced species have become established in some areas (Blatz 1991).

Where the probability of extinction has been increased through the gradual loss or degradation of particular habitat types, restoration ecology may provide the answer (Kusler & Kentula 1990). Our increasing understanding of ecological relationships within particular habitats, such as seagrass beds, is becoming sufficient to restore or rehabilitate areas following their degradation by man or other disturbances. Where conservation fails, habitat restoration may become the major tool available to prevent the ultimate demise of marine species.

Conclusions

At present it is difficult to apply the endangered species concept to marine organisms other than mammals and perhaps some reptiles. While actual extinction appear to be infrequent in the marine environment, local and ecological extinctions of key species are more common and may have a major impact on marine habitats and ecosystems. The vast majority of marine organisms are rare, undescribed, and perhaps unimportant, but not necessarily endangered. Without an enormous taxonomic and monitoring effort we will not know the full complement of species or the extent of any extinction problem. It is clear that the research emphasis and management guidelines developed for terrestrial organisms will not be appropriate for marine species with 'open' populations, particularly those guidelines based on a minimum viable population size. Although the biological characteristics of 'potentially vulnerable' species are recognised, we cannot categorise marine species in terms of the likelihood of extinction.

However, there are a number of population-orientated management strategies that should be implemented as a precautionary measure, including establishing protected areas for endemic species with small geographic ranges or restricted breeding sites, protecting long-lived, large and wide-ranging marine species, and implementing population enhancement strategies for excessively exploited species. These areas should be in addition to networks of marine reserves established as a 'catch-all' strategy for protecting the majority of species of unknown importance. Biological theory for conserving marine species will not progress without further information on the interactions between key species and other members of marine communities, developing methods for detecting and monitoring rare species and gaining a greater understanding of metapopulation structure and larval dispersal. It is only by a combination of immediate precautionary measures and vital research that we meet our responsibilities, not only to protect our endemic marine species, but also those widely distributed species that are under enormous pressure in other parts of their range (Veron 1993).

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Acronyms

AAT	Australian Antarctic Territory
ABARE	Australian Bureau of Agricultural Resource Economics
AC	Antarctic Convergence
ACPC	Australian Circumpolar Current
AGSO	Australian Geological Survey Organisation
AIMS	Australian Institute of Marine Science
ANCA	Australian Nature Conservation Agency
ANPWS	Australian National Parks and Wildlife Service (now ANCA)
ASEAN	Australian and South-East Asian Nations
BM	Bureau of Meteorology
CALM	Conservation and Land Management
CCAMLR	Commission for the Conservation of Antarctic Marine Living Resources
CITES	Convention on International Trade in Endangered Species
CSIRO	Commonwealth and Industrial Research Organisation
DNA	Deoxyribose Nucleic Acid
DOC	Dissolved Organic Carbon
EAC	East Australian Current
EEC	Exclusive Economic Zone
EIO	East Indian Ocean
FRV	Fisheries Research Vessel
GBR	Great Barrier Reef
GBRMPA	Great Barrier Reef Marine Park Authority
GIS	Geographic Information Service
IUCN	International Union for the Conservation of Nature
IWC	International Whaling Commission
I-WP	Indo-West Pacific
LC	Leeuwin Current
mya	million years ago (sometimes as 'mybp' or 'million years before present')
NPWC	National Parks and Wildlife Conservation
Ne	Effective Population Size
nm	nanometres
PCBs	Polychlorinated Biphenyls
PSP	Paralytic Shelfish Poisoning
PSU	Practical Salinity Units
RAOU	Royal Australasian Ornithologists Union
RV	Research Vessel
SAC	Subantarctic Convergence
SCUBA	Self-Contained Underwater breathing Apparatus
SIR	Seabird Islands Report
SPREP	South pacific Regional Environment Program
STC	Subtropical Convergence
TBT	Tributyltin