OVERVIEW OF THE CONSERVATION OF AUSTRALIAN MARINE INVERTEBRATES

A REPORT FOR ENVIRONMENT AUSTRALIA

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BY

WINSTON PONDER

PAT HUTCHINGS

REBECCA CHAPMAN[1]

AUSTRALIAN MUSEUM

6 College Street, Sydney, NSW 2010 Australia

[1] Now Port Stephens Fisheries Centre, NSW Fisheries, Private Mail Bag 1, Nelson Bay, NSW 2315.

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EXECUTIVE SUMMARY

INTRODUCTION

This volume is the marine counterpart of Yen and Butcher's (1997) overview of the conservation of non-marine invertebrates. These animals represent the great bulk of marine biodiversity and the consequences of not properly managing and conserving them will be profound. Conservation as a whole suffers from an imbalance in favour of vertebrates, but marine conservation, in general (with the exception of some marine mammals, birds and reptiles) is typically (and given where humans live, not surprisingly) seen as less of a priority than its terrestrial counterpart. One of several reasons for the general lack of interest and action regarding marine invertebates is the lack of accessible information about them and issues relating to their conservation. In developing this report we have been mindful of these matters and have attempted to provide not only an overview of the issues but a resource that will be a useful starting point for the study and conservation of marine invertebrates in this region.

This overview encompasses all aspects of marine invertebrate conservation in Australia and its territories, including the current state of knowledge, conservation issues and approaches, details of threatening processes, recommendations for conservation strategies, information resources, gaps and shortfalls and recommendations, including research needs, necessary to overcome these shortcomings. Appendices detailing collection resources and marine societies are also provided.

Scope, limitations and time frame of the overview

This report was completed in draft format in July 2000, after almost two years of compiling data and information. Individual draft chapters were reviewed by many relevant experts (see Acknowledgements). The availability of the draft report was then widely advertised in the scientific literature and Environment Australia distributed copies upon request. A list of people and agencies that reviewed the draft report is given in the acknowledgements. In the meantime, largely due to administrative reorganisation in Environment Australia, the final production of the report lapsed for more than a year and a half. A contract for the incorporation of the comments and the updating of literature was finally let in mid May 2002 with a completion date of the 30th June, 2002. Given this short time frame, we have been selective in the new literature and information incorporated into the final report and, with limited time for literature searching, have undoubtedly overlooked some important contributions. Appendix 1 has not been updated since July 2000 - it contains details on museum collections and the degree to which these collections have been identified and databased, as well as details on the number of taxonomic specialists. While we have tried to ensure that the other information in the report is as up to date as possible, time constraints have not allowed these updates to be as comprehensive as we would have wished. Some matters with respect to the latest legislation, regulations and policy papers may have been overlooked or will have changed since the completion of this report and users should consult the relevant web pages of the various agencies and government departments for the latest information.

Winston Ponder and Pat Hutchings, 23 June 2002

SUMMARY OF MAIN FINDINGS

1. Australian marine environment

1.1. Australia's marine jurisdiction, including the Exclusive Economic Zone (EEZ), is about twice the size of the Australian mainland and ranges from the sub-Antarctic to the tropics.
   1.1.1. It includes a wide range of habitats including estuaries, coastal lagoons, supralittoral, intertidal and subtidal habitats, the continental shelf and slope and abyssal depths.
   1.1.2. The diversity of substrates is huge, including soft sediments, rocky reefs, coral reefs and vegetated substrates (including seagrasses, saltmarshes, mangroves and algal beds).
1.2. Organisms are found throughout the water column from the sea surface to the seafloor and into the substrate.
1.3. There is a high degree of connectivity in marine ecosystems, including the integration of inshore shelf waters and offshore waters, and the land–sea interface. Thus, artificial administrative/ political boundaries are not a good a basis for management.

2. The invertebrate fauna

2.1. Invertebrates comprise all members of the animal kingdom except vertebrates. They are not a natural grouping but consist of many major groups of vastly different organisms.
2.2. The seas contain all but one of the known animal phyla and invertebrates comprise the great majority of marine biodiversity.
2.3. Invertebrates range in size from microscopic to several metres in length and some colonial organisms (e.g., corals, sponges) are particularly conspicuous.
2.4. All exploited marine taxa depend on invertebrates either directly or indirectly and marine ecosystems would collapse without their services.
2.5. Invertebrates, especially corals, are a major source of tourist income, and many others (e.g., prawns, abalone, scallops, oysters, lobsters, squid) are commercially important, as are products from some (e.g., pearls).
2.6. Of the known fauna, a large percentage of the invertebrates found in Australian waters (including the EEZ) are endemic to the region.

3. State of knowledge

3.1. The state of taxonomic, biological and ecological knowledge regarding marine invertebrates is generally poor. It is most comprehensive in shallow coastal waters in SE Australia and least known in deeper waters. Knowledge varies with location, habitat and taxonomic group.
   3.1.1. There are large gaps in our understanding of even the relatively well-studied macrofaunal groups while many taxa are very poorly known to virtually completely unstudied.
   3.1.2. Many more marine invertebrate taxa remain undescribed than have names.
   3.1.3. Reasons why our marine invertebrate fauna is so poorly known include:
      3.1.3.1. Many studies of marine organisms typically focus on fishes with, at best, only the largest of the invertebrates being considered.
      3.1.3.2. There are very few experts on marine invertebrates in Australia, despite the diversity of the fauna.
      3.1.3.3. Little funding is available for research. The lack of biological information frequently necessitates the use of exemplars (often from the northern hemisphere) when attempting to extrapolate biological features or predict ecological outcomes.
3.2. The available knowledge is not readily accessible, the few guidebooks dealing with only a small fraction of the common species and most of the literature is in relatively obscure scientific publications. For most groups there is not even and up to date, authoritative list of species available.

3.3. The intertidal and shallow water faunas are best known, while the deep-sea fauna is virtually unknown. Most parts of the Australian marine environment are poorly sampled or unsampled for invertebrates, especially the deep-sea, offshore islands, seamounts and banks.
    3.3.1. In general, the faunas in tropical ecosystems are more poorly known than temperate ones. Coral reefs are relatively well studied compared with most other ecosystems, but this is only true with regards to corals and fish, not for other invertebrates or inter-reefal areas.
    3.3.2. The microscopic fauna in all habitats is very poorly studied, especially the interstitial fauna (meiofauna).
3.4. The majority of data relating to marine invertebrates resides in museum collections.
3.5. There is great variation in the data (and thus our knowledge) available between groups of organisms, regions and habitats.
3.6. There is a need to synthesise existing data and collate biological data with physical/oceanographic data.

4. Impediments

4.1. There is a serious lack of resources in the provision of taxonomic studies and services
    4.1.1. Funding for taxonomic studies has declined, as has the number of taxonomists working on marine invertebrates (in museums and universities) and many currently employed are approaching retirement.
    4.1.2. University courses have reduced appropriate courses at undergraduate level.
    4.1.3. Consequently, there are usually significant difficulties with identification.
    4.1.4. Few keys and guides are available to identify Australia's marine invertebrates and those available are restricted to only a few groups.
    4.1.5. There are no checklists for many groups.
    4.1.6. There are very few specialists in Australia and several significant groups have no specialists
4.2. There is a serious lack of information about virtually all marine ecosystems and communities, including their composition, natural variability, biological processes within them etc.
4.3. There is little or no information on the ecology and basic biology of most marine invertebrates, even for many abundant, ecologically or commercially important taxa.
4.4. The lack of infrastructure is impeding research effort.
    4.4.1. A major problem (for marine science in general) is the very small number of research vessels available to Australian scientists. The high cost and high demand on the very limited facilities available makes it almost impossible for most "basic" offshore and deep-sea research work to be undertaken.
        4.4.1.1. There is also little funding available to utilise the existing research vessels and these are difficult to access by non-CSIRO scientists.
    4.4.2. Much ecological research is focussed around urban centres and marine stations on the GBR. Islands and external territories (with the possible exception of Antarctica) are, overall, not well studied.
        4.4.2.1. This is in large part due to a lack of accessible marine stations in most bioregions, a lack of which also hinders research and teaching.
        4.4.2.2. Other stations located around Australia include fisheries research stations and field stations for particular universities and access to these by outside workers and students can be difficult. With the exception of the fisheries research stations, all, including those on the Great Barrier Reef (GBR), lack access to guaranteed long-term funding to ensure that facilities are maintained and upgraded over time.

4.4.2.3. Research in areas lacking a marine station markedly increases costs and precludes many research activities.

4.4.3. There is a serious lack of access to existing information. Knowledge sharing and access to information are key issues that need to be addressed as quickly as possible by the facilitation of programs that will increase public access through the production of printed and web-based information.

4.4.3.1. While there is a considerable amount of information about Australia's marine invertebrate fauna, much of this is only accessible to a few experts.

4.4.3.2. Some programs (such as The Global Biodiversity Information Facility - GBIF) are global in scope but Australian input will be vital (the Australian component of GBIF is ABRS's ABIF, which currently receives very little funding).

4.4.3.3. Museums are struggling to maintain reasonable curatorial standards and do not have the resources to database their collections (the repository of most of the basic information on marine invertebrates) so that information can be electronically available via the WWW for:

- Use by decision makers and the community at large.
- The identification of sampling gaps so that surveys could be more effectively planned.
- The provision of an historical record.

4.4.3.4. University, museum and other relevant institutional libraries, are continually forced to make cuts in journal and book acquisitions so it is often difficult to obtain specialist literature.

5. Consequences of our lack of knowledge

5.1. While a large number of threats (ranging from local impacts to global; e.g. global warming) have been recognised as impacting on the marine invertebrate fauna, in reality it is difficult to assess the magnitude of the problem because:

5.1.1. Changes to the fauna in most locations have not been adequately documented.

5.1.2. The dynamics of natural variation are not well understood.

5.2. Loss of components of the invertebrate fauna may lead to losses of processes and functions with the eventual possible collapse of the ecosystems.

5.2.1. Flow on effects could occur throughout the marine system with impacts on commercial stocks, loss of tourism and recreational uses.

5.3. Managing marine ecosystems will be largely guesswork without better knowledge. The objective of an adequate conservation policy for marine biodiversity cannot be realised without a much better knowledge of the components of that diversity, the habitats it occupies and what its biological requirements are.

5.3.1. The stated aim of marine protected areas is to maintain biodiversity but decisions are made regarding the placement and management of these areas in ignorance of the composition and biological requirements of the majority of the fauna.

5.4. Many potential resources are currently under-utilised as a result of our ignorance of the fauna.

6. Threats and conservation

6.1. Threats to the marine environment are at very different scales ranging from local disturbance to the worldwide impacts of global warming. Thus, strategies for dealing with identified threats must often be multifaceted and range from local to global in scale.

6.1.1. Many threatening processes (and solutions to them) are still very poorly understood.

6.1.2. Synergistic effects are probably common.

6.1.3. Threats can be indirect and complex.

6.1.4. Solutions would often require substantial changes to current practices (e.g., reduction of sediment load in nearshore environments would require changes in farming practices; drastic alteration of communities on the continental shelf would require substantial changes in some commercial fishing practices).

6.1.5. Threatening processes that affect marine invertebrates typically impact generally on all marine life forms (e.g., pollution, habitat modification through development), but some have more serious impacts on invertebrates (e.g. dredging, benthic trawling).

6.1.5.1. For bottom communities trawling is extremely damaging through its destruction of the epifaunal communities (sponges, corals, echinoderms, molluscs, crustaceans etc.) on the seafloor. Much of the Australian continental shelf is probably already heavily impacted by this activity and the communities markedly changed as a result, probably to the detriment of the sustainability of fish stocks.

6.1.5.2. Scallop dredging destroys epifauna and shallow infaunal communities and has been shown to be unsustainable.

6.1.6. Most obvious impacts related to pollution, developments etc. are in coastal areas.

6.2. Offshore oil and gas exploration and extraction are currently minor impacts compared with the fishing industry, and tends to be more rigorously controlled.

6.2.1. Offshore mining for sand or minerals can be very damaging at local scales.

6.2.2. The extraction of minerals from the deep-sea is potentially very damaging to a largely unknown habitat and fauna.

6.3. Conservation measures must consider the interconnectiveness of the coastal region with the land and freshwater systems.

6.3.1. The often vulnerable, narrow transitional habitats (particularly semi-terrestrial areas), and their faunas, tend to be ignored or forgotten by both researchers and management agencies, who are commonly divided according to a terrestrial / marine dichotomy.

6.4. There are considerable differences in the levels of conservation concern for, and the legislative recognition of, invertebrates between the Commonwealth, states and territories. This lack of a consistent or coordinated approach to marine biodiversity conservation in Australia is a serious hindrance to implementing national strategies.

6.4.1. As with terrestrial ecosystems, there are many different agencies that have jurisdiction over the marine environment, or parts of it leading to often ill-informed *ad hoc* decisions and the inability to develop comprehensive, long term, co-ordinated strategies.

6.4.1.1. While the complete removal of inter-departmental/ inter-agency/state-Commonwealth boundaries and rivalry is probably unrealistic, increased levels of cooperation would greatly increase efficiency.

6.4.1.2. Problems include conflicting approaches to, or uses of, marine resources, even within single agencies (e.g., exploitation vs conservation).

6.5. Due to the difficulty of dealing with the conservation of all threatened marine invertebrates on a species basis, there is a need to focus on protecting and managing identified threatened *systems* at a variety of scales, from assemblages and communities (including habitats) through to larger scales such as "ecosystems", bioregions, etc.

RECOMMENDATIONS

1. Policy

1.1. While the Australian Government has adopted the Oceans Policy, the Marine Science and Technology Plan and the Coastal Marine and Planning Program which all relate to the conservation of marine biodiversity (of which invertebrates are the substantive part), funding and resources for the implementation of these polices needs to be made available.

2. Conservation

2.1. The taxon approach for conserving marine invertebrates is generally neither a practical nor cost effective strategy for the great majority of taxa. However, it can be a useful approach in some circumstances, such as for:

- 2.1.1. Taxa harvested (including by collectors) or impacted indirectly by other exploitative activities (can be managed by specific controls on numbers taken or methods and/or effort employed);
- 2.1.2. Taxa that have narrow geographic ranges (once identified, specific measures can be implemented); and
- 2.1.3. Taxa that live in highly specialised environments threatened by specific, manageable, threatening processes (targeted reduction in, or cessation of, impact(s) may be possible).

2.2. Commonwealth, State and Territory agencies should attempt to coordinate threatened species listing and management through the *Environment Protection and Biodiversity Conservation Act 1999* (EPBC Act), with the goal of moving towards uniform threatened taxon legislation and a single national threatened taxon list.

- 2.2.1. The listing of threatened species is not practical for many marine invertebrates, where high heterogeneity and poor knowledge do not generally provide sufficient data to assess their status according to current IUCN or similar criteria.
- 2.2.2. A category of "insufficiently known" may need to be used to enable listing of those taxa suspected to be at risk, but which lack adequate quantitative data to assign them with certainty to another category.
- 2.2.3. Australian taxa listed by IUCN should be assessed and considered for listing by the relevant agencies.
- 2.2.4. Expert panels should be established to recommend appropriate conservation strategies for major taxonomic groups, especially those in which taxa have been identified as being vulnerable or at risk.

2.3. We recommend that, in general terms, encouraging the adoption of habitat-based conservation strategies, based on IMCRA's, are more effective than taxon-based approaches.

- 2.3.1. However, if the focus is restricted mainly to readily identifiable, high profile habitats, a large proportion of marine invertebrate diversity will be neglected.
- 2.3.2. Conservation of large areas encompassing a range of habitats is the most desirable strategy.
- 2.3.3. Such areas should be carefully placed to maximise their inclusiveness of taxon diversity and to cover geomorphological and environmental regimes.
- 2.3.4. A minimal requirement would be one or more such marine protected area(s) in each bioregion. Duplication is essential to ensure effective monitoring.
- 2.3.5. Conserved habitats may deteriorate over time given likely anthropogenic impacts so long-term monitoring is necessary.

2.4. Effective management of threats that affect marine invertebrates will require a coordinated approach from management agencies.

2.4.1. Because trawling is identified as a serious threat to the benthic epifauna of coastal and offshore (Continental Shelf and Slope, seamounts, banks etc.) areas we recommend that:
2.4.1.1. Gear be modified to decrease damage;
2.4.1.2. Restrict the areas where trawling is allowed;
2.4.1.3. Multiple no-go areas (e.g., as part of marine protected areas) that extend across the shelf and slope should be set up in each bioregion. Such areas need to be effectively policed with adequate deterrents in place.

2.4.2. More attention should to be given to preventing habitat destruction- and changes to catchments which then impact downstream in terms of changed hydrography and increased rates of run off.

2.4.3. The tourism industry should be even more aware that it too impacts marine habitats by its activities. These impacts need to be continually reviewed and the activities causing them revised.

2.4.4. Environmental impact statements concerning mining activities should give greater attention to impacts on the benthic and pelagic invertebrate communities likely to be affected.

2.4.5. The aquaculture industry needs to be effectively regulated to ensure that it does not impact adversely on natural habitats and the invertebrates they contain.

3. Research

3.1. Basic research is necessary for the gathering of adequate information for the formulation of informed conservation and management strategies.

3.1.1. This report highlights the large gaps in our knowledge base - even in the dominant invertebrate groups, and very large parts of Australian waters (including the EEZ) that remain unsampled. To obtain a better understanding of the marine invertebrate fauna the following are necessary:
3.1.1.1. Increased basic taxonomic research (will require increased funding, ideally through ABRS) on marine invertebrates, especially in those currently poorly known groups.
3.1.1.2. Determine biodiversity baselines by surveys and inventories.
3.1.1.3. Identify areas of high diversity and endemism (and hence of conservation significance) by accessing data in museum collections, by survey and by phylogenetic and genetic research.
3.1.1.4. Increased general and specialist inventory of marine invertebrates in Australian waters, especially in those areas (or for those groups) currently extremely poorly sampled – e.g., in deep-waters and the tropics (especially NW Australia), or the meiofauna, and other small-sized marine animals from most habitats.
3.1.1.5. Such surveys should include all bioregions and transitional zones.

3.1.2. Increased basic research (and therefore funding) on the basic biology (feeding, breeding, habitat preferences, behaviour etc.) of marine invertebrates, especially the ecologically important groups.
3.1.2.1. Greater encouragement of whole-animal studies on marine invertebrates in our universities.

3.1.3. Increased basic research (and therefore funding) on the ecology of marine invertebrates.
3.1.3.1. Encouragement and funding of ecological studies on marine invertebrates and marine ecosystems in universities.
3.1.3.2. Test the robustness of using surrogates (such as physical features – e.g., sediment, or other biota such as marine vegetation or other animals – e.g., corals) as the basis for predicting benthic invertebrate communities.

3.1.3.3. Examine the effects of anthropogenic changes in ecosystems (e.g., increased nutrients, reduced freshwater runoff, effects of global warming etc.), especially in bays, estuaries and shallow coastal waters.

3.2. Assessment and monitoring are activities that provide essential information for informed management.

3.2.1. They require well-formulated methodology with rigorous scientific standards.

3.2.2. In order that assessments can be expedited, baseline inventories should be conducted.

3.2.2.1. Faunal inventories using rigorous sampling protocols and covering a wide range of invertebrates should be conducted in representative habitats (especially on major bays and estuaries) in major biogeographic areas around Australia. These will serve as benchmarks for assessing change in faunal composition, assisting in the detection of introduced taxa and community change due to anthropogenic impacts.

3.2.3. Survey for introduced species should be against a background of comprehensive faunal inventory with the involvement of specialist collectors and taxonomists.

3.2.4. Initiate long term monitoring sites in locations around Australia, in at least one location representative of each of the major bioregions.

3.2.4.1. There is a need to establish that marine parks really work. Long term monitoring is needed with comparisons using control sites in the same general area.

3.2.4.2. Set up a national register of coastal wetlands so that changes can be monitored.

3.2.4.3. Community groups could play an important role in monitoring changes in benthic communities but, to be effective, such work must be undertaken with the involvement of appropriate scientists.

3.2.5. Given that there are relatively few marine ecologists or other marine biologists capable of undertaking detailed assessments, this expertise should be developed at regional levels as a matter of urgency.

3.3. Marine research infrastructure is generally inadequate.

3.3.1. The provision of additional facilities for all of facets of marine research should be a high priority for the Commonwealth Government, especially given the aims and objectives of the current Government marine policy objectives.

3.3.1.1. The lack of access to research vessels is a major impediment to offshore and deep-sea marine research in Australia.

4. To maximise the range of research outcomes, Commonwealth research vessels should be more accessible to scientists from universities and agencies than they are at present.
5. Outcomes from individual cruises should be maximised by ensuring that sampling and other activities are of benefit to as wide a range of scientists and research areas as possible.

5.1.1.1. There is a chronic shortage of marine research stations in most parts of Australia (with the exception of the GBR).

6. There should be research stations in representative areas around the Australian coast.
7. They should be collectively regarded as a national facility.
8. All such stations, including those established on the GBR, need to have guaranteed sources of funding – the current ad hoc system prevents long term planning and regular updating. The recent financial support for research stations on the GBR run by Universities will help in the short term, but funding should be given to all such facilities regardless of which Institution runs them.

9. Management

9.1. There is a need to resolve the differences in levels of conservation concern and recognition of invertebrates between the Commonwealth, states and territories to overcome the lack of a consistent or coordinated approach to marine biodiversity conservation in Australia.

9.1.1. Develop procedures to increase cooperation and dialogue between all the management authorities and agencies involved directly or indirectly in marine systems.

9.1.1.1. Develop cooperation between land and marine based agencies because terrestrial inputs are critical to the health of coastal marine invertebrates.

9.1.1.2. Agencies should be aware of, and attempt to reduce, the cumulative effects of impacts - the "tyranny of small decisions" - made by different authorities that have little or no communication.

9.1.2. Some agencies with a primary role of exploitation of resources are also charged with the conservation of the marine environment. This will lead to conflicts of interest so, where possible, these functions should be separated.

9.1.3. Transitional marine-non-marine habitats (particularly semi-terrestrial areas) and their faunas need to be incorporated in management strategies.

9.2. There is a need to review the effectiveness of the controlling regulations intended to abate threats from over-harvesting – e.g., bag limits; restricted seasons etc., and the effects that these moderated activities are having on associated taxa.

9.2.1. Regulations controlling harvesting/collecting should not unnecessarily inhibit activities by school groups, hobbyists, interested parents with children etc.

9.3. Develop consultation processes where all available data and expertise is used to maximise input.

9.3.1. Regulations should consider traditional uses and allow access to marine invertebrates by indigenous people.

9.4. Lack of, or minimal involvement in, invertebrate studies, or consideration of them, by state and Commonwealth agencies responsible for marine and fisheries research needs to be addressed.

9.4.1. Staff or consultants with appropriate expertise in marine invertebrates should be employed to advise on issues relating directly or indirectly to marine biodiversity.

9.5. It is vital that coastal planning instrumentalities consider the impacts of development proposals on all aspects of the marine environment (including marine invertebrate habitats and communities) and introduce regulations that minimise these.

9.5.1. While developments in enclosed bays and estuaries are likely to cause the most damage, all developments leading to the loss of supralittoral, intertidal and shallow subtidal habitats should undergo assessments that include:

10. Their direct impact on marine invertebrates and their habitats;
11. The consequences of these impacts at ecosystem and species levels in the general area.

12. Education and Community Involvement

12.1. There is an urgent need to develop public awareness programs regarding the need to conserve marine invertebrates, and the nature and consequences of threats to the marine environment. This could be achieved by:

12.1.1. An increase in content on invertebrates, and their importance for our well-being, in school curricula and university courses.

12.1.2. Development of tools and material that will assist in highlighting marine conservation issues in educational arenas.

12.1.3. Training of teachers and teacher trainees in issues relating to invertebrates and their conservation.

12.2. Attitude surveys should be conducted to assist in the development of campaigns to promote awareness of marine invertebrates and of the need for their conservation.

12.3. Conservation and public interest would by enhanced by abandoning the word "invertebrates" because it encompasses such a huge, totally artificial grouping of animals with vastly different morphology, biology and ecology.

12.3.1. Where possible, specific, readily identifiable taxon or ecosystem labels should be used.

12.4. Reverse reduction in, or sidelining of, basic invertebrate biology and diversity courses in universities.

12.4.1. There is a trend in universities utilise the expertise in museums leading to a healthy synergy between these institutions with students exposed to museum expertise and research. However, the development of reciprocal funding arrangements is essential to ensure the long-term viability of such arrangements.

12.5. Encourage the consideration of invertebrates in local conservation issues or planning.

12.5.1. Develop information and material that will empower local conservation or other interest groups or individuals to be advocates for invertebrate causes and issues.

12.6. Development of resources (see below) that will give the community access to relevant information.

12.7. Establish a field studies network in Australia, using as a starting point available facilities run by schools, education groups, national parks, and universities.

12.7.1. Establishment of a number of suitable teaching facilities in coastal locations (ideally in conjunction with marine research stations) so that they can participate in marine studies courses.

13. Improving access to information

13.1. Provision of information utilising the worldwide web should be given a very high priority.

13.1.1. Australian museums and university collections already contain a huge amount of biodiversity information that is under utilized and difficult to access by planners/managers etc.

- o Web-based museum databases that provide basic information need to be linked, initially at a national level and, eventually, to other similar international databases.
- o Provision of funding to facilitate databasing of collections in state museums.

13.1.2. Development of ABRS's ABIF facility is a means of providing single source web-based information on marine invertebrates.

- o Completion of ABIF to at least to checklist stage as a first step. These should be regularly updated and should serve as a means of ensuring that taxonomic consistency is achieved.
- o Incorporation of biological data, illustrations etc. to ABIF.
- o Link with point data from museum collections for dynamic distributional data.

13.2. Provision of information, ideally via the worldwide web, on the ecology and biology of invertebrates and the ecosystems that they live in.

13.2.1. The production of identification guides (including interactive keys (web based or on CD), handbooks etc.) for at least the major groups of shallow-water marine invertebrates should be encouraged and supported.

13.2.2. Provide a searchable, comprehensive literature database on the ecology, biology and taxonomy of marine invertebrates.

PART 1 – INTRODUCTION

CHAPTER 1 – OVERVIEW

1.1 Outline and scope of the document

1.1.1 Aims

This volume is intended to be a companion to Yen and Butcher's (1997) overview of the conservation of non-marine invertebrates. As with that work, we see one of our major roles as addressing the “perceptual and practical imbalance” in the current approach to conservation, and facilitating the conservation of marine invertebrates in Australia and its Exclusive Economic Zone (EEZ). Not only is there an imbalance in favour of vertebrates over “the other 99%”, but also an imbalance in favour of the terrestrial environment at the expense of the marine realm.

This overview of marine invertebrate conservation in Australia and its territories encompasses:

- The current state of knowledge;
- The conservation issues and approaches;
- Details of threatening processes;
- Recommendations for conservation strategies;
- Information resources, gaps and shortfalls;
- Recommendations, including research needs, necessary to overcome these shortcomings;
- An extensive bibliography;
- A list of the institutions, individuals and conservation and professional societies that provided information for this volume;
- Appendices detailing collection resources and a list of relevant societies.

1.1.2 Scope and definitions

The scope of this document is huge, both in terms of the geographic area involved and the diversity of organisms and habitats covered. Australia has one of the world’s longest national coastlines. It also has one of the largest marine jurisdictions, approximately twice the size of the Australian mainland, ranging from the sub-Antarctic to the tropics. However, these are measurements only in surface area – marine organisms live in all available habitats extending throughout the water column to the floor of deep ocean trenches. Australia also has a unique and extraordinarily diverse invertebrate fauna, much of which is still unknown or only poorly known to science but is considered to be mega-diverse. Consequently, this overview must be, of necessity, somewhat superficial in some areas.

We define "**marine**" as encompassing all oceanic and coastal environments, including estuaries, brackish or saltwater coastal lagoons, mangrove and saltmarsh habitats (see extended definition below).

We define "**invertebrates**" as all members of the kingdom Animalia other than the Subphylum Vertebrata of the Phylum Chordata. Thus, invertebrates are not a natural grouping phylogenetically but consist of many groups of vastly different organisms (see below). The use of this term has several unfortunate connotations (see also Chapter 9), including:

- The perception that invertebrates and vertebrates may be roughly equivalent groupings; in fact, the invertebrates actually comprise about 95-99% of all animals; and
- The use of the prefix "in" implies inferior status (Lunney and Ponder 1999).

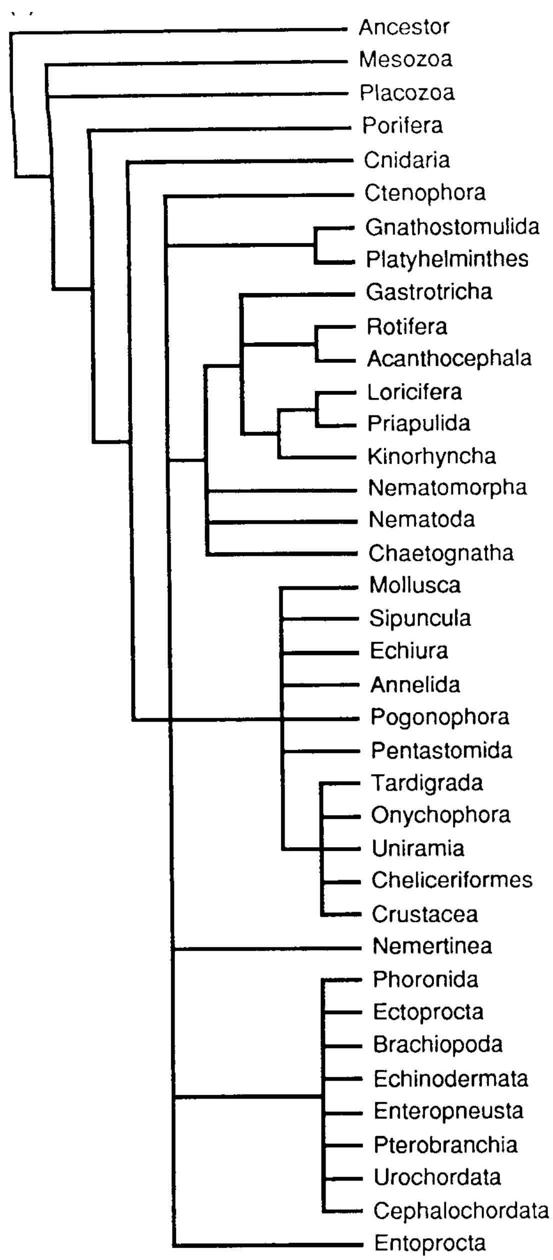
Despite the fact that they comprise most of animal diversity, and a significant proportion of "biodiversity" as a whole, conservation organisations have unfortunately given invertebrates little or no priority. This may be because "invertebrates" are:

- Widely viewed as insignificant, small-sized organisms of little or no interest;
- Seen as too poorly known or too difficult;
- Unlikely to attract public sympathy, money or votes; or
- Believed to be catered for if vertebrate and plant habitats are conserved.

Invertebrates

The term "invertebrates" is not a natural grouping, being a "grab bag" for over thirty metazoan animal phyla with very different features and evolutionary histories delineated by the absence of a backbone. In contrast, the vertebrates (mammals, birds, reptiles, amphibians and fish) are a phylogenetically related grouping – the Vertebrata. Marine invertebrates include animals as diverse as molluscs (e.g., snails and clams, seaslugs, squid and cuttlefish), crustaceans (e.g., crabs, barnacles, shrimp), many different types of worms comprising several phyla (e.g., bristle worms, flatworms, acorn worms, spoon worms), cnidarians (jellyfish, sea anemones and corals), and echinoderms (sea stars, sea cucumbers, sea urchins etc.). They include several groups that, because of their sessile habitat and colonial organisation, are not recognised by many people as even being animals or are even often thought to be plants (e.g., corals, bryozoans and sponges). Invertebrates range in size from less than a millimetre to several metres in length, the latter including enormous animals such as the giant squid and the nemertean worm *Lineus longissimus* that reaches 30 m in length. We do not regard protozoans and other unicellular groups as invertebrates, although the former group at least is sometimes included (e.g., Yen and Butcher 1997). It can be argued that the use of such a catchall term is detrimental to invertebrate conservation and this idea is developed further in Section 9.1.1. An outline of each of the major groups of invertebrates found in Australian waters is given in Section 2.2.

Figure 1.1: The relationships of invertebrate phyla are still unresolved in detail and are the subject of considerable current scientific debate. This strict consensus tree is from Eernisse et al. (1992).



Invertebrates are geologically ancient, with a history stretching back to the Pre-Cambrian. Knowledge of the taxonomy and phylogenetic relationships of the invertebrates is far from complete. One interpretation of the relationships between the different invertebrate groups is shown in Figure 1.1. However, it must be emphasised that invertebrate phylogeny is in a constant state of flux, with many conflicting ideas about the diagnosis of, and relationships between, groups. Molecular studies are increasingly being used to compliment analyses made based on morphological studies, developmental data etc.

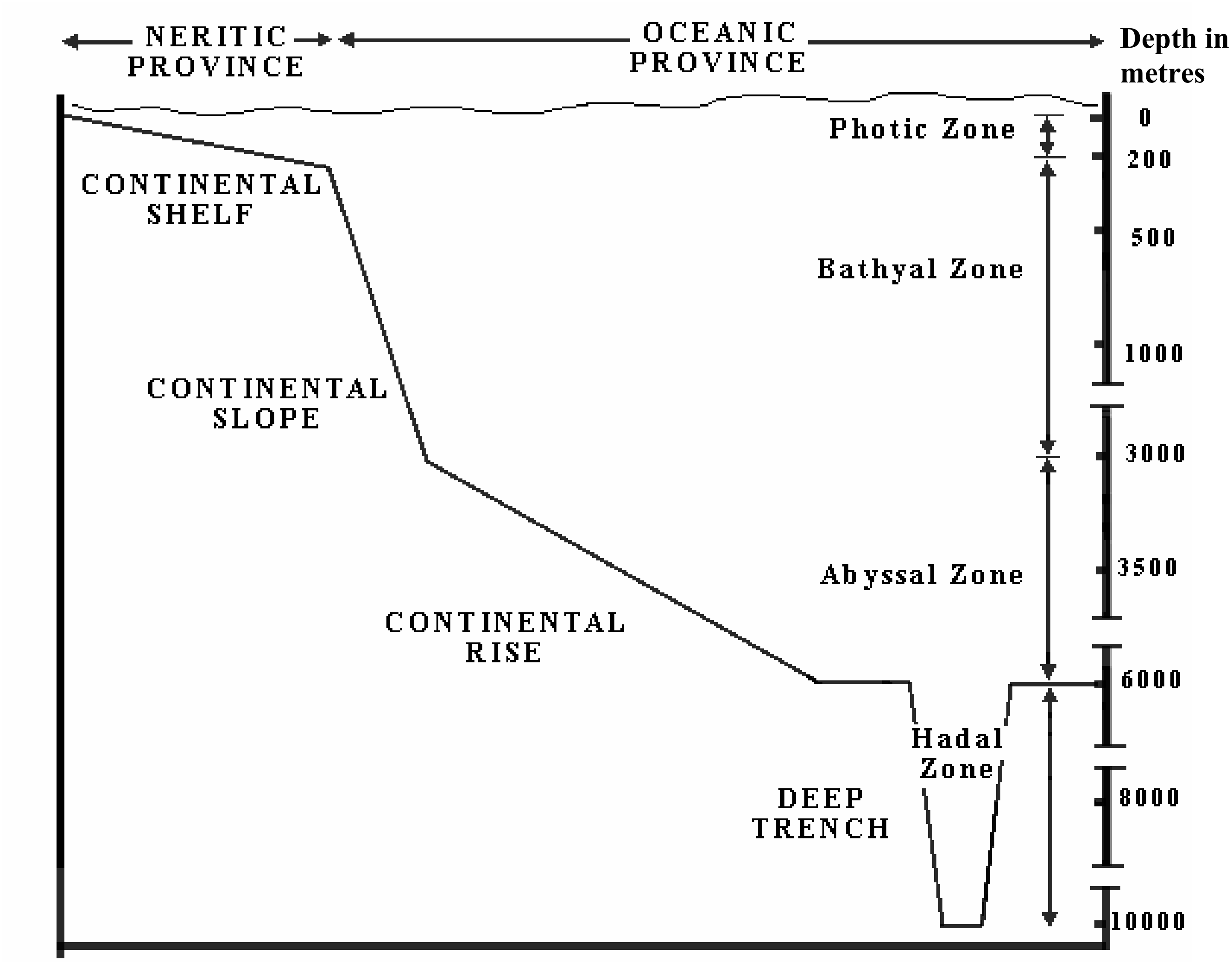
The marine environment

The sea comprises more than 99% of the biosphere permanently inhabited by plants and animals. It covers more than twice the area of the terrestrial realm in a permanently inhabited layer more than 100 times as thick (the average depth being nearly 4000 metres – Norse 1997, with a maximum depth of about 5000m in Australian waters and about 11,000m in the world's deepest trenches). While the general concept of the marine environment is fairly clear (the sea; salty water), the lack of a clear demarcation line between marine and terrestrial environments (on the shore), or between marine and freshwater environments (e.g., in estuaries and coastal lagoons) necessitates some definition of the habitats to be included in this overview.

We consider all habitats with a reasonable degree of marine influence, ranging from intertidal shores and estuaries to the deep (abyssal) sea, and including semi-terrestrial habitats such as the supralittoral zone, saltmarshes and the shoreward side of mangroves, as well as brackish environments found in estuaries, coastal lagoons and other water bodies with both freshwater and tidal influences. These transitional habitats (particularly semi-terrestrial areas), and their faunas, tend to be ignored or forgotten by both researchers and management agencies, who are commonly divided according to a terrestrial / marine dichotomy. Hence, it is seen as important that they be included in the present overview. It also should be stressed that the boundaries between terrestrial and marine environments or between marine and freshwater are not static, changing during a tidal cycle, seasonally and between years. For example, the terrestrial boundary of saltmarshes changes over time with fluctuations in sediment deposition. While some of the fluctuations are natural, others are induced by anthropogenic impacts such as changing freshwater flows of rivers, changing sediment supply or sea level rises due to global warming. A diagram of the different regions of the seabed and oceans, together with some of the terminology commonly used to describe them, is provided in Figure 1.2.

Marine invertebrates are found in all marine habitats ranging from shorelines and estuaries down to the barely explored habitats of the abyssal sea that reaches depths of up to 11,000 m in some of the deepest ocean trenches. In addition to well-known habitats such as rocky shores and rock pools, coral reefs and seagrass beds, marine invertebrates can be found in the interstitial spaces between grains of sand, around hydrothermal vents, on floating debris such as algal mats and driftwood, attached to other animals, on the peaks and slopes of sea mounts, and swimming or floating in the water itself.

Figure 1.2: Schematic diagram of the different regions of the seabed and oceans (based on Levinton 1995; Hammond and Synnot 1999).

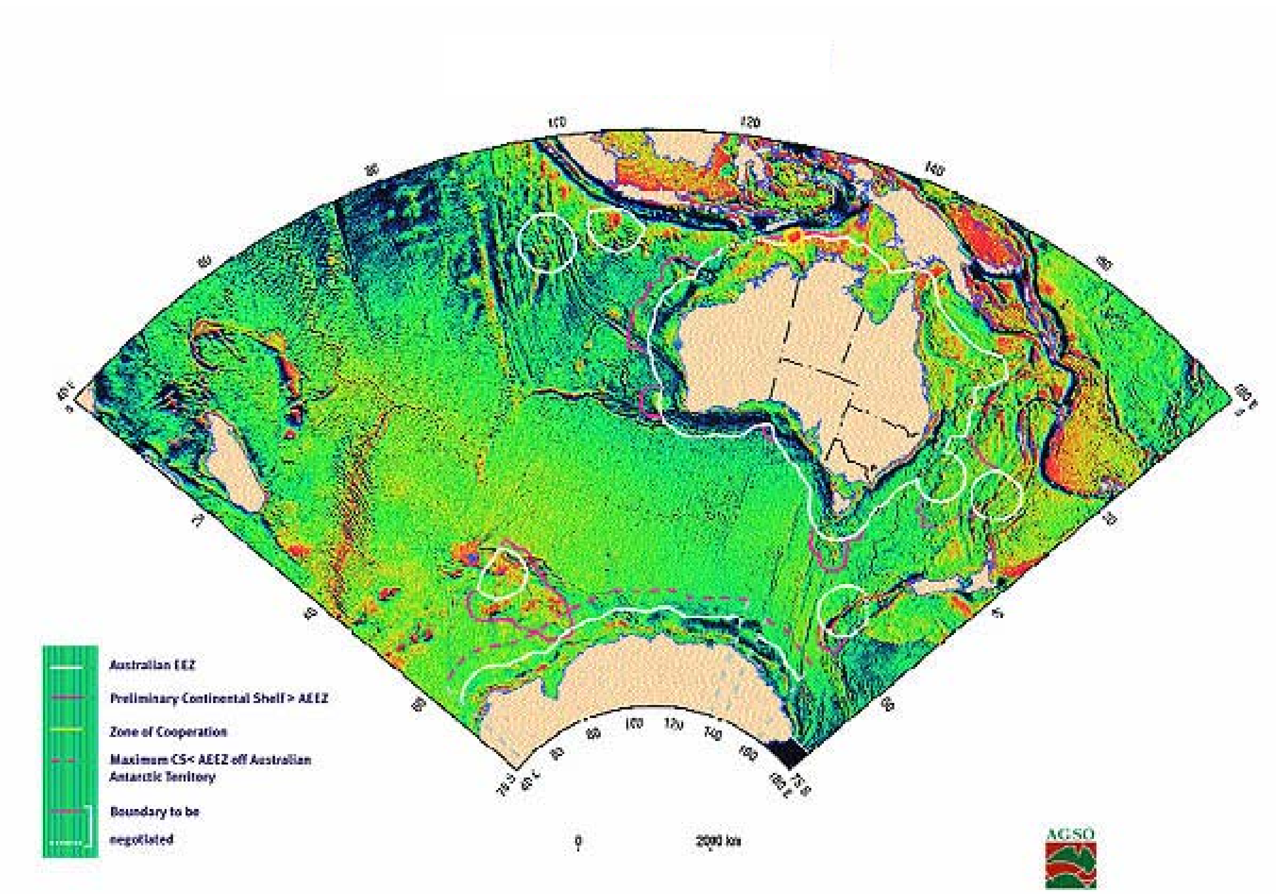


Australia and its territories

Australia's coastline is one of the world's longest national coastlines. There are about 30 000 km of mainland coast, including Tasmania, with an equal amount contained in surrounding islands and in the several hundred estuaries and coastal lagoons, lakes and bays, the total coastline measuring around 70 000 km (ABS 1992).

Australia has the right to explore and exploit resources within its Exclusive Economic Zone (EEZ), a zone that extends up to 200 nautical miles from the coast of Australia and its offshore islands and territories (except where reduced by proximity to an adjacent jurisdiction, e.g., New Zealand, Indonesia). Australia also has a claim over that part of the continental shelf extending beyond the EEZ outer boundary[1]. The total Australian Marine Jurisdiction (AMJ), which includes the EEZ, covers a total *surface area* of around 16 million km^2 – roughly twice the size of the Australian mainland. However, a large proportion of the extra territory within the AMJ surrounds the Australian Antarctic Territory rather than the Australian mainland. A preliminary map of the Australian Marine Jurisdiction is given in Figure 1.3.

Figure 1.3: Australia's marine jurisdictional zones (preliminary) (from Commonwealth of Australia 1998b)



[1] A case to confirm Australia's rights over the 5.1 million square kilometres of the shelf that extend beyond the EEZ as part of the AMJ must be lodged with the United Nations Commission on the Limits of the Continental Shelf (UNCLOS) by November 2004 (Marine Science and Technology Plan Working Group 1999). Under UNCLOS, Australia must show that it can look after this extra territory responsibly if it is to uphold its claim. Australia must prove that it is acting appropriately within the EEZ and that it has the scientific understanding to extend this stewardship throughout the AMJ (Luntz 1999).

Included within Australian territorial waters are:

- Australian Antarctic Territory
- Macquarie Island
- Heard Island and McDonald Islands
- Christmas Island
- Cocos (Keeling) Islands
- Lord Howe Island
- Norfolk Island (including Philip Island)

A brief description of each of these areas, with an introduction to their fauna, is provided in Section 2.3.

Australia has sovereign rights within its EEZ over its natural resources and jurisdiction over offshore installations, marine scientific research and the protection and preservation of the marine environment (Newton 1999). In addition to the EEZ, Australia has sovereign rights on its 'Legal Continental Shelf' over non-living resources (e.g., oil, gas, minerals) of the seabed and 'subsoil', as well as the '*sedentary living organisms*' (i.e., mainly benthic invertebrates). While the Legal Continental Shelf overlaps with the EEZ, it also extends considerably beyond it in some places, although its outer limit can never extend beyond 350 nm, or 100 nm beyond the 2500 m water depth contour (Newton 1999).

1.1.3 Approaches

As stated above, the major role of this review is to complement the overview of non-marine invertebrates by Yen and Butcher (1997). In line with Yen and Butcher, this overview is divided into three parts. Part 1 provides an introduction and some background information on marine invertebrate conservation and the marine invertebrate fauna; Part 2 covers approaches to marine invertebrate conservation ranging from taxa to systems and threatening processes, and Part 3 looks at information and implementation.

Part 1 begins by defining the aims and major concepts (this chapter), and then gives a summary of the state of knowledge of the Australian marine invertebrate fauna (Chapter 2) before discussing the issues involved in the conservation of these animals (Chapter 3).

The range of possible approaches to conservation is covered in Part 2 which begins with taxa (typically species) as the primary focus (Chapter 4), followed by systems at different scales ranging from species assemblages or communities to ecosystems (Chapter 5). Another approach, the identification and management of the threatening processes that impact on marine invertebrates, is covered in Chapter 6.

Part 3 looks at the existing situation and future requirements in terms of knowledge and the research base (Chapter 7), and legislation, policy, administration and implementation (Chapter 8). This Section concludes with a discussion of the essential issues of education, provision of better information, and promotion of the need to conserve marine invertebrates (Chapter 9).

Recommendations are presented in all but this chapter and are summarised in the Summary of Recommendations at the beginning of this document.

1.2 Marine invertebrates and biodiversity

1.2.1 What is biodiversity?

Biodiversity is defined by the Biodiversity Convention as "the variability among living organisms from all sources including *inter alia*, terrestrial, marine, and other aquatic ecosystems and the ecological complexes of which they are part; this includes diversity within species, between species and of ecosystems". Thus, biodiversity is not just species diversity, although it is often thought of as such. Species are but one level in the hierarchy and it is important to consider other levels such as genetic diversity and higher-taxon-level diversity (e.g., numbers of classes, phyla etc), as well as functional diversity, ecosystem diversity etc. The economic values of ecosystems, ecosystem function and biodiversity are also only now being realised and various attempts have been made to translate these into dollar figures (see overview by Daily et al. 2000).

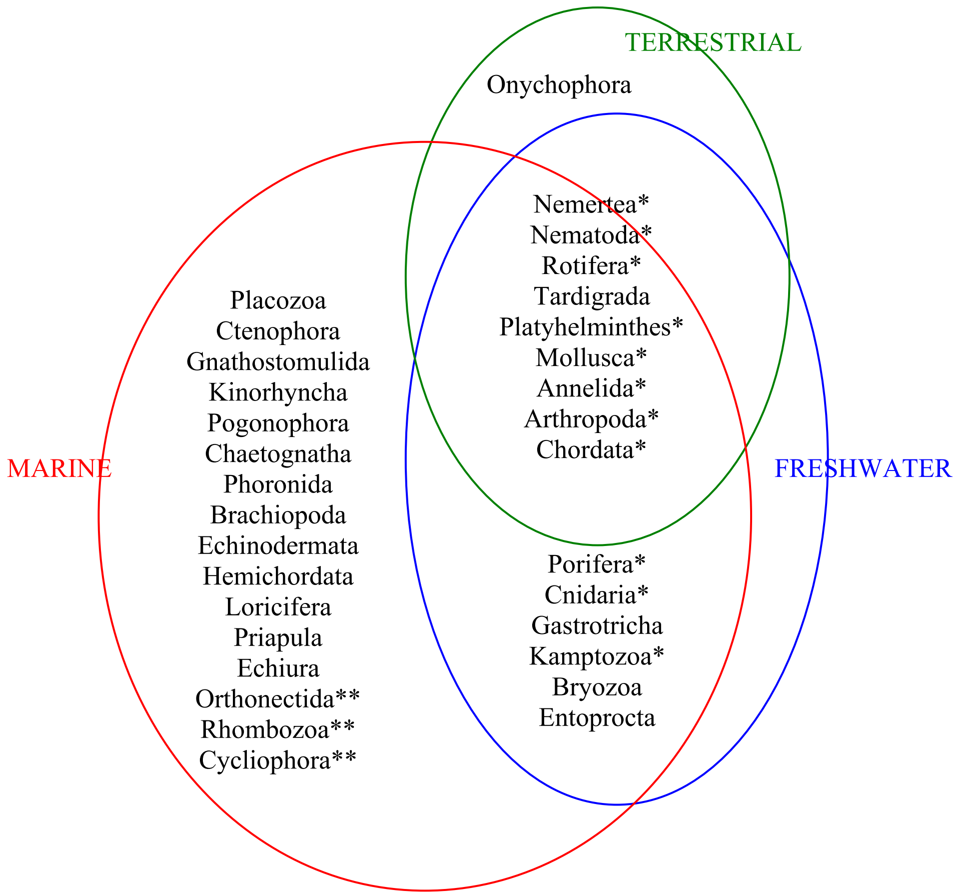
Useful summaries of biodiversity in relation to marine organisms are provided by Gray (1997a) and GESAMP (1997), who discussed genetic, species, phyletic, functional, community / ecosystem and habitat diversity. Patterns in marine biodiversity, and the processes responsible for these, are the subject of a book edited by Ormond et al. (1997).

As pointed out in the Interim Marine and Coastal Regionalisation for Australia (IMCRA Technical Group 1998), biodiversity can be understood, conserved and managed at a range of spatial and temporal scales. At the macro-scale (e.g., major oceanic and pelagic ecosystems), it is defined by large-scale oceanographic processes (e.g., currents, upwellings), trophodynamics, coastal physiography and basin topography. Biodiversity also occurs at the micro- or pica-scales of ecosystems (e.g., open coasts and gulfs), habitats (e.g., reefs, estuaries and bays) and biological communities (e.g., mangroves, seagrasses, kelp forests and coral reefs). At these scales, patterns in biodiversity may be dominated by small-scale physical processes such as the type of substratum, cyclones, storm events, tidal range and changes in wave exposure, or by biological processes such as competition and predation. It is at this scale, and smaller ones, that human impacts are particularly obvious. Smaller scales can be identified that affect distinct communities within larger ecosystems, impacts relevant to these scales perhaps operating within a few square metres.

1.2.2 How diverse are marine invertebrates?

Although fewer species appear to inhabit the ocean than the land, based on numbers of *currently known* species, consideration just of the numbers of species alone can be a misleading measure of biodiversity. At higher taxonomic levels (class, phylum etc.), marine ecosystems have a significantly higher degree of diversity (e.g., Ray 1985; Earle 1991; Ray and Grassle 1991; Williamson 1997) (see Figure 1.4).

Figure 1.4: Distribution of invertebrate phyla by habitat (based on Ray and Grassle 1991).
***Some members of these phyla are symbiotic. **These phyla are exclusively symbiotic.**



Examples of large-scale diversity are readily apparent in most marine habitats. For example, a single small boulder may be covered with representatives of at least 10 invertebrate phyla - sponges, corals (or other cnidarians such as hydroids or anemones), nematodes, brachiopods, bryozoans, arthropods, molluscs, polychaetes, sipunculans, crinoids etc. (Earle 1991).

In the sea, as on land, most animal life is small. Many biologists tend to ignore the small-sized taxa, reflecting another human bias. Humans are among the upper 5% of living organisms in size, yet most of the essential ecological processes are accomplished by microscopic organisms (Wilson and Peter 1988; Earle 1991; Ponder 1992; New 1995b).

Reaka-Kudla (1995) compared the biodiversity of coral reefs and rain forests and noted that there were 193 000 described macroscopic marine animal species. The ambitious *Census of Marine Life*[2] (Malakoff 2000) aims to document all marine life but at present

[2] http://www.coreocean.org/Dev2Go.web?anchor=coml_home_page

the number of marine invertebrate species cannot be accurately estimated because many groups, particularly those that are very small or obscure, have not been well studied and even the better known groups still have many undescribed species.

Tens of thousands of Australian marine invertebrates have already been described, many more thousands are known and await description, while very many more still await discovery. Even within those relatively well-known groups, there may be many sibling species awaiting discovery – those species difficult or impossible to distinguish based on morphological characters. Such taxa are common, a large number of abundant, well-studied or economically important marine taxa have recently been shown to be complexes of sibling species (Knowlton 1993). While the unravelling of sibling species complexes may involve using sophisticated techniques, for many groups of marine invertebrates all that is required is careful work by a competent taxonomist using morphological information. Hutchings (1998a), for example, indicated that many polychaete taxa previously considered to be very widespread are actually species complexes. In the more poorly known groups there are numerous readily distinguishable taxa awaiting description.

While it is generally accepted that there are many more species in terrestrial ecosystems, due to the large numbers of certain groups of insects, recent studies (e.g., Grassle 1991; Grassle and Maciolek 1992; Poore and Wilson 1993; Bouchet 2000; Bouchet et al. 2002) have shown previously unsuspected very high species-level diversity in marine ecosystems, although the jury is still well and truly out on even the right order of magnitude for these estimates (see Section 7.6.2).

1.2.3 How diverse are marine ecosystems?

The large number of fundamentally different life forms may confer high ecological complexity to marine and coastal ecosystems, this relating to how ecosystems are structured functionally (Ray 1991). Hypotheses about marine environments and ecosystem function (for a recent review see Loreau 2000) and comparisons with terrestrial systems are discussed by Ray and Grassle (1991). Norse (1997) argued that the marine ecosystem diversity exceeds that of the land and Knowlton (2001) and others have argued that coral reefs are the marine equivalent to rainforests in species diversity. The sea has equivalent habitats to forests, grasslands, montane and insular ecosystems, caves and hot springs, in addition to ecosystems without terrestrial analogues, such as the sea-air interface, the underside of pack ice and the water column. All these habitats are home to diverse assemblages of life-forms (including neuston, plankton, nekton) as well as functional guilds such as filter-feeding substrate dwellers entirely or largely absent on land. Marine ecosystems include those with the highest measured primary production (NE Pacific intertidal kelp beds Leigh et al. 1986), but they also include vast, sparsely inhabited spaces in which there is no autochthonous primary production, as well as ecosystems based wholly on chemosynthesis rather than photosynthesis.

1.2.4 Patterns of biodiversity in the marine environment

Besides alleged large-scale patterns of biodiversity (e.g., latitudinal and depth clines (but see below, this section)), biodiversity patterns relate to environmental gradients at smaller scales (e.g., ecotones – see Section 5.2.3). Most marine habitats, like many of those on land, are not discrete units but are the intersections of multiple environmental gradients (Ray 1991).

Biogeographic studies are based on patterns of taxon distribution. However, attempts to develop pictures of marine diversity are hampered by the small number of key studies, the different sampling methods they employ, the different measures of diversity utilised and the varying levels of taxonomic resolution (Clarke and Crame 1997). The notion of a latitudinal diversity cline (e.g., Pianka 1966), with increasing diversity towards the tropics, was previously well entrenched. There is some supporting evidence, particularly in the Northern Hemisphere (e.g., Thorson 1957; Clarke and Crame 1997; Gray 1997b), particularly for some taxa (bivalves, Stehli and Wells 1971; gastropods and mangroves, Huston 1994; e.g., sponges, corals, molluscs, echinoderms and decapod crustaceans, Poore 1995a; taxa with calcareous skeletons, such as gastropod and bivalve molluscs, Foraminifera (Protista) and hermatypic corals, Clarke and Crame 1997). However, the same trends are not necessarily so obvious in the Southern Hemisphere or in other taxa. For example, Price et al. (1999) found little evidence to support the notion of increasing diversity towards the tropics in Atlantic asteroids. Poore (1995a) noted that the diversity of marine amphipod crustaceans in the southern hemisphere was greatest in the lower latitudes, Hutchings (1998a) indicated that some of the highest polychaete diversities for soft sediments have been recorded from southern Australia and Hooper et al. (1999; in press) noted the complete absence, if not the inverse, of latitudinal biodiversity gradients in sponges. Thus, evidence for a cline in latitudinal diversity in non-calcareous marine taxa remains equivocal, and there is no evidence "for a cline in with-in habitat diversity for shallow-water soft-bottom infaunal communities" (Clarke and Crame 1997). Data from the deep-sea (e.g., Rex et al. 1993) appear to confirm the idea of increasing diversity with decreasing latitude although patterns are far from clear-cut (Gray 1997b). However, the Antarctic has high diversity for many taxa (Clarke 1992; Arntz et al. 1997), and diversities recorded from southern Australia (Poore and Wilson 1993; Coleman et al. 1997) are as high as the highest values for soft sediments anywhere. Thus, while it seems that there is a cline of increasing diversity from the Arctic to the tropics, the cline from the Antarctic to the tropics is far less well established if it occurs at all (Gray 1997b).

A shallow to deep-sea increase in diversity was originally suggested by Sanders (1968) who collected samples from different habitats ranging from the boreal to the tropics and from estuaries to the deep-sea slope. While much new data have been obtained since Sanders' study, this has mainly been from the deep-sea (e.g., Rex 1981; Grassle and Maciolek 1992). In the western North Atlantic, the most thoroughly sampled ocean region of the World, species diversity is low on the shelf, increases to a maximum at intermediate depths, and then decreases in the abyssal plain (Rex et al. 1997). However, analyses of diversity in other deep basins of the Atlantic suggest that this pattern may not be universal (Svavarsson et al. 1990; Blake and Grassle 1994). Because our knowledge of large-scale diversity patterns in the deep-sea is still based on a few major taxa and on very limited sampling and geographic coverage (Rex et al. 1997), it is uncertain as to whether the observed patterns are universally applicable.

Roy et al. (1998) discussed the causal hypotheses for latitudinal trends in biodiversity and tested these using data on Recent and fossil marine snails from the Atlantic and Pacific coasts of America. They found that their data did not agree with three of the major hypotheses – Rosenzweig's habitable space hypothesis (in this case, area of continental shelf); Rapoport's rule (species packed more tightly in the tropics, with smaller range sizes and more specialised niches); and the idea that the tropics have been undisturbed during ice ages and therefore have had more time to develop higher diversity. Instead, they found a significant relationship between the number of snail species and the temperature of the ocean surface along coasts (especially outside the tropics), and a probable influence of seasonality in nutrient supply, associated with the stability of the water column.

Pelagic systems are the largest on earth but have relatively few species, typically with vast distributions. Zooplankton diversity, in general, decreases with increasing latitude, both for oceanic and for neritic species (Pierrot-Bults 1997). The greatest concentration of zooplankton is in the upper 200 m of the water column, but maximum species richness occurs at about 800-900 m (Pierrot-Bults 1982; Angel 1993). Pelagic diversity patterns are largely determined by the large-scale physical and chemical environment of the ocean, such as oceanic circulation, and the observed latitudinal and bathymetric gradients do not necessarily correlate with factors such as productivity (Angel 1993, 1997).

The biogeography of Australian marine organisms is discussed in Section 2.4.2.

1.3 Conservation

1.3.1 What is conservation?

Conservation is defined (in an anthropocentric and utilitarian way) by the World Conservation Strategy (IUCN 1980) as "The management of human use of the biosphere so that it may yield the greatest sustainable benefit to present generations while maintaining its potential to meet the needs and aspirations of future generations". Other definitions are more straightforward; for instance, Erzinclioglu (1990) defined it simply as "the carrying of the maximum genetic diversity into the next century".

Conservation – i.e. protecting biological diversity and ensuring that essential ecological processes and life-support systems are maintained – is a key objective in the National Strategy for Ecologically Sustainable Development (Commonwealth of Australia 1992), National strategy for the conservation of Australia's biological diversity (Commonwealth of Australia 1996) and Australia's oceans policy (Commonwealth of Australia 1998a).

1.3.2 Similarities and differences between marine and terrestrial conservation

Comparisons between marine and terrestrial conservation from various perspectives have been made by several writers (Kenchington and Agardy 1990; Ray and Grassle 1991; Angel 1993; Norse 1993, 1997; Hutchings and Ponder 1999). While there are some obvious differences that can be emphasised, there are few fundamental differences. Norse (1997), for example, pointed out that some principles from terrestrial conservation biology will probably apply to the marine realm and gives the following examples: small populations are at greater risk than large ones; large species with low fecundity are at special risk; endangerment, phyletic distinctness and ecological importance are useful criteria for establishing priorities for species conservation; removing guilds such as high-level predators or grazers can profoundly affect organisms at lower trophic levels; greater structural heterogeneity allows species diversity; and certain areas (including areas of high diversity, high endemism, or high productivity, spawning areas, nursery grounds, and migration corridors and stopover points) merit special protection.

However, Norse (1997) argued that some conservation concepts and mechanisms that apply in terrestrial systems are not always applicable to the sea without modification because marine and terrestrial species and ecosystems function differently. He gave as examples:

- Planktonic dispersal stages in many marine taxa, with (potential) dispersal distances greater on average than for terrestrial species. Thus, the existence of allochthonous population sources and sinks can have implications for conservation and management.
- Ex situ conservation of marine species is not a viable option for the foreseeable future.
- Marine ecosystems are most productive and diverse at the edges of the sea, where human population densities, growth and impact are greatest.
- Time scales in the marine realm are different - the dominant primary producers are phytoplankton that can have doubling times of days, rather than large plants with doubling times of months or years.
- Producers and consumers of organic materials are often geographically distant, whereas on land materials tend to be produced and decomposed in close proximity; thus human activities that affect oceanic biogeochemistry may not become manifest for a long time.

1.3.3 Invertebrate conservation

This document is concerned primarily with the conservation of marine invertebrates. General issues relating to the conservation of invertebrates have been discussed by a number of writers in recent years but most of these approach the subject from a terrestrial perspective and are usually mainly focused on terrestrial arthropods and insects in particular (e.g., New 1993; New 1995b, etc. – see also Section 9.1; Yen and Butcher 1997). As outlined above, the following chapters discuss in detail various aspects of the marine invertebrate fauna, its conservation and the threats it faces.

CHAPTER 2 – THE AUSTRALIAN MARINE INVERTEBRATE FAUNA

2.1 State of knowledge of Australian marine invertebrates

The main points are:

- Taxa - state of taxonomic, biological and ecological knowledge: there are many knowledge gaps in even the relatively well-studied macrofaunal groups whereas many other taxa are very poorly known to virtually completely unstudied. This lack of knowledge results in:
  - A serious taxonomic impediment; and
  - The necessity to use exemplars, often (most inappropriately) from the northern hemisphere, when attempting to extrapolate biological features or predict ecological outcomes.
- State of knowledge of fauna by habitat:
  - Intertidal and shallow water fauna best known;
  - Deep-sea fauna is virtually unknown;
  - Coral reefs are relatively well studied compared with most other ecosystems, although only with regard to the most conspicuous taxa (hard corals and fish), not other invertebrates or inter-reefal areas;
  - Microscopic fauna in all habitats very poorly studied, especially the meiofauna;
  - Commensal and parasitic taxa often ignored and generally very poorly known.
- State of knowledge by geographical location:
  - Much ecological research is focussed around urban centres and marine stations on the GBR;
  - Islands and external territories are, overall, not well studied.

The vast expanse of ocean in the Australian Marine Jurisdiction, extending from Antarctica to the tropics, contains a large proportion of the southern hemisphere's marine biological diversity, much of which is endemic to Australian waters. Due to the size of the area involved, and the limited resources allocated[3], even information concerning the nature of the sea floor is limited. Our understanding of the biological diversity of marine invertebrates and extent of the marine resource that they represent is very limited (e.g., Marine Science and Technology Plan Working Group 1999). This lack of knowledge is highlighted, for example, by the relatively recent investigation (since the late 1980s) of a large number of seamounts south of Tasmania (Koslow and Gowlett-Holmes 1998; Koslow et al. 2001). Information about Australian marine invertebrates is generally poor and very patchy, being largely concentrated in a few highly visible, relatively common, or commercially important taxa. Even for the relatively well-studied dominant macrofaunal groups, there remain numerous gaps in the understanding of their taxonomy, biology and ecology; many of the minor groups are hardly known at all.

[3] Marine research comprised about 1.3% of R & D expenditure in Australia in 1996/7 - and yet marine resources contribute about 9% of gross national product in 1995/6, a figure that is rapidly increasing each year (Marine Science and Technology Plan Working Group 1999).

The cost and difficulty of accessing marine habitats has meant that accessible intertidal and shallow sublittoral systems are best known, whereas the fauna of deep-sea habitats remains largely unknown. Some habitat types – such as coral reefs – have been relatively well studied with regards to a few groups and processes, but there much remains to be done (particularly for the largely ignored but ecologically important inter-reefal areas). In all habitat types, the macrofauna has been best studied, whereas smaller components of the fauna – particularly cryptic fauna, the microscopic animals inhabiting the interstitial spaces between grains of sediment (i.e., meiofauna) and the myriad of commensal and parasitic taxa– are generally very poorly known or completely unknown.

Issues of accessibility have also led to something of a geographical bias, with research effort being largely concentrated near the coast, in the southeastern part of the mainland (especially in the vicinity of population centres), and in the Great Barrier Reef region. There is a significant lack of data from remote areas, such as north and northwestern Australia, the outer continental shelf and slope and even the outer slopes of the GBR, the deep-sea, the seamounts and around the islands and territories. However, even in the better-studied regions of Australia, there are very few locations where the invertebrate fauna has been sufficiently surveyed to enable reasonable baseline data to be obtained. The exceptions include Port Phillip Bay in Victoria and Botany Bay and Jervis Bay in NSW, each of which has been the subject of fairly extensive surveys, mainly in response to proposed large-scale developments. On the other hand, the only list of marine invertebrates from Sydney Harbour is that of Whitelegge (1889), and only recently has a survey been undertaken of the port of Sydney Harbour for introduced species by the Australian Museum (Australian Museum Business Services 2002). Aside from the issue of taxonomic knowledge, almost nothing is known of the biology or ecology of the vast majority of marine invertebrates.

2.1.1 The taxonomic impediment

Lack of taxonomic knowledge hampers advances in biological and ecological research, and this is particularly true in the marine environment (Marine Science and Technology Plan Working Group 1999) and has led to many ecologists identifying fauna only to higher taxonomic levels (family, order, class, or even phylum). This 'taxonomic impediment' exists because:

- Many groups of marine invertebrates are so poorly known and difficult to work with that they cannot be identified even to morphospecies with any certainty.
- The number of undescribed taxa far exceeds those named in many groups.
- Most groups have only one or a few authorities worldwide that can identify the taxa with any certainty; some groups currently have no accessible or active authoritative workers. Even when authorities exist, they are often unwilling to work outside their particular geographic region of expertise.
- The number of taxonomists in marine invertebrates is currently at its lowest level in decades and is declining[4] (see below).

[4] This decline in the number of taxonomists is a global phenomenon and applies to all groups, even the better studied taxa (vertebrates, vascular plants). However, it is perhaps more critical for the little-known

This "taxonomic impediment" to environmental monitoring and ecological understanding has been identified many times over the last few years in reviews of marine science and policy (e.g., in "Biodiversity Conservation" - Issues Paper 7 of Australia's Ocean Policy (Ward et al. 1997) and "Environmental Indicators for National State of the Environment Reporting on Estuaries and the Sea" (Ward et al. 1998a).

While the taxonomic impediment and the decline in the numbers of systematists are worldwide phenomena (e.g., Holden 1989; Nash 1989; Cotterill 1995; Cracraft 1995; Cotterill 1997; Hutchings 1999a), unlike other so-called developed countries, the Australian fauna is particularly poorly known, especially in comparison with Europe and the USA. For many groups, particularly the "minor" ones, there are no dedicated Australian taxonomists and there is consequently a reliance on overseas experts. Even the larger, most conspicuous groups have very few active workers in Australia (see Section 7.2.1). The problem is further exacerbated by the quantum of difference between North American and European versus Austral-Asian magnitudes of marine biodiversity.

The ABRS (see below) carried out a survey in the early 1980s on the state of taxonomic knowledge of the Australian fauna. Richardson (1984) presented the results of this survey, which included, but was not limited to, marine invertebrates. These results highlighted the seriousness of the "taxonomic impediment" as it relates to certain segments of the Australian fauna. The survey was distributed to taxonomists from around Australia and asked for information on 3971 families known or expected to occur in Australia. The responses showed that:

- For vertebrates and insects information was available on basically all families;
- For other groups, more than 40% of families were without data, this being especially true for the 'lower phyla' (cnidarians, sponges and "protozoans");
- For non-vertebrate coelomate groups, only about half of the species estimated to be present had been described;
- For acoelomate groups, it was not even possible to obtain accurate estimates of the number of families, let alone species, in Australia.

It should be emphasised that these conclusions related to the Australian fauna as a whole; the taxonomic impediment is generally far more significant for invertebrates than vertebrates, and for marine organisms compared to terrestrial ones.

Further highlighting the skewed distribution of taxonomic knowledge and research effort towards vertebrates and terrestrial groups, Richardson (1984) noted that the *Directory of Australian Taxonomists* (Bureau of Flora and Fauna 1981) listed:

- 29 mammal taxonomists (i.e. one for each family),
- 73 insect taxonomists (one for each 8 families), and
- 18 taxonomists studying helminthes, cnidarians, sponges and "protozoans" (one for each 86 families).

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groups (including most marine invertebrate phyla) given that there have never been sufficient experts or resources available for their study, even before current downward trends, and a diminishing capability to effectively train and recruit successors.

This situation has, if anything *deteriorated* since 1984 when Richardson's overview was published.
For instance, ABRS conducted a survey of the number of taxonomists employed in museums, herbaria and universities, which showed some fairly disturbing trends (Richardson and McKenzie 1992; Hutchings 1999a). For example:

- There were about 450 practising taxonomists (in zoology and botany) in 1991 in Australia compared to 28,000 in North America – which has about half of our biodiversity. 57% of those in Australia did not have full time paid positions as taxonomists.
- In 1974 there were 193 practicing taxonomists in Australian universities, compared to 64 in 1991. As taxonomic training is primarily carried out in universities, this has serious implications for the future.
- Many museum taxonomic specialists have already retired and many more are approaching retirement age. For instance, in 1991 the average age of museum taxonomists was 44.3 years, compared with 38.8 in 1974. As people retire or leave, they are often not being replaced. In 1974 taxonomists under 30 constituted 12% of the workforce in museums, in 1991 they constituted only 3% (Richardson and McKenzie 1992; Hutchings 1999a).

With little taxonomic expertise or training now provided in universities and ever declining funding for museums, there is a prevailing pessimism regarding the likelihood of their replacement by a new generation of systematists (e.g., Hutchings 1999a).

Australian Biological Resources Study (ABRS)

The Australian Biological Resources Study (ABRS) is an excellent but poorly funded Commonwealth program with the objectives of coordinating work on the collection, description and classification of Australian plants and animals, determining their distributions and publishing this information. It provides the main source of taxonomic research funding in Australia and also produces the Zoological Catalogues and (previously) the Fauna of Australia volumes. It is recently undergone a major reorganisation with the result that much of its output is now web-based, revolving around the online Australian Biological Information Facility (ABIF)[5], with the aim of making this information more readily and widely accessible.

Biology versus quantitative ecology

The need for ecological data, long term monitoring and assessment is briefly discussed in Chapter 7. While there is a justifiable emphasis in contemporary studies on the collection of quantitative ecological data there are very few studies being undertaken on the biology (natural history) of Australian marine invertebrates. With the exception of some commercial species, data being used to surmise the biology (e.g., feeding, life history, life span, role in ecosystem) of Australian taxa are often based on one or a few studies often carried out on (presumed) related taxa on the other side of the world.

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[5] http://www.ea.gov.au/biodiversity/abrs/abif/index.html

2.2 Marine invertebrate taxa found in Australia

The sea has a much greater high-order diversity (phyla, classes and orders) than terrestrial or freshwater habitats (Figure 1.4). In terms of diversity, size and numeric dominance, the major groups of macroinvertebrates are the crustaceans, molluscs, and polychaetes. Echinoderms are also of large size and ecologically important but less diverse. On hard substrates, such as rocky and coral reefs, sessile organisms such as sponges, bryozoans, hydroids and ascidians are often abundant or even dominant. There are many different worm phyla, ranging from extremely abundant and ecologically important groups (e.g., polychaetes, nematodes), to those that are little known (e.g., Loricifera, first described in 1983). The marine meiofauna have pivotal nutrient recycling roles. Despite this, they comprise the least known component of the fauna, consisting of minute, specialised representatives of many phyla, including some exclusively meiofaunal.

In this section, a brief overview of the diversity and state of knowledge of each of the major groups of Australia's marine invertebrate fauna is provided. This includes a brief description of each major group and – where available – numbers of described and estimated total numbers of species, and a short discussion of biogeographical patterns, endemism and trophic roles. This account is intended to highlight the great diversity of our fauna, as well as summarising the relative state of knowledge for each of these groups. For example, while the Nematoda is a huge group of undoubted ecological and biological significance with many thousands of parasitic and free-living taxa, this group remains very poorly known in Australia compared with most of the larger free-living taxa.

2.2.1 Porifera (Sponges)

The phylum Porifera (sponges) comprises the simplest multicellular animals, with little (or no nervous) co-ordination between individual cells, with huge numbers of different types of cells that do not form well-defined tissues or organ systems. They are predominantly marine with a small number of freshwater taxa. A general account of sponges is given by Bergquist (1978), and a synopsis of the higher taxonomy by Hartman (1982- albeit greatly out of date).

The **Porifera** is divided into three classes, the **Demospongiae** (siliceous sponges), **Calcarea** (calcareous sponges) and **Hexactinellida** (glass sponges). Worldwide, there are about 5000-6000 described species, although three times this number may be extant (Hooper and Wiedenmayer 1994).

The Zoological Catalogue of Australia (Vol. 12) records over 1350 Australian species, including representatives from extra-territorial waters (Hooper and Wiedenmayer 1994). The ABRS on line catalogue[6] has updates of this fauna, and includes about 1700 described species. About another 4000 species exist in museum collections (J. Hooper,

[6] http://www.anbg.gov.au/abrs/abif-fauna/vols.www.htm

pers. comm.). An estimated 500-600 species occur in the Antarctic and sub-Antarctic regions, and 25 or so in the Christmas and Cocos Islands; Norfolk and Lord Howe Islands have not yet been systematically investigated (J. Hooper pers. comm.). A ‘living fossil’ Mesozoic sponge fauna has previously been found on the Norfolk Rise off New Caledonia; there is a need to investigate deeper water species (200-500m), especially those associated with seamounts, to determine the presence, magnitude and affinities of this fauna in Australian waters.

Sponges are cosmopolitan, and are found from tidal areas to the abyssal depths (>6000 m). Adults are sessile and most attach themselves to any suitable surface, although many bore into rocks or calcareous substrates such as coral reefs or shells (Laverack and Dando 1987). The shape may be massive (i.e. an irregular lumpy shape without a regular outline), branched or encrusting, although this provides no indication of phylogenetic relationship as it can vary greatly within species and is determined in large part by the physical environment (Bergquist and Skinner 1982). For instance, encrusting species can live attached to rocks in relatively shallow water on the open coast, while the larger, upright forms are unable to withstand violent water movement and are found in depths below about 20 m or under ledges and in pools. In some cases these ecomorphotypes consist of a single species with a large morphological plasticity, conferring a capability to survive across a broad range of environmental extremes. In most cases, however, species appear to have very strict niche requirements and are found only in very narrow-range physical and geochemical regimes (J. Hooper pers. comm., Hooper and Wiedenmayer 1994).

Being sessile filter feeders, sponges flourish in areas where currents are strong. They are ecologically important, with large sponges being host to multitudes of commensal invertebrates (e.g., crustaceans, molluscs, worms and echinoderms) as well as huge numbers of micro-organisms, including bacteria and Cyanobacteria (Bergquist and Skinner 1982) and many species of the Archaeal Domain (e.g., Fuerst et al. 1999). Some species have lost their filter-feeding abilities (lacking an important sponge character, choanocyte cells - see below) adopting instead a carnivorous mode of feeding (e.g., Vacelet and Boury-Esnault 1995), but this is uncommon and appears to be confined to some deep-water species where carnivory has become a secondary adaptation.

Sponges are also of economic significance both as undesirable colonisers of man-made structures (and destructive pests of oysters, e.g., Wesche et al. 1998), and because of their pharmaceutical potential (Hooper et al. 1998). Interest in the phylum has escalated during the last decade or so, largely as a result of the rapidly growing marine natural products industry and the abnormally high proportion of “biologically active” members compared with other phyla (Hooper and Wiedenmayer 1994; Hooper et al. 1998).

With probably less than 30% of the fauna described, the discovery and documentation of poriferan diversity will be a major challenge (Hooper et al. 1999). There are also considerable difficulties with sponge identification and taxonomy, given that morphological characters are conservative and morphology is plastic, yet the current systematics is almost completely morphologically based. There are also problems with existing museum collections, as it is often impossible to equate many preserved specimens with living examples, and much of the material is unsuitable for use in molecular studies having been either preserved in formalin or dried. Yet, the few data from molecular studies indicate that allopatric 'sibling' species barely distinguishable morphologically are genetically very distinct (J. Hooper, pers. comm). The development of a stabilised, workable systematics based on objective phylogenetic criteria is thus an important objective.

The mechanisms for dispersal and recruitment of sponges also need to be investigated – for example, most species allegedly have limited demersal larvae, and appear to be very restricted in their distributions, yet about 5-15% of tropical morphospecies are widely distributed across the Indo-West Pacific Ocean (Hooper and Lévi 1994; Hooper 1998).

The Queensland Museum has identified contemporary collections (i.e., documented in situ) of 131 species of marine sponges from the Sydney region collected for the shallow subtidal 'storm water' studies (AWT Ensight) and deeper water sewerage outfalls (EPA). A further 69 species are known from the Byron Bay region (QM surveys), although many of these could not be assigned to a known taxon given the poor descriptions of earlier authors (Hooper et al. 1999). By way of contrast, 760 species have been recorded from the southern GBR, 45% of which are endemic (J. Hooper, pers. comm.).

The existence of sponges that lack the single-most important synapomorphy of the phylum, choanocyte cells, presents a conundrum in terms of defining what constitutes a sponge (Vacelet 1999). It is conceivable that each of the major poriferan groups, extant and extinct (Demospongiae, Calcarea, Hexactinellida, Stromatoporoidea, Archaeocyatha, Verticillatida and so on) might legitimately constitute taxa recognisable as distinct phyla, with some preliminary support from genetic evidence (J. Hooper pers. comm.), whereas based on grades of morphological construction all these groups are now presently recognised within a single phylum Porifera (e.g., Wood 1990).

Demospongiae (Siliceous Sponges)

This class comprises the siliceous sponges. While mostly marine, a few have invaded brackish and freshwater habitats. They are found from the intertidal to hadal (> 6000 m) depths. The Demospongiae account for the majority of described sponge species – about 5000 – worldwide, with probably three times as many yet to be described, and about 1300 'valid' species described so far for Australian waters (Hooper and Wiedenmayer 1994). Approximately 1000 southern Australian species, belonging to about 200 genera, have been described, although many are clearly synonyms (Bergquist and Skinner 1982). The fauna is not very well known and it is still difficult and often impossible to put a name on any but a few dozen species, due partly to the lack of knowledge of sponges as a whole, but mainly to a lack of knowledge of the Australian fauna in particular (Bergquist and Skinner 1982). In contrast to many invertebrate groups, the deeper water (abyssal, hadal) fauna is possibly better known than that of shallow waters (Hooper and Wiedenmayer 1994) as a result of the early exploration of the deeper seas during global expeditions. In addition, there is an allegedly relatively lower diversity of macrobenthic species in deeper seas than in shelf, shallow-water and intertidal regions, and an alleged 'cosmopolitanism' of the deeper water species (as compared to confirmed genetic heterogeneity between often closely adjacent sister species or populations of morphospecies in shallower habitats) (J. Hooper pers. comm.). However, with more deep-water collections the so called "widely" distributed sponge taxa are being found to be geographically heterogenous with distinct southern provincial trends (J. Hooper, pers. comm.).

Calcarea (Calcareous Sponges)

This class comprises the calcareous sponges, an exclusively marine group most common in relatively sheltered waters of less than 1000 metres. They are predominant in temperate regions, and in the tropics they are associated mainly with coral reefs. There are about 400 described species worldwide, with 163 'valid' species recognised in Australian waters pre-1930s (Hooper and Wiedenmayer 1994), and nine new species published more recently (Wörheide and Hooper 1999). Another approximately 50 new species from the Great Barrier Reef currently await formal description (J. Hooper pers. comm.). Moreover, genetic analysis of allegedly widely distributed taxa (eg GBR, Vanuata, Fiji) confirm the existence of genetically distinct sibling species some very cryptic, emphasizing that substantial genetic differences are not necessarily manifested at the morphological level in this highly enigmatic class.

Hexactinellida (Glass Sponges)

This Poriferan class comprises the glass sponges. These are exclusively marine and predominantly found in waters deeper than 200 m, although in boreal and austral seas many species may extend into shallower waters. While there are about 600 named species worldwide, there are only an estimated 450 living species and only 35 species recorded from Australian territorial waters to date (Hooper and Wiedenmayer 1994). This fauna is virtually unknown in Australian waters given the poor record of deep-water exploration.

2.2.2 Cnidaria (Corals, Hydroids, Jellyfishes, Sea Anemones, Bluebottles etc.)

Cnidarians are exclusively aquatic, predominantly marine, radially symmetrical animals with tentacles encircling a mouth at one end of the body. They include the hydroids, medusae ("jellyfish"), sea anemones and corals. All cnidarians possess nematocysts (stinging capsules) used primarily for food capture and defence, but also to assist in locomotion or adhesion to the substrate. Members of the phylum are highly polymorphic, the two main forms being the polyp (or polypoid or hydroid form) and the medusa. Although there are exceptions, the polyp stage is typically sessile, benthic, usually colonial and can reproduce asexually. Members of colonies are connected through extensions of the body cavity and different members may be specialised for different functions such as feeding, defence and reproduction. Medusae are nearly always solitary, free-swimming and sexual, thus aiding (where they occur as part of the life history) in dispersal (Southcott and Thomas 1982).

All but a few known species of cnidarians are carnivorous, though many rely heavily on symbiotic algae in their tissues for much of their nutrition. They prey largely on planktonic animals and plants that touch their stinging tentacles. While most medusoid forms (jellyfish) are harmless to humans, some species, especially of cubomedusae, are highly venomous, and their sting may be fatal.

The phylum is divided into 3-4 classes, the **Hydrozoa**, **Scyphozoa**, **Cubozoa** (sometimes included within the Scyphozoa) and the **Anthozoa**. In the Hydrozoa, polyps are generally more conspicuous than medusae; in the Scyphozoa and Cubozoa the reverse is the case; in the Anthozoa the animal is polypoid and there are apparently no medusae. However, there are many exceptions to these generalisations, such as hydrozoans that lack a polyp stage (Barnes et al. 1988). The generalised term "jellyfish" refers to the medusoid phase in the life cycles of the Scyphozoa and Cubozoa as well as some Hydrozoa.

Hydrozoa (Hydroids, Hydrocorals, Bluebottles etc.)

The class Hydrozoa includes the hydroids and hydromedusae. Alternation of generations occur in most genera, although one or the other generation may be suppressed or lacking. Medusoids are often retained on the polyp, polyps are usually colonial, reproduced by budding; and the medusae are mostly small and transparent (Brusca and Brusca 1990).

The class is divided into seven orders, including Hydroida (hydroids and their medusae), Milleporina and Stylasterina (hydrocorals), and Siphonophora (siphonophorans), as well as some lesser-known medusoid and polypoid orders.

The so-called **hydrocorals** are colonial hydrozoans that have a hard calcified supporting skeleton. The millepore hydrocorals are commonly known as fire or stinging corals (they can inflict a serious sting). Hydrocorals are essentially warm water animals and are extremely rare in southern Australia (Watson 1982). The **hydroids** are best known from the sessile colonial stage, which is more easily collected than the usually small medusae, which may measure only a few millimetres in diameter. The colonies take many growth forms, including flower-like (on a stem), tree-like or feathery. Hydroids are best represented in cool temperate southern Australian seas, where they are known from the intertidal zone to the deep-sea. They may comprise 20% of the total species of sessile organisms in some reef communities, although due to their generally small size, they form only a minor part of the total biomass of living reef organisms (Watson 1982). As sessile carnivores, they are most abundant in places where there is a good supply of water-borne food, such as reefs in fast-flowing tidal channels (Watson 1982). The **siphonophores** are hydrozoan colonies composed of both polypoid and medusoid individuals, with as many as 1000 zooids in a single colony (Brusca and Brusca 1990), and include some of the most dramatic examples of polymorphism among polyps. This order contains a great variety of unusual and poorly understood forms, including the famous 'Portuguese man-of-war' or 'blue bottle' (*Physalia*), which consists of a gas float (modified medusa) and trailing stinging tentacles arising from clusters of variously modified polyps (Barnes et al. 1988). *Physalia* is an important stinger along the open oceanic beaches of Australia, and occasionally in more sheltered waters (Southcott 1982). The group also includes *Velella,* commonly known as the 'By - the - Wind Sailor', which often washes ashore on ocean beaches with *Physalia*.

About 2,700 hydrozoan species have been described worldwide. Most families are represented in Australian seas. Many genera are confined to either tropical or temperate zones, although some are cosmopolitan circum-tropical or circum-temperate. Approximately 600 species have been described from the waters around the Australian mainland, although perhaps two-thirds as many again have yet to be described (J. Watson pers. comm.). Around 120 species have been described from the Antarctic region (J. Watson pers. comm.). While the Christmas Island fauna is only poorly known, and that of the Cocos Keeling island group completely unknown (Watson 1996), there are perhaps 20 described species for that region (J. Watson pers. comm.). There are many endemics, more so than in any comparable area except South Africa, as far as is known (P. Cornelius pers. comm.).

Knowledge of the taxonomy, biology and ecology of some hydroids is still poor although some information on the southern species is provided in Watson (1982) and western Australia (Watson 1996). However, little is known of the Hydromedusae in Australian seas, the group being without a dedicated Australian taxonomist and much in need of research (J. Watson pers. comm.).

Scyphozoa (Jellyfish)

In the scyphozoans the medusoid stage (the "jellyfish") dominates, the polypoid being small and inconspicuous or even absent. In general, scyphozoan medusae are larger, more complex and more conspicuous than the hydromedusae, although there are exceptions, notably among the siphonophores (Southcott 1982).

Scyphozoan jellyfish occur worldwide, although there are more species in tropical seas. Almost all appear to have wide geographic ranges; their long life spans and drifting habit are presumed to promote pan-ocean genetic mixing and low speciation, but also phenotypic variation. Some moderate-sized mid-water oceanic species are assumed to occur almost worldwide, with enormous total biomass (P. Cornelius pers. comm.).

Scyphozoans occur at all depths from shallow to deep-waters, but are most speciose coastally (P. Cornelius pers. comm.). The majority are planktonic but they may also be demersal or attached. Some have more restricted habitats, being found, for instance, only at the water-air interface, among seaweeds, or on or near the sea bottom (Southcott 1982).

Scyphozoans play a significant ecological role as pelagic predators and carnivores. They are often generalists with voracious appetites, capable of consuming any organisms, living or dead, in the water column (L. Gershwin pers. comm.). In addition, some species bloom in massive numbers, which can have a significant impact on the larvae and micro food sources of other species. Blooms can also, in some cases, have a considerable economic impact, as seen for example in the losses to Tasmanian salmon fisheries in January 1999 and the obstruction of power plants in the northern Kimberley during September 1999 (L. Gershwin pers. comm.).

There are approximately 200 species of scyphozoan jellyfish known worldwide (Brusca and Brusca 1990). Currently around 50 species have been recorded from Australian waters, almost all of which have wide geographic ranges and none of which is endemic (P. Cornelius pers. comm.). Currently only two nominal species, in just one genus, are known to be endemic to the Southern Ocean, with ranges extending from the South Atlantic to about the Falkland Islands (P. Cornelius pers. comm.).

Estimates of the proportion of the fauna that has been described vary among specialists. While P. Cornelius believes that "conspicuous forms…, excepting in the deep-sea, have probably mostly been described by now", L. Gershwin "strongly suspect(s) there is a huge undescribed medusan fauna in Australian waters", having recently found 38 new species of large and conspicuous medusae, mostly in museum collections, that were previously misidentified or unidentified. She also has purported evidence of many more taxa, either in anecdotal descriptions, photos, or fragments, and believes that many species are probably endemic to Australia, having been previously misidentified as northern hemisphere taxa. If this is the case, endemicity of coastal species could, in fact, be very high; at least 50% for Australian waters and perhaps almost 100% for Antarctic waters (L. Gershwin pers. comm.). However, P. Cornelius notes that while some local 'races', and some apparently well-marked species, may be genuine, so far no such full species have been recorded locally in the Australian literature, although "molecular taxonomy may elucidate the frequency of morphologically very similar 'sibling species' in some genera" (P. Cornelius pers. comm.). In any case, it appears that the scyphozoan fauna of Australia (whether widespread or endemic) is, as yet, incompletely documented. In northern Australia, for instance, it is likely that several widespread tropical species have yet to be recorded. P. Cornelius added two species "the size of footballs" to the Australian faunal list in a visit to northern Australia of only a few weeks, and noted that many quite large species are virtually or completely unknown – "an absurd situation in a rich country in which jellyfish are of such high profile to bathers" (P. Cornelius pers. comm.).

Existing research knowledge was summarised by Arai (1997) but this did not deal with taxonomy. The most recent monographs are by Mayer (1910) and Kramp (1961) and both have many shortcomings. Southcott (1982) summarised the southern Australian fauna. P. Cornelius[7] (in London) is undertaking a world revision with G. Jarms (Hamburg) doing the coronate (mostly deep-sea) species.

One species of Australian jellyfish *Catostylus mosaicus* is edible and there is a small fisheries for them and recently their reproductive biology has been studied with reproduction occurring throughout most of the year, although at some locations may cease during the winter months (Pitt and Kingsford 2000b, a).

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[7] Recently retired.

Cubozoa (Box Jellyfish)

The Cubozoa, treated either as a separate class (Cubozoa) or as an order (Cubomedusae) within Scyphozoa, are commonly known as box jellyfish or sometimes, "sea wasps". Medusae are small (15-25 cm tall), largely colourless and square in cross-section. They have a number of tentacles that in large species may be two metres in length or more (Southcott 1982), and a sting that, in some cases, is fatal to humans (Edmonds 1989). For this reason they have attracted much research and publicity, particularly in tropical regions.

Cubozoans occur in all tropical waters but are especially abundant in the Indo-West Pacific region (Brusca and Brusca 1990). There are approximately 17 known species worldwide, with about five species undescribed. Around nine species have been recorded from Australian waters, with possibly two more still to be described (P. Cornelius pers. comm.). Cubozoans are typically recorded as pan-Indo-Pacific. Some have been recorded from the Atlantic also, although the influence of possible transport, in historical times, in ballast water or on ships' hulls is unclear. None occur in the Antarctic region. While one or two have been described uniquely from Australia (or nearly so), this almost certainly reflects lack of collecting (P. Cornelius pers. comm.).

Knowledge of cubozoans is scant, and the group was not included in Arai's (1997) monograph. The taxonomy is chaotic, with probable multiple redescriptions across wide geographic ranges. Molecular taxonomy may elucidate the frequency of morphologically very similar 'sibling species' in some genera (P. Cornelius pers. comm.). There is a great need for a better understanding of the group (as for other stinging cnidarian medusae) in Australia and other tropical Pacific regions, in both taxonomy and other aspects of biology, to promote identification, first aid, clinical treatment, public awareness, and prevention (P. Cornelius pers. comm.). A very useful, albeit clinical analyses of both scyphozoan and cubozoan biodiversity, is presented by Williamson (1985; Williamson 1996).

Anthozoa (Corals and Sea Anemones)

In the Anthozoa the polyp is dominant, though larger and more complex than polyps in other classes, and there is no medusoid phase. There are both solitary and colonial forms. The class includes the familiar sea anemones, stony corals and octocorals. Three subclasses are recognised; Ceriantipatharia (solitary semi-burrowing anemones and black corals), Hexacorallia (=Zoantharia; comprising the sea anemones and scleractinian or stony corals), and Octocorallia (=Alcyonaria; comprising the soft corals, gorgonians or sea fans, seapens etc.). Much of even the southern Australian fauna remains undescribed. See Edgar (1997) for a list of the older literature.

Ceriantipatharia (Burrowing Anemones)

The only Australian representatives in this primitive group are solitary, semi-burrowing anemones of the order Ceriantharia. They live in a soft mucous tube and are typically found on sandy or muddy bottom in deeper water (15-50 m) in places with water movement. Due to the burial of the base some 60 cm or more below the substrate surface, they are difficult to collect without benthic suction apparatus. A number of species occur in Australia but as of the early 1980s, these had not yet been identified (Thomas and Shepherd 1982b). Two new species were recently described from Sydney Harbour (Carter 1995).

Hexacorallia (=Zoantharia) (Anemones and stony corals)

The Hexacorallia, members of which all display a six-rayed symmetry (i.e. six tentacles), includes the anemones and the stony or scleractinian corals.

Sea anemones – orders **Zoanthidea**, **Corallimorpharia** and **Actiniaria** (the sea anemones proper) – are familiar, solitary anthozoans and occur throughout the world in coastal waters. They live at all depths on the sea floor and are abundant on most rocky shores. A few species live in sand or mud but they are generally attached to rock or shell below the surface. They should be considered carnivores rather than suspension feeders, since they actively capture plankton (mostly tiny crustaceans) with their tentacles after paralysing them with the nematocysts. However, many have zooxanthellae and a few species are also detritus feeders, using their tentacles to trap drifting organic matter (Thomas and Shepherd 1982b). Some animals have commensal relationships with anemones; these include small fishes which gain protection from living among the tentacles, cleaning shrimps, snapping shrimps, brittle stars, etc. Anemones also attach to other animals (scyphomedusae, ctenophores and the shells occupied by hermit crabs), offering protection and gaining from being moved around to new feeding areas (Barnes et al. 1988).

The sea anemone fauna consists of approximately 950 described species worldwide. The Australian anemone fauna is rich, with around 200 actiniarian and 20 corallimorpharian anemone species recorded from Australian waters, and perhaps a third as many again yet to be described (D. Fautin pers. comm.). Anemones have been described from the Antarctic region (c. 60 described, with possibly 20 undescribed species), Christmas and Cocos Islands (c. 40 described species), Norfolk (c. 20) and Lord Howe (c. 20) Islands. An estimated one-fifth of species are probably endemic to Australian waters (D. Fautin pers. comm.).

Sea anemones are moderately well known in some parts of the world but Australia's sea anemone fauna is very poorly known. Although some papers have been published on Australian sea anemones, Australia has never had a dedicated anemone taxonomist, and there is not even a rudimentary overview of the fauna (D. Fautin pers. comm.).

The *stony or scleractinian corals* (order **Scleractinia**) consist of polyps connected laterally and have a hard calcareous 'exoskeleton', into which the polyps can retract, providing support and protection. Most species form sessile colonies of small polyps 1-20 mm in diameter, but there are also solitary corals, which typically have much larger polyps and can be sessile or free-living (Jackson 1991). Corals are well known and important as one of the primary reef-builders in coral reefs, although there are also ahermatypic (non-reef-forming) species. Coral growth is indeterminate, with no indication of senescence. Some corals may grow up to five or ten metres in diameter and live for centuries or millenia (Jackson 1991). Colony shapes range from simple encrusting sheets, to massive mounds, to various erect forms most commonly branching (like small trees) or foliaceous (horizontal or vertical plates attached at one end). Most corals can propagate asexually by breaking apart into clonal fragments, which may disperse short distances across the reef, as often occurs during cyclones. The other second important means of reproduction and dispersal is by sexually produced larvae. Most species spawn gametes fertilised in the ocean and develop into drifting planktonic larvae, but around a quarter of coral species are brooded by their parents (Jackson 1991). As with other cnidarians, corals have nematocysts and are carnivorous, using their tentacles to capture small planktonic organisms and other suspended organic material. The reef building (hermatypic) species also contain symbiotic zooxanthellae (minute algae) in their tissues which fix carbon photosynthetically and transfer some of it to the host coral, substantially increasing the growth and calcification of the coral (e.g., Trench 1979; Muscatine 1990).

The requirement for light by the autotrophic zooxanthellae limits reef building to clear, shallow waters. Although the great majority of stony corals are found in tropical and subtropical seas, a few – particularly the ahermatypic species – are found in temperate waters (Shepherd and Veron 1982), and can be abundant enough to form reef-like structures. Scleractinians show a variety of distribution patterns. Some genera – such as *Acropora* – occur in reef systems worldwide but the vast majority are found either in the Indo-Pacific region (a global centre of coral diversity) or the Caribbean. Cornell (2000) discuss the factors that determine coral diversity.

Around 390 scleractinian species have been described from the waters surrounding the Australian mainland, 25 from the Christmas and Cocos (Keeling) Islands, 60 from Norfolk Island and 83 from Lord Howe (Harriott et al. 1995). Southern Australian species were reviewed by Cairns and Parker (1992). None have been recorded from the Antarctic or sub-Antarctic regions (J. Veron pers. comm.). Very few species are cosmopolitan; many are Indo-Pacific and others have more restricted distributions. The Australian fauna was discussed in detail by Veron (1986), who provided an overview of the systematics, biogeography and evolutionary history of the tropical Australian and Indo-Pacific coral fauna. The major patterns in faunal distributions were summarised briefly by Veron (1995a).

By far the largest extant genus of reef-building corals, and among the most widespread, is *Acropora*, the staghorn corals. This group includes the most obvious and important corals on coral reefs throughout the world. The world fauna comprises 114 described species, including 63 species recorded from Australian waters. A comprehensive revision of the group is provided by Wallace (1999a; Wallace 1999b).

Corals are the subject of considerable popular and scientific interest, due in large part to the productivity, biodiversity and economic (e.g., fisheries and tourism) values of the coral reef ecosystem, as well as the threats facing it, particularly from global warming (see Section 6.7.1) and pollution from terrestrial runoff (see Section 6.6.1). Undoubtedly, hermatypic scleractinarian corals are the most significant and high-profile of all marine invertebrates – the only taxon to be recognised as a world heritage icon forming one of the "wonders of the natural world", the Great Barrier Reef. Consequently, they are generally well researched and well known, and there are probably relatively few undescribed species (J. Veron pers. comm.). Nonetheless, the taxonomy and identification of scleractinian corals remain difficult due to morphological variation, even within a single colony, and in response to habitat.

Octocorallia (Alcyonaria) (Octocorals, Soft Corals, Sea Fans, Sea Pens and Sea Whips)

The octocorals are easily distinguished from all other Anthozoa by the eight-rayed symmetry of their polyps, each of which bears eight tentacles. Colonies consist of polyps united to various degrees by a common tissue mass, the coenenchyme, with an internal skeleton of calcareous structures. There are an extremely wide variety of colony shapes, from a few polyps extending from a membranous base, to fleshy encrusting masses many square meters in area, to huge tree-like colonies several metres tall (P. Alderslade pers. comm.).

Octocorals are cosmopolitan and found from intertidal to abyssal depths, but are most diverse in shallow, warm water, even occurring in estuarine conditions if the salinity is not too low (P. Alderslade pers. comm.). As adults they are sessile, the majority attaching themselves to solid structures such as rocks, harbour piles, jetties, mollusc shells and the like (Grasshoff and Verseveldt 1982). Sea pens, however, are obligate colonisers of soft bottom environments, anchoring in the substrate with a muscular foot or peduncle. In general octocorals feed on planktonic organisms (both animal and plant), but many shallow water forms supplement this by hosting zooxanthellae, while some others appear to rely totally on these photosynthetic symbionts. They use both sexual and asexual reproduction strategies, sexual reproduction including broadcast spawning, internal fertilisation, and brooding (P. Alderslade pers. comm).

Currently only 3 octocorallian orders are generally recognised: **Coenothecalia**, blue coral, **Alcyonacea,** the soft corals and gorgonians (sea fans and sea whips), and **Pennatulacea**, the sea pens (P. Alderslade pers. comm.).

Bayer (1980) reviewed the systematic status of octocorals of the world, and more recently groups of Australian taxa have been researched by Alderslade (e.g., Alderslade 1998). Fabricius and Alderslade (2001) have produced a generic level field guide for the broader Indo-Pacific region. Worldwide, there are an estimated 2500 described octocoral species, with probably a similar number yet to be described (P. Alderslade pers. comm.). Around 700 species have been recorded from Australian continental waters, close to half of which may be Australian endemics, and about 10 species from Australian Antarctic and sub-Antarctic regions. Octocorals have also been collected from the Christmas and Cocos (Keeling) Islands and Lord Howe Island. The diversity in these regions is unknown, but an estimate for the Christmas and Cocos island region is 250 species, and for Lord Howe 50 species (P. Alderslade pers. comm.).

In general, the world's octocorals are very poorly known, despite much interest in the potential for useful biologically active compounds. There are only 3 scientists, worldwide, employed full-time to research the taxonomy of the group, and only one is stationed in the diverse Indo-Pacific. Only the taxonomy of the Mediterranean and Caribbean octocorals is close to complete. In Australia and its territories, there are estimated to be a large number of genera and species still undescribed. Most of the Australian tropical and subtropical coastal regions have never been surveyed, and apart from a few isolated areas of the Tasman seamounts, the deep-water regions around the country are virtually untouched. The biology and ecology are likewise poorly known, with most studies having been conducted on Caribbean and Red Sea fauna (P. Alderslade pers. comm.) although recent work by Fabricius and De'ath (2001) is beginning to rectify this for Australian species.

2.2.3 Ctenophora (Comb Jellies)

The members of this phylum are known commonly as comb jellies. They are carnivorous and exclusively marine. While they resemble some cnidarian (hydrozoan) medusae, and were formerly placed together with the Cnidaria in the phylum Coelenterata, they have a number of important distinguishing features. For instance, virtually all lack nematocysts, but possess specialised cells called colloblasts, which on contact throw out long threads with adhesive tips with which they can entangle small prey. They also have comb rows; eight radially-arranged bands of cilia partly fused in transverse rows like tiny combs (Southcott and Thomas 1982). They are also notable for their luminescence (Barnes et al. 1988).

The Ctenophora is a small group with only about 100 species known worldwide (Barnes et al. 1988). Approximately five to ten species have been described from Australian waters, and nine from the Antarctic / sub-Antarctic region (L. Gershwin pers. comm.), with the most recent local review being that of O'Sullivan (1986). However, Australian ctenophores are poorly studied, and examination of museum collections has revealed numerous new species, most of them apparently endemic (L. Gershwin pers. comm.). There may in fact be 50 or more undescribed species from coastal Australian waters, plus a number from the Antarctic region (L. Gershwin pers. comm.).

2.2.4 Platyhelminthes (Flatworms, Flukes and Tapeworms)

The Platyhelminthes are a large group comprising around 25,000 species (Edgar 1997). They have bilaterally symmetrical, soft, flattened bodies that lack segmentation. They lack a circulatory system, their digestive system consists of a single opening (mouth) leading to a body cavity and they usually have separate sexes. The group consists of four major classes, **Turbellaria**, **Monogena**, **Trematoda** and **Cestoda**, the last three wholly parasitic. The parasitic groups can have very complex life cycles involving two to several hosts. In general the group is very poorly known.

Turbellaria (Flatworms)

Turbellarian flatworms inhabit a variety of marine, freshwater and moist terrestrial habitats, and range from interstitial species in marine muds and sands to benthic and pelagic species. The class is very diverse, with several thousand species having been described worldwide (L. Cannon, A. Faubel pers. comm.). In Australia, around 200 marine species have been recorded and up to 50 from the Antarctic / sub-Antarctic region. However, the fauna is generally very poorly known and it has been estimated that an enormous number of species (e.g., 1-2,000 for Australian waters; 50,000 worldwide) remain to be discovered and/or described (L. Cannon pers. comm.). The number of species occurring in the waters surrounding the Christmas, Cocos, Norfolk or Lord Howe Islands is not known (L. Cannon pers. comm.), though presumably they do occur there.

Data on other aspects of the biology are likewise unknown or very sketchy (L. Cannon, A. Faubel pers. comm.). There are currently only a few specialists worldwide dealing with the taxonomy of free-living Platyhelminthes, and large areas of the world have not been studied (E. Schockaert pers. comm.).

There are 12 recognised orders of Turbellarians. Among the more common or more important of these are the Polycladida, Tricladida, Rhabdocoela, and Acoela. A guide to the families and genera is provided by Cannon (1986).

The **Polycladida** (polyclads) are a diverse group of relatively large turbellarians, with a flattened body up to 15 cm in length. They are predominantly marine, benthic and free-living, although a few are pelagic or symbiotic. They are common in the littoral and sublittoral zones throughout the world, but especially in the tropics; in polar regions there are only about a dozen species (Prudhoe 1982; Brusca and Brusca 1990). They may be found in a variety of benthic habitats and pelagic species may live among surface plankton or floating weed (Prudhoe 1982a). In Australia, they are found throughout the Great Barrier Reef region on sublittoral coral reefs (A. Faubel pers. comm.) but are also found on all other Australian coasts. Polyclads are carnivorous and readily feed on dead or dying animals; many species also feed exclusively on sessile animals such as bryozoans and alcyonarians (Prudhoe 1982a).

Despite being conspicuous inhabitants of shallow water reefs, surprisingly little is known about either their biology or diversity (Newman and Cannon 1994). Around 150 species have been recorded from Australia and the Indo-Pacific, with probably thousands of species yet to be described (L. Newman pers. comm.). From 1989 to 1993, Newman and Cannon documented 134 species of polyclads (over 90% new) from two locations in the southern GBR, indicating that the diversity of tropical marine polyclads is much greater than was previously thought (Newman and Cannon 1994). No species have been described from any of the islands or territories, although they must occur there, including a small number in the Antarctic / sub-Antarctic region (L. Newman pers. comm.). Southern species are documented by Prudhoe (1982a; Prudhoe 1982b).

Polyclads form associations with soft corals, brittle stars, molluscs, crustaceans, echinoderms and corals, are important predators of commercial bivalves and are known to accumulate many pharmaceutically important compounds and yet remain very poorly known (Newman and Cannon 1994; L. Newman pers. comm.).

The **Tricladida** (triclads) are marine or freshwater, with some terrestrial species. Most are free-living, including the familiar planarians. The marine taxa of the world are reviewed by Sluys (1989) and those of Australia by Sluys and Ball (1989). Nevertheless, the Australian marine triclad fauna is very poorly studied, with few species having been described and few records available on their distribution. For instance, there are only four described species of the suborder Maricola from Australia (three endemic), and six from the Antarctic and sub-Antarctic regions, from a world total of 70 described species (R. Sluys pers. comm.). The faunas of Christmas, Cocos (Keeling) and Lord Howe Islands are unknown (R. Sluys pers. comm.).

The **Acoela** (acoels) are small (1-5 mm) common flatworms which inhabit marine and brackish water sediments; a few are planktonic or symbiotic (Brusca and Brusca 1990). Little is known of the Australian acoel fauna, although a couple of papers, based mostly on specimens collected opportunistically from hard and soft corals, have been published (Winsor 1988, 1990).

The **Rhabdocoela** are an extremely large and diverse group with both free-living and symbiotic taxa found in marine and freshwater habitats. Relatively little is known of this or any of the remaining free-living marine turbellarian orders present in Australian waters, although some groups (particularly the Rhabdocoela and Proseriata) have been studied by E. Schockaert. Of these groups, there are around 15-20 species recorded from Australian waters, of a total world fauna of approximately 6500 known species. However, about 200 species have actually been observed, and there are undoubtedly many more (E. Schockaert pers. comm.). For instance, sampling in a limited area of coast near Townsville and between Sydney and Brisbane has resulted in the collection of about 150 species, most of them new, and most yet to be described; this may only be a fraction of the total undescribed fauna (E. Schockaert pers. comm.).

The other relatively common or important orders are (Brusca and Brusca 1990):

Nemertodermatida – small turbellarians inhabiting subtidal marine muds and sands; one genus is parasitic in sea cucumbers
Catenulida – elongate, freshwater and marine
Macrostomida – small and predominantly interstitial; marine and freshwater
Lecithoepitheliata – a small group of about 30 species worldwide
Prolecithophora – small, free living, marine and freshwater
Proseriata – most are free living, marine

Turbellarians form a component of the marine meiofauna, and considerable work has been done recently in Australia on these previously understudied animals by Dr Anno Faubel (Leigh Winsor pers. comm.). Although little published information is currently available, Faubel et al. (1994) described the Macrostomida from eastern Australia.

Monogenea (Flukes)

Monogeneans are parasitic flukes with a reduced or absent oral sucker and a life cycle involving only one host. Most are ectoparasitic, usually on fishes but occasionally on turtles, copepods, squids and some terrestrial animals; a few are endoparasitic in warm-blooded vertebrates (Brusca and Brusca 1990). Whittington (1998) suggests that as their are approximately 25,000 species of fish worldwide, and as the Monogenea are believed to be among the most host-specific of parasites, and if each species of fish are host to a different species of monogenean, there could be almost 25,000 monogenean species, of which between 3000 to 4000 are described. Australia has about 3500 species of fish and thus potentially 3500 species of host specific monogeneans, although to date only about 300 + species have been recorded from Australia. At Heron Island, where there has been a focus of sustained research on these parasites, only about 85 species have been described from 40 of the most common, easily caught species of fish, indicating that several species may be found on a single host. Although many monogeneans have been recorded from Heron Island, many have only been identified to genus, family or even to class (Lester 1989).

Trematoda (Flukes)

Trematodes are another class of parasitic flukes. Those of the subclass Digenea have two or three hosts during the life cycle, the first intermediate host being a mollusc or, occasionally, a polychaete, and the final a vertebrate; those of the subclass Aspidogastrea mostly have a single host (a mollusc), with a second, when present, being a fish or turtle.

Cribb (1998) has reviewed the diversity of the Digenea from Australian animals , both marine and terrestrial. Asexual reproduction occurs in the first intermediate hosts, in generations of sporocysts, rediae and then to the sexual generation (cercariae). The cercariae leave the intermediate host and infect the definitive host or may encyst in a second intermediate host to be finally eaten by the definitive host. Digeneans occur in numbers where molluscs are abundant. Curently the approx. 600 species (all habitats) are included in about 360 genera with 82 families in Australia with many vertebrates acting as final hosts. Cribb (1998) suggests that based on current knowledge and extrapolation there may over 6000 species of digeneans in Australia.

Cestoda (Tapeworms)

Tapeworms are exclusively endoparasitic, in the guts or coelomic cavities of various vertebrates. General references on marine parasites, including parasitic flatworms, include Rohde (1993) and Rohde et al. (1995). L. Cannon at the Queensland Museum has a database (ASPIC) of all Australian museum holdings of parasites. Beveridge and Jones (2002) have reviewed the biodiversity and biogeographical relationships of the Australian cestode fauna.

The Australian cestode fauna is poorly documented with 342 species described to date (Beveridge and Jones, 2002). The best-studied group for these internal parasites is the elasmobranchs (although only about 32% of species examined). Currently all but two of the orders of cestodes are found in Australia. Some endemic species occur in Australian fish and a high percentage of known cestodes occur in a single host species suggesting that considerable numbers of cestode species remain to be documented.

2.2.5 Nemertea (Ribbon Worms, Proboscis Worms)

Nemerteans – commonly called ribbon worms – are unsegmented, usually flattened, solid-bodied (i.e. 'acoelomate') worms with highly extensible bodies; many can stretch easily to several times their contracted lengths. Larger species are often leaf or ribbon shaped. They vary greatly in size, ranging from less than a millimetre to more than 30 metres in length, though most species (even the larger ones) are less than about 6-8 mm wide (Gibson 1997). The phylum includes the longest of known animals: *Lineus longissimus*, from European waters, regularly attains 30 m and some individuals can probably achieve twice this length when fully extended (Barnes et al. 1988).

Nemerteans are predominantly benthic and marine, although a few freshwater and terrestrial species are known. A few are planktonic, and some are symbiotic in molluscs or other marine invertebrates (Brusca and Brusca 1990). They typically dwell under stones and boulders, burrow into softer muddy or sandy sediments, or live amongst algae, between colonial sessile invertebrates or in rock and coral crevices and cavities (Gibson 1997). Some species are common members of the intertidal benthos, but most are found in shallow coastal waters, while a few have been trawled from extreme depths of 3000 to 4000 metres or more (Gibson 1997). Almost all species are predatory, capturing organisms ranging in size from protists to molluscs, arthropods and fish (Barnes et al. 1988).

Worldwide, approximately 1200 species and 250 genera are currently recognised as valid (Gibson 1997; R. Gibson pers. comm.). Around 60 species have been recorded from Australian waters, 24 from the Antarctic / sub-Antarctic region, and although none have yet been recorded from the Christmas, Cocos (Keeling), Norfolk or Lord Howe Islands, they presumably do occur there. There is estimated to be a very large undescribed fauna, consisting of perhaps 200 or more species in Australian waters, and 5,000-10,000 worldwide (R. Gibson pers. comm.).

There are huge gaps in our knowledge of the nemertean fauna from most parts of the world, with few serious nemertean taxonomists globally (see review of world fauna by Gibson 1995) and none based in Australia. There is a reasonable (though not complete) knowledge of the nemertean fauna of some parts of the world – such as northwest Europe, the USA, some of Brazil, parts of New Zealand, some coastal areas of Russia and Japan, and parts of Australia (Gibson 1979a, 1979b, 1981a, 1981b, 1982, 1983; and southern WA,Gibson 1990, 1993; Sundberg and Gibson 1995; Burgman and Lindenmayer 1998). However, for most parts of the world, nemertean records are either completely unknown or sporadic (R. Gibson pers. comm.).

2.2.6 Gnathostomulida

Members of the phylum Gnathomulida are minute (< 1 mm length), transparent meiofaunal worms and the phylum was only named in 1969 (Barnes et al. 1988; Brusca and Brusca 1990). This late discovery was not due to rarity of these animals, which can achieve densities of 600 000 m^{-3}, but to previous lack of investigation of the anoxic layers of marine sediments, and the extent to which these animals distort on preservation.

Gnathostomulans are found interstitially in marine sands, often occurring in high densities in anoxic sulphide-rich conditions, from the intertidal zone to depths of hundreds of metres (Brusca and Brusca 1990). They feed on bacteria and protists occurring either free in the interstitial water or in association with particles of organic detritus (Barnes et al. 1988). They have been found worldwide, with about 80-100 described species (Brusca and Brusca 1990). However, as with other meiofaunal groups, virtually nothing is known of the Australian fauna.

2.2.7 Gastrotricha

The gastrotrichs constitute a phylum of small marine, brackish or freshwater animals with a more or less transparent, flattened body. They may reach 3-4 mm in length, although most species are less than 1 mm. Many bear a superficial resemblance to rotifers or even large ciliate protozoans (Brusca and Brusca 1990).

Many gastrotrichs live in the interstitial spaces of loose sediments. Others are found in surface detritus or among the filaments of aquatic plants; a few are planktonic (Brusca and Brusca 1990). There are around 450 known species worldwide but virtually nothing is known of the Australian fauna.

2.2.8 Nematoda (=Nemata) (Round Worms, Threadworms)

The nematodes, or threadworms, are one of the most abundant and successful metazoan phyla, being known from virtually every habitat in the sea, freshwater and on land. There have been estimates that 80% (Bongers 1988) or 90% (Jairajpuri and Ahmad 1992) of all metazoa are nematodes. However, most of the enormous literature on the nematodes deals with the parasitic species of economic or medical importance (Brusca and Brusca 1990).

Free-living marine nematodes are among the most common and widespread groups of animals, occurring from the shore to the abyss. Most free-living species are very small, of the same order of size as the gastrotrichs and gnathostomulans. Wherever they are found, they are often the most numerous metazoans, in both numbers of species and individuals;

some environments yield as many as three million threadworms per square metre (Brusca and Brusca 1990). Ecologically, some are generalists, but many require very specific habitats. Many free-living nematodes live interstitially and feed on bacteria or protists ingested as a suspension or on detritus. Some, however, are predators, especially of other nematodes and microorganisms, and have developed a variety of piercing tooth or jaw-like plates (Barnes et al. 1988).

Despite their abundance, the free-living nematodes are poorly known, and thus their importance in marine benthic systems is little appreciated (Brusca and Brusca 1990). Much of the ecological work on nematodes involves identification of the fauna only to family level.

Worldwide, there are about 20,000 described species of nematodes (both free living and parasitic, from marine and non-marine habitats) worldwide (Boucher and Lambshead 1995), but probably many times that number undescribed. Around 5000 free-living marine species have been described, and there are probably at least 20,000 in total (W. Nicholas pers. comm.). Lambshead's (1993) estimate of many millions of species, based on the diversity of several studies of the benthic fauna of the deep Atlantic, has not received much support, being a huge extrapolation (W. Nicholas pers. comm.).

Greenslade (1989) published a checklist of the free-living marine nematodes of Australia and Macquarie and Heard Islands. The sublittoral nematode fauna of Australia is virtually unknown. The littoral and estuarine fauna is better known, with taxonomic records going back to the 1890s. Recent studies on the fauna of mangroves and estuaries include Hodda and Nicholas (1985; Hodda and Nicholas 1986a; 1987) and Nicholas et al. (1991; 1992). Faubel et al. (1994), Blome and Riemann (1994) and Nicholas and Hodda (1999) also discussed the free-living nematodes of the sandy beach meiofauna of southern Queensland and NSW, including recent species descriptions.

There is little information on the affinities of the Australian nematode fauna and little agreement on how worldwide the distributions of species are. There is a tendency to ascribe all the specimens found to species described from Europe and North America because these are the only published descriptions (W. Nicholas pers. comm.).

2.2.9 Nematomorpha (Horsehair Worms)

Members of this phylum are commonly called horsehair worms on account of their thread- or hair-like shape. The nematomorphs are somewhat similar to nematodes in body plan, and generally measure 1-2 mm in diameter and up to a metre in length (Brusca and Brusca 1990). The majority of species occur in freshwater or moist soil, but members of one genus (*Nectonema*, in the monogeneric class Nectonematoida) are pelagic in coastal marine environments, their larvae parasitising decapod crustaceans.

In total there are about 230 described species in the phylum worldwide (Barnes et al. 1988; Brusca and Brusca 1990). Little is known of Australian nematomorphs and there are few identified in the AM marine invertebrates collection (P. Berents pers. comm.). G. Poiner at Oregon State University works on Australian nematomorphs.

2.2.10 Kinorhyncha

The kinorhynchs, sometimes called the Echinodera ("spiny neck") or Echinoderida because of their spiny cuticle, are a little-known phylum of marine meiofauna. Since their discovery in the mid-1800s, about 150 species have been described, nearly all less than 1 mm in length. Most live in marine sands or muds, from the intertidal zone to a depth of several thousand metres. Some are known from algal mats, holdfasts, and sandy beaches, while others live on hydroids, ectoprocts, or sponges. They are probably deposit feeders (Barnes et al. 1988; Brusca and Brusca 1990). As with other members of the marine meiofauna, little or nothing is known of the Australian species.

2.2.11 Loricifera

Loriciferans are a little-known phylum of very small (225-383 µm long) interstitial animals that, like the gastrotrichs, bear a superficial resemblance to rotifers or to priapulan larvae. They were first described in 1983, based on a single species found in marine shell gravel. A few dozen species have since been described or are in the process of being described, and they are now known to depths of several thousand metres where they live in mud.

These tiny animals cling tightly to sediment particles, or possibly other organisms, and are therefore not susceptible to standard extraction techniques used to collect interstitial marine species. The larvae have been observed alive and are free-living. The adults are probably also free-living, but the biology of this newest phylum is poorly known (Barnes et al. 1988; Brusca and Brusca 1990).

2.2.12 Priapula

This is a small but characteristic phylum of marine worms whose affinities are unclear (Barnes et al. 1988). They are cylindrical or cucumber-shaped and range from 0.55 mm to 20 cm in length. Priapulans are uncommon in most parts of the world's oceans. The larger priapulans are infaunal burrowers in relatively fine marine sediments and appear to be restricted to boreal or cold temperate seas. The meiofaunal species may burrow or live interstitially among sediment particles. The majority prey on various soft-bodied invertebrates such as polychaete worms (Brusca and Brusca 1990). Their burrows are also the home for commensals.

Worldwide, there are about 15 extant species (Theroux and Wigley 1998). Priapulans are present in Australia, being common for example on intertidal beaches in northern NSW and Queensland, although little is known of them.

2.2.13 Rotifera

The phylum Rotifera (=”wheel bearer”) consists of minute animals characterised by a crown of cilia in the anterior part of the body. They may reach 2-3 mm but are mostly less than 1 mm long. Most live in freshwater or in damp soil but there is one marine genus (*Seison*, in the monogeneric class Seisonidea) which live on the gills of crustaceans (Brusca and Brusca 1990).

2.2.14 Sipuncula (Peanut Worms)

Sometimes called “peanut worms”, members of this phylum are soft-bodied, unsegmented, worm- or sac-like animals. They are exclusively marine. The body consists of two parts, a slender, extensible introvert and a thicker, sausage-shaped trunk. Sipunculans range in length from 1 cm to about 50 cm, but most are 5-10 cm long (Brusca and Brusca 1990).

Sipunculans are found in great abundance in some habitats and have a wide distribution in many parts of the world ocean (Theroux and Wigley 1998), being found from the intertidal zone to depths of over 5000 metres. They are usually reclusive, either burrowing into sediments or living beneath stones or in algal holdfasts. Some inhabit abandoned gastropod shells, polychaete tubes and other such structures. In tropical waters sipunculans are common inhabitants of coral and beach rock communities, where they often burrow into hard calcareous substrata (Brusca and Brusca 1990). In southern Australia they occur mainly in the roots of seagrass beds, in burrows in limestone reefs, under rocks, in thick masses of the tubeworm *Galeolaria* and in beds of mussels (Edmonds 1982a). Their burrows may be shared with a number of commensal associates that share food resources and shelter (Manning and Morton 1987; Morton 1988). They are predominantly non-selective detritivores or deposit feeders (Brusca and Brusca 1990), though others are suspension feeders or graze on the fine layer of algae that grows on marine surfaces (Edmonds 1982a).

Taxonomically they are a small group, comprising about 250 species (worldwide) in 17 genera in a single class and order (Barnes et al. 1988; Brusca and Brusca 1990). Edmonds (1980) revised the Australian species. Current knowledge on Australian sipunculans is reviewed by Edmonds (2000b).

2.2.15 Echiura

The echiurans are also soft-bodied, unsegmented, sac-like worms related to sipunculans and resemble them in several ways. They have an extensible proboscis with a food-collecting tip. Many are quite large; the trunk may be from a few mm to 40 cm long, but the proboscis may reach lengths of 1-2 metres (Brusca and Brusca 1990).

Echiurans live exclusively in marine or brackish-water habitats where they are known from intertidal regions to about 10,000 m. Like the sipunculans, they have a sedentary, deposit-feeding lifestyle, typically burrowing in sand or mud, or living in surface detritus or rubble. Some species inhabit rock galleries excavated by boring bivalves or other invertebrates (Brusca and Brusca 1990). They may also share their burrows with a range of commensals (Boxshall and Humes 1987; Morton 1988). Most echiurans feed on epibenthic detritus. There are about 150 known species. Only a few species are known from Australia (Edmonds 1982b). Edmonds (1987) listed and described the Australian species, and current knowledge on echiurans is reviewed by Edmonds (2000a).

2.2.16 Annelida (Segmented Worms)

The phylum Annelida contains the segmented worms. The body has a coelomic space, a gut running through the length of the body, a closed circulatory system, well-developed nervous system and specialised excretory structures (Brusca and Brusca 1990). There are about 20,000 named species (Edgar 1997), representing only a small proportion of the total living species.

The phylum consists of two main groups, the **Polychaeta** (Bristle Worms) and the **Clitellata** (earthworms and their allies and leeches). The relationship between these two groups is currently in a state of flux, and although the Annelida is considered monophyletic, Rouse and Fauchald (1997) found weak support for the group.

Polychaeta (Bristle Worms)

Polychaetes (commonly known as “bristle worms”, “sand worms”, “tube worms” etc) are multisegmented worms typically with segmental parapodia and chaetae (bristles) arranged in distinct bundles. They exhibit a wide range of sizes and species vary from less than 1 mm for some interstitial species to over 3 metres for some Australian beach worms (Family Onuphidae). They also exhibit a great range of morphological and functional diversity, as well as reproductive strategies (Beesley et al. 2000; Glasby et al. 2000a; Rouse and Pleijel 2001).

Polychaetes are mostly marine, although a few species have successfully invaded freshwater and fewer still are parasitic or terrestrial (albeit living in moist habitats) (Hutchings 1998a). In the marine environment, they occur from the intertidal zone to abyssal depths of the ocean, as well as in brackish waters, in all types of habitat and substrata. They range from those that inhabit soft sediments, to others that bore into hard substrates such as limestone, corals and mollusc shells, to interstitial and planktonic forms. Some have colonised very specialised habitats such as deep-sea hydrothermal vents. Many species produce a tube either of cemented mud or sand grains and some (serpulids) secrete a calcareous tube. Others move through the sediment without a permanent tube, just with a fine mucous sheath. Some live individually but many are gregarious, forming dense colonies (e.g., Sydney "coral", *Galeolaria caespitosa*, and some of the sabellariids). Polychaetes exhibit a wide range of feeding strategies, including carnivory, herbivory, omnivory, mud swallowing, filter feeding and surface deposit feeding (Glasby et al. 2000a).

In soft substrates, they often dominate the benthic community, both in terms of numbers of individuals and numbers of species, and contribute significantly to the benthic productivity as many species exhibit high rates of turnover (Hutchings 1998a). Some species are able to successfully exploit organically polluted areas; these are often highly opportunistic, with some such as the *Capitella capitata* species complex able to rapidly build up numbers after an area has been depopulated, for instance after an oil spill (Grassle and Grassle 1974; Hutchings 2000b). Others can live in sediments highly contaminated with heavy metals (Ward and Hutchings 1996).

The Polychaetes are a diverse group, with about 13,000 species belonging to over 80 families having been described worldwide, although only 8000 of these are considered as 'reasonable (Hutchings and Fauchald 2000). Snelgrove et al. (1997) suggested that polychaete biodiversity may be as high as 25 000 to 30 000 species. Although Rouse and Fauchald (1997) have recently designated a group of clades to which they have assigned families, it is common practise to refer to individual polychaete families rather than clades (Rouse and Fauchald 1997; Rouse and Pleijel 2001, purposely avoid the use of ranks). The majority of families occur worldwide and most are represented in Australian waters. The group was check listed by Day and Hutchings (1979) and an ABRS catalogue of the Australian fauna has recently been completed with over 1000 known species, including new species still to be formally named (Hutchings and Johnson 2001). Certainly many more remain to be described, especially from more remote areas and deeper waters. In addition, there are an estimated 150 or so described species occurring in Australian, Antarctic and sub-Antarctic territorial waters (C. Glasby pers. comm.).

Chisholm and Kelley (2001) speculated that the activities of eunicid polychaetes might provide the initial firm substrate for reef-building corals to grow by accumulating pieces of hard substrate and gluing them together.

Pogonophorans are a deep-water (>200m) group of around 145 species that lack a gut and live in long chitinous tubes. They have been the subject of taxonomic and phylogenetic debate (Brusca and Brusca 1990) since their discovery in 1900, having previously being treated as a separate phylum (Pogonophora), but are now considered members of the Polychaeta, and are referred to as the siboglinids. One group, known as the Vestimentifera, are found associated with deep-sea hydrothermal vents. 146 species of pogonophorans are known worldwide from all oceans, but they are not well known from Australian waters; only a few collections of unnamed species of pogonophorans have been made, and as yet no members of Monilifera or Vestimentifera have been recorded (Southward 2000) from Australia.

Our knowledge on the biogeography of Australian polychaetes is restricted to recently revised families, and several patterns of distribution appear to be present. A few species occur throughout Australia, others have either a northern or southern distribution, and some have very restricted distributions (Glasby et al. 2000b). While many polychaete genera have widespread distributions, individual species often have more restricted ranges. Glasby and Alvarez (1999) recently examined the distribution of selected polychaete families in the Southern Hemisphere and found some evidence of Gondwanan distributions, with the highest level of species endemism – 67% – occurring in southern Australia, a similar pattern to that seen in molluscs and crustaceans (e.g., Poore 1995a).

In general, as for many other marine invertebrate groups, Australian polychaetes are much less known than the relatively well-studied faunas of Europe and North America. In Australia, about half a dozen families are well-known taxonomically, with the remaining 45-50 poorly known. There are major gaps in our knowledge of deep-water forms, meiobenthic forms, and the fauna of parts of northern Australia and the sub-Antarctic and Antarctic regions (C. Glasby pers. comm.). The recent Fauna of Australia volume on Polychaetes and Allies (Beesley et al. 2000) highlights both our knowledge and gaps in the systematics and ecology of the Australian polychaete fauna. It also lists the literature detailing the extensive more recent studies on Australian polychaetes.

Clitellata (Earthworms and Leeches)

The Clitellata consist of two groups usually treated as separate classes within the Annelida – the Hirudinea (leeches) and the **Oligochaeta** ("earthworms"), both with many marine taxa. The clitellates have the first segment similar in structure to the rest of the body, with segments lacking parapodia (although chaetae are often present) and the peristomium forming a complete ring.

Oligochaeta (Earthworms etc.)

These worms have elongate cylindrical bodies and differ from polychaetes in lacking appendages (although they have minute chaetae) and a distinct head. While they are best known because of the diverse earthworm fauna, there are also many small freshwater and marine taxa. Marine oligochaetes are particularly abundant in shallow, nutrient-rich sediments.

Marine oligochaetes have been described from Victoria (Erséus 1990a), Darwin (Coates and Stacey 1997; Erséus 1997; Healy and Coates 1997), and southwestern Australia (Coates 1990; Erséus 1990b; Coates and Stacey 1993; Erséus 1993).

Erséus (1997) suggested that the genera of tubificids found in the tropics and subtropics are widely distributed in the world, and that endemicity occurs at the specific level. He also suggested that the distribution of genera differs around Australia with the genera present being determined by the available habitats (see the above papers for further discussion). While oligochaetes are present in most coastal habitats they are often not collected by routine benthic surveys using 0.5mm sieves or larger. Far smaller mesh size and other sorting techniques are required to adequately sample this fauna.

Hirudinea (Leeches)

Leeches have flattened bodies and a sucker at each end. They lack chaetae or appendages. Leeches live on the land, in freshwater and in the sea, and are parasites on marine vertebrates such as fish and turtles. The marine leech fauna of Australia is poorly known.

Dr Gene Burreson, of the Virginian Institute of Marine Sciences, USA is currently working on Australian marine leeches as part of a world revision and has a PEET grant from NSF, he is also collaborating with Dr Ron Davies of the University of Calgary, Canada in preparing a catalogue of Australian leeches for ABRS.

2.2.17 Kamptozoa (=Entoprocta)

Kamptozoans are tiny, sessile, mainly marine, suspension feeding animals, which bear a superficial resemblance to bryozoans (with which they were originally placed). However, they differ in fundamental ways, such as their adult and larval body plan, and their closest living relatives may be polychaetes. They reproduce asexually by budding, and are therefore usually found as clonal aggregations of separate zooids or as colonies of interconnected zooids. The zooids have a characteristic bending movement, which has given rise to both the scientific name (Greek: *kamptozoa* = bending animals) and the common name (nodding heads).

Except for one freshwater species, all kamptozoans are marine. A few species can tolerate brackish conditions and dwell in bays and harbours. Kamptozoans are known from the low intertidal zone to deep-water (Wasson and Shepherd 1997).

About 150 species have been described worldwide, but kamptozoan diversity probably exceeds 500 species (Nielsen 1989). While they are widespread and are quite abundant in some microhabitats, most of the world's kamptozoans are poorly characterised or not known at all, because most species are tiny and easily overlooked (K. Wasson pers. comm.). Little is known of Australian kamptozoans, with published reports and museum specimens of this group being very scarce (Wasson 1994), and only a few detailed taxonomic investigations (Wasson 1995). There are 18 described species recorded from Australian waters, 14 endemic to the Australian / New Zealand region, plus 17 known undescribed species; there may be 100 or so species as yet undescribed (K. Wasson, pers. comm.). Wasson (1997) provided a key to 10 described species known from southern Australia.

2.2.18 Phoronida (Horseshoe Worms).

Phoronids – or 'horseshoe worms', as they are sometimes called – comprise one of the least diverse and least familiar of the larger-bodied marine phyla. Their closest relatives are the bryozoans and the brachiopods, and the three are sometimes grouped as the lophophorate phyla because they all possess a probably homologous structure called a lophophore. Phoronids are elongated and worm-like in general appearance and lack appendages. They secrete and inhabit a membranous tube, from which the lophophore, used in suspension feeding, normally protrudes (Shepherd 1997a).

The different species of phoronids occur in a wide range of habitats and their tubes may be found embedded in substrate or adhering to hard surfaces such as rock or mollusc shells. Some live commensally. The ability of most to reproduce asexually results in them being clustered in small to large groups. Despite the small size of the phylum, phoronids may be abundant in shallow marine sediments at certain localities (Shepherd 1997a).

Worldwide, there are only about 12 known species in two genera (Brusca and Brusca 1990), seven recorded in Australian waters (Emig and Roldán 1992). The Australian species have been reviewed by Emig et al. (1977) and Emig and Roldán (1992).

2.2.19 Bryozoa (Moss Animals, Sea Mats, Lace Corals)

Bryozoans – known commonly as "moss animals", "sea mats", or (for some forms) "lace corals" – are aquatic, sessile, colonial animals most frequently found attached to hard substrates. Colonies of bryozoans are generally small and inconspicuous, but morphologically are very variable (Bock 1982). They are comprised of a few to many millions of individual zooids, and colonies range from millimetres to metres in size, although the individual zooids are rarely larger than a millimetre (Waggoner and Collins 1999). The majority of living bryozoans are encrusting, forming flat sheets that spread out over the substrate, but others grow upwards into the water column and may be massive, sheet-like, branchlike or reticulated. They are often mistaken for hydroids, corals, or even seaweeds (Waggoner and Collins 1999).

The majority of bryozoans are marine, although there is one group of freshwater forms. They are typically attached to hard substrates; rocks, shells, ship's hulls, pilings, outer surfaces of other animals or algal fronds (Theroux and Wigley 1998; Waggoner and Collins 1999), and some form colonies on sediment. Most are sessile and immobile, but a few colonies are able to creep about, and a few species of non-colonial bryozoans live and move about in the spaces between sand grains (Waggoner and Collins 1999). One species forms floating golf-ball sized colonies in the Antarctic (Peck et al. 1995). All species are filter feeders, so they thrive in waters rich in microplankton (Bock 1982). They feed on small microorganisms, including diatoms and other unicellular algae, trapped by the ciliated feeding tentacles, or lophophore. Bryozoans form the prey of grazing organisms such as sea urchins, molluscs and fish, and are subject to competition and overgrowth from sponges, algae, and tunicates (Waggoner and Collins 1999).

Bryozoans are important in the fossil record, sometimes being so abundant that they form limestone rocks (Bock 1999; Waggoner and Collins 1999). Recent species have been found at depths of up to 8200 metres (Waggoner and Collins 1999), although most inhabit much shallow water, and are common components of the littoral and sublittoral benthos (Bock 1982). There are at least 5000 described living species, with several times that number of fossil species (Waggoner and Collins 1999). In Australia, bryozoans are numerically and taxonomically abundant (Campbell in press), comprising the dominant organisms in some marine benthic (and fossil) assemblages (Gordon 1999a). Australia and New Zealand together have the richest bryofauna in the world, closely approached by that in the Philippine-Indonesian area (Gordon 1999a), but it is very poorly known (P. Hayward pers. comm.). There are over 900 described and known undescribed species in Australian waters, and more than 200 from Antarctica and the sub-Antarctic islands (D. Gordon pers. comm.). However, since the continental shelves – especially those of

Western Australia – have yet to be fully explored for bryozoans, these figures must be regarded as conservative. The Australian deep-sea bryofauna is likewise poorly known (Gordon 1999a). The recent fauna is possibly 60-70% described (Gordon 1999a), with at least 500 (D. Gordon pers. comm.), and perhaps as many as 2,000-3,000 (P. Cook, P. Bock pers. comm.) undescribed species for all Australian territorial waters. The southern fauna is the most distinctive, with a high proportion of endemic species and geologically ancient taxa (P. Hayward pers. comm.). Perhaps 50% of all species, plus many genera and some families, are Australian endemics (Gordon 1999a; D. Gordon, P. Cook, P. Bock pers. comm.). There are probably many, as yet unsampled, areas similar to Port Phillip Heads, which has a more diverse fauna than that of Europe (P. Cook, P. Bock pers. comm.).

Bryozoans are economically important as fouling organisms on man-made structures such as boats and water pipes (Bock 1982). Some species, particularly those resistant to antifouling paints, are important components of the fouling population and provide a suitable settlement surface for other types of fouling organisms. Some bryozoans can become pests in commercial shellfish production (Bock 1982). Pollard and Hutchings (1990) listed five non-indigenous bryozoan species introduced to Australia, most probably through hull fouling. On the other hand, bryozoans are an important source of marine natural products, producing a remarkable variety of chemical compounds, some of which may find uses in medicine. For instance, one compound produced by a common marine bryozoan (the drug bryostatin 1) is currently being assayed as an anti-cancer drug (Waggoner and Collins 1999). Bryozoans are also ecologically important as providers of habitat and food for other species.

While many species of Australian bryozoans appear to have a cosmopolitan distribution, which could be explained in certain cases as due to dispersal by shipping, further revision may prove that some of the local records of overseas species have been based on incorrect identifications (Bock 1982).

2.2.20 Brachiopoda (Lamp Shells)

Commonly called lamp or lantern shells, the brachiopods are solitary, benthic, sessile or burrowing marine organisms with a bivalve shell and pedicle (stalk). On account of their bivalve shell they may superficially resemble bivalve molluscs, but are totally different in their anatomy and are not related to molluscs. While they are now uncommon and often considered to be nearing extinction, they have a long and rich fossil history, having been extremely abundant and diverse during the Palaeozoic (Collins 1995).

Although they are known from all ocean depths, brachiopods are most abundant on the continental shelf. Since substrate appears to be the principal factor governing distribution, the range of latitude, longitude and depth occupied by any one taxon may be extensive. Brachiopods usually live attached to the substrate by a pedicle, or by the cementation of one valve to the substrate itself, or burrowing with an elongated pedicle sheathed in sand (Brusca and Brusca 1990). They are filter-feeders, using their ciliated lophophore to capture fine phytoplankton (e.g., diatoms), and dissolved and colloidal material.

The phylum is divided into two classes – the Articulata and the Inarticulata – on the basis of how the two valves of the shell are joined (by a tooth-and-socket hinge, or simply by muscles, respectively). Most inarticulate species are widely distributed and cosmopolitan, whereas articulate genera are locally more common but more restricted in distribution. These differences appear to be related to the reproductive strategies of each group; inarticulate larvae being pelagic with a probable life span of up to six weeks, whereas articulate larvae are brooded in the lophophore and settle and metamorphose within hours of their release, mostly on or close to members of the parental population (Richardson 1997).

There are approximately 330 living brachiopod species, which represent only a small fraction of the more than 12,000 species that were once extant (Brusca and Brusca 1990). The Australian fauna is, with the possible exception of Japan, more diverse than that in any other part of the world, containing about 38 species including representatives of all living orders and most families (Richardson 1997). While brachiopods are generally described as being a sedentary group living predominantly on hard surfaces, in Australia only nine species are exclusively sedentary and restricted to hard substrates. The majority (29 species) are found on soft sediments of the continental shelf, and have the capacity to move in response to changes in levels of sediment (Richardson 1997).

The Australian brachiopod fauna is poorly known and has been little studied, most work having been undertaken by Dr J. R. Richardson, who concentrated on the southeastern Australian fauna, from the 1970s to the early 1990s (R. Craig, pers. comm.). The known taxa are now catalogued and this information will be available on the ABIF website in late 2002. They have no commercial value, are not targeted by shell collectors, and, with the exception of *Lingula*, are considered inedible (Richardson 1997).

2.2.21 Mollusca (Snails, Slugs, Limpets, Squid, Octopuses, Cockles, Oysters, Clams, Chitons etc.)

Molluscs are soft-bodied animals without an internal skeleton, with no standard shape and most have a hard, protective shell. The general structure consists of four body parts; a head (carrying the mouth and sense organs), a muscular foot (used for locomotion or attachment), a visceral mass (containing the major internal organs) and a mantle or pallium that secretes the shell (when present). They are the second largest phylum of animals (after the arthropods) and are economically one of the most important groups of invertebrates. The Australian fauna has recently been extensively reviewed (Beesley et al. 1998) so we only provide a very brief outline of the group.

The molluscs show great structural and ecological variability and are found in a wide range of habitats. They may burrow in soft substrates (e.g., many bivalves, some snails), attach themselves to hard surfaces (e.g., mussels, oysters), clamp to rocks (e.g., limpets, chitons), crawl or glide (most snails, slugs), or swim, some by jet propulsion (e.g., squids). They range from herbivorous to carnivorous and from microphagous to active predators to parasites. Most (with the notable exception of the bivalves) feed using a row of teeth attached to a moveable ribbon - the radula.

The economic significance of molluscs, including their role as pests and disease carriers and the commercially exploited species such as abalones, scallops, pearly oysters, mussels, squid and abalone, are reviewed in Boray and Munro (Boray and Munro 1998).

The **Mollusca** are generally classified into seven or eight classes. Of these, the **Bivalvia** (clams, oysters, mussels) and **Gastropoda** (snails, limpets, slugs) are also found in freshwater habitats, while only the Gastropoda have colonised the land. Other classes are the **Cephalopoda** (squids, octopuses), **Scaphopoda** (tusk shells), Polyplacophora (chitons), **Monoplacophora** and **Aplacophora** (spicule worms).

Approximately 120,000 species of molluscs have been described to date, and the Australian marine molluscan fauna comprises around 10,000 described species with perhaps the same number yet to be described. Most mollusc families are represented in the Australian fauna. While the diverse, large shelled groups are relatively well known because their shells can be collected, a great many smaller species are poorly known. Around two-thirds of described species are tropical (with less than 10% endemic), and one-third temperate. Approximately 10% of the larger species are endemic to either the east or west coast (F. Wells pers. comm.), and a large proportion (about 95%) of temperate Australian species are endemic. The shallow seas of southern Australia support a unique regional molluscan fauna having its origin largely in the faunas of the southern basins formed in the Tertiary after the separation of Australia from Antarctica (Wilson and Allen 1987; Ludbrook and Gowlett-Holmes 1989a).

Some endemic genera are relicts of once widespread Tethyan groups that have become extinct elsewhere; these include the single representatives of the endemic families Trigoniidae, Campanilidae and Diastomatidae, and are most conspicuous in the southwest. Other endemic genera have lineages in the southeast dating back to the early Tertiary suggesting that they are relicts of the ancient Palaeoaustral fauna (Wilson and Allen 1987; Poore 1995a).

A recent major, comprehensive family-level treatment of the Australian Mollusca can be found in Beesley et al. (1998), Volume 5A in the Fauna of Australia series produced by ABRS. In addition, several groups of marine molluscs are available on the ABIF web site and a major program is currently underway to checklist all of the Indo-West Pacific molluscan taxa (Middelfart 2000).

Aplacophora (Spicule "Worms")

The Class Aplacophora contains worm-like marine molluscs without a shell and covered by a cuticle with numerous calcareous spicules. There are two distinct groups, often treated as separate classes, Neomeniomorpha and Chaetodermomorpha (=Solenogastres and Caudofoveata, respectively). Most species are small, often less than 5 mm in length, but some reach 300 mm. They are mostly found on the continental shelf and in the deep-sea.

These worm-like molluscs have been poorly collected in Australia, but even so, 11 families, 17 genera and 33 species have been described from the continental shelf of Australia and Macquarie Island (Scheltema 1998). There is undoubtedly a large undescribed fauna, with most of the large collections from the upper continental shelf off southeastern Australia having not yet been identified (Scheltema 1998). Aplacophorans have also been described from the Australian Antarctic Territory, but none have been reported from Lord Howe, Cocos (Keeling), Christmas or Norfolk Islands, although a few species are known from the vicinity of Heard Island (Scheltema 1998). The fauna was recently catalogued (Scheltema 2001). There are no Australian workers on this group.

Polyplacophora (Chitons)

Polyplacophora, the chitons or coat-of-mail shells, have a flattened body and a broad foot, and a shell of (usually) eight articulated calcareous plates or valves. These plates allow the animal to either fasten itself tightly to an irregular surface or roll itself into a ball if detached, and like limpets, they are able to withstand considerable force before they can be dislodged. Chitons are usually found attached to hard surfaces such as rocks or seaweed blades, particularly in intertidal or shallow coastal waters, although some groups are found in deeper waters. Most chitons are generalist grazers (herbivores and omnivores), feeding on algae or small encrusting animals. A review of the group and the Australian families can be found in Beesley et al. (1998) and there is a catalogue of the Australian fauna by K. Gowlett-Holmes (2001).

Chitons occur worldwide, but are most diverse in Australasia and along the tropical Pacific coasts of America. There are around 750 living species worldwide, of which at least 180 are known from Australian waters (Kaas et al. 1998). This includes around 15 species known from the Antarctic and sub-Antarctic, and 10-15 from each of Christmas, Cocos (Keeling), Norfolk and Lord Howe Islands (K. Gowlett-Holmes pers. comm.). Almost all temperate species are endemic. Most shallow-water tropical species are also known from Indonesia, New Guinea and neighbouring waters, although those occurring in deep-waters are mostly endemic (K. Gowlett-Holmes pers. comm.).

Tropical chitons are relatively poorly known, particularly the deep-water species, of which there may be as many as 40 undescribed. In general, the biology and ecology of chitons are inadequately known, although from the little work done, Australian chitons appear to display greater diversity in feeding habits than recognised elsewhere, and comprise a much more significant component of rocky reef faunas than in other parts of the world (K. Gowlett-Holmes pers. comm.). Despite the diversity of the Australian fauna, there is only one Australian specialising in this group, although she is not employed to undertake this work.

Monoplacophora

Monoplacophorans are a class of simple, primitive, mostly tiny bilaterally symmetrical molluscs with a simple cap-shaped shell. The few living species are known from the deep-sea and none are yet recorded from Australian waters (a reflection of our poorly sampled our deep-sea fauna), although a few living monoplacophorans have been found in Antarctica and New Zealand (Ponder 1998a).

Gastropoda (Snails, slugs, whelks, limpets)

The gastropods form by far the largest and most diverse class of molluscs, comprising more than half of all mollusc species. They include the sea and land snails and slugs, limpets, abalones, cowries, whelks and cones. Most have a single coiled shell but in some the shell is limpet-like, rudimentary, or lost. Gastropods use a wide variety of feeding methods and occur in many types of marine, freshwater and terrestrial habitats. While the majority of marine gastropods are benthic, a few groups have adopted a wholly planktonic mode of life, two groups being swimming carnivores. A comprehensive overview of Australian gastropods and each of the families is provided in Beesley et al. (1998).

Gastropoda have traditionally been divided into three subclasses, the Prosobranchia, Pulmonata and Opisthobranchia, but this classification has now been substantially revised (Ponder and Lindberg 1997) with the following groups (rank unassigned) currently recognised: Patellogastropoda (true limpets); Vetigastropoda (top shells, abalones, turban shells, keyhole limpets, slit shells etc.), Neritopsina (nerites), Caenogastropoda (many of the marine snails, including periwinkles, whelks, cowries, cones, balers, etc.) and Heterobranchia (land snails and slugs, seaslugs etc.). Patellogastropods and vetigastropods are all marine, while the neritopsines and caenogastropods are mostly marine, including a few freshwater and terrestrial groups. These groups comprise most of the "prosobranch" gastropods, although a few "prosobranchs" are now included in the Heterobranchia, which also encompasses the opisthobranchs and pulmonates. In all, there are around 6300 described species of marine gastropods in Australian waters and perhaps another 35 – 40% undescribed, with most of the southern fauna being endemic. There is one full-time taxonomist working on marine gastropods (a specialist in Opisthobranchia) in Australia, although three other scientists employed in museums carry out some taxonomic studies on marine gastropods.

A small, deep-water group of limpets (Cocculinoidea), comprising at least two families, are of uncertain affinities, as are a number of families found mainly associated with hot vent habitats.

Patellogastropoda (True Limpets)

Australia has a rich fauna of true limpets, with typically several taxa present in most intertidal areas, with the highest diversity on temperate rocky shores. In all, there are about 40 species in five families in Australian waters, including a couple known only from the sub-Antarctic region (Lindberg 1998). The group has an important functional role as grazers and scrapers on intertidal rocky shores. Many of the Australian species are illustrated and described in Wilson (1993).

Vetigastropoda (Top Shells, Abalone, Keyhole Limpets, Turban Shells, etc.)

Vetigastropods are diverse and conspicuous members of the intertidal and sublittoral fauna around Australia, being dominated by families such as Trochidae (top shells, Trochus), Turbinidae (Turban shells, cat's eyes) and Fissurellidae (slit and keyhole limpets), the former two families containing commercially exploited *Trochus* and *Turbo* respectively. Another vetigastropod family, the Haliotidae, is commercially valuable, comprising the abalones, of which there are around 25 Australian species, all but a few non-commercial tropical species being endemics. There are also eight other families of less well-known and/or inconspicuous snails. Vetigastropods are exclusively marine and most are grazers specialised to feed on a wide range of substrates including detritus, algae, and colonial animals. A few taxa, like *Bankivia*, a colourful snail abundant in the surf zone off temperate ocean beaches, are filter feeders. Some taxa, such as some trochids, are very abundant in the rocky intertidal and there is a diverse but very poorly known fauna of small and minute taxa. Many of the larger Australian species are illustrated and described in Wilson (1993). There is no Australian scientist specialising in the taxonomy of vetigastropods.

Neritopsina (Nerites)

This group is comprised of six families in Australia (Beesley et al. 1998), two terrestrial. Members of the major family, the Neritidae, are conspicuous on most intertidal shores in temperate and tropical Australia and other tropical areas. They are mostly marine, feeding on algae and detritus, there being only about 24 Australian species (Scott and Kenny 1998), most illustrated and described in Wilson (1993). There is no Australian scientist specialising in the taxonomy of neritopsines.

Caenogastropoda (Winkles, Whelks, Cowries, Cones, Balers etc.)

This is the largest group of marine snails. There are 88 families recognised in Australia (see Beesley et al. 1998 for details) in two orders, most of them marine, and these show a wide range of shell morphologies (coiled shells, worm-like shells, limpets, a few shell-less), feeding strategies (grazing, predation, parasitism, filter feeding) and habits. Most are benthic crawlers, some burrow, others are sessile, and a few are permanently attached to the substrate. One group (Heteropoda) are highly modified swimmers and active carnivores, and are permanent members of the zooplankton. Another group (Janthinidae) are pelagic drifters that feed on siphonophores. The diversity of this group is considerable and while accurate figures are not available there are several thousand species found in Australia. While this group includes the largest snails (balers and some large whelks), the majority are small or even minute (to about 1mm in size) and these taxa are generally very diverse but poorly known. The caenogastropods include cowries, cones, balers (volutes), mitres and several other groups of interest to shell collectors. Many of the larger Australian species are illustrated and described in Wilson (1993; Wilson 1994).

The group includes whelks, winkles, and many other common snails found in the intertidal and shallow sublittoral and a huge, very poorly known fauna of small to minute species. There are currently only two Australian scientists undertaking some taxonomic studies on marine caenogastropods in Australia, although one is primarily working with freshwater taxa and the other is mainly doing ecological studies.

Heterobranchia (Seaslugs, Bubble Shells, Air-breathing Limpets etc.)

This morphologically very diverse group includes forms with coiled or limpet-like shells to a wide variety of shell-less slugs. They range in size from less than 1mm to several tens of cms in length. The most primitive members of the group were previously included in the "Prosobranchia" and contain a diverse group (12 families) of marine taxa mainly with coiled shells, some of very small size and very poorly known.

The seaslugs (including nudibranchs) and bubble shells comprise the **Opisthobranchia**, with most being benthic although two groups ("pteropods") and two species of seaslugs are pelagic. While a few opisthobranchs are detritus feeders or herbivores, most are carnivorous, preying upon sessile organisms such as sponges, hydroids, actiniarians, and bryozoans. They are found in almost every marine habitat. 76 families in nine orders are recognised in Australia (Beesley et al. 1998). Estimates of the opisthobranch fauna of Australia range as high as 5000 species (Rudman and Willan 1998), with temperate southern Australia having more than 300 species with probably at least another 100 species awaiting discovery (R. Burn pers. comm.). However, assiduous collecting has been undertaken in only a few areas and many of the smaller taxa in particular are very poorly known. About 50 species have been recorded from the Christmas and Cocos Islands (C. Bryce pers. comm.), but the fauna is undoubtedly much larger. Guides to the species of Australian opisthobranchs include Willan and Coleman (1984) and (for Western Australia) Wells and Bryce (1993). There are currently two museum-employed specialists on opisthobranchs in Australia and a third scientist also has an interest in the group. A considerable amount of information is available on opisthobranchs on the Seaslug Forum[8].

The **Pulmonata** comprise marine groups but the majority are freshwater or terrestrial. The marine members comprise six families in three orders, including the mangrove onchidiid slugs (Onchidiidae), air breathing limpets (Siphonariidae and Trimusculidae) abundant in the rocky intertidal, and the estuarine ear shells (Ellobiidae) and mud snails (Amphibolidae), found abundantly associated with mangroves and saltmarsh habitats. There are currently no Australian scientists undertaking research on the systematics of marine pulmonates. A calalogue of the Pulmonata will be available through the ABIF website in late 2002

Scaphopoda (Tusk Shells)

Commonly known as tusk or tooth shells, these are a small class of bilaterally symmetrical, marine burrowing molluscs with a long curved, tubular, tapering shell open

[8] http://www.seaslugforum.net/

at both ends. They inhabit all types of unconsolidated sediment from the shallow sublittoral to abyssal depths. All are benthic micro-carnivores, feeding on detritus or microscopic animals such as foraminiferans. Worldwide there are 11 families and 500-600 living species; Australia has more than 150 named species (fossil and extant) from eight families (Palmer and Steiner 1998), of which 46 were named as recently as 1997 (Lamprell and Healy 1998b). The Australian fauna has recently been catalogued by K. Lamprell and J. Healy (2001).

Bivalvia (Oysters, Clams, Mussels, Cockles etc.)

The bivalves are aquatic, laterally compressed, typically bilaterally symmetrical molluscs with a shell composed of two hinged valves. They are an economically important group including the scallops, oysters, cockles, mussels and clams – all major sources of food for humans – as well as less obvious forms such as the destructive shipworms. An overview of the bivalves and a review of the Australian families is provided by Beesley et al. (1998) and many of the Australian species are described and illustrated in Lamprell and Whitehead (1992) and Lamprell and Healy (1998a).

A few bivalve families are found only in freshwater, but most are marine. They are nearly all sedentary, and many burrow in sediment; some, like the shipworms (Family Teredinidae - world fauna reviewed by Turner 1966) can bore into wood, while others (e.g., Pholadidae) bore into rock. Many others are epifaunal, living attached by proteinaceous threads (byssus) or cement to the surface of stones or other organisms. Most are ciliary suspension feeders, although some are deposit feeders; a few are microcarnivores and some others rely partially or wholly on symbiotic bacteria in the gills for their nutrition.

Approximately 1640 bivalve species have been described from Australia, with about 1000 known from southern Australia. Another 500 or so may still be undescribed. While the majority of the 84 families known from Australia are cosmopolitan, there is high (90%+) endemism for southern Australia at the specific level. While the southern Australian fauna is relatively well known, the tropical bivalve fauna is still poorly understood. There is only one Australian taxonomist employed in a museum that is publishing on marine bivalves. Some additional work is being carried out through a couple of small grant-funded projects.

Although the phylogeny of bivalves is still in a state of flux, five subclasses are recognised in Beesley et al. (1998). These are Protobranchia (Nut Shells, Date Shells etc.), comprising two orders and six families; Pteromorphia (Ark Shells, Oysters, Pearl Oysters, Mussels, Scallops etc.) comprising five orders and 23 families; Palaeoheterodonta (Brooch Shells; Freshwater Mussels) comprised of two orders and two families; Heterodonta (Clams, Cockles, Shipworms, etc.) made up of two orders and 42 families, the largest group; and Anomalodesmata (Lantern Shells, etc.) with one order and 11 families.

Cephalopoda (Squids, Cuttlefish, Octopuses, Nautilus)

The Cephalopoda are a group of highly organised, exclusively marine molluscs that include the familiar squids, octopus, cuttlefish and nautilus. With the exception of the Pearly Nautilus (*Nautilus*), which has a heavily built chambered external shell, cephalopods either have a greatly reduced internal shell (squid and cuttlefish) or no shell at all (octopuses). The head bears a pair of well-developed eyes (which, except for *Nautilus*, show development analogous with that of the vertebrates) and one or two circles of tentacles (numbering 8 or10 in most forms) around the mouth. They are adapted for swimming (although some are secondarily sedentary), with a foot modified as a funnel for jet propulsion which is used primarily as an escape response. Most cephalopods are active predators, capturing prey with their arms and tentacles, injecting it with salivary toxin, and tearing it apart with beak-like jaws. They also have much more highly developed nervous, circulatory and reproductive systems than the other molluscs, and their large and complex brain and ability to perceive and respond quickly to cues make them the most highly developed and intelligent of all invertebrates. Cephalopods vary in size from small planktonic forms to giant squid such as *Architeuthis* that reaches a total length of 18 m (Zeidler and Norris 1989).

The Cephalopoda consists of about 130 known living genera and about 650 species, divided into four orders. A review of the group, and of the 34 Australian families, is provided in Beesley et al. (1998). Relatively little is known about the Australian fauna and the systematics of some groups, such as the octopuses, is confused. Except for a few commercially important species such as *Nototodarus*, virtually nothing is known about their biology. As they often occur in large numbers, they are a potential food resource and deserve more attention. Many more species are likely to be found, particularly in the deeper waters of the continental shelf and slope (Zeidler and Norris 1989). At present few researchers are studying cephalopods in Australia; only two (M. Norman - octopods, and A. Reid - sepiids) are actively working on cephalopod taxonomy, and neither are currently supported by full time research funding (A. Reid pers. comm.). There is a recent catalogue of Australian cephalopods by C.C. Lu (2001) that updates the previous checklist of Lu and Phillips (1985). There is also a popular guide to the cephalopods of Australia (Norman and Reid 2000).

Nautilida (Pearly Nautilus)

This group is well represented in the fossil record but only five living species, belonging to two genera, *Nautilus* and *Allonautilus*, are known and, for this reason, they are considered “living fossils”. Their most distinctive feature is a chambered external nacreous shell. Living species are confined to the tropical Indo-Pacific, occurring from the Philippines to New Caledonia, Samoa and northern Australia. Live populations of two species have been documented from the Great Barrier Reef and the Coral Sea, with one of these also having been found off Western Australia; drift shells of a third species have also been found on the GBR. The presence of drift shells in other parts of Australia suggests that *Nautilus* may be widespread on Australian deep reef slopes (Saunders 1998). The natural history of *Nautilus* is discussed in Ward (1987).

Sepioidea (Cuttlefish, Dumpling Squids)

Includes the cuttlefish (Sepiidae) and dumpling squids (Sepiadariidae and Sepiolidae). Cuttlefish inhabit the continental shelf and upper slope to a maximum depth of about 500 metres. They are very common in shallow inshore waters, and their internal shells (cuttlebones) are familiar objects on beaches. Cuttlefish are primarily bottom-dwellers over a range of habitats including rocky, sandy and muddy bottoms to seagrass, seaweed and coral reefs. The total world fauna of cuttlefish is currently 112 described species. Over a third of the world fauna – at least 40 species – have been described from Australian waters (A. Reid pers. comm.), giving Australia the largest fauna of cuttlefish in the world. None of these species are cosmopolitan, and most of those occurring on the southern, eastern and western margins of the continent are endemic to these regions. No cuttlefish have been described from the Antarctic or Sub-Antarctic and none are expected to be found there (A. Reid pers. comm.). A synopsis of the Sepiidae in Australian waters was published by Lu (1998). The three other families in this group (including the two named above) known from Australia contain a small number of small-sized species.

Teuthoidea (Squids)

Comprises the shallow water (Suborder Myopsida) and oceanic (Suborder Oegopsida) squids. Shallow water squid usually come into inshore waters to breed in vast, closely packed schools and at such times are fished commercially. The oceanic squid include a wide variety of species from very small to the largest known, and occur from the surface to great depths. Australian records are relatively scant and it is only in recent years, with an increasing interest in offshore and deep-sea fisheries, that more specimens have been obtained. The most recent species checklist for most families is Lu and Phillips (1985).

Octopoda (Octopuses)

The octopuses are probably the most specialised of all the cephalopods and are considered the most neurologically sophisticated. They have lost all trace of a shell, although the Paper Nautilus (*Argonauta*) produces a secondary shell as a brood chamber. Most octopods are benthic and shallow-water, but a number are specialised for a pelagic habit while others are deep-sea forms. Most are solitary, usually hiding under rocks or deep in rock crevices during the day and coming out at night to feed. Generally they are carnivorous. Knowledge about Australian octopods was scant but recent work has shown the existence of a diverse fauna with many endemic species (e.g., Stranks 1988; Norman 1992). The largest family, Octopodidae, includes around 29 species described from Australian waters and eight from the Antarctic and sub-Antarctic region, although there are various undescribed species (M. Norman pers. comm.). The other octopod families are small with few described species (Lu 1998). The mid- to deep-water fauna has been barely surveyed or sampled, and though many species are harvested as bycatch, and some specifically targeted, there is negligible biological information available for most species (M. Norman pers. comm.).

2.2.22 Arthropoda (Jointed Limbed Animals - Crustaceans, Insects, Spiders etc.)

The arthropods constitute by far the largest and most abundant animal phylum, with around a million described species and several million more (especially among the insects) yet to be named. They have colonised virtually every habitat on land, in freshwater and the oceans, although only the crustaceans are well represented (i.e. diverse, numerous and ecologically important) in the sea. Arthropods share certain characteristics such as a segmented body, each segment bearing a pair of jointed appendages, and a hardened outer skeleton (cuticle) that is generally shed during successive moults to allow growth. The major divisions of living arthropods, usually given subphylum status, are **Cheliceriformes** (comprising the Chelicerata – horseshoe crabs, spiders, scorpions, mites, ticks etc.; and the **Pycnogonida** – "sea spiders"), **Myriapoda** (centipedes, millipedes and symphylans); Hexapoda (insects and their flightless allies), and **Crustacea** (crabs, lobsters, shrimps, as well as many diverse, generally small-sized groups such as ostracods, copepods and amphipods).

Cheliceriformes (Horseshoe Crabs, Spiders, Mites, Ticks, Sea Spiders etc.)
Chelicerata (Horseshoe Crabs, Spiders, Mites, Ticks)

The only living fully marine group in this class are the "horseshoe crabs" (Subclass Merostomata, Order Xiphosura), of which there are five described living species (Brusca and Brusca 1990), but with an extensive fossil history. None are known from Australian waters.

There are a few spiders found on Australian shores - two species of the Anyphaenidae (both southern Australia/Tasmania) and two members of the Desidae – an undescribed species from tropical Australia and one from Tasmania and Victoria (M. Gray pers. comm.).

The mites (Order Acarina) are the largest group of chelicerates and are mostly terrestrial. One family, Halacaridae, has many marine taxa with faunas from southwestern Australia (Bartsch 1993a, 1993b, 1993c) and Queensland (Bartsch 2000; Otto 2001b; a and references therein). Some members of the family Oribatidae, the largest terrestrial family, are also found in intertidal or shallow marine habitats (Edgar 1997).

Pycnogonida (Sea Spiders)

The pycnogonids are a relatively small and little-known class of marine arthropods which, though not closely related to the true spiders, are commonly referred to as "sea-spiders" due to their spider-like appearance. Pycnogonids have extremely reduced bodies in which the abdomen has almost disappeared, while the legs are long and clawed. They are found in all oceans but are more abundant in temperate waters and range from the intertidal zone down to abyssal depths. Although some do swim, most are bottom-dwellers. Intertidally they live among algal mats or beneath boulders but are rarely seen due to their typically small size and cryptic colouring (Staples 1997). Pycnogonids are predators on a variety of invertebrates including polychaetes, anemones, bryozoans, sponges and hydroids (Staples 1997). There are about 600 known species (Theroux and Wigley 1998), including 41 shallow-water species recognised from southern Australian waters, but many others remain to be described (Staples 1997). The only revision of Australian species is very dated (Clark 1963).

Uniramia (Insects, Centipedes, Millipedes)

Millipedes are found in the supralittoral zone but are probably members of the truly terrestrial fauna whereas some centipedes are restricted to this habitat. However, insects and centipedes have specialised members of this community.

Chilopoda (Centipedes)

A few centipedes are found exclusively in the supralittoral and upper littoral zone in Australia and elsewhere (e.g., Jones 1998).

Insecta

In contrast to their extraordinary abundance and diversity on land, the insects are poorly represented in the marine environment. The only fully marine insects are tropical, these being the sea-skaters (*Halobates* spp.) and the limnichid beetle (*Hyphalus*) (Cheng 1976; Matthews and Queale 1997). The sea skaters (or “coral treaders”; *Hermatobates*) live on the surface of the water. The majority are found on coastal waters, but a few live on open oceans; all of the latter occur in the Pacific with three species from the Australian region (Anderson and Weir 2000). In the temperate waters of southern Australia there are no insects which complete their life-cycle entirely in marine waters (other than tide pools) or on continuously submerged substrates, with the exception of the marine caddisfly *Philanius plebeius* on the NSW south coast (Matthews and Queale 1997).

However, a wide variety of insects live in semi-marine environments (particularly the supralittoral zone), and other species complete only part of their life cycle in marine environments (e.g., rock pools). On one hand, some aquatic insects (or those with aquatic larvae) can tolerate varying salinities, and encroach on estuarine, marsh or mangrove waters or tide pools. However, these have generally not become adapted to living in the sea itself; even mangrove mosquitos, for instance, cannot survive in waters inundated regularly by the tides, occurring only in still waters. On the other hand, terrestrial insects, mostly beetles and higher Diptera, have encroached by way of the dunes and other open areas behind beaches and are often found associated with stranded seaweed and dead animals. These include both specialist littoral insects, i.e. species which either habitually complete their life cycle in the littoral zone or which find a major proportion of their food there as larvae or adults; and those which occur there casually as part of a wider foraging range (Matthews and Queale 1997). While low in diversity compared with terrestrial habitats, the insect fauna of these shorelines can play an important ecological role. For instance, the larvae of various Diptera (flies) are important in the breakdown of strandline seaweed (D. McAlpine pers. comm.). One of these groups, the kelp flies (Colelopidae), have two species on Macquarie Island (also found in NZ) and another 11 in Australia and Tasmania, with only two found in the tropics (McAlpine 1991).

There is an insect fauna associated with mangroves, although there are few data and probably many undescribed species (Hutchings and Saenger 1987). Detailed studies have been carried out only on termites, mosquitos and biting midges, because of their economic or medical importance, particularly the potential role of the latter two groups in disease transmission. The biting midges (Ceratopogonidae) are very widespread and abundant, with probably well over twenty described mangrove-associated species, seven of them common (Hutchings and Saenger 1987). Most do not require free water, the larvae being part of the substrate infauna exclusively above mid-tide level.

Crustacea

The crustaceans are a large and extremely diverse group of arthropods. They vary a great deal in shape and form, making it difficult to define easily recognised traits common to them all and some have highly modified bodies (e.g., barnacles, parasitic isopods and copepods). They range in size from tiny interstitial and planktonic forms less than a millimetre in length to crabs with leg spans of 4 m. Crustaceans are found at all depths in every marine, brackish and freshwater environment. They also occur on land (primarily Isopoda and Amphipoda) but are considerably less abundant and diverse in terrestrial ecosystems. They are ecologically important in a variety of trophic roles, as grazers, scavengers, predators and as prey for larger organisms. They are also economically very significant, with many species fished commercially for seafood. For instance, the western rock lobster, *Panulirus cygnus*, forms the basis of the largest single-species fishery in Australia, valued at over $260 million per year, while prawns collectively account for a further $400 million (ABARE 1999).

However, detailed biological and ecological knowledge of crustaceans is limited and mainly confined to larger decapods of commercial significance (e.g., some prawns, crabs and rock lobsters). Despite their popularity as seafood, there is little popular literature (Jones and Morgan 1994). Rock lobster fisheries are very valuable and undertaken around much of Australia (e.g., Phillips and Crossland 1990; Pitcher 1990; Brown and Phillips 1994; Hobday et al. 1996; Frusher et al. 1997; Punt and Kennedy 1997; Donohue 1998; Frusher et al. 1998; Montgomery et al. 1998; Anonymous 1999; Chen and Montgomery 1999; Liggins et al. 1999; Donohue 2000; Donohue and Barker 2000). The Western Australian Rock Lobster industry is discussed in more detail in Section 6.10.4. Similarly, prawns are a very important industry in Australia (e.g., Wallner and Phillips 1988; Turnbull 1990; Wang and Die 1996; Penn et al. 1997).

While warmer waters have more species of decapod crustaceans (crabs, hermit crabs, lobsters, shrimps, prawns etc.), many peracarids (amphipods, isopods and others) are more diverse in southern Australia (Poore 1995a). A popular guide to Australian crustaceans is provided by Jones and Morgan (1994) and these authors (Jones and Morgan 1993) also provide a checklist of the crustaceans of Rottnest Island.

J. Lowry is co-ordinating the relevant specialists around the world to develop interactive keys which are being placed on Crustacea Net[9], a web-based resource.

Most workers recognise five major and two smaller classes (e.g., Martin and Davis 2001), Remipedia and Cephalocarida, as well as the **Branchiopoda** (many are in freshwater), the **Maxillopoda** (barnacles, copepods etc.), the **Ostracoda** and the enormously diverse **Malacostraca**, which contains the majority of well-known crustaceans. The following discussion treats only the five larger classes.

There are more than 30,000 described living species of crustaceans, with probably several times that number yet to be described. The Zoological Catalogue of selected groups of the Malacostraca has recently been published (Davie 2002a, 2002b) including the Phyllocarida, Hoplocarida, Eucarida and the Decapoda- Anomura.

Branchiopoda

Branchiopods are mostly small, freshwater animals, although a few may inhabit hypersaline lakes or marine lagoons. The order Anostraca (‘fairy shrimps’) has some marine members (Brusca and Brusca 1990) and the few marine taxa of Cladocera (‘water fleas’) are listed by Smirnov and Timms (1983).

Maxillopoda

The maxillopods are a variable group and there is still some controversy over its composition. The Cirripedia (barnacles), Copepoda, and Branchiura (Fish Lice) are discussed here.

***Cirripedia*:** The cirripeds (barnacles) are a group of sessile marine and estuarine crustaceans that attach to hard surfaces including rocks, jetty piles, hulls of boats, other invertebrates, and even whales. The animal is enclosed in a shell composed of a variable number of calcareous plates. Barnacles feed with their modified feathery legs that sweep through water like sieves. The most commonly seen barnacles on Australian shores are members of the order Thoracica, the stalked or goose barnacles (Suborder Lepadomorpha) and the acorn barnacles (Suborder Balanomorpha), and they often form dense bands on hard shores. Members of the order Acrothoracica are minute and burrow into calcareous substrata, including corals and mollusc shells, while members of the other two orders are parasitic. Worldwide there are around 1,000 described cirriped species, mostly free-living (Brusca and Brusca 1990). Around 460 species of thoracicans are known worldwide; about 200 have been recorded from Australia (checklist by Jones et al. 1990), and 85 from the Antarctic and sub-Antarctic regions, although perhaps twice this number may be extant (D. Jones pers. comm.). Comprehensive regional descriptions of species are provided by Jones (1990a; 1993). Some groups (e.g., littoral and shallow-water balanids) are generally well-known through studies in Europe and North America, but in Australia these groups, as well as others such as the deep-water, cryptic and epizoic groups, are not well known (D. Jones pers. comm.).

[9] http://www.crustacea.net/intro.htm

Copepoda**:** Copepods are generally small (e.g., 0.5-10 mm long) crustaceans that lack a carapace and have a distinct head, usually bearing a small central eye. They form an important component of the marine plankton, and are also common in most inland waters. Some are extremely modified as parasites, especially on fish. The majority of free-living copepods, and those most frequently encountered, belong to the orders Calanoida, Harpacticoida, and Cyclopoida, although these also contain some parasitic members. The calanoid copepods, as a group, are extremely important as primary consumers in freshwater and marine food webs, being largely plankton feeders. Most harpacticoids are benthic, and are often reported as detritus feeders, but many feed predominantly on microorganisms living on the surface of detritus or sediment particles. Cyclopoids are mostly planktonic.

Worldwide there are about 9,000 described copepod species (Brusca and Brusca 1990). Most available literature is dated (see summary in Edgar 1997, p. 531). In general, however, the Australian fauna is rich but poorly worked. Marine harpacticoids are being worked on by J. Walker and some other marine copepods by V. Harris, a retired worker.

Branchiura**:** This group, commonly known as “fish lice”, are ectoparasites on marine, freshwater and brackish water fishes. They have flattened, broadly oval carapaces and attach to the outside of the host by means of two suckers. They feed by piercing the skin and sucking blood or tissue fluids. There are about 130 described species. Two of the four genera occur in Australia, but very little is known of this group and there are no Australian workers.

Ostracoda**:** The ostracods are rarely seen by the general public because they are mostly very small. They do not have a widely accepted common name although the terms ‘mussel shrimps’ or ‘seed shrimps’ are sometimes used. They have a bivalved carapace that encloses the body, and range in size from 0.1 to about 30 mm in length. They are one of the most successful groups of crustaceans, being abundant worldwide in virtually all aquatic habitats. Most species are benthic crawlers or burrowers, although some are planktonic, and a few are terrestrial in moist habitats. Some are commensal on echinoderms or other (larger) crustaceans. Species can be filter feeders, herbivores, predators or scavengers. They are a large group, comprising about 2,000 living species, but are relatively poorly known. Ostracods are common, diverse and widespread in Australia and, because of their hard carapace, are extremely abundant as fossils.

Comprehensive illustrated accounts of shallow water species include McKenzie (1967), Yassini and Wright (1988) and Yassini and Jones (1995) and papers on tropical (Kornicker 1996) and deepwater temperate species (Kornicker and Poore 1996a). There is a large Australian fauna but currently few Australian workers are investigating marine taxa, and some Australian taxa are being worked on by overseas workers. Parker and Lowry are currently preparing a monograph on 70 new species of scavenging cypridinids ostracods from eastern Australia.

Malacostraca

The majority of well-known crustaceans, especially the edible species, belong to this class. There are usually three subclasses recognised, Phyllocarida, Hoplocarida and Eumalacostraca (e.g., Martin and Davis 2001).

Phyllocarida

The Phyllocarida, containing the single order Leptostraca, comprises less than 20 species worldwide. These animals, which have a bivalved carapace, all are marine, and most are epibenthic. They extend from the intertidal zone to a depth of 400 metres and most feed by stirring up bottom sediments. Only a few species have been recorded from Australian waters (P. Davie pers. comm.).

Hoplocarida

The Hoplocarida contains a single order, Stomatopoda. Stomatopods, also known as mantis shrimps and prawn killers, are relatively large (2-30 cm long) crustaceans characterised by a distinctive pair of very enlarged second thoracic limbs, used in hunting, either for crushing or as spears. Most are found in shallow tropical to warm temperate marine environments. They are found in burrows excavated in soft sediments, or in cracks and crevices, among rubble or in other protected spots, in coral reefs, estuaries, mudflats, and weed beds, to a depth of 1500 m. They are raptorial carnivores, preying on fishes, molluscs, cnidarians, and other crustaceans. Worldwide, there are around 440 described species (S. Ahyong pers. comm.). Genera were reviewed by Manning (1980). Recent taxonomic work by Ahyong (2000; Ahyong 2001), based mostly on existing collections, has nearly doubled the known Australian fauna to 113 species. In fact, the true number is probably much higher, since many species are difficult to sample using the standard methods on which most collections are based (S. Ahyong pers. comm.). Stomatopods have been recorded from Lord Howe, Norfolk, Christmas and Cocos (Keeling) Islands but do not occur in the Antarctic (S. Ahyong pers. comm.). Little or nothing is known of the biology or ecology of most stomatopods. Their burrows are utilised by commensals (Morton 1988).

Eumalacostraca

Eumalacostraca is by far the largest and most diverse group of crustaceans, and contains most of the large and well-known species. There are 13 orders, but several are either extremely inconspicuous or unknown from large parts of Australia. Those frequently found in the marine environment and more generally known include the members of the Superorder Pericarida[10], the Amphipoda (e.g., sand fleas, skeleton shrimps), Isopoda (e.g., pill bugs and fish lice, sand lice, etc.), and the common, but more poorly known, Cumacea, Tanaidacea, Mysidacea, Euphausiacea (krill etc.) and the huge order Decapoda (shrimp, lobsters, crabs etc).

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10 A Zoological Catalogue of Australia (Vol. 19.2a) on the peracarids and syncarids (freshwater) is currently in press (Lowry and Poore in press). A Zoological Catalogue of Australia volume on the Eucarida and the Hoplocarida (stomatopods) has recently been published by Davie (2002a; Davie 2002b).

Amphipoda: Amphipods and isopods belong in the group of crustaceans that brood their young. Amphipods are generally small, most species being less than 5 mm in body length, although some giant deep-sea species reach 25 cm, and one group of planktonic forms exceeds 10 cm. Amphipods inhabit almost every marine and most freshwater environments and often comprise a large portion of the biomass in an area. They are diverse, with more than 7000 known species worldwide, and ecologically important. They exhibit a wide range of feeding strategies, including herbivory, scavenging, carnivory, suspension feeding and parasitism. The Australian fauna comprises around 700 species, with an estimated 1500 or more still undescribed (J. Lowry and P. Berents pers. comm.). The majority of described species are known from southern Australia. Little is known of the northern fauna or that of offshore islands such as Christmas, Cocos (Keeling), Norfolk or Lord Howe. Species-level endemicity is high and most species have relatively restricted distributions, even amongst tropical species (e.g., Myers 1997).

There are four suborders. The Gammaridea (sand fleas) are found in virtually all intertidal and subtidal marine habitats, and are particularly common among algae, under stones and in seagrass beds. A few are pelagic, usually in deep oceanic waters. Some are semiterrestrial on sandy beaches (e.g., beach hoppers); there are also terrestrial and freshwater forms. Gammaridean amphipods are frequently the most numerous and diverse of the benthic crustaceans and are very important as food items for larger crustaceans and fish. A key to world families and genera, and a list of all species, is provided by Barnard and Karaman (1991) and recent monographs on Australian family level groups include Barnard (1972; 1974), Barnard and Drummond (1978; 1979; 1982), Moore (1981; 1987), Myers and Moore (1983), Lowry and Poore (1985), Myers (1988) and Poore and Lowry (1997). The known marine benthic gammaridean fauna (about 600 species) is currently being revised by Lowry, Berents and Springthorpe in a series of electronic monographs being published by the Australian Museum[11]. These monographs include illustrated, interactive keys to all Australian families and all known species. The Talitridae (beach hoppers, land hoppers) inhabit intertidal and supralittoral habitats (as well as wet forest floors). The beach faunas are particularly diverse in southern Australia but are largely undescribed. Work in progress on the Tasmanian fauna has found 22 species, only six described (A.. Richardson, pers. comm.). Sub-Antarctic (including Macquarie Island) species are described by Lowry and Fenwick (1983) and Lowry and Stoddart (1983).

The Caprellidea (skeleton shrimps) are exclusively marine and estuarine. They are mainly predators highly modified for clinging and are usually found attached to algae, seagrasses, hydroids and bryozoans. The Australian fauna is currently being monographed by I. Takeuchi and J. Guerra-Garcia[12]. One family (Cyamidae) is parasitic on whales. The Hyperiidea live a very specialised existence (as parasitoids) on pelagic jellyfish, salps etc. This association is not well understood and little is known of their biology. The worldwide hyperiidean fauna consists of about 240 species, with around 120 species described from Australian waters (reviewed by Zeidler 1992, 1998) and 70 from

[11] Online at www.crustacea.net.

[12] Department of Conservation Ecology, Ehime University, Japan and Laboratorio de Biologia, Universidad de Salvilla, Spain.

Antarctic and sub-Antarctic waters. Nearly all knowledge on Australian hyperiideans is based on collections from eastern Australia and very little is known from other areas or from deep-water (>200 m). There are probably still several undescribed species (W. Zeidler pers. comm.) in Australian waters, associated with rare gelatinous species. Ingolfielledeans are a rare and highly modified group of amphipods known from freshwater and marine habitats in various parts of the world. Lowry and Poore (1989) described two marine species from southern Australia and at least one other undescribed species is known from southwestern Australia.

Isopoda: Isopods are a diverse group of small crustaceans, including species known as pill bugs and fish lice, sand lice, and sea centipedes. They are generally less common than amphipods but are nonetheless frequently encountered. Isopods are of a similar size to amphipods but are flattened dorsoventrally rather than laterally. They range in length from less than a millimetre to more than 40 cm in length, the largest being the huge benthic *Bathynomus*. They are common inhabitants of nearly all environments, including land, and some are partly or exclusively parasitic. This group is especially important in deep benthic environments where it is often the most abundant crustacean taxon (Poore et al. 1994). Their feeding habits are extremely diverse; many are herbivorous or omnivorous scavengers, but direct plant feeders, detritivores, and predators are also common. Several thousand[13] species have been described worldwide; 773 are known from Australian waters, and 130 from the Australian Antarctic and sub-Antarctic regions, including about 20 from Macquarie Island (G. Poore pers. comm.). However, the extant fauna is probably much larger, with 3000 or more undescribed species (G. Poore pers. comm.). For instance, a diverse fauna is recognised from southeastern Australian continental slopes but it contains mostly undescribed taxa. Many areas of the Australian coast have never been collected, especially in Western Australia, and the surrounding deep-sea has no collections. As with most marine invertebrates, the southern Australian and Antarctic faunas show a high degree of endemism (about 95%), whereas many species from tropical regions are shared with the whole or part of the Indo-West Pacific. Tropical species are especially poorly known (G. Poore pers. comm.). The fairly extensive literature on Australian isopods includes significant papers such as Bruce (1986), Cookson (1990), Poore and Lew Ton (1993), Harrison and Ellis (1991) and Cohen and Poore (1994). S. Keable is developing an on-line key to the marine families of Australian isopods on Crustacea Net. The Australian deep-sea (i.e. depths below 3000m), a realm dominated by the suborder Asellota, is virtually unsampled. The Asellota can have as many as 100 species in a single sample and regional faunas can exceed 10,000 species (G. Wilson pers. comm).

Cumacea: The cumaceans are small crustaceans, usually 0.5-2 cm (Brusca and Brusca 1990), but up to 4 mm (Jones and Morgan 1994) long, and have a large bulbous anterior end and a long abdomen. They are mostly marine, living primarily in soft sediments; the few freshwater species are unknown in Australia. There are about 850 known species and they are distributed worldwide. Hale published a series of papers on the southern Australian fauna (Hale 1927, 1929) and (Mühlenhardt-Siegel 1999) has discussed the biogeography of sub-Antarctic and Antarctic taxa. There is a large Australian fauna but

[13] Bruce (2001) gives a figure of 9-11,000 named isopods (marine and non marine).

there are currently no Australian workers investigating the group, although some Australian taxa are being worked on overseas (e.g., Gerken 2000) and Tafe and Greenwood have published on the cumacean fauna of Moreton Bay (Tafe 1996a, 1996b).

Tanaidacea: Tanaids are mostly small (0.5-2 cm in length) and have an elongated body with a short and inconspicuous carapace. They are mostly marine and benthic, being known from all ocean depths, although a few live in brackish or almost fresh water. They often live in burrows or tubes; a few species live in small gastropod shells. Tanaidaceans are usually filter feeders on detritus or plankton, but some are predators. Like cumaceans, they are not rare but they are small and infrequently seen. About 850 species are known (Brusca and Brusca 1990) and they are found worldwide. Two papers by Sieg (1980; 1993) provide some information on the Australian fauna and Larson (2000; 2001) described some Australian species. About 20 species have been described from Australian waters, most of them endemic, although possibly ten times this number may be extant (G. Edgar pers. comm.). Tanaidaceans have not yet been recorded from Lord Howe, Norfolk, Christmas or Cocos (Keeling) Islands (G. Edgar pers. comm.), although they are expected to occur there. The group is poorly known, although they play an important ecological role, being one of the most numerous macrofaunal groups in many soft sediment habitats (G. Edgar pers. comm.).

Mysidacea: Mysids (sometimes called opossum shrimps) are shrimp-like crustaceans often confused with the superficially similar euphausiaceans. Their distinguishing features include possession of a pair of distinct balance organs (statocysts). They are generally pelagic or demersal, but are known from all ocean depths (Fenton 1985). Some species are intertidal and burrow in the sand during low tides. Most are omnivorous suspension feeders, eating algae, zooplankton, and suspended detritus. Mysids are not uncommon, but are rarely seen. Worldwide, nearly 700 species are known (Brusca and Brusca 1990). Approximately 100 species have been recorded from Australian waters, plus around 6 from the Antarctic and six from the Cocos (Keeling Islands); there may be a few dozen more undescribed (S. Talbot pers. comm.). Mauchline (1980) has reviewed this group worldwide and the Australian taxa are currently being reviewed by S. Talbot. J and J Greenwood are currently working on the mysids of Queensland.

Euphausiacea (Krill): These shrimp-like crustaceans range in length from about 4 to 15 cm. All are pelagic and they are known from all oceanic environments to depths of 5000 m. Most species are distinctly gregarious and, when they occur in huge schools (krill), they provide a major source of food for larger animals such as baleen whales, squids, fishes, and even birds. Krill densities often exceed 1,000 animals/m3. Krill are increasingly the target of fisheries, particularly in the Antarctic. Euphasiaceans are generally suspension feeders, although some are predators or detritivores. Baker et al. (1990) provide a guide to the world species. Fifty five species have been recorded from Australian waters (P. Davie pers. comm.).

Decapoda: Decapods (Crabs, Shrimps, Lobsters): possess a well-developed carapace and five pairs of thoracic legs, which give them their name. They are an extremely diverse group, with around 10,000 known species, and are among the most familiar eumalacostracans, supporting a variety of major and minor fisheries in Australia and worldwide. They occur in all aquatic environments at all depths, and a few spend most of their lives on land. Many are pelagic, but others have adopted benthic sedentary, errant, or burrowing lifestyles. Decapods are often categorised as natant (swimming) or reptant (walking), although these terms are no longer used as taxonomic groupings. Many reptant decapods, including the Thalassinidea (mud shrimps/ghost shrimps/marine yabbies), Astacidea (true marine lobsters), and Palinura (rock lobsters and shovel-nosed lobsters) have a dorsoventrally flattened abdomen terminating in a strong tail fan. Feeding strategies include nearly every habit imaginable: suspension feeding, predation, herbivory, scavenging, and more. Approximately 2100 species of decapods have been described from Australian waters, and several dozen from each of the islands and territories. About a third as many again are yet to be described. They show the greatest diversity in the tropics, although this is also where the least work has been done (P. Davie pers. comm.). Relatively good biological knowledge is available for many commercially important species, but information on other species is generally poor. There is currently only one paid museum taxonomist in Australia working full-time on decapods. A discussion of the distribution of decapods from SE Australia is given by O'Hara and Poore (2000). There are also ABRS catalogues on selected groups of decapods (Davie 2002a, 2002b).

The suborder Dendrobranchiata contains about 450 species worldwide (107 in Australian waters; P. Davie pers. comm.), including the penaeid and sergestid prawns. Many are quite large, over 30 cm long. The sergestids are pelagic and all marine, whereas penaeids are pelagic or benthic, and some occur in brackish water; some are of major commercial importance in the world’s prawn (or shrimp) fisheries. The Australian penaeids were reviewed by Grey et al. (1983). The suborder Pleocyemata comprises all the remaining decapods, including several kinds of prawns as well as the crabs, crayfish, lobsters and a host of less familiar forms. Infraorder Caridea (including the carid prawns and snapping shrimps) are generally smaller than the penaeids and less known, although the Moreton Bay fauna was described by Wadley (1978) and Holthuis (1993) provides a key to the genera. They are most diverse in the tropics and there are nearly 2000 living species worldwide, with 555 species recorded from Australia (P. Davie pers. comm.). Snapping (alpheid) shrimps are described by Banner and Banner (1972; 1975; 1982). Infraorder Stenopodidea (stenopodid shrimps) are usually only a few centimetres long. Most species are tropical and associated with benthic environments, especially with coral reefs. Many are commensal, and the group includes 20 or so species of cleaner shrimps (e.g., *Stenopus*), found on tropical reefs, which remove parasites from fishes. Six stenopodid species are known from Australian waters (P. Davie pers. comm.). Infraorder Thalassinidea includes the animals known as mud shrimps, ghost shrimps, and marine yabbies. Most are marine sediment burrowers or live in coral rubble. Important reviews include Poore and Griffin (1979) and Poore (1994). Seventy five species are known from Australia (P. Davie pers. comm.). Infraorder Astacidea contains the true marine lobsters (16 species in Australia; P. Davie pers. comm.) as well as the freshwater crayfish. Infraorder Palinura (44 Australian species; P. Davie pers. comm.) is represented in Australia by two major families – the Palinuridae (rock-lobsters) and Scyllaridae (shovel-nosed lobsters, including Moreton Bay and Balmain bugs). All are marine and found in a variety of habitats. Infraorder Anomura (270 Australian species; P. Davie pers. comm.)

contains a variety of decapods including hermit crabs, porcelain crabs, mole and sand crabs, and a few other crab-like taxa. Most are marine, although a few freshwater and semi-terrestrial species are known. The abdomen of these is either soft and asymmetrically twisted (as in hermit crabs), or short, symmetrical and flexed beneath the thorax (as in porcelain crabs and others). Hermit crabs typically inhabit empty gastropod shells or other "houses" not of their own making. Infraorder Brachyura includes the so-called true crabs, with a symmetrical, reduced abdomen flexed beneath the thorax and chelate, usually enlarged first pereopods (Brusca and Brusca 1990). Brachyurans are the dominant decapod crustaceans on most marine coasts, and they show immense diversity and variety. A few freshwater and semi-terrestrial species occur in the tropics. There is a considerable literature on Australian decapod crustaceans, which is summarised in Jones and Morgan (1993) and Edgar (1997). There are 939 described species known from Australian waters (P. Davie pers. comm.).

2.2.23 Echinodermata (Sea Urchins, Starfish, Brittle Stars, Sea Cucumbers etc.).

The **echinoderms** are a well-defined group of exclusively marine animals characterised by a basic radial symmetry, usually pentamerous (i.e. with five sections or arms), a calcite skeleton, and a water vascular system with hydraulic tube feet. The tube-feet are highly flexible muscular tubes used variously for locomotion, adhesion, food capture and transfer of food to the mouth; some are sensory and some may assist in respiration (Thomas 1982). The ability to regenerate missing or damaged parts is well developed among echinoderms. Most echinoderms are free-living and benthic, with a more or less sedentary habit. Five classes are recognised, the **Crinoidea** (feather stars and sea lilies), the **Holothuroidea** (sea cucumbers), the **Echinoidea** (sea urchins etc.), the **Ophiuroidea** (brittle stars) and the **Asteroidea** (starfish).

Around 6600 living species of echinoderms, belonging to five extant classes, occur worldwide. Rowe and Gates (1995) recorded 1154 species from Australian waters, representing about 18% of the world fauna. However, these authors also noted that the vast majority of these species are known from shallow coastal or reefal waters (<30 m), and predicted that significantly more species, perhaps over 2000, will eventually be identified from Australian waters. 52 echinoderm species, including 4 endemics, have been recorded from seas around Macquarie Island and the Macquarie Ridge (O'Hara 1998a). About 13% of species are endemic to tropical Australia while the temperate fauna has about 90% endemism (Wilson and Allen 1987), although a tropical origin for a major component of the Australian modern echinoderm fauna is generally accepted (Rowe and Gates 1995). Patterns of distribution for echinoderms in SE Australia are given by O'Hara and Poore (2000).

The Australian echinoderms are catalogued by Rowe and Gates (1995) and an ABRS Fauna of Australia volume on echinoderms is currently in preparation. Details of taxonomic literature for each group can be found in Rowe and Gates (1995), but of major importance is the monograph of the Australian fauna by Clark (1946). The five classes currently recognised are briefly described below.

Crinoidea (Feather Stars and Sea Lilies)

In the crinoids (feather stars and sea lilies) the basic five-rayed form has developed by branching into ten or more feather-like arms. These echinoderms are common in both temperate and tropical waters but, due to their cryptic habits, often pass unnoticed. Some inhabit rocky reefs, being abundant on open coasts to depths exceeding 75 m. Others live attached to algae or seagrasses (often being cryptically coloured to match their host), or attached to bryozoans, sponges or coral, often in deeper water (Shepherd et al. 1982). They also have an important fossil record. Crinoids are suspension feeders, consuming minute crustaceans such as amphipods and copepods, larvae of marine animals, protozoans and detrital matter.

There are about 600 species worldwide, and more than 130 species belonging to 58 genera in 16 families have been described from Australian waters (Rowe and Gates 1995). Rouse (pers. comm) is currently undertaking a phylogenetic analysis of this group.

Holothuroidea (Sea Cucumbers)

Holothurians, commonly known as sea cucumbers or (in the case of edible species) *trepang* or *bêche de mer*, are elongated, soft-bodied echinoderms basically pentamerous but often with a secondarily imposed bilateral symmetry. The skeleton is reduced to minute calcareous plates or spicules that number from a few to millions (Rowe 1982).

Holothurians usually live upon, or buried in, the substrate. The members of one order (Dendrochirotida) are benthic suspension feeders, using the tentacles to pass food to the mouth; the other orders are deposit feeders, ingesting organic matter with mud or sand. They may play an important ecological role through their bioturbation of sediments during feeding (e.g., Uthicke 1999; Uthicke 2001). Some species are economically important, being harvested commercially for food.

There are about 1400 species worldwide, and at least 210 species, belonging to 69 genera in 15 families, have been described from Australian waters (Rowe and Gates 1995).

Echinoidea (Sea Urchins, Heart Urchins and Sand Dollars)

The Echinoidea comprise animals known commonly as sea urchins, heart urchins and sand dollars. They have an external skeleton of interlocking, spine-bearing calcite plates surrounding the body organs. This skeletal shell or *test* may be spherical, subspherical, oval, heart-shaped or discoidal. Echinoids are particularly common in shallow water and on the continental shelf or slope, but they also occur in the abyssal depths. They are found in habitats ranging from rocky areas to sand and mud, variously modified spines and tube feet enabling them to cling to rock surfaces or to burrow into soft sediments. Some species bore into soft rock or coral or graze on endolithic algae in these substrates and can be important agents of reef erosion, especially in areas where over fishing has occurred (Pari et al. 1998). Those living in more exposed conditions (e.g., shallow areas of strong water movement) often have strong spines with which they can wedge themselves tightly into rock crevices; other more delicate species with very short spines live in calmer waters, such as among algae or seagrasses. Sea urchins are mostly non-selective scavengers, browsing on whatever is readily available, including both plant and animal material. The sand- and mud-burrowing forms eat organic matter in the substrate (Baker 1982a). They are ecologically important, being widely considered the major grazers in temperate subtidal systems (Jones and Andrew 1990). Some species keep large areas free of foliose algae, these habitats being known as 'sea urchin barrens'. They are also of some economic importance in various parts of the world, such as Japan and Southeast Asia, where they are a traditional seafood delicacy. In parts of Australia, they are harvested commercially, and "recreational" harvesting occurs. Some species may have indirect effects, through competition for habitat, on commercially important species such as abalone (Baker 1982b).

There are about 800 echinoid species worldwide, and more than 200 species, belonging to 95 genera in 32 families, have been described from Australia (Rowe and Gates 1995).

Ophiuroidea (Brittle Stars)

The ophiuroids – commonly known as brittle or basket stars – have a body consisting of a hollow central disc from which five (or occasionally six or seven) slender arms radiate out like the spokes of a wheel. They are often brilliantly coloured with red, blue or black pigments. They are found from intertidal rock pools to the abyssal depths in all oceans, but are especially diverse in the tropics. Some bury themselves in mud or sand but most live under rocks, among seaweed, or in association with other marine invertebrates such as sponges, hard corals, and bryozoans (Baker 1982a). Brittle stars are of no direct economic importance, but they do play a significant role in the ecology of coastal waters, since they are often preyed upon by fish, and they scavenge organic detritus.

There are about 2000 species worldwide, and more than 320 species belonging to 95 genera in 15 families, have been described from Australia (Rowe and Gates 1995).

Asteroidea (Starfish)

The asteroids – commonly known as sea stars or starfish – are, together with the sea urchins, the best-known representatives of the phylum. Their body consists of a central disc from which the arms extend laterally, although in some species the arms and disc merge giving a polygonal appearance. The ability to regenerate missing or damaged parts is well developed among asteroids (and among echinoderms generally). Sand or mud-dwelling species are often omnivorous, feeding on small molluscs, ophiuroids, worms, micro-organisms and detritus. Some voracious species feed on large molluscs such as scallops, mussels, cockles and abalone and scavenge moribund animals as well. Most other species live on rocky substrates where they feed on encrusting sponges, ascidians and bryozoans (Zeidler and Shepherd 1982), but some are herbivores. Some species are important pests. For instance, the Northern Pacific seastar (*Asterias amurensis*) is a significant pest in Tasmanian waters, having probably been accidentally introduced as larvae in ballast water (see Section 6.5.2). The crown-of-thorns starfish (*Acanthaster plancii*), which feeds on live coral, has been the subject of a great deal of attention because of the damage caused during periodic population explosions on the Great Barrier Reef (see Section 6.8.2).

There are about 1800 species known worldwide, and around 280 species (belonging to 111 genera in 24 families) have been described from Australia.

2.2.24 Chaetognatha (Arrow worms)

This small phylum contains about 60 species worldwide. They are small, elongate, predatory swimmers that live in the plankton and feed on copepods and other small invertebrates. A key to the Australian taxa is provided by Thompson (1947), but they have not been extensively studied since.

2.2.25 Hemichordata

This phylum contains worm-like, soft-bodied coelomate animals with a body divided into three distinct regions: a proboscis, collar and trunk. Although only a small group, they are of considerable zoological interest because their pharynx is perforated by gill clefts, a structure unique among the invertebrates and otherwise found only in the phylum Chordata (which includes the Vertebrates).

Hemichordates are marine and benthic, inhabiting either roughly U-shaped, loosely coiled burrows in sand or mud, or collagenous tubes attached to or spread over a solid substratum. They are widely distributed from the polar regions to the tropics, the majority being found in the mid to lower intertidal zone or shallow offshore areas. Some can be fairly dominant members of the macrofauna on shores and reefs. However, others favour deeper muddier habitats on the continental shelf, and some have been recorded from abyssal depths and one in association with hydrothermal vents (Burdon-Jones 1998).

There are two classes, the Enteropneusta and the Pterobranchia. The **enteropneusts** (“acorn worms”) are all burrowing forms, which secrete a soft, usually delicate mucous tube. They feed on organic material contained in sand and mud and may also collect organic particles in suspension using mucous secreted by the proboscis. They are often found among the roots of eelgrass (*Zostera* and *Heterozostera*) or *Posidonia* though very occasionally they occur under stones near low tide mark among the holdfasts of large, brown algae or in turfs of coralline red algae. The **pterobranchs** comprise very few species, in three genera, of small (usually less than 5 mm long) colonial, tube-dwelling animals attached by a tube to a solid object (Shepherd 1997b).

There are 16 genera and 94 known species worldwide, with many more species recognised but yet to be described. The Australian fauna was catalogued by Burdon-Jones (1998). Seven genera and 12 species, as well as four or more undescribed species, are found in Australian waters, but only three are endemic (Burdon-Jones 1998). In

Australia they have been recorded from shores and offshore reefs and neighbouring islands of all states except the Northern Territory (Burdon-Jones 1998).

2.2.26 Tunicata (=Urochordata)

Tunicates are marine, filter-feeding organisms. Their name is derived from the tunic or test, an outer protective coat of an acellular cellulose-like material that is secreted by the epidermis (absent from Class Appendicularia). While regarded as invertebrates, they have features that suggest a relationship with chordates, and formerly were included as a separate class within the Chordata.

The tunicates are divided into three classes; the sessile **Ascidiacea** (sea-squirts) and the pelagic **Thaliacea_**(salps, doliolids and pyrosomids) and **Appendicularia** (larvaceans). The Australian tunicate fauna is catalogued by Kott (1998).

The ascidians are the most familiar and best-studied tunicates, partly because they are easily obtainable worldwide – many species can be collected on the shore and live well in aquaria – and partly because the group has much interested zoologists ever since Kowalewsky (1867) showed that their tadpole larvae allied them to the Chordates (Bone 1998). The more delicate pelagic forms are difficult to obtain and maintain in the laboratory, so that whilst their distribution, morphology and taxonomy have long been studied from fixed material collected in plankton net tows, their physiology, behaviour and ecological impact are less well known, with few zoologists having had the opportunity to examine them alive. Recent work by open-ocean divers from submersibles and workers at favoured shore locations has enabled direct observations that have greatly increased knowledge of these animals. In particular, pelagic tunicates may be important in facilitating the downward flux of carbon and nitrogen from the surface to the benthos (Bone 1998).

Ascidiacea (sea-squirts)

The ascidians (sea-squirts) are solitary or colonial sessile organisms in which the larval tail is lost upon metamorphosis. Ascidians exist either as separate individual zooids (solitary), aggregations of zooids vascularly connected at their bases (social), and/or as colonies made up of many physiologically interdependent and typically smaller modular units (compound) (Brusca and Brusca 1990). These three morphologies are associated with a suite of other life-history characters. Most notably, the social and compound forms have a mixed mode of reproduction, using asexual reproduction to generate colonies of genetically identical zooids but using internal fertilisation to produce larvae that have free-swimming periods which can be measured in minutes and that are brooded until they are competent or near-competent to settle. In contrast, solitary forms are typically broadcast spawners, and are assumed to be widely dispersed (Ayre et al. 1997b). Solitary ascidians range from individuals of less than 1 mm found in the meiofauna up to large, often stalked species 20 cm or more in length. Colonies consist of small asexually replicated individuals, and range from small cushions 1 cm or less in diameter to large sheets no more than 2-3 mm thick to very massive lobed structures to a metre or more high. There are different sorts of colony and degrees of organisation of the zooids in a colony (Kott 1997).

Ascidians are found in most marine habitats from intertidal to hadal depths, and often dominate epifaunal assemblages. They usually live attached to stones, weed, shell particles or rocky substrates by adhesive secretions of their test, or are fixed in or on soft and shifting sediments by root- or hair-like extensions of the test. Algae, hydroids, bryozoans, sponges and other ascidians often grow over the outside of the test, especially of solitary ascidians, and so camouflage and further protect the animal. While a few abyssal ascidians have evolved a carnivorous habit, the great majority are filter-filters, straining bacteria, phytoplankton and organic detritus from the incurrent stream of water (Kott 1998). The filtering rate is often extremely high; for instance, Kott (1976) showed that a small solitary ascidian of about 2 cm diameter filtered nearly a litre of water per hour. As one of the most important components of the filter feeding fauna in most benthic habitats, they almost certainly play a significant role in the energy cycle of marine communities, by concentrating the energy from the small organic particles, bacteria and phytoplankton that they filter so efficiently, and passing it onto their predators (Kott 1997). Large intertidal ascidians are widely collected as bait. In NSW these are known as cunjevoi and often form a distinct zone on the lower shore of exposed rocky shores.

The ascidians are the largest and most diverse class of tunicates. Approximately 2000 described species are known worldwide (P. Mather pers. comm.), and Kott (1998) records about 750 extant species in the Australian fauna. About 160 species have been described from the Antarctic and sub-Antarctic region, and several recorded from each of Christmas, Cocos (Keeling), Lord Howe and Norfolk Islands, although in all cases there are undoubtedly a number of additional taxa yet to be described (P. Mather pers. comm.).

Kott (1997) showed that the southern Australian ascidian fauna has both temperate and tropical elements. The temperate element includes indigenous species known only from the southern coast (114 species), those that extend up the eastern and/or western Australian coast (33 spp), and temperate (possibly Gondwanan) species known also from South Africa or New Zealand, or those that are circumpolar (16 spp). The tropical element includes western Pacific species extending around Australia or down its east coast (31 species), and pan-tropical and cosmopolitan forms (16 spp).

The taxonomy of the group is relatively well understood, being the focus of a four-volume monograph by P. Kott (1985; Kott 1990a, 1990b, 1992a, 1992b; Kott 2001). However, there is still a need for taxonomic work on species from continental slope locations, the Northwest Shelf, temperate habitats on eastern and western coasts and the Southern Ocean. Field studies on natural history, including life history and observations that will contribute to species recognition in the field, are also much needed (P. Mather pers. comm.). In addition, genetic evidence for colonial forms indicates that sibling species may be common (e.g., Davis et al. 1999; A. Davis pers. comm.).

Thaliacea (Salps, etc.)

The Thaliacea consists of the salps, doliolids and pyrosomids. All show alteration of generations, have an external cellulose test, and move through the water jet-propelled by the ciliary feeding stream. They are among the most prolific and conspicuous components of the zooplankton in all the oceans of the world, and in certain seasons (e.g., autumn and spring in Australian eastern coastal waters) some species occur in vast swarms that exclude most other zooplankters from surface waters (Kott 1998). The geographic ranges of most species are vast, being defined by ocean currents rather than geographic areas, and no indigenous species are known (Kott 1998).

Salps are tubular animals that swim continuously by rhythmic muscular contractions. The test is transparent and usually colourless and the individuals are surrounded by it. They are ubiquitous in the world oceans, usually being found at low densities throughout the oceanic regions, although they periodically occur in dense swarms in continental shelf and slope areas. Swarms may cover large areas – one recorded off the Californian coast covered some 9000 square kilometres (Deibel 1998) – and can last for several months. Salps filter copious quantities of seawater, taking almost all particles in the size range 0.004-1.0 mm in diameter, and have a rapid growth rate and short generation time. They are important in planktonic food webs, having an important grazing impact on phytoplankton and in turn being consumed by a variety of predators including heteropod molluscs, jellyfish, tuna and some seabirds (Kott 1997). They also play a role in the downward flux of matter. Yet, despite their ecological importance, the large salp biomass is, at any time, made up of relatively few species (Kott 1997). Thirteen genera and 23 species have been recorded from Australian waters (Kott 1998).

Pyrosomas are colonial animals, consisting of permanent, tubular, hollow colonies of asexually replicated zooids that are independent but remain embedded side-by-side in a common transparent tunic. They are common in tropical and warm temperate waters. They are mainly caught in the epipelagic and upper mesopelagic layers, although some have occasionally been observed at greater depths (Godeaux et al. 1998). Only two species have been recorded from Australian waters (Kott 1998).

Doliolids are small, free-swimming, barrel-shaped organisms with 8-9 circular muscle rings around the body, which is open at each end. They are pelagic animals and have a cosmopolitan distribution but are most abundant in the euphotic zone of warm continental shelf waters, rarely being found poleward of 64° N or S (Deibel 1998). As with the other thaliaceans, they may form dense swarms. They are not diverse, with only two species recorded from Australian waters, although both of these are common components of the eastern Australian gelatinous plankton (Kott 1998).

Appendicularia (Larvacea)

Appendicularians, or larvaceans, are small free-swimming planktonic tunicates with a body consisting of a short trunk and a muscular tail supported by a notochord. Most are only a few millimetres long, with a trunk of normally 1-8 mm and a tail usually several times longer, but some may be much larger, up to 8-9 cm (Fenaux 1998). All secrete and inhabit a complicated mucous filtering house with many chambers, which is normally discarded every few hours.

Appendicularians occur in all oceans, although most species are warm water forms. While the majority are found in surface waters, an increasing number of deep-sea (mesopelagic and even bathypelagic) species have been described in recent times (Fenaux 1998).

Appendicularians are amongst the commonest members of the zooplankton community, often being the second or third most abundant group in plankton samples, but due to their fragility, are often severely damaged or destroyed when collected with plankton nets. Consequently there have been relatively few studies attempting to assess their environmental impact. Still, as one of the few groups of animals able to retain very small organisms (down to 0.0001 mm, and usually less than 0.002 mm), they play an important role as grazers in planktonic food webs (Kott 1997). In turn they are eaten by many planktonic carnivores such as medusae, ctenophores, siphonophores, chaetognaths and fish, being a particularly important source of food for the larvae and juveniles of the latter. In addition the discarded houses, being rich with mucous and trapped organisms, comprise a rich source of food for a range of planktonic and benthic organisms (Kott 1997).

The Zoological Catalogue of Australia (Vol. 34) records 22 described species from Australian waters (Kott 1998). As with the thaliaceans, the geographic ranges of most species are great, and so far no indigenous species are known.

There has been no systematic study of the group in Western Australian waters or in the tropical or southern coastal waters of the Australian continent.

2.2.27 Cephalochordata (Lancelets)

The cephalochordates, also known as lancelets or acraniates, are small (<50 mm), entirely marine, primitive chordates normally considered to be the sister group to the vertebrates. They are elongate and pointed at both ends with a fold of skin that forms a continuous dorsal, caudal and ventral fin (Richardson 1998). Like the tunicates, they are jawless mucous filter feeders.

Cephalochordates are found in shallow temperate and tropical seas, where they burrow in sand, gravel and shell fragments, although they do not seem to tolerate fine sediments which may interfere with filtration (B. Richardson pers. comm.).

The Cephalochordata is a small phylum, with only about 36 species described worldwide (B. Richardson pers. comm.). The Australian fauna consists of two genera and eight species, of which two are temperate-water endemics, two are tropical-water endemics and four have an Indo-West Pacific distribution (Richardson and McKenzie 1994; Richardson 1998). They have also been described from Christmas, Cocos (Keeling) and Lord Howe

Islands but are not known from Norfolk Island and do not occur in the Southern Ocean. No species are known from the waters off the southwestern quarter of the continent (between Ceduna and Shark Bay), and since good collections of other bottom-dwelling species have been made in this area, it seems likely that their absence is a real phenomenon, although the causes (and the consequences) are unknown (Richardson and McKenzie 1994).

Only incidental observations on the distribution and ecology of these animals have been made in Australia. The taxonomy and distribution of Australian cephalochordates are discussed in Richardson and McKenzie (1994) and the fauna is catalogued by Richardson (1998).

2.3 Marine invertebrate fauna of the States, Territories and Islands

The following discussion refers only to the fauna of the littoral and Continental Shelf since very little is known about the deep-sea faunas. The States and Territories have jurisdiction to three nautical miles (nm) from the shore, the Commonwealth to 200 nm. The review below is geographically based, with states and territories being the units, rather than jurisdiction.

2.3.1 States and mainland territories

Overviews of the Australian marine environment were provided by Bunt (1987) and Poore (1995a) and the zoogeography of the marine fauna of Australia has been reviewed by Wilson and Allen (1987). The latter recognised a Northern Australian Zone (part of the Indo-West Pacific Region), a Southern Australian Region and Overlap or Transition Zones on the east and west coasts. The Transition Zones have many endemics, as does the Southern Australian Region (O'Hara in press), while the northern coast has a much lower level of endemism. Earlier schemes have recognised various faunal provinces around Australia (see, Wilson and Allen 1987; Section 2.4.2).

There are no comprehensive reviews of the marine invertebrate fauna of Australia. The ABRS Fauna of Australia series (a family-level overview and synthesis of available information) includes a volume on the Mollusca (Beesley et al. 1998) and one on Polychaetes and their allies (Beesley et al. 2000). The Zoological Catalogue of Australia series (a species-level directory of taxonomic, geographic and biological data) includes catalogues on Porifera (Hooper and Wiedenmayer 1994), Echinodermata (Rowe and Gates 1995), Hemichordata, Tunicata, Cephalochordata (Wells and Houston 1998) and syncarid and pericarid Crustacea (Lowry and Poore in press), various groups of Decapods (Davie 2002a, 2002b), Polychaeta (Hutchings and Johnson 2001) and some classes of Mollusca (Wells 2001) with some others in preparation. Checklists of many taxa, in some cases enhanced (the above groups, plus Bivalvia and Gastropoda (in part)), and in others to family-level only, are available as part of the Australian Faunal Directory on the ABIF website[14]. Some details of the resources available for other groups are given in the overview of the main groups of marine invertebrates earlier in this chapter.

General, semi-popular guides to the marine invertebrate fauna are few, the most comprehensive being restricted to the temperate fauna. The two most important of these are Dakin's (1952) Australian Seashores, which has been reprinted and revised many times, the most recent revision being by Isobel Bennett in 1987 , and Edgar's (1997) *Australian Marine Life*. Bennett and Pope (1953; 1960) provided accounts of descriptive "ecology" of southern shores. The three volumes of "Marine invertebrates of southern Australia" (Shepherd and Thomas 1982, 1989; Shepherd and Davies 1997) also provide an overview of the major groups found in temperate Australian waters.

Accessible guides to groups of marine invertebrates which encompass most or all of Australia include crustaceans (Jones and Morgan 1994), molluscs (Allan 1962), "prosobranch" gastropods (Wilson 1993; Wilson 1994), bivalves (Lamprell and Whitehead 1992; Lamprell and Healy 1998a), nudibranchs (Willan and Coleman 1984), corals (Veron 1986, 1995a; Veron 1999), and staghorn corals (Acropora) (Wallace 1999a, 1999b).

We have not attempted to list reviews of various taxa, although some of the more important are given under each of the taxa outlined in this chapter. A detailed, recent bibliography for identification resources for the marine invertebrates of temperate Australia is given by Edgar (1997).

New South Wales

The fauna of NSW is essentially warm temperate with elements from the subtropical fauna shared with southern Queensland and the cool temperate fauna of Bass Strait and the south coast. There are a few NSW endemics, the main area of endemism being in the north of the state. The NSW coast is largely an alternating series of headlands, sandy beaches and estuaries. There are many coastal lagoons, especially from the central part south and there are three very important inlets in the vicinity of Sydney (Broken Bay, Sydney Harbour, Botany Bay and Port Hacking) that all have quite different characteristics and faunas and all are unique on the Australian coast. The only other large embayments are Jervis Bay and Twofold Bay in the southern half of the state, both of these being more open than those in the vicinity of Sydney. Lord Howe Island (see below) is administratively part of New South Wales.

Relevant Institutions. Several major universities have a focus on marine systems and there is a Special Research Centre on Ecological Impacts of Coastal Cities at the University of Sydney and a Centre for Marine and Coastal Studies at the University of NSW. In addition, there is a recently established National Marine Science Centre at Coffs Harbour, a co-operative venture between The University of New England and Southern Cross University. Thus, there are potentially good resources available to study marine

[14] Australian Biodiversity Information Facility – ABIF Fauna is at http://www.environment.gov.au/abrs/abif-fauna/

invertebrates in NSW. There are several small university-operated marine stations and those at Arrawarra (University of New England) and Jervis Bay (University of Canberra) encourage general use. Small teaching facilities are also available in various locations in the state and there is a major fisheries laboratory at Port Stephens that undertakes research work on aquaculture of crustaceans and molluscs as well as housing the Conservation Section of NSW Fisheries. Most of the work relevant to marine invertebrates being conducted in universities is primarily ecological, with genetics and physiological studies also being undertaken.

The Australian Museum is the oldest and largest natural history museum in Australia and has very large marine invertebrate collections, including one of the world's largest of some groups. Eight scientists specialise in the taxonomy of various groups of marine invertebrates (mainly crustaceans, polychaetes, molluscs). There are no other significant collections of marine invertebrates in the state.

Published Resources. There is no comprehensive list of the marine invertebrates of NSW, or a recent list for any major group other than those catalogued by ABRS. Whitelegge's (1889) list of the marine fauna of Port Jackson is, amazingly, the only attempt to list all the marine taxa found in Sydney Harbour, although a recent survey by the Australian Museum for Sydney Ports Corporation for introduced species, sampled intensively within the port area (Australian Museum Business Services 2002). Extensive species lists of the invertebrates of some areas have been produced as a result of short-term surveys – these include Jervis Bay (CSIRO 1994), Twofold Bay (Hutchings et al. 1989), Botany Bay (Wilson 1998), and the shelf off Sydney (Jones 1977).

Other state-based checklists include the molluscs (Iredale and McMichael 1962), and Hutchings and Murray (1984) produced a taxonomic account of the polychaetes from the Hawkesbury River and some other estuarine areas in central and southern New South Wales. Pope (1945) produced a key to the barnacles of Sydney and Roberts and Davis (1996) provided an ecological assessment of major sponge taxa on temperate coastal reefs of Sydney. An interactive key (on CD Rom) is available for the estuarine (and freshwater) molluscs of NSW (Ponder et al. 2000), and one to the polychaetes will be available by late 2002 (Wilson et al. 2002). Yassini and Wright (1988) investigated the distribution and ecology of ostracods from Port Hacking.

Conservation Studies. The ecological and conservation significance of the subtidal rocky reef communities of northern New South Wales were studied by Harriott et al. (1999) who suggested areas that should be considered for marine parks. The Solitary Islands in northern NSW have been the focus of studies on their coral communities (eg., Harriott and Banks 1995; Smith and Harriott 1998). Parker (1995), in a report for the National Parks Association of NSW, reviewed the status of coastal marine and estuarine conservation in NSW, including assessments of waters adjacent to over 60 terrestrial protected areas, and proposed a number of new marine reserves. Marine parks have been declared in the Solitary Islands, Jervis Bay and Lord Howe Island and management plans are in various stages of development for each of these (EA 2001b respectively; 2001a; NSWMPA 2001a, b, 2002). NSW Fisheries has recently identified candidate sites for aquatic reserves for the conservation of rocky intertidal communities in the Hawkesbury Shelf and Batemans Shelf Bioregions (Otway 1999b). A similar exercise has been carried out for estuarine aquatic reserves in the same area (Frances et al. 2001).

Studies on the effects of sewage effluent in the state have been the subject of several studies (see Section 6.6.1) and many ecological studies have been conducted on intertidal rocky shores (e.g., see Underwood and Chapman 1995 for review), mangroves, saltmarshes and seagrasses (see Sections 5.3.2), and soft bottom benthos.

Victoria

Victoria has a cool temperate fauna. The shallow water hard shore fauna in the eastern-most corner of the state is disjunct with that in the middle part of the state (from Wilson's Promontory to the west) due to the very long stretch of soft shore habitat across the Gippsland coast. In addition, there are two very large embayments, Port Phillip Bay and Western Port that both contain important, reasonably well-known faunas.

Relevant Institutions. There are no generally accessible marine stations in Victoria, although there is a fisheries facility at Queenscliffe. The Victorian universities undertake some, mainly ecological, marine-based work involving invertebrates. Museum Victoria has extensive collections of, and several scientists involved in research on, marine invertebrates (currently crustaceans, echinoderms, and polychaetes).

Published resources. Reports on studies in Western Port include Smith (1975), Coleman et al. (1978) and Coleman and Cuff (1980). Results of the zoobenthos studies in Port Phillip Bay are reported in MMBW and FWD (MMBW 1973), Poore et al. (1975), Poore and Kudenov (1978) and Poore and Rainer (1974; Poore and Rainer 1979). Decapod crustaceans from the Port Philip survey were described by Griffin and Yaldwyn (1971) and the ostracods by McKenzie (1967). Wilson et al. (1998) described changes in the benthos over the two decades since the Port Phillip surveys were undertaken. Epibenthic studies in Port Phillip Bay carried out by Cohen et al. (2000), found that seven of the 63 epibenthic organisms were introduced species, and many of these were widespread and abundant (35% of all species) suggesting that they are likely to be having significant effects on the ecology of the Bay.

Important guides to the marine fauna include molluscs (Macpherson and Gabriel 1962), stony corals (Cairns and Parker 1992); and the Marine Research Group of Victoria (1984) has produced an “atlas” of some of the common, large coastal invertebrates. The Marine Invertebrates of Southern Australia handbook series (see under South Australia below) is also relevant. Bennett and Pope (1953) described the intertidal zonation patterns on exposed rocky shores. Erséus (1990a) has described marine oligochaetes from Victoria.

A large number of studies have resulted from the Museum of Victoria’s Bass Strait surveys, and many more are likely to be published over the coming decades. Detailed information on the taxonomy of selected groups from Bass Strait and off the shelf include studies on sponges (Wiedenmayer 1989; Hooper and Wiedenmayer 1994), isopods

(Poore et al. 1994) and hydroids (Watson 1994). Selected families of polychaetes from the area have been incorporated into taxonomic revisions undertaken by Hutchings and her co-workers (Hutchings and Glasby 1988; Hutchings and Peart 2000; Hutchings and Peart 2002).

Conservation Studies. Conservation of marine invertebrates in Victoria was discussed by Norman and Sant (1995), and O'Hara (1995) discussed their conservation at San Remo. Recently, O'Hara and Barmby (Museum Victoria) carried out an inventory of rare and vulnerable marine animals in Victoria for the Department of Natural Resources and Environment (O'Hara and Barmby 2000). This project identified marine invertebrate taxa (belonging to these three major taxonomic groups) that might be considered as priorities for conservation management, because of their small extent of occurrence, small area of occupancy within Victoria, specificity to vulnerable habitats, etc. O'Hara (in press) has discussed the endemism of decapods, echinoderms and crustaceans along the Victorian coast. Several of these are short-range endemics restricted to isolated populations in vulnerable habitats. O'Hara suggests that the narrow-range endemics in this region are more vulnerable to widespread processes that alter habitat than localised disturbance.

Tasmania

Tasmania's marine fauna is cool temperate with many taxa shared with the rest of the southern coast of Australia. There is some endemism, especially in the southern part of the island.

Relevant Institutions. The University of Tasmania undertakes work on the ecology and biology of marine invertebrates. Both the Tasmanian Museum (Hobart) and the Queen Victoria Museum (Launceston) have marine invertebrate collections, although there are no research workers specialising principally in marine taxa (one has done some generalised work on several taxa).

In addition, the CSIRO's Division of Marine Research is based in Hobart, as is the Australian Antarctic Division. The University of Tasmania runs a basic laboratory at Maria Island, and the Department of Primary Industries, Water and Environment (Fisheries) has a Marine Research laboratory at Taroona, near Sandy Bay.

Published Resources. Semi-popular treatments of the molluscs include Richmond (1990; Richmond 1992). Molluscs were checklisted by Kershaw (1955) and, earlier, May and Macpherson (1958) provided an illustrated index of all species of molluscs and brachiopods recorded from the state at that time. Dartnall (1980) provided a guide to the echinoderms, and Cairns and Parker (1992) reviewed the stony corals. Bennett and Pope (1960) described intertidal zonation patterns on exposed rocky shores, and Dartnall (1974) provided a review of the littoral biogeography of Tasmania and the Bass Strait region. The Marine Invertebrates of Southern Australia handbook series (Shepherd and Thomas 1982, 1989; Shepherd and Davies 1997) is also relevant.

The fauna from a series of sea mounts south of Tasmania has been reported on by Koslow and Gowlett-Holmes (1998). Koslow et al. (2001) reviewed the impacts of trawling on the community structure of these seamounts.

Conservation Studies. In the early to mid 1990s, scientists from the University of Tasmania and the Parks and Wildlife Service carried out a systematic marine biological sampling program at shallow rocky reef sites around the State. These data were used to complete a coastal marine bioregionalisation and for preliminary identification of suitable sites for marine protected areas (Edgar et al. 1995; Edgar et al. 1997).

The Tasmanian Seamounts Marine Reserve is divided into two management zones, a Highly Protected Zone (from a depth of 500 m to 100 m below the sea-bed) in which fishing and mining is banned, and a Managed Resource Zone (from the sea surface to a depth of 500 m) in which long line fishing is permitted. A Plan of Management for this reserve is in preparation (Environment Australia Marine Group 2000).

South Australia

South Australia has a varied coastline which includes the deep embayments of the Gulf St Vincent and the Spencer Gulf, the large Kangaroo Island and several other islands. The fauna is cool temperate and has a number of endemics.

Relevant Institutions. The University of Adelaide and Flinders University undertake ecological studies in marine invertebrates and the South Australian Museum has large collections of marine invertebrates and scientists specialising in polychaetes, crinoids and marine parasites while W. Zeidler (retired) mainly works on amphipods.

SARDI (South Australian Research and Development Institute) undertakes research work related to resource management (e.g., shell fisheries) and biodiversity.

Published Resources. South Australia is unique in Australia in having a handbook series that covers the fauna (and flora) of the state. This includes volumes on crustaceans (Hale 1927, 1929), molluscs (Cotton and Godrey 1938; Cotton 1959, 1961, 1964; Ludbrook 1984; Shepherd and Thomas 1989), sponges, cnidarians, bryozoans, echinoderms, polyclad flatworms, sipunculans, echiurans, and polychaetes (Shepherd and Thomas 1982); insects, brachiopods, tunicates, nemerteans, kamptozoans, pycnogonids, phoronids and hemichordates (Shepherd and Davies 1997). Cairns and Parker (1992) reviewed the stony corals. However, for many groups of crustaceans the most recent handbook is Jones and Morgan (1994).

Conservation Studies. Edyvane (1996) discussed the role of marine protected areas in temperate ecosystem management, and Edyvane completed a report series on marine biodiversity conservation in South Australia for Environment Australia (e.g., Edyvane 1998). The latter document audited the existing system of MPAs in the State, identified poorly conserved habitat types and prioritised bioregions according to the need for further MPAs.

A number of technical reports arising from marine benthic surveys and biogeographical studies in South Australian waters have been produced by SARDI (e.g., Edyvane and Baker 1996a for the St Francis Isles and Investigator Isles,). Hutchings et al. (1993) studied the infauna of marine sediments and seagrass beds of the Upper Spencer Gulf near Port Pirie and subsequently related these communities to levels of heavy metal concentration in the sediments (Ward and Hutchings 1996).

Western Australia

Western Australia's huge coastline ranges from the cool temperate south coast to the tropical Kimberley coast. There are also offshore reefs (Scott Reef, Ashmore). Consequently the marine invertebrate fauna is extremely diverse. There is a tropical fauna in the north that is primarily Indo-west Pacific, although having a number of endemic elements. The middle section of the western coast has a mixed fauna (Wilson and Allen 1987), while the southern section is temperate to cool temperate with many endemic species. However, many tropical species "ride" the Leeuwin current southwards to Esperance and into the Gulf (Pearce and Walker 1991).

Relevant Institutions. The universities undertake studies on marine invertebrate ecology and genetics and the Western Australian Museum has extensive collections from the state and several scientists specialising in some groups of marine invertebrates (molluscs, crustaceans, sponges, echinoderms; although L. Marsh (echinoderms) is now retired). The CSIRO Marine Laboratory at Perth is a joint venture between CSIRO and the WA Government. There are no accessible marine laboratories in the state. WA Fisheries has a hut on Rat Island in the Abrohlos, AIMS had a facility at Dampier which is now closed and the University of WA has a station on Rottnest Island which can accommodate both marine and terrestrial studies.

Published Resources. Popular guides to the marine invertebrate fauna of the state include molluscs (Wells and Bryce 1988, 1993), and the stinging and venomous marine invertebrates of the state (Marsh and Slack-Smith 1986). Faunal reports (including the larger-sized invertebrates) have been prepared by the Western Australian Museum from various offshore islands including Rowley Shoals (Berry 1986c; Berry 1986b), Christmas Island (Berry and Wells 2000b), Montebello Islands (Berry and Wells 2000a), Kimberley region (Wells 1989), offshore reefs (Wells 1990; Wells 2000) and Ashmore and Cartier Reefs (Wells 1986; Berry 1993).

Workshops on the marine fauna and flora have been held in Albany (Wells et al. 1991a), Rottnest Island (Wells et al. 1993; Walker and Wells 1999 – for seagrasses), and Abrolhos Islands (Wells 1997c) and the Dampier Archipelago (Wells et al. in press) with many papers in the proceedings resulting from these workshops containing valuable contributions on various groups of marine invertebrates. The influence of the warm, south-flowing Leeuwin Current on the biota of the southwestern Australian shelf has also been the focus of much attention (Morgan and Wells 1991; Cresswell 1996). For example, intertidal platforms at the western end of Rottnest Island support a number of tropical taxa whose larvae are carried southwards on the warm Leewin current. The Leewin curent and its influence on the coastal climate and marine life were the subject of a published symposium (Pearce and Walker 1991).**]**

There is little published information on the subtidal, soft-bottom communities in western Australia. Among the few studies are several on molluscs from local areas including estuaries (Wells and Threlfall 1980) and sublittoral and shelf communities (Glover and Taylor 1997; Glover and Taylor 1999). Slack-Smith (1990) documented bivalves of Shark Bay. Other papers of note include revisions of the hermit crabs (Morgan 1989; Morgan and Forest 1991) and various papers by Hooper and Fromont on the WA sponge faunas pertaining to particular families (Fromont 1998, 1999).

The Kimberley area has only been sampled by the Western Australian Museum during about the last ten years (the first trip was in 1988 and there have been five trips since then). Previously there were no data, apart from some days of sampling done by the British passing through in the 1890s. Hutchins et al. (1996) published the results of a marine biological survey of the Muiron Islands and the eastern shore of Exmouth Gulf, Western Australia.

The south coast of Western Australia is still not particularly well sampled and, in particular, the area east of Esperance is poorly known[15]. Along the west coast, even Shark Bay, which has been listed as a World Heritage site, is still poorly known, with recent Western Australian Museum fieldwork finding hundreds of new records (F. Wells pers. comm.).

Offshore from Western Australia lie several reefs that have been the subject of some faunal studies. Surveys of marine invertebrates and fishes of these have been published in popular and technical reports; macromolluscs (Wells 1990; Wells et al. 1990), general report on Cartier and Hibernia Reefs, Timor Sea (Russell and Hanley 1993); and some scientific publications (e.g., sponges – Hooper 1994). Berry (1993) reported on surveys of Ashmore Reef and Cartier Island, and Berry (1986a) on Rowley Shoals, Scott Reef and Seringapatam Reef. Coral records from these areas are incorporated in recent monographs by Veron (1986; Veron 1999) and Wallace (Wallace 1999a; 1999b).

The North West Shelf is better known, in part as a result of studies associated with petroleum development. Surveys of the benthos of the Northwest shelf and slope were carried out by CSIRO fisheries between 1985 and 1988 (Wallner and Phillips 1988; Wadley and Evans 1991) and USSR RV 'Akademik Oparin', recognising a unique fauna containing several endemic genera (Hooper 1986; Hooper and Krasochin 1989; Hooper 1991, 1996). Material from these surveys is held in the Western Australian Museum, Northern Territory Museum and Queensland Museum, but the majority remains unworked. Hooper et al. (in press) provide biodiversity estimates for the region. Ward and Rainer (1988) recorded 357 taxa of epibenthic crustaceans, including 308 decapods, from these surveys, and some of the "deep-water" decapod crustaceans were recorded by Morgan and Jones (1987). A review and bibliography of research and data relevant to

[15] Dr F. Wells is organising a marine biology workshop to be held in Esperance in early 2003.

marine environmental management on the Northwest Shelf have been produced by AIMS and CSIRO Marine Research (Heyward et al. 1999; Heyward et al. 2000) and are available on the web[16]. A new survey of shelf and slope communities will be commencing shortly.

Watson (1996) described the distribution and biogeographic relationships of the hydroid fauna of the Australian west coast. Hooper and Lévi (1994) and Hooper (1991; Hooper 1996 and Hooper et al. [, in press #3014) assessed the large-scale biodiversity of some sponge taxa throughout Australasia, including the largely unique WA fauna, and its relationship with other regional Australasian and Indo-west Pacific populations.

Conservation Studies. A recent review of marine habitats on the Western Australian coast has provided recommendations for conservation areas (CALM 1994). Recently Heyward (1999) assessed the biodiversity values of the Montebello Island region. Various studies on the environmental impacts of offshore gas and petroleum drilling on the Northwest Shelf have been carried out (e.g., Pendoley 1992).

Northern Territory

Much of the coast of the Northern Territory is dominated by river estuaries and mangroves with some coral and rocky shores. It has a diverse tropical fauna that is generally poorly known.

Relevant Institutions. The Museum and Art Gallery of the Northern Territory has a good collection of marine invertebrates and three scientists specialising in this area (molluscs, polychaetes and alcyonarians; a crustacean worker is now retired, and recently an honorary sponge worker has been appointed). The Northern Territory University has programs in aquaculture and tropical biology and ecology, including research into mangrove ecology. There are no field stations but recently funding has been obtained for the development of a joint ANU and AIMS facility in Darwin, the Arafin-Timor Research Facility (ATRF).

Published Resources. A marine workshop held in Darwin in 1993 produced a series of papers on the fauna of Darwin Harbour (Hanley and Russell 1997). The Beagle, Records of the NT Museum of Arts and Sciences, contains descriptions and inventories of several hundreds of marine invertebrates, including sponges, soft corals and gorgonians, crustaceans, molluscs, polychaetes etc., and there are several Technical Reports containing overviews of the NT fauna, geomorpholgy, geochemisty and other physical processes of the NT marine province, published by Larson et al. (1988).

There are few checklists or semipopular accounts of the marine invertebrate fauna, although a handbook on the hard corals of Darwin Harbour is currently in press (Wolstenholme et al. in press) and one has recently been published on the genera of octocorals (Fabricius and Alderslade 2001) for the Indo-Pacific.

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[16] Review of research: http://environ.wa.gov.au/nws/documents/review_jan2000.html
Bibliography: http://203.12.67.130/NWS/bibliography.htm

A few groups of marine invertebrates have been documented for the Darwin area including terebellid polychaetes (Hutchings 1997), fiddler crabs (Hagen and Jones 1989), hermit crabs (Morgan 1987), scleractinian corals (Wolstenholme et al. 1997), and sponges (Hooper et al. 1997, which also includes a synopsis of the sponge literature for northern Australia). Bruce and Coombes (1995) recorded 61 species of palaemonoid shrimps (Crustacea, Decapoda, Caridea) from the Cobourg Peninsula, this being about one third known from northern Australia, and Blackburn (1974) listed the molluscs from that locality.

A recent baseline study of Port of Darwin for introduced marine species (Russell and Hewitt 2000) contains species lists for some groups of marine invertebrates.

Conservation Studies. Storrs and Finlayson (1997) produced an overview of the conservation status of wetlands of the Northern Territory, including coastal saltmarshes and mangroves, for the Parks and Wildlife Commission of the Northern Territory. J. R. Hanley documented the benthic invertebrates of mangrove communities in the Darwin area (Hanley 1993a) and the Gulf of Carpentaria (Hanley 1993b).

Queensland

Queensland's extensive coastline ranges from the warm temperate in the south to the tropical eastern coast, along which extends the Great Barrier Reef (GBR). There are also offshore reefs in the Coral Sea, several Coral Sea seamounts, and numerous islands in Torres Strait. The northern coast includes the muddy shores of the western side of Cape York and much of the Gulf of Carpentaria.

Associated with the diverse habitats in Queensland – but particularly with the GBR – is an exceptionally diverse marine invertebrate fauna. The northern coast has a fauna partially shared with the rest of northern Australia and New Guinea (Benzie 1998) but also containing a highly regionalised and apparent endemic fauna for some taxa (e.g., Benzie 1998; Hooper et al. 1999; Hooper et al. in press; Kennedy and Hooper in press). Recently GBRMPA has compiled much of the available data on reefal and inter-reefal areas as part of its Marine Representative Areas Program (Day et al. in press). The fauna associated with the GBR has primarily Indo-west Pacific origins, although there are also some endemic elements, especially in the coastal waters inside the reef. The coastal areas inside the GBR are relatively poorly known for marine invertebrates compared with the GBR itself. Nevertheless the faunas of these areas appear to have a higher level of endemism than those of the GBR, and also have very high diversities (see for example Birtles and Arnold 1983; Cannon et al. 1987; Birtles and Arnold 1988; Hooper et al. 1999). There are also endemic elements associated with the Coral Sea seamounts. The southern part of Queensland has the large embayment of Moreton Bay that contains an invertebrate fauna quite different from that of the sheltered muddy areas inshore of the GBR, in addition to a significant apparently endemic component (Davie and Hooper 1997; Davie 1998). This area and the coastal areas south of Gladstone lack emergent offshore reefs although with several significant submerged coral reefs. They contain some endemics but are primarily a transition zone between the Solanderian and Peronian faunas (those found in NSW versus in the tropics).

Relevant Institutions. Two of the universities in Queensland, The University of Queensland and James Cook University, Townsville (the latter also with a campus in Cairns), have been centres for marine biology for many years. Queensland University has recently expanded its School of Marine Studies and its research interests are heavily focussed in Moreton Bay and around Heron Island. In addition, the University of Southern Queensland and the University of Central Queensland in Rockhampton both have a strong marine component, and similarly Griffith University, with Brisbane and Gold Coast campuses, has a strong marine ecology group. These and the other universities mainly undertake studies on marine invertebrate ecology and biology and are major partners in 17 National Cooperative Research Centres (CRCs). The Queensland Museum (including the Museum in Brisbane and the Museum of Tropical Queensland in Townville) has extensive collections of sessile and other marine invertebrates, including parasitic and meiofaunal worm phyla. The QM's marine research capabilities presently comprise 10 scientists specialising mainly in marine protozoans, sponges, soft corals, gorgonians, hard corals, parasitic and free-living marine worm phyla[17], decapod crustaceans, ascidians, hemichordates and bryozoans). The Museum of Tropical Queensland, in Townsville, contains one of the world's most significant Hexacorallia collections and many deep-water species and has benthic collections from numerous benthic surveys carried out in the GBR lagoon. The Northern Fisheries Research Centre (DPI) in Cairns has a strong seagrass research group. CSIRO Marine Research, Cleveland, have been proactive in their extensive surveys of the inter-reef fauna of the GBR in particular, concerning the effects of fishing on the macrobenthos (Pitcher 1997; Pitcher et al. 1997; Poiner et al. 1998; Pitcher et al. 1999), and are also heavily involved in prawn and demersal fin fisheries. The Australian Institute of Marine Science (AIMS) is a focal point for major research programs on the GBR, coastal mangrove systems and oceanography of the northeast Australian marine biome as well as coral reef ecology. In addition, there is the CRC for Coral Reefs based in Townsville that is a partnership of AIMS. JCU, DPI, GBRMA as well as commercial interests on the reef, and it is tending to concentrate on answering management issues of the GBR province.

Published Resources. Several popular and semi-popular books include information on the invertebrates of the GBR, including early works such as Roughley's *Wonders of the Great Barrier Reef*, first published in 1936 (Roughley 1936) the *Reader's Digest Book of the Great Barrier Reef* (Talbot and Steene 1984), and Bennett's (1986) *The Great Barrier Reef*. The extensive reports of the Great Barrier Reef Expedition, which took place from 1928 to1929, were published over many years and cover many groups of marine invertebrates. A narrative of the expedition and of the corals of the GBR is given by Yonge (1930). There is a popular guide to the corals by Deas and Domm (1976), and a major monographic treatment of the staghorn corals by Wallace (1999a). Mather and Bennett(1984) edited a coral reef handbook on the fauna, flora and geology of Heron Island and adjacent reefs. A number of books dealing with corals and coral reefs worldwide (e.g., those by Veron 1986; Veron 1995b; Veron 1999) also cover the GBR,

[17] L. Cannon, who is about to retire.

and Wallace (1999a) monographed the staghorn corals (genus *Acropora*). Opisthobranchs are covered by Marshall and Willan (1999).

Davie (1998) edited a field guide to the wildlife and habitats of Moreton Bay and the Queensland coast, and the University of Queensland has also published a book on Moreton Bay and its catchment (Tibbets et al. 1997).

General access to field stations, such as Lizard Island and Low Islands Research Station in the northern part of the GBR, Orpheus in the Palm Group off Townsville and Heron Island and One Tree Island at the southern end of the GBR, have enabled Australian and overseas scientists to undertake many studies that have substantially improved our knowledge of the biology and ecology of reef invertebrates. Others have produced important reviews of otherwise poorly known invertebrate groups. Among the many possible examples, Ryland and Hayward (1992) and Hayward and Ryland (1995) reported on the Bryozoa of Heron Island, and Marshall and Willan (1999) on the nudibranchs of Heron Island and Wistari Reefs. There are also important field stations on One Tree Island (University of Sydney), Low Isles (University of Queensland) and Orpheus (James Cook University), all on the GBR, and a substantially rebuilt facility on North Stradbroke Island (University of Queensland), off Brisbane. All of these stations are available to non-University staff.

Conservation Studies. The GBRMPA has recently reviewed the status of the GBR (Lucas et al. 1997), produced a State of the Reef report (Wachenfeld et al. 1998) and is undertaking a substantial review of bioregional planning of the GBR. Long-term monitoring of coral cover, crown-of-thorns etc. is carried out by AIMS and the results are placed on their web site, including up to date information on the current bleaching episode.

2.3.2 Coral Sea

The Coral Sea is an area of ocean adjoining the NE coast of Australia and includes the waters of the GBR and Torres Strait. Its western boundary is with the Arafura Sea, and to the north lies the Solomon Sea. The eastern boundary includes Vanuata, Nokanhui in New Caledonia and Norfolk Island. To the south lies the Tasman Sea. In 1969 and 1973 Australia proclaimed the Coral Sea Islands Territory which applies to all islands within an area limited by the GBR to the west, by the 157º 10 meridian to the east and the 12º latitude to the north and 24º latitude to the south, this being an area of 350,000 km^2. It includes a number of islands and reefs briefly described below. Between the relatively shallow Coral Sea platform and New Guinea lies the Coral Sea Basin up to 4000m deep. A complex area where the Norfolk and Lord Howe rises meet lies between the southern GBR and New Caledonia where reefs such as Chesterfield and Frederick are found, although these latter reefs lie outside Australian jurisdiction.

Torres Strait Protected Zone

The Torres Strait is located between the tip of Cape York and Papua New Guinea. It consists of over a hundred islands and reefs which have evolved from four major origins; volcanic, alluvial, coral cays, and flooded land bridges which were once part of the Great Dividing Range. Eighteen are currently inhabited.

The Torres Strait Treaty was entered into by Australia and Papua New Guinea in 1985. It is concerned with sovereignty and maritime boundaries in the areas between the two countries and the protection of the way of life and livelihood of traditional inhabitants and the marine environment.

In the Australian component of the TSPZ, the Protected Zone Joint Authority (PZJA) manages (among others) those fisheries which Australia and Papua New Guinea have agreed to jointly manage, including prawns, pearl shell, tropical rock lobster. The PZJA agreed in October 1996 that all commercial fishing in Torres Strait shall come under PZJA management, including (from April 1999) crab, trochus and bêche de mer.

Relevant Institutions. None, although there are plans by James Cook University to establish a field station in the area.

Published Resources. GBRMPA undertook a Torres Strait Baseline Study (e.g., Dight 1991; Dight and Gladstone 1994). Pitcher et al. (1992) mapped the distribution of seagrasses, substratum types and epibenthic macrobiota in Torres Strait, with particular reference to pearl oyster abundance.

Conservation Studies. The impact on commercially important invertebrates in the Torres Strait from pollution caused by heavy metals from the Fly River is described by Turnbull(1990), Phillips and Crossland (1990), Murphy et al (1990) and Denton and Heitz (1990).

Islands and reefs of the eastern Coral Sea

A number of reefs occur to the east of the GBR on the Coral Sea platform - a relatively shallow sea which extends about 300 kms east of the GBR and is therefore included within the Coral Sea Territory. These include Heralds Beacon Island, Osprey Reef, the Willis Group and 15 other reefs and island groups. Several studies of the marine invertebrates of these reefs have been published, or are currently in progress (e.g., Reitner et al. 1999). Osprey Reef is currently being studied with regards to rates of bioerosion of coral substrates (Tribollet in press). However, it is fair to say that our knowledge of the invertebrate fauna of the Coral Sea reefs and seamount fauna is rudimentary and that museum collections from these areas are sparse.

The Kermadec Islands lie on the southern edge of this area and share faunal elements with Lord Howe and Norfolk Islands. In the Kermadec Islands, Cole et al. (1992) studied abundance patterns of subtidal benthic invertebrates and fishes, and the coastal molluscan fauna of the northern islands has recently been listed by Brook (1998).

Queensland Plateau

This area occurs on the western side of the Coral Sea and includes the lower shelf and upper bathyal areas of the NE continental margin of Australia. The Queensland Plateau is the area of the Coral Sea containing Osprey Reef in the north down to Holmes Reef etc. in the mid section, to Saumarez Reef in the south and out to Wreck Reef in the east. It is a different slab of continental crust than the east coast, and should be treated separately from the GBR reefs from which if differs geologically. Preliminary explorations reveal a relect fauna below the old Miocene (22.5-5 MYA) and Pleistocene (2.5MYA) sea level stands (150-200m depth) (J. Hooper, pers. comm.). These remnant living fossils are of great interest in terms of biodiversity and geobiological sciences. Davies (1994) considered that the Queensland Plateau was the cradle of the GBR, an external gene pool providing the GBR with a supply of coral larvae after catastrophic events (such as sea level low stands) enabling quick recovery and rebuilding of coral reefs along the Australian NE coast with rising sea levels. Coral reefs have existed on the Queensland Plateau since the Miocene even during times of the lowest sea leves (up to 200 mm below present). However little sampling has occurred in the area. Similar areas geologically on the Norfolk Ridge and the slopes of New Caledonia have revealed a relict fauna and many living fossils and endemics, and presumably these will also occur on the Queensland Plateau (J. Hooper, pers. comm.)

2.3.3 Tasman Sea – Lord Howe, Norfolk Is, Elizabeth and Middleton Reefs and Tasman Sea Mounts.

Lord Howe Island and Elizabeth and Middleton Reefs are situated on the western margin of the Lord Howe Rise which is approximately 320 km wide and 1575 km long. It is separated from the southern GBR and the Chesterfield reefs by the Chesterfield Trough and from New Caledonia by a narrower trough of similar depth (3500-4500 m). Lord Howe Island is situated near the middle of a roughly oval-shaped submerged volcanic platform (27 by 18 km) with general depths of 20-60 m. This is only one of at least 20 known large volcanic peaks in the Tasman Sea.

There has been little attempt to sample the shelves and deeper areas around these areas by Australian research workers, largely due to the scarcity of suitable ship facilities and financial support for these activities. In contrast, the New Zealand Oceanographic Institute has undertaken a number of sampling cruises in these areas that have sampled shelf and deeper areas. Unfortunately, however, this material remains largely unworked.

Lord Howe Island

Lord Howe, 600 km off the coast of northern NSW, is administratively part of that state. It has a population of about 400, plus tourists, and is surrounded by the southernmost coral reefs in the world where temperate and tropical faunas uniquely mix. The significance of the region was recognised by its inclusion on the UNESCO World Heritage List in 1982. Lord Howe and Ball's Pyramid lie atop two closely adjacent seamounts, both surrounded by a shelf 3-6 nm wide that, at about 200 m, falls off very steeply to depths greater than 2000 m.

The marine biota of Lord Howe is known to have a number of endemic species in shallow water and on the shelf, but the deeper water fauna is unknown. There has never been a full-scale survey of the marine invertebrates of Lord Howe Island, although there is some information (reviewed by Pollard and Burchmore 1985). By far the best known group is the corals (Veron and Done 1979; Harriott et al. 1993; Harriott et al. 1995; Pichon 1995), Lord Howe having the world's southern-most coral reefs. Harriott et al. (1993) undertook a baseline study to determine the status and structure of the "marine benthic (coral) communities" and compared their findings with published records. They also looked at coral recruitment and *Acanthaster* abundance. A total of 59 coral species were recorded during their study (including 19 not previously recorded from Lord Howe), bringing the total to 83 species of corals known from the island (c.f. 356 from the GBR), a very high diversity considering the latitude and isolation of the island (Harriott et al. 1993; Harriott et al. 1995). Other groups of marine invertebrates have not been synoptically reviewed and are rather poorly known. For example, there are five described species of ascidians known from the island but perhaps 50 occur there (P. Mather pers. comm.). A number of endemic marine molluscs are known (Allan and Iredale 1939; Iredale 1940; Ponder 1981).

Ponder et al. (2001b) reported on the marine invertebrates on the Lord Howe Island shelf based on collections in the Australian Museum. The following text is taken from that report. They found 3 crustaceans, 16 annelids, 12 echinoderms, 1 sipunculid, 1 bryozoan, 360 molluscs and 7 brachiopods. These records do not cover the entire fauna as some groups are clearly under-represented and others not recorded. A single sample from 2738 m consisted of very fine foram ooze and clay and contained only 8 molluscs of which only one could be attributed to a named species. Of the 207 species of Mollusca recorded from the shelf, 130 are known to have a range outside the EEZ surrounding Lord Howe Island and Ball's Pyramid. Of the remainder, 13 are endemic to Lord Howe and one to Ball's Pyramid, while the identity of the remaining taxa is not sufficiently well known to ascertain their status. Thus, at least 10% of the 143 sufficiently well known taxa are apparently endemic to the Lord Howe Island shelf. Only three of these 13 endemic taxa are named, all occurring in shallow water. The remaining 10 taxa are known only from the dredge samples on the shelf. There are four (21%) probable endemics of the 19 identified non-mollusc invertebrate taxa. The overall endemism is relatively high (11.3 %), 9.4 % of which are known only from the samples on the shelf. Although the level of endemism in the shallow water fauna is not well understood, for those groups (corals, fishes, echinoderms) where it is known, it is considerably lower. Because commercial trawling has apparently not occurred on the shelves around Lord Howe and Ball's Pyramid, they will be relatively pristine compared with other shelf areas in Australia. This, combined with the relatively high endemism in the fauna, indicates the high conservation value of the area.

Relevant Institutions. Stationed on the island is a representative of NSW Marine Parks who has been assisting in the development of management plans for NSW Marine Park that is surrounded by protected Commonwealth waters. Recently a small laboratory has been established on the island, with preference given to research focussing on management issues. The Australian Museum has significant collections of marine invertebrates from the island.

Published Resources. Pichon (1995) noted the glaring lack of data on the benthic biota from the Commonwealth waters around Lord Howe and Ball's Pyramid. Ponder et al. (2000) reported on the invertebrates on the shelf of Lord Howe Island and Balls Pyramid and Lowry (1989) gave an overview of a survey of the benthic invertebrates from the Lord Howe Rise. There are also published reports on the corals (see above).

Conservation Studies. The area has recently been proclaimed as a Marine Park and a Plan of Management is currently being developed (EA 2001b, a).

Norfolk Island (including Philip Island)

Norfolk and Philip Islands lie about 1676 kms to the NE of Sydney at 29 02 S 167 56 E, and 1063 km from Auckland. They are small islands (Norfolk is about 8 kms long, with a coastline of 32 kms) surrounded by a narrow shelf that forms the top of a volcanic sea mount. Philip Island is uninhabited and Norfolk has a population of about 2000. The island only has a couple of landing points, being largely surrounded by basaltic cliffs. Philip Island is 6.24 km from Norfolk and is much smaller, being only 258 ha in area. Sea temperatures range from about 24.5 C to 17 C. The island is administered by its own Legislative Assembly.

There do not appear to be any studies on the conservation of marine invertebrates of these islands, nor are there any published reviews of note.

The marine fauna of Norfolk Island is surprisingly understudied. There are some corals, but most of the island, and nearby Philip Island, is surrounded by steep rocky shores. Nothing of a synoptic nature has been published on the marine invertebrate fauna and much basic work remains to be done. For example, there are ten described ascidians known, but around 50 remain undescribed (Mather pers. comm from surveys by Neville Coleman). The fauna is allegedly similar to that on the Kermadec Islands and Lord Howe Island, being a mixture of temperate and tropical elements with some local and regional endemics (e.g., Iredale 1940).

Elizabeth and Middleton Reefs

Middleton and Elizabeth Reefs lie about 500 kms to the east of Coffs Harbour and consist of two coral reef-capped seamounts. Environment Australia is responsible for the management of these reefs. The Australian Museum carried out a survey of Elizabeth and

Middleton Reefs (Hutchings 1992c) that revealed a limited number of endemic species and gave detailed species list for corals, molluscs, echinoderms and decapod crustaceans. Other invertebrate groups have been sorted to family or below and are available for study in the Australian Museum.

The Elizabeth and Middleton Reefs Marine National Nature Reserve (Commonwealth) was declared in 1987. The second Plan of Management for the Reserve came into force on 15 March 1994, and is valid for ten years (unless revoked)[18].

Tasman seamounts

These sea mounts and others (e.g., those south of New Caledonia, north of Elizabeth and Middleton Reefs) have revealed the existence of previously unknown, rare or endemic species (Pichon 1995). However, data is scarce, these potentially very interesting faunas being very poorly studied (see also Section 5.3.1).

An Australian Museum cruise on RV *Franklin*, on which some sampling was undertaken on some of the Tasman Seamounts, was reported on by Lowry (1989). Richer de Forges (1992; 1993) reported on deep-sea crabs found on this cruise. The seamounts are under Commonwealth jurisdiction.

2.3.4 Indian Ocean – Christmas and Cocos Is

These islands lie in the Indian Ocean and have typical tropical Indo-West Pacific faunas.

Christmas Island

Christmas Island (10º 25’22”S, 105º 39’59”E), about 2800 km west of Darwin and 360 km south of Java, is a mid-oceanic atoll only 19km long and 135 km^2 in area, which rises steeply from deep oceanic waters. Its 80 km coastline consists of almost continuous steep limestone cliffs, with a few shallow bays and small sand and coral beaches, and only one anchorage and landing point (Flying Fish Cove). The island, which has an encircling coral reef, has a very narrow continental shelf dropping off to a depth of about 5000 m within 200 m of the shore (DTRS 2000b). There are thus relatively limited areas of habitat for shallow water reef taxa. Its marine biota consists almost entirely of widespread Indo-West Pacific forms. The line defining the EEZ to the north of Christmas Island is shared with Indonesia.

The island has been mined for phosphate since 1895. A national park of 1,600 hectares was declared in 1980. In December 1989, this (the Christmas Island National Park) was extended to cover over 60% of the island, including much of the coastline out to 50 m from the island shores, to protect the wave pools and limited areas of reefs surrounding the island. At its first meeting, the Christmas Island National Park Advisory Committee, which includes, *inter alia*, representatives of the Christmas Island community, unions and mine management, proposed that it be further extended to 150 m from shore to include

[18] http://www.ea.gov.au/coasts/mpa/

the total reef platform. This proposal is currently under consideration. The Commonwealth Government is responsible for the marine resources of the island.

Published Resources. A popular book is available on the molluscs (Wells et al. 1990) and the Western Australian Museum undertook a survey for Environment Australia of elements of the marine fauna, including some groups of invertebrates (Berry and Wells 2000b). Hicks (1980) provided a natural history bibliography of the island.

Conservation Studies. The Western Australian Museum has produced a report (Berry 1988) for Environment Australia, who funded a survey of the marine fauna of the island which contains species lists for some invertebrate groups. The Institute of Marine Ecology at the University of Sydney also carried out a study (also for Environment Australia) on the impact of harvesting of marine invertebrates and reef fish in the Christmas Island National Park (Institute of Marine Ecology 1993).

Cocos (Keeling) Islands

The Cocos (Keeling) Islands (12 05 S 96 53 E) consist of two small, mid-oceanic atolls, lying about 3685 km west of Darwin and 2768 km NW of Perth. The nearest land masses are Christmas Island, 900 km to the northeast, and Java, 1200 km to the north. The group consists of 27 small coral islands in two separate atolls with a total land area of only about 14 km^2. The southern atoll consists of a horse-shoe chain of 26 islands around a large lagoon. The northern atoll (North Keeling), about 24 km from the main group, is a single uninhabited island and relatively undisturbed. It is an important breeding site for a variety of seabirds and is managed as Pulu Keeling National Park by Parks Australia (Environment Australia). The majority of the population lives on two of the islands, Home Island and West Island. A range of marine resources has been traditionally fished for food (Caton et al. 1998).

Published Resources. Atoll Research Bulletin Nos. 399-414 (Woodroffe 1994) deal with the ecology and geomorphology of the Cocos (Keeling) Islands, including summaries of marine habitats (Williams 1994), hermatypic corals (Veron 1994), marine molluscs (Wells 1994), echinoderms (Marsh 1994), barnacles (Jones 1994) and decapod crustaceans (Morgan 1994). Faunal studies of the corals, echinoderms, molluscs and crustaceans were made by Gibson-Hill in the 1940s, but, except for one survey by Maes (1967) on molluscs there were no modern studies of marine invertebrates prior to those contained in this 1994 volume (Williams 1994). Several groups of marine invertebrates remain unstudied, for example there are only two described species of ascidians known from Christmas and Cocos Islands, whereas there may be as many as 50 undescribed species (Mather pers. comm)

Conservation Studies. Lincoln Smith et al. (1995) reported on the management of the harvesting of marine organisms.

2.3.5 Antarctic and sub-Antarctic

The Antarctic marine invertebrate fauna is, in general, better known than most of the Australian fauna, having been the focus of attention of several early expeditions and many subsequent studies undertaken by scientists embarking on programs funded by several countries. This applies even to groups relatively poorly known in Australia. For example, there are about 160 described ascidians, of which 150 are endemics, with only around 20 still undescribed (P. Mather pers. comm.) and there are 135 described and 86 undescribed (87 endemic) bryozoans (D. Gordon pers. comm.).

Guides to various Southern Ocean marine invertebrate taxa are provided in various ANARE Research Notes and Reports (e.g., Kott 1957; Kramp 1957; Kirkwood 1982, 1983; Hosie and Cochran 1994).

Australian Antarctic Territory

The Australian Antarctic Territory occupies a large part of the western half of Antarctica (nearly six million km^2, or about 42% of the 13.9 million km^2 of Antarctica, in addition to about 2.2 million km^2 of ocean[19]; H. Stagg, H. Brolsma pers. comm.). The continent is surrounded by a "cold ring ocean" which developed 20-30 million years ago, making it one of the most discrete and thus zoogeographically isolated marine ecosystems in the world (Dayton 1990; Arntz et al. 1994). The marine systems are characterised by low, but relatively constant water temperatures, permanent or persistent ice cover in winter and occasionally during the entire year, as well as strong seasonality in light regime and primary production (Hempel 1985). The Antarctic, in contrast to the Arctic (and enclosed ocean surrounded by land masses) is a continental landmass surrounded by a deep shelf (500 to 800 m). Since the Antarctic continent is covered by ice, very little freshwater inflow occurs.

Since the end of the last century, successive expeditions have provided information on the main characteristics of Antarctic marine benthos, such as the low diversity of higher taxa, the high degree of endemism, gigantism, or particular developmental types. In particular, the predominance of non-pelagic development in polar marine invertebrates has been noted. While Antarctic marine invertebrates, as those elsewhere, display a great variety of life-history traits and reproductive strategies they tend to have a higher proportion of direct developing (including brooding) species than faunas from other areas, a factor probably leading to increased speciation (Poulin and Féral 1996).

While the Antarctic marine invertebrate fauna is rather well known compared with most other areas dealt with in this report, in recent years the Australian Antarctic research program has done little to advance our knowledge of the invertebrate fauna of the Australian Antarctic Territory, despite extensive marine surveys being undertaken).

Relevant Institutions. Australian Antarctic bases include Mawson (67º 36' S 62º 52' E; established 1957), Davis (68º 35' S 77º 58' E; established 1957) and Casey (66º 17' S 110º 32' E; established since 1969). There are also a number of bases in Antarctica operated by

[19] It is impossible to calculate a precise area for the Antarctic EEZ because of the inaccurate coastline

other nations. Australian Antarctic studies are undertaken by the Australian Antarctic Division based in Hobart[20].

Published Resources. Early studies include those provided by expeditions, including the Australasian Antarctic Expedition. More recent reviews of the fauna include molluscs (Powell 1960; Dell 1990), echinoderms (Piepenburg et al. 1997; Dahm 1999), etc. Both Everitt et al. (1980) and Kirkwood and Burton (1988) described the benthos from near Davis Station in eastern Antarctica.

General reviews of Antarctic benthos are provided by Dell (1972) and Arntz et al. (1994), who discuss aspects of the zoogeography, biodiversity, ecology, biology and conservation of the fauna.

There are many recent studies on Antarctic macrobenthos conducted outside the Australian Antarctic Territory (e.g., Galéron et al. 1992; Piepenburg et al. 1997; Gutt and Starmans 1998; Sahade et al. 1998; Starmans et al. 1999; Barnes and Arnold 2001; Clarke 2001) and regional differences are apparent. Assemblages of suspension feeders dominated by sponges and bryozoans are prevalent on the shelf of the eastern Weddell Sea, but almost absent in the Bellingshausen and Amundsen Seas also in eastern Antarctica. These assemblages seem to be restricted to areas where bottom currents provide favourable feeding conditions. Mobile deposit feeders are more abundant in both these regions where there is a soft bottom substrate with presumably slow bottom currents (Starmans et al. 1999). Recent reviews of other groups of marine invertebrates from the Antarctic include sea spiders (Pantopoda, Pycnogonida) (Kuznetsov and Turpaeva 1997), corals (Keller and Pasternak 1996), planktonic copepods (Razouls 1994) and McClintock et al. (1997) reviewed the chemical ecology of Antarctic marine invertebrates. The biogeography and ecology of the Southern Ocean decapod fauna were reviewed by Gorny (1999).

Conservation Studies. The dominant species of euphasiid crustacean in Antarctica is the Antarctic krill, *Euphasia superba*. This and various other species of euphasiids (krill) are fished commercially by Ukraine, Poland and Japan and are used for human consumption and aquaculture. International concerns about how the potential overfishing of krill might affect other Antarctic wildlife has led to the signing of an innovative treaty, the Convention for the Conservation of Antarctic Marine Living Resources (CCAMLR). The international agreement on the krill fishery is part of the Antarctic Treaty System, and came into effect in 1982. The Convention takes an ecosystem approach to fisheries management and recognises the place of krill at the centre of the Antarctic food web. The agreement has set limits to catches that take into account the needs of the other animals in the ecosystem. Limits are conservative, even though scientists still do not know the full extent of the krill resource or have a complete understanding of their biology (Rockcliffe and Nicol 1999) despite numerous studies in recent years (e.g., McClatchie et al. 1994; Siegel and Loeb 1995; Godlewska 1996; Brierley et al. 1997; Opalinski et al. 1997).

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[20] http://www.antdiv.gov.au/

Impacts from research work, while generally small, may be significant in some areas. Gutt (2001) pointed out that that in the Weddell Sea, where long term fisheries research has been carried out, the area of sea floor impacted by icebergs was about 2000 times greater than that affected by research trawling and that the use of less damaging technology (underwater cameras, etc.) was increasing.

Cripps and Shears (1997) described the impact of a minor diesel fuel spill on the marine environment at an Antarctic research station while Lenihan et al. (1995) reviewed anthropogenic and natural disturbances to marine benthic communities in Antarctica.

Development of a conservation strategy for the Antarctic was the focus of the 1993 Fenner Conference on the Environment (Handmer and Wilder 1993). Abbott and Benninghoff (1990) stress the need for long-term observations and monitoring to detect and determine the significance of changes in Antarctic ecosystems.

Macquarie Island

Macquarie Island lies at 54˚29’S, 158˚58’E in the Southern Ocean about 800 nm SE of Tasmania, midway between Tasmania and the Victoria Quadrant of the Antarctic continent. The island is small (34 km long and to 5.5 km wide) and, together with small rocky outcrops to the north and south, lies on the central section of the Macquarie Ridge which runs south from New Zealand. Water temperatures usually vary from 2-7˚C (mean monthly sea temperature 7.2 ºC in Jan. and 2.8 ºC in July) and, although the Antarctic convergence lies only 40 km to the southeast, the island is the southern most ice-free shore in the eastern section of the Southern Ocean.

Relevant Institutions. Macquarie Island is governed by Tasmania. The whole island has been a Tasmanian state nature reserve since 1972 and has been managed by the Tasmanian Parks and Wildlife Service. An ANARE base on Macquarie Island has functioned continuously since 1948. In 1996 Macquarie Island was nominated (jointly by the Commonwealth and Tasmanian governments) by the Australian Government for inscription on the World Heritage Register, mainly for its unique geological features.

Published Resources. 52 echinoderm species are recorded from off Macquarie Is and the Macquarie Ridge, at least four of them endemics (O'Hara 1998a) and 64% of the molluscs are endemic (Register of National Estate Database). Lowry et al. (1978) reported on a biological expedition to Macquarie Island and a popular account was provided by Bennett (1971).

Conservation Studies. The effects on, and recovery of, benthic communities following a small oil spill was described by Smith and Simpson (1995; 1998) and Simpson et al. (1995). In addition, Smith and Simpson (1998) and Smith (2000) have investigated the impact of sewage disposal from the station on the invertebrate fauna.

A brief benthic survey (Butler et al. 2000) was carried out east of Macquarie Island using video surveys and epibenthic sleds. This area had a steep rocky sea floor with patches of silty sediment at shallow depths. Benthic macrofauna is sparce, with only 102 species of epibenthic invertebrates collected with at least 10 new. Based on this very limited survey, it appears that this is a biogeographic contact zone. Some species have ranges extending to the Australian and New Zealand regions, others extend to the south, still others have affinities in island groups to the west or wide circumpolar ranges. They suggested that any future expansions of the marine park at Macquarie should include portions of the Macquarie Ridge to the north of the island that appear to support a wide range of benthic biotopes. They also suggested that management arrangements for the Park should preclude bottom-contact fishing gear and mining activities that disturb the seafloor.

Heard and McDonald Islands

Heard and McDonald Islands lie in the southern Ocean about 2500 nm SW of Fremantle and approximately midway between South Africa and Australia. Heard Is (53 01 S 73 23 E) is a volcanic island about 43 km by 21 km with an active volcano. The McDonald Islands are a group of three small rocky islands about 26 nm west of Heard Island.

Relevant Institutions. The administration of these islands has been the responsibility of the Commonwealth Minister for Science since 1972. A station was built on the island in 1947 but closed in 1955, although a basic camp has been maintained.

Published Resources. A review of the shallow water macrofauna is provided by Edgar (2000). There is a report on the echinoderms (O'Hara et al. 1999a) and a few of the common molluscs are detailed by Dell (1964).

2.3.6 Summary – state of knowledge by States and Territories

Table 2.1 (next page) provides an overview of the state of knowledge of the marine macro-invertebrate faunas of Australian states and territories.

2.4 Biodiversity, endemism and distribution patterns in the Australian marine invertebrate fauna

The Australian marine invertebrate fauna is among the most species-rich and diverse on earth (Wilson and Allen 1987). For instance, Australia has the highest global levels of biodiversity – compared to other geographic areas of similar size – for several marine invertebrate taxa (Zann 1995) including bryozoans (Bock 1982), ascidians (Kott 1985, 1990a, 1990b, 1992a, 1992b), crustaceans (Poore 1990), molluscs (Beesley et al. 1998) and sponges (Hooper et al. in press). For example, Hooper and Lévi (1994) estimated that our territorial waters contain around one-third of the world's sponge diversity (~5000 species out of an estimated 15,000 species worldwide).

Table 2.1: Summary table of the general state of knowledge of the better known groups (larger molluscs and crustaceans, echinoderms, polychaetes), based on a subjective score between 0-3.

0 - unknown or virtually unknown; 1 - very poorly known (one or a few collections, mainly unworked); 2 - reasonably well collected and described, with few if any biological or ecological studies; 3 - well collected and described, with some biological and ecological studies. NA - not applicable

	Intertidal-shallow	Shelf	Slope	Abyssal
NSW	3	2-3	2	1
Victoria	3	2-3	2	1
Tasmania - east and north coast	3	2-3	2	1
Tasmania - west coast	2	1-2	1	1
South Australia	2-3	2-3	2	1
Southern Western Australia	2-3	2	0-1	1
Mid-Western Australia	2-3	2	0-1	0
Scott Reef etc.	2	1	0	0
Northwestern Australia	1-2	1	1	0
Northern Territory	2	2	?	NA
Queensland – north coast	1	1	NA	NA
GBR region Queensland	2-3	2	1	1
Southern Queensland	2-3	2	1	1?
Queensland Plateau	1	0-1	0	0
Coral Sea	1	0-1	0	0
Lord Howe Is	1-2	1	0	0
Tasman Seamounts	NA	1	0	0
Middleton and Elizabeth Reefs	2	0	0	0
Norfolk Id	1-2	0-1	0	0
Cocos Keeling Ids	2	0	0	0
Christmas Is.	2	0	0	0
Antarctic Territory	NA	2	0-1	0
Macquarie Id	3	0?-1	0	0
Heard Id	1	0?	0	0

This diversity has been attributed (e.g., Wilson and Allen 1987) to a combination of factors, including:

- The long history of isolation which has fostered the development of the highly endemic southern temperate fauna, which also contains Gondwanan elements and recruits from the Southern Ocean;

- The location of the northern Australian coast in the enormous Indo-West Pacific region, a centre of diversity for many taxa (the Australian continental shelf fauna acquires recruits and genetic diversity from this region);
- The extent and wide latitudinal range of Australian waters, from tropical to temperate to polar waters;
- The enormous coastline (including the longest stretch of south-facing coastline in the Southern Hemisphere (Edyvane 1996) and the extensive and stable continental shelf, providing a great diversity of habitats for colonisation and speciation;
- The characteristic low nutrient status of Australia's coastal waters, with low nutrient regimes generally promoting biological diversity and co-evolutionary strategies to rapidly harvest, use and recycle limited resources (Edyvane 1996).

2.4.1 Geological history and origins of the fauna

A summary of the geological history of the Australian continent, and the way it has affected the biogeography and diversity of shallow-water marine biota, can be found in Poore (1995a). Newman (1991) discusses a number of hypotheses, relating to geological and biological events, about the origins of endemism in the marine fauna of the Southern Hemisphere. Only a few key points are summarised here.

The present distribution patterns of many taxa in the Australasian region reflect the interaction of geological (tectonic), oceanographical and biological events associated with the final breakup of Gondwana and the subsequent dispersion of the continental fragments during the early Cainozoic, around 125 million years ago (mya) (Zinsmeister 1982).

The three most important tectonic events affecting the biogeography of Australia’s coastal and shelf fauna since this time were (Poore 1995a) (1) the separation of Australia from other Gondwanan continents before and during most of the Mesozoic; (2) separation from eastern Antarctica 40-35 mya enabling circum-Antarctic oceanic circulation to develop (causing major cooling) but also enabling the mixing of previously separated faunas; and (3) the collision of Australia with southeast Asia around 20 mya, leading to the evolution of island arcs that provided additional opportunities for faunal exchange. Although our region has had much the same configuration for the last 2 million years or so, Bass Strait and Torres Strait have opened and closed several times due to around 200m fluctuations in sea level during the Pleistocene, this affecting the continuity of the fauna and resulting in east-west species pairs (Dartnall 1974; Poore 1995a).

2.4.2 Distribution patterns and major faunal regions

The biogeographical patterns in the Australian fauna reflect the tropical and temperate characteristics of the northern and southern faunas, and latitudinal transition between these two extremes along the east and west coasts. The marine biogeographical regions recognised have been based on physical and biotic characteristics, including information on some of the larger-sized, better known invertebrates, but relatively little attention has been paid to most invertebrate groups due to lack of data. Wilson and Allen (1987), for example, discussed their biogeographic scheme in the light of distribution patterns of fishes, molluscs, corals and echinoderms around the Australian coast.

The northern tropical waters form part of the Indo-West Pacific biogeographic region and support an extremely rich and diverse species assemblage. In contrast, the temperate southern Australian oceans support fewer species with very high species endemicity while the east and west coasts have a mixture of transitional and endemic faunas (Wilson and Gillett 1971; Wilson and Allen 1987; O'Hara and Poore 2000). The boundary between the tropical and warm-temperate provinces coincides approximately with 18-20ºC winter minimum surface temperature, this boundary shifting due to variations in the East Australia Current and the Leeuwin Current (Poore 1995a). The present faunal composition results from contributions from two very different early Tertiary biotas, the pan-Pacific Tethyan biota (the forerunner of the Indo-West Pacific biota), and the temperate Palaeoaustral fauna (Poore 1995a).

Tropical region

The tropical Indo-west Pacific Region has a much richer diversity than that of other tropical regions (Briggs 1974) with the centre of diversity being in the central part (between the Philippines, Indonesia and New Guinea). The northern Australian coast is adjacent to this region of highest diversity, and the fauna contains many widespread tropical taxa. However, there is some endemism, including differences between the northeastern and northwestern coasts, due to the closure of Torres Strait during some of the Pleistocene (Wilson and Allen 1987), and its maintenance is assisted by southerly flowing currents on the east and west coasts (Poore 1995a). The shallow water and extensive muddy areas may also differentiate this area.

Temperate region

The temperate marine fauna of southern Australia is derived from three major elements - a small cosmopolitan element, the southern Austral fauna, and the tropical fauna (Wilson and Allen 1987). While a few relictual tropical elements (Newman 1991) remain, especially in the southwest, the temperate Austral element represents the great majority of the fauna since the early Tertiary. Because of its long isolation, this fauna is highly endemic (Poore 1995a).

The temperate marine fauna can be subdivided into warm temperate or transitional (Wilson and Allen 1987) zones on the east (Peronian Province) and west (West Australian Province) coasts and cool temperate or “transitional warm-cold temperate” (Edyvane 1996) on the southern coast (Flindersian Province). This latter area has very high endemicity (e.g., 95% of molluscs and 90% of echinoderms, Poore 1995a) while the transitional warm temperate areas contain mixtures of southern and northern elements as well as some endemics (Wilson and Allen 1987).

Biogeographic provinces

A well-supported set of biogeographic zones or provinces, that realistically reflect faunal turnover, could be used to assist in the selection of "representative areas" for conservation and management. While there have been several attempts to produce a scheme of biogeographic provinces for the Australian coast, to date these have been based on limited data (see reviews by Wilson and Allen 1987; and Poore 1995a). In addition, the concept of biogeographic provinces (or zones) is contentious, there being few generally agreed definitions or principles. However, it is clear that most biogeographic boundaries are intimately related to temperature regimes and ocean currents (e.g., Gaylord and Gaines 2000).

Most early biogeographic schemes proposed between the 1930s and 1970s recognised five or six provinces (Whitley 1932; Bennett and Pope 1953; Knox 1963; Briggs 1974). Wilson and Allen (1987) greatly simplified these earlier schemes into a Northern Australian Tropical and a Southern Australian Warm Temperate Province with broad zones of overlap on both east and west coasts. However, such a scheme is clearly a gross oversimplification if locality-based conservation is to realistically reflect faunal turnover. For example, Hooper et al. (1999) showed, using NE Australian sponge faunas as models, that within a region apparently containing only two provinces (Solanderian and Peronian), there were at least five major biogeographic provinces, with a sixth one likely but still equivocal pending further data (Hooper et al. in press).

Considerable complexity can exist which is not necessarily latitudinally based - for example, the inshore and offshore areas of the GBR have very different faunas as a result of different habitats, sediment, wave exposure etc. The southward flowing Leeuwin Current brings offshore warm, low salinity water south around Cape Leeuwin and east into the Great Australian Bight. While offshore islands are bathed by this current, the coastline is not affected by it until the Cape Naturaliste-Cape Leeuwin area but this area is also affected by the cold West Wind Drift (Morgan and Wells 1991).

Bioregionalisations in Australia

A biogeographic or regional ecosystem classification for the marine environment was first developed by relevant Commonwealth, State and Territory management agencies in 1985 (CONCOM 1985; ACIUCN 1986). The CONCOM classification was specifically developed to assist with the establishment of a National Representative System of Marine Protected Areas. However, the classification was generalised, broad scale and lacked sufficient detail to assist bioregional conservation planning.

The 1985 national regionalisation was based on Ray's (1976) work and was endorsed by the Council of Nature Conservation Ministers (CONCOM) as a basis for planning the development of a system of national marine protected areas (CONCOM 1985). In 1986, the Australian Committee for the World Conservation Union (ACIUCN) modified the CONCOM regionalisation in their proposal for a national representative system of coastal and marine protected areas (ACIUCN 1986). This latter proposal 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. These represent 14 coastal (shallower than 200 m) coastal zones plus 18 oceanic zones and external territories. However, some (e.g., Poore 1995a) have argued that the boundaries between the coastal zones did not reflect suspected biogeographic boundaries and were thus inappropriate for environmental management. This latter argument has been adopted in a recent attempt to incorporate bioregional and biogeographic evidence into the GBRMPA Representative Areas Program for the planning and management of the Great Barrier Reef World Heritage Area (Kerrigan et al. in prep.; Day et al. in press).

In 1993 ANZECC established the National Advisory Committee on Marine Protected Areas (now the Task Force on Marine Protected Areas) to coordinate the development of the National Representative System of Marine Protected Areas (Commonwealth of Australia 1994). Part of this strategy has been the development of an *Interim Marine and Coastal Regionalisation for Australia* (IMCRA) as a framework for planning resource utilisation and conservation. This has resulted in a revised system of marine regions produced from regional frameworks developed by the Commonwealth, States and Northern Territory marine management and research agencies and coordinated by the Biodiversity Group, Environment Australia.

IMCRA (Interim Marine and Coastal Regionalisation for Australia)

The following is a summary of Version 3.3 of the Interim Marine and Coastal Regionalisation produced by the IMCRA Technical Group (1998)[21].

The process began in 1992 with Commonwealth support for a range of biogeographic projects in the States and Northern Territory and, in 1995, the development of regionalisation projects for Commonwealth waters began. The scheme is based on the best available information and is able to be progressively revised as new data and information become available.

IMCRA was developed in three stages:

- A meso-scale regionalisation was developed by the inshore waters working group (coastline and state territorial waters);
- Provincial to meso-scale regionalisations were developed by the offshore waters working group (incorporating the Australian EEZ); and
- A synthesis of both into several integrated biogeographic regionalisations at different scales.

The ANZECC Taskforce on Marine Protected Areas held a workshop in Sydney in 1994 to develop a single, meso-scale regionalisation for Australia. The workshop summarised recent regional classifications developed by State/Territory agencies for coastal and nearshore marine environments around Australia (Muldoon 1995). In several States, analytical multivariate procedures have been used to classify patterns of nearshore ecosystem diversity (e.g., Oritz and Burchmore 1992; Hamilton 1994; Edgar et al. 1995;

[21] http://www.ea.gov.au/coasts/mpa/nrsmpa/imcra.html

Edyvane and Baker 1995; Stevens 1995; Edyvane 1996) and input from these studies was an important component. As a result of this workshop, the Commonwealth funded a project in 1995 to develop a single, ecosystem-level regionalisation of Australia's coastal and marine environments, to be known as the 'Interim Marine and Coastal Regionalisation of Australia' (IMCRA). The project, coordinated by Environment Australia (Biodiversity Group), identified a collaborative approach between Commonwealth and State/Territory agencies, based on the previous successful terrestrial regionalisation of Australia, known as the Interim Biogeographic Regionalisation of Australia (IBRA) (Thackway and Cresswell 1995; Thackway and McCrae 1995).

The IMCRA classification was intended to reflect the range of methodologies and expertise available and the highly variable quality and quantity of data. Thus, both a qualitative, expert or `delphic' approach and quantitative, analytical methods were employed.

IMCRA utilises the following hierarchical structure, which recognises the need to consider ecological patterns and process that occur at continental, regional, local and site scales.

>1000s of km	Macro scale	Continental provinces
100s–1000s of km	Meso scale	Regions
10s–100s of km	Micro scale	Local units
<10 km	pica scale	Sites

Various State and Northern Territory agencies were funded to develop meso-scale biogeographic regionalisations and Commonwealth agencies were funded to develop regionalisation products at the provincial and meso-scale for the whole EEZ and the coastal zone, enabling the integration of State and Northern Territory regionalisations.

Physical versus biological attributes (surrogates)

Various environmental data sets (both biological and physical) were compiled for use in developing the regionalisation of the marine environment. In the absence of detailed mapping of habitats, the general approach adopted was to derive surrogates for coastal and marine environments. This process usually involved compiling the best available biological and physical data and information. Biological data sets included sponges, fishes, corals and sea grasses, while physical data sets included bathymetry, coastal geomorphology, sediments, currents, water chemistry and water temperature. This information was classified into "ecologically meaningful regions comprising similar combinations of environmental attributes". However, these biological data sets apparently have not been used generally in the bioregionalisation process. Corals - an almost entirely tropical group, and sponges, a relatively very poorly known and collected group other than in NE Australia, as the only other invertebrates, appear to have been used in the GBR bioregionalisation but not elsewhere (J. Hooper pers. comm.). It seems to us strange, given the objectives of the exercise, that the large amounts of distributional data reasonably easily accessible for some other taxa (e.g., certain groups of molluscs, echinoderms, decapod crustaceans and polychaetes) are not being utilised. A considerable body of data exists (mostly in museum collections) on the distributions of such taxa around the coast. While the geographic and taxonomic coverage is uneven, and the usefulness for determining biogeographic boundaries is has been questioned because the data were not collected for this purpose (e.g., Poore 1995a), the sheer volume of these data should, in our view, provide much better answers than are otherwise currently possible. The absence of comparative quantitative data on the relative abundance of species in many taxa for most of the coast has also been seen as an impediment (Poore 1995a). Given the logistical problems of acquiring reliable quantitative data, it is unlikely that these data will be acquired in a reasonable time frame given likely budgetary constraints.

Shallow versus deep-waters

Shallow waters generally have more detailed data and information on the species and habitats compared with deeper waters. The IMCRA document states, "In shallow waters, dive survey teams are often used to survey and map the boundaries of benthic habitats and to collect as comprehensively as possible, samples of flora and fauna." However, it is not clear whether this was part of the IMCRA process or not and exactly what fauna was involved, and at what level of resolution it was identified to.

In deeper waters, data and information are usually collected by remote sensing techniques, with perhaps a limited number of physical samples over large areas. In deep-waters, biological data and information, in contrast to physical data sets, are usually not detailed, comprehensive or contiguous in space or time[22].

The web document on which this text is based notes that "Caution needs to be applied when attempting to match the meso-scale and provincial regions with observed patterns in the distribution of fauna and flora, or to explain broad patterns of biota in terms of the IMCRA regions. The meso-scale and provincial regions act as surrogates for broadscale ecosystems and habitats and have not been extensively or rigorously tested."

Marine Habitat Mapping

Marine habitat mapping is an important aspect of micro-scale regionalisation. The States and Territories, in collaboration with CSIRO Division of Marine Research, have been undertaking a major national project to map all of continental Australia's shallow waters. Using Landsat Thematic Mapper imagery coupled with strategic field verification are used to map eight categories of habitat. To date extensive areas of southern Australia have been mapped.

Smaller-scale bioregionalisations

The development of a Marine and Coastal Regionalisation for Australia involves the development of meso-scale biogeographic regionalisations by various State and Territory

[22] Largely due to the consistent very limited resource-base available in Australia for such sampling (e.g., in comparison with New Zealand).

agencies to integrate with the provincial-level regionalisation developed by the Commonwealth for the whole EEZ and the coastal zone.

Examples of such smaller-scale regionalisations include the Tasmanian case and the regionalisation of the Great Barrier Reef area by GBRMPA.

Tasmania

As part of the Tasmanian reef bioregionalisation, quantitative surveys of plants and animals were made at over 150 shallow rocky reef sites around the Tasmanian coastline and Bass Strait islands. Data were analysed using several different methods (overlap of species ranges, multidimensional scaling and ecotone analysis) to produce a state bioregionalisation (Edgar et al. 1995; Edgar et al. 1997). Reef communities in the northern Bass Strait area were found to be distinctly different from those occurring further south, and were considered to reflect a division between two biogeographical provinces. These two areas were each divisible into four biogeographical regions (bioregions), which occurred along the northern, northeastern, southeastern, southern and western coasts of Tasmania, and around the Kent Group, the Furneaux Group and King Island in Bass Strait (Edgar et al. 1997). This regionalisation was also used to maximize the conservation value of sites within a proposed system of representative marine protected areas (MPAs) around Tasmania (Edgar et al. 1997).

Great Barrier Reef bioregionalisation

The GBRMPA Bioregionalisation Program (Kerrigan et al. in prep.; Day et al. in press) used expert knowledge and a dataset of over 1.5million points to classify, identify and recognise areas of unique, important, representative or other heritage values. These data points included taxonomic records of most marine invertebrate phyla and fishes that were used in the GIS assessment of the GBR. However, while the entire Queensland Museum marine database (J. Hooper pers. comm.) was used, the very large datasets for GBR taxa in the Australian Museum were not requested. While the use of sponges and corals as major components in the bioregionalisation of the GBR is appropriate, both groups being well collected in this region, the utilization of other well-collected taxa would provide an even more rigorous assessment.

Biodiversity in particular locations

Analysis of the biodiversity of particular areas is limited by the taxonomic impediment; Poore (1995a) states "rarely are more than 40% of the true species complement known to science". The few quantitative attempts in Australia to obtain data on numbers of species are rarely comparable due to differences in the habitat chosen, its size, methods, and on the effort and skills of the taxonomists involved. Some examples are:

- In the Great Barrier Reef lagoon, Birtles and Arnold (1988) recorded 103 species of echinoderm and 196 species of mollusc from 4 sites[23]; Hooper et al. (1999) used records of 913 species of sponges (from a database of nearly 2000 species) to assess

[23] Their sampling methods used a rather coarse mesh size so many small-sized taxa were not collected. There are probably more than 8,000 species of molluscs on the GBR with more than 1200 species collected from one station on the Swains Reef (W.F.P., pers. observ.).

biodiversity (and biogeography) at regional- and provincial-scales; Kennedy and Hooper (in press) used records of 247 species of sponges to examine biodiversity relationships amongst reefs at the micro- and pico-scales;
- On the Northwest Shelf, Ward and Rainer (1988) reported 308 species of decapod crustaceans (both this and previous study taxonomically limited);
- In the estuarine system of the Gippsland Lakes, Poore (1982) found 90 species, a figure 30% lower than in estuarine systems studied by others (perhaps affected by relative sizes of estuaries, degree of marine influence, biogeographic history or differences in methodology);
- In Port Phillip Bay, Poore et al. (1975) collected 713 macrobenthic species from 430 samples (43 m^2) of sandy and muddy habitats;
- In Western Port, Coleman et al. (1988) found 572 species of invertebrates from fewer samples, but a total of 2000 has been estimated from all habitats;
- In eastern Bass Strait, Parry et al. (1990) turned up 353 species in a survey of 1.2 m^2 of sea floor, but more detailed work at the same place has discovered about 800 species (of macroinvertebrates) in 10 m^2 (unpubl).) where even the lower figure is much greater than in most other parts of the world;
- On the southeastern Australian slope, Poore and Wilson (1993) and Poore et al. (1994) have summarised the number of isopod crustacean species, with 359 species having been identified, many more than found in similar studies in the northern hemisphere;
- Along the Victorian coast, Coleman et al. (1997) found 60 258 individuals and 803 species (of macroinvertebrates) in 104 samples (a total of 10.4 m^2 from shallow water (11-51m) in three areas). Few species were highly abundant and 51% of species collected were apparently undescribed.

In addition to this list, we report some figures in Section 7.6.2 for high diversity in molluscs.

2.5 Main issues and recommended actions

Issues

Large components of the marine invertebrate fauna are still poorly known or completely unknown due to:
- The chronic lack of resources for basic taxonomic, biological and ecological studies on the vast majority of marine invertebrates in Australia;
- Lack of, or minimal involvement in, invertebrate studies, or consideration of them, by state and commonwealth agencies responsible for marine and fisheries research; and
- Weakening of basic invertebrate biology and diversity courses in many universities.

Recommended actions

Facilitation of programs that will rapidly increase our knowledge of Australian marine invertebrate taxa, and the access to that knowledge, are needed. These include:
- Completion of the ABIF facility at least to checklist stage as a means of providing single source web-based information on marine invertebrates.

- Electronic access to the material stored in museum collections (databasing of marine invertebrate collections) and the linking of the resulting databases into a national, distributed facility (which can also be linked to taxa in the ABIF website) providing real-time distribution maps etc.
- Facilitation of existing knowledge and taxonomic identification aids to the well-known goups of marine invertebrates, enabling these to be accurately and efficiently identified (support of taxonomic programs, training, dissemination of guides, keys, interactive identification systems etc.).
- Encouragement of biological, systematic and ecological studies on marine invertebrates in universities.
- Encourage biodiversity agencies to become involved in training by the provision of supplementary funding.
- Utilise a larger representation of marine invertebrates in exercises like the bioregionalisation of Australia. This would not only enhance the result but also provide an incentive for comprehensive biogeographic studies on marine invertebrates.

On a longer-term basis, filling the large gaps in our knowledge of the taxonomy, composition, biology and distribution of the biota can be facilitated by:

- Provision of resources and facilities to enable the collection of material from areas unsampled or poorly sampled, including:
    - Provision of oceanographic vessels for offshore and deep-sea sampling.
    - Additional marine stations around the Australian coastline and upgrading existing ones.
- Encouraging and facilitating taxonomic, systematic, biological and ecological studies on targeted groups of marine invertebrates.

PART 2 - CONSERVATION

CHAPTER 3 – MARINE INVERTEBRATE CONSERVATION: THE ISSUES

3.1 Marine conservation

Given the highly visible destruction of many terrestrial and freshwater ecosystems, the problems of the conservation of marine organisms and habitats have generally been seen as a lesser priority (e.g., Irish and Norse 1996; Murphy and Duffus 1996). Although there have been some notable exceptions such as the conservation of marine mammals (especially whales and – in Australia – dugongs), turtles, and coral reefs, marine issues, in general, have been a significant gap in the focus of conservation biology. Conservation-related research on marine ecosystems and organisms has not kept up with the output of similar research on terrestrial environments (Irish and Norse 1996; see Chapter 9). Marine systems (oceans, coastal waters, and estuaries) constitute by volume most of the Earth's habitat for plants and animals. Much of biotic diversity occurs in the sea with 15 of the 33 or so living animal phyla occurring exclusively in the sea, and almost all others having marine members. Many of the smaller marine taxa are even more poorly known than the commonly used example of the mega-diverse terrestrial arthropods, and available data suggest that oceanic species diversity has been greatly underestimated. Ecosystem diversity in the sea is undoubtedly higher than on land and "among the ecosystem services provided to humankind, marine biodiversity is vital as a protein source, a future medicine cabinet, a storm surge bulwark, and a regulator of global atmosphere and climate…." (Murphy and Duffus 1996).

Arguments for marine invertebrate conservation are aided by the fact that marine invertebrates that cause or transmit disease are few and of little importance compared with non-marine invertebrate pests of medical significance for human beings (Fitter 1986). Although some marine invertebrates are capable of inflicting harm on humans (e.g., bluebottles, cone shells, blue-ringed octopus; see for example Edmonds 1989), these species are relatively few compared with the great diversity of marine invertebrates, many of which are a current or potential source of food and numerous important medical compounds (see Section 3.2.5).

Invertebrates make up a major proportion of all marine biodiversity. One argument for conserving biodiversity in its entirety – or as close to its entirety as possible – is simply our ignorance about the vast majority of organisms and ecosystems. Our taxonomic knowledge – which would allow us to answer questions as simple as "how many species are there?" – is so sparse for many groups that answers cannot be given even to within an order of magnitude. Even worse, information on the biology and ecology of the vast majority of species is virtually non-existent, the exceptions being a handful of commercially important species. We have a long way to go in understanding the complex interactions between species which form the basis of marine ecosystems, or the true value of the "ecosystem services" that they provide. We also have little understanding of the factors that determine the distribution and abundance of marine invertebrates. Until we know more, we need to exercise the precautionary principle by conserving as much biological diversity as possible.

The conservation of biodiversity has become a key issue over the last couple of decades. In 1982 the World Wildlife Fund (WWF), the International Union for the Conservation of Nature (IUCN) and the United Nations Environment Program (UNEP) published the World Conservation Strategy for the Conservation of Living Resources, which introduced the concept of ecologically sustainable development (ESD). A conference on biodiversity held in 1986 under the auspices of the Smithsonian Institution and the National Academy of Sciences further set the scene for biodiversity conservation, as did the resulting volume (Wilson and Peter 1988). However, there has been an overwhelmingly terrestrial – and to a large degree vertebrate – bias, with invertebrates, and in particular marine invertebrates, having been largely left off the agenda. Because concerns about biodiversity loss and marine conservation have only recently converged (Norse 1996), marine conservation biology lags behind its terrestrial counterpart by about two decades (Norse 1997), and research in this field still forms a small proportion of the total conservation-oriented output[24]. Even among the marine biota, the focus has been on the charismatic megafauna (e.g., dugongs, whales, turtles), not invertebrates. For instance, while the Great Barrier Reef Marine Park Authority spends about 35% of its budget (i.e. c. $315,000) in the Conservation, Biodiversity and World Heritage unit, on species conservation, to date most of this has been spent on dugongs, whales and turtles (J. Day pers. comm.). Likewise, while the oldest protected areas on land date from the last century[25], the development of a representative system of Marine Protected Areas to preserve marine biodiversity has only been seriously advocated in the last few decades (Kelleher and Kenchington 1992; Parker 1995). The development of Marine and Estuarine Protected Areas (MEPAs) in Australia and worldwide is outlined in Section 5.5.3.

3.2 Why conserve marine invertebrates?

Many of the animals regarded as insignificant by most people are the most essential in sustaining ecological processes and systems. E.O. Wilson (1987) succinctly described invertebrates as "the little things that run the world" and stated that if invertebrates disappeared, the world as we know it would cease to exist.

[24] For instance, of the 60 or so panellists involved in the above conference, only two were marine scientists, with a third having partial involvement in this area (Earle 1991). Irish & Norse (1996) scanned all existing editions of the journal *Conservation Biology* and found only 5% of papers dealing with marine environments, compared to 67% with a terrestrial focus. Only 13% of the 388 papers of an international conservation conference held in Sydney in 1999 dealt with marine matters and only 5% with marine invertebrates (see also Chapter 9). France & Rigg (1998) reviewed 2524 articles published from 1987 to 1995 in five international conservation and ecology journals to assess patterns and imbalances in biodiversity research. They found that much of biodiversity research was narrowly focused on the implications of forest habitat loss and terrestrial megafauna while marine biodiversity was largely ignored, as were many other areas of research.

[25] E.g., Yellowstone National Park (US), est. 1872; Royal National Park (NSW), gazetted 1879.

The reasons for conserving biodiversity can be divided into utilitarian values and intrinsic values (Office of the Chief Scientist 1992; Burgman and Lindenmayer 1998). Utilitarian values, often emphasised because they can be defined in practical terms, can be divided into consumptive use values, productive use values, service values, scientific and educational values, and cultural, spiritual, experiential and existence values (including aesthetic, recreational and tourist use) (Burgman and Lindenmayer 1998). Realistically, only a tiny proportion of invertebrate biodiversity is ever likely to have major economic benefits for humans, but it is frequently argued that an important reason for conserving a broad range of biota is to ensure that opportunities to develop such resources are not lost. However, economic criteria for conservation are opportunistic in their practical application. For example, rare species, often the most vulnerable, are also the ones least likely to have an important perceived economic value (Ehrenfield 1988). Ethical considerations – which take account of the intrinsic values of biodiversity (whether from an ecocentric or biocentric perspective) – are an alternative system for the valuation of conservation (Burgman and Lindenmayer 1998).

A very large proportion of all biodiversity (marine, terrestrial and freshwater) consists of species of invertebrates. Their crucial ecological roles and abundance make them indispensable for life as we know it. Faunal diversity is essential to maintain the biological balance of the biosphere, and invertebrates play a vital role in this (e.g., Council of Europe 1987; Wilson and Peter 1988). The need for invertebrate conservation has been recognised by the Australian government for some time as evidenced by the Council of Conservation Ministers Australian Statement on Invertebrates (CONCOM 1989), as well as having at least some international recognition - for example IUCN Recommendation No. 41 and the Committee of Ministers Council of Europe Charter on Invertebrates (Appendix to Recommendation No R(86)10) (Council of Europe 1987)[26]. The Charter on Invertebrates, for example, recognised that:

> "The natural fauna of invertebrates is diminishing continually and many species have either disappeared or are in the process of disappearance because of man's action, without man even having been aware of their existence or having studied their characteristics and possible uses" (Council of Europe 1987).

Yen and Butcher (1997) discussed in detail the many reasons why invertebrate conservation is important, as have many others (e.g., Wilson 1987; Wilson and Peter 1988; Wilson 1992; Kellert 1993; New 1995b, etc.; Yen and New 1995b). Recent discussion on this matter relating to Australian invertebrates can also be found in several papers in the proceedings of the three meetings on Invertebrate Biodiversity and Conservation (Ingram et al. 1994; Yen and New 1995a; Ponder and Lunney 1999). While much of the argument regarding invertebrate conservation has focused on terrestrial invertebrates (e.g., Wilson 1992; Yen and Butcher 1997; Horwitz et al. 1999), there are pressing reasons to also be concerned about the conservation of marine invertebrates (Norman and Sant 1995; Irish and Norse 1996; Schinner et al. 1996; Carlton 1996b; Davis et al. 1999). The following discussion will focus on the reasons why marine invertebrate conservation is important although, regrettably, largely overlooked.

[26] these documents are reproduced in Yen & Butcher (1997), Appendix 2.

Conservation of marine invertebrates can be justified for much the same reasons given for other biota, including non-marine invertebrates (e.g., Kellert 1993; Yen and Butcher 1997; Horwitz et al. 1999). These reasons can be summarised as follows:

- **Ecological** – e.g., ecosystem stabilisation, energy and nutrient transfer, provision of habitat; and **biogeochemical** – e.g., climate stabilisation, re-mineralisation.
- **Scientific** – as subjects for study in a wide range of disciplines, use in environmental assessment and monitoring etc.;
- **Utilitarian/Economic** – values may be *direct* – e.g., food (fisheries), industrial and medicinal products, decorative products, building materials, tourism revenue – or *indirect* (derivatives of ecological functions) – e.g., waste decomposition, shoreline protection, food for fishes[27] and other animals of economic or other significance, etc.
- **Cultural** – education, recreation, aesthetics, cultural significance;
- **Ethical**.

Each of these reasons is briefly expanded below.

3.2.1 Ecological functions and services

Invertebrates are an integral part of marine ecosystems, and play a number of roles that help to support the function and stability of the food chains and ecosystems upon which we, and other animals, rely. For instance,

- They play an important role in the cycling of nutrients,
- Are essential for the breakdown of plant matter and other detritus,
- Form the basis of many food chains (including those supporting commercial fisheries),
- Provide habitat for other species (e.g., coral reefs),
- Regulate populations of other organisms (plant and animal) through predation, parasitism and herbivory, and
- Help maintain water quality by filtering large amounts of water during feeding.

Marine invertebrates also provide many important *ecosystem services*, including production of food, stabilization of shorelines, trapping and removal of excess nutrients and pollutants, and cycling of nutrients and organic matter (Snelgrove et al. 2000; Austen et al. 2002 and references therein).

There has been considerable debate over the extent to which the provision of “ecological" or "ecosystem services” (i.e. maintenance of ecological processes and functioning ecosystems) depends upon biological diversity (e.g., Ray 1991; Schulze and Mooney 1994; Eakin et al. 1997; Naeem and Li 1997; Wardle 1998; Schläpfer et al. 1999; Duarte 2000), but it is not our intention to cover this debate here - simply to summarise some of the important functions and roles played by marine invertebrates. The relationships

[27] For example, see Bulman (2001) who analysed the diet of 102 fish species showing that they eat a large range of benthic and pelagic invertebrates as well as pelagic fish.

between the provision of "ecological services" and biodiversity, the effects of biodiversity loss on ecosystem function, and the extent of species redundancy in nature are all discussed more fully in Section 5.4.4.

Important ecological functions for marine invertebrates include:

- ***Breakdown of plant matter and detritus***. Marine invertebrates play an important role in the recycling of material and nutrients, and many are key links in detritus-based food chains, especially in coastal habitats such as wetlands and seagrass beds where large quantities of plant material are produced. In mangroves, for instance, sesarmid crabs consume 30-80% (depending on forest type and intertidal elevation) of the annual litter fall and thus have an important role in controlling the rate of re-mineralisation of detritus within forests and the export of particulate matter from the forests to other nearshore habitats (Robertson 1991). The other major component of litter is wood, which is broken down relatively rapidly by teredinid molluscs (shipworms); more than 90% of the weight loss from decomposing trunks of *Rhizophora* species during the first four years of decay is through ingestion by teredinids (Robertson 1991). In shallow coastal habitats, many of the decomposition processes occur predominantly within the sediment (Austen et al. 2002), and largely involve micro-organisms. Macrofauna have an impact on the rates of these processes through their bioturbatory and sediment reworking activities, which bring oxygen deeper into the sediment and irrigate the sediment (Austen et al. 2002), and by burying organic matter (Levin et al. 1997). Different species have different modes of reworking and of bioturbation so that the diversity of the bioturbators has a direct effect on the rates of decomposition (e.g., Pelegri and Blackburn 1995).
- ***Energy and nutrient transfer between trophic groups***. The role of invertebrates in energy and nutrient transfer and the maintenance of trophic structures is well known (e.g., Mazzella et al. 1992; Kellert 1993). In most aquatic ecosystems, energy and nutrients assimilated by plants are often transferred to other consumer levels by invertebrates. Many belong to low trophic levels and form either the principal food source of, or the basis of food chains leading to, higher-trophic level predators including fish, birds and marine mammals. Well known examples include zooplankton in the open ocean, and krill as the basis of Antarctic food chains; however, benthic invertebrate communities in shallower waters are also extremely significant, and their importance in marine food webs leading to commercially exploitable yields of fish has been widely recognised (Newell et al. 1998). As noted by Newell et al. (1998), studies in the North Sea suggest that approximately half of the net primary production by phytoplankton falls to the sea bed as a detrital input to the benthic community, with additional input from faecal material resulting from the proportion consumed by pelagic herbivores (e.g., copepods and euphausiids). The benthos is thus heavily implicated in carbon flow in coastal systems and is of increased importance in shallow waters where benthic algae and seagrasses largely replace phytoplankton as the primary carbon source.
- ***Provision of habitat for other biota***. Corals, sponges and other encrusting organisms, such as bryozoans, serpulid worms, mussels and oysters, provide habitat for a wide range of other animals and algae. Coral reefs, for example, provide a range of niches

for a wide diversity of plants and animals, including perhaps one-third of all fish species (Goreau et al. 1979). Large sponges are host to multitudes of commensal invertebrates, such as crustaceans, molluscs, various groups of worms and echinoderms, as well as huge numbers of micro-organisms, mainly bacteria and blue-green algae (Bergquist and Skinner 1982). Such habitat interactions can be very complex. For example, dense populations of filter feeding organisms (such as tunicates, barnacles, mussels or tubiculous polychaetes) can modify local water currents thus influencing recruitment and the supply of food.

- ***Ecological services***. Invertebrates perform essential roles in various ecological interactions, including herbivory, predation, parasitism, mutualism and competition (Kellert 1993). For example, without herbivores algae would dominate many shallow water marine ecosystems. Predation and parasitism also serve to regulate populations. While direct interactions such as these play an important role in structuring communities, indirect effects are also important. For example, high rates of sediment disturbance by burrowing organisms can have significant deleterious effects on suspension feeders and tube builders (Posey 1983; Wilson 1983a).
- ***Maintaining environmental health***. Invertebrates play a very important role in the maintenance of the health of marine ecosytems. For example, filter feeders help maintain water quality, scavengers remove dead animals and infaunal marine invertebrates improve sediment quality. The oyster population of Chesapeake Bay (USA) currently filters a volume of water equal to that of the entire bay about once a year, but before a human-induced decline in the oyster population, it filtered a volume equal to the entire bay about once a week (Newell 1988). The filtering capacity of bryozoans in seagrass meadows in Western Australia was estimated by Lisbjerg and Petersen (2000), as was the rate of turnover by the infauna of the soft sediments in Port Phillip Bay by Poore et al. (1974) and Bird (1999), where the turnover rates are measured in days. Burrowing marine infauna, such as polychaetes (Hutchings 1998a) and holothurians (Uthicke 1999), are involved in the maintenance of sediment oxygenation and quality in the much the same way that earthworms, termites and other invertebrates modify and improve soils in the terrestrial environment. Uthicke (1999) showed that two holothurian species at Lizard Island, Queensland have the potential to rework about 4600 kg dry wt yr^{-1} for every 1000 m^2, which is equivalent to approximately the upper 5 mm of sediment in this area. Such bioturbation destabilises sediment stratification, enhances aeration and distributes organic material dissolved in the interstitial water into the water column. Likewise, burrowing crabs are important in the aeration of soil in mangrove stands, and hence critical to the growth and survival of mangrove trees (e.g., Robertson 1991; Smith et al. 1991; Robertson and Alongi 1995). Experimental removal of crabs has been shown to result in significantly increased levels of soil sulphide and ammonium, and significantly less cumulative forest growth and reproductive output, compared to control plots (Smith et al. 1991). Burrowing by crabs seems to be an important process, affecting soil aeration that in turn affects the productivity and reproductive output of *Rhizophora* (Smith et al. 1991).
- ***Ecosystem stabilisation***. All plants and animals contribute something to the maintenance of the ecosystem, even if they are just part of a food chain. Diverse ecosystems are commonly seen as more stable – i.e. possessing greater resistance to

perturbation and greater resilience (ie., capacity to recover from disturbance) (Austen et al. 2002) – than more simplified ones, probably due in part to the existence of a degree of "functional redundancy" (Snelgrove et al. 1997; Warwick and Clarke 1998). While this idea is still controversial[28], it is clear that major disturbances, such as the extinction of key species, are likely to have serious impacts. The loss of species from highly interrelated systems may cause a cascade of further extinctions, an obvious example being the dependence of host-specific commensals or parasites. Just how many species can be removed from any particular ecosystem before it collapses is unknown, and would certainly vary considerably depending on the ecosystem being considered.

Also generally ignored and largely unexplored in the marine environment (though beyond the scope of this document) are the roles of microbial processes and marine viruses in ecosystem services and function. Marine viruses, for example, are important for their biogeochemical and ecological effects – including nutrient recycling, system respiration, particle size-distributions and sinking rates, bacterial and algal biodiversity and species distributions, algal bloom control, dimethyl sulphide formation and genetic transfer (Fuhrman 1999). These processes warrant a review in their own right, although this whole area of research has been largely neglected in Australian waters.

3.2.2 Biogeochemical functions

The processes involving the cycling of key materials and nutrients, such as nitrogen, carbon, etc., as well as those of more limited availability, are referred to as biogeochemical functions. These processes can have vital secondary consequences such as climate stability and shoreline protection. The primary producers such as marine phytoplankton, together with bacteria, are responsible for fixing atmospheric carbon and nitrogen into organic form, but marine invertebrates contribute to the flow of these materials through the system, via trophic consumer chains, and their roles as detritus feeders, decomposers etc. Coral reefs also store carbon in the form of calcium carbonate.

The major biogeochemical functions of marine invertebrates include:

- ***Nutrient cycling***. Planktonic and benthic invertebrate communities are important in the recycling of nutrients. For example, salps and pyrosomids can have a high grazing impact on phytoplankton populations and, through defecation, play an important role in the downward flux of matter (Andersen 1998). This faecal material is enriched in several elements (e.g., carbon, nitrogen, calcium and aluminium) and can markedly contribute to the rapid vertical flux of particles and the distribution of trace elements.

[28] The idea that simple ecosystems are less stable than complex ones is controversial. Early studies (e.g., Elton 1958) suggested that this was so, but some later studies came to the opposite conclusion – possibly because of confusion regarding the different meanings of 'complexity' and 'stability' (see Pimm 1984 for a discussion). Some recent evidence supports the idea of a positive relationship between diversity and stability (e.g., Loreau 2000) – for instance, increased diversity was found to reduce variation in the biomass and species density in experimental microcosms (Naeem and Li 1997). Other evidence suggests that this is not always the case; some apparently stable ecosystems, such as the Baltic, being relatively species-poor (Elmgren and Hill 1997).

- ***Effects on climate***. The marine biological carbon pump has an important effect on the level of atmospheric CO^2 (Broecker 1974; Knox and McElroy 1984; Sarmiento and Toggweiler 1984; Sarmiento and Bender 1994). While phytoplankton are primarily responsible, zooplankton play a major role in determining phytoplankton abundance.
- ***Carbon sink***. Calcium carbonate also acts as a carbon store, and coral reefs in particular are important in this regard (Wood 1998). The structural mass of coral reefs also protects shorelines and greatly modifies the habitats associated with them.

3.2.3 Scientific values

Both as subjects for study and tools for research in a range of fields, marine invertebrates contribute to our knowledge of ecosystems and environments, genetics, physiology, biology, medicine, chemistry, physics and other areas.

As noted by Murphy and Duffus (1996) "the oceans have …been the primary stage upon which the history of life on Earth has been played. The major expansions of Earth's diversity and the most wide ranging extinctions have taken place in the seas, leaving an unparalleled legacy of knowledge for us to unlock." Yen and Butcher (1997) point out that non-marine invertebrates have played, and will continue to play, important roles in scientific research. This is doubly true for marine invertebrates. Some examples of how marine invertebrates have markedly contributed to science are:

- ***Contributions to our understanding of the environment***. They have helped us understand how our environments function; the many ecological principles derived from work on rocky shores being an excellent example. In nature the diversity and abundance of marine invertebrates make them ideal subjects for ecological studies and experiments.
- ***New drugs and other useful compounds***. The potential for the discovery of new products, including drugs, has hardly been realised and yet some very important discoveries have already been made (see Section 3.2.5 below).
- ***Experimental research, such as neurophysiology***. Many invertebrates are excellent experimental animals (Council of Europe 1987; CONCOM 1989; Brakefield 1991). They have been vital tools for research in behavioural and neurological studies, genetics, phylogenetics, physiology, biology, medicine, chemistry and physics (Yen and Butcher 1997). For instance, the giant axons (large diameter nerve fibres) of squids are widely used in neurophysiological research, and much of our neurophysiological knowledge stems from this research (Dunning and Lu 1998, p. 517). Other marine invertebrates (octopuses, sea hares (*Aplysia),* decapod crustaceans) have also figured prominently in similar work on nerve function, neurosecretion etc. (e.g., Amit 1990). Early development studies and embryology have used echinoderm larvae extensively.
- ***Fossil and climatic record***. The hard skeletons of a number of important marine invertebrates have resulted in a rich fossil record, and they have provided the data on which much of our understanding of the evolution and diversity of life, and past extinctions, is based. For example, corals, which can live many hundreds of years, can

provide records of past climates and environmental change. Changes in coral skeletal structure, determined by taking cores from massive coral species, can indicate rainfall patterns and seawater temperatures (e.g., Isdale 1984, 1986; Lough and Barnes 1990, 1995; Isdale et al. 1998).

3.2.4 Environmental assessment and monitoring

While the very considerable diversity and complexity of invertebrates are sometimes cited as making them too intractable to study, this variety provides an opportunity to map patterns of diversity and to monitor environmental quality at a scale that can be much more meaningful than using vertebrates (Miller and Holloway 1991; Yen and Butcher 1997). The rapid response of many taxa to disturbance, the much smaller scales of species turn-over and the abundance and small size of many of them enable easy quantitative sampling and provide good data for statistical analyses. The use of aquatic (including marine) invertebrates in monitoring the environment, either as indicators of environmental health or degradation (e.g., Cao et al. 1996; Deeley and Paling 1999; Moverley and Hirst 1999) or as "bio-indicators" of pollution (e.g., Hodda and Nicholas 1986b; Bilyard 1987; CONCOM 1989; Ellis and Pattisina 1990; Wilson et al. 1993; Oehlmann et al. 1996; Scanes 1996) is well established. The use of sophisticated sampling design and well-constructed experiments in marine ecology have been used very successfully in many recent studies to address many of the highly relevant questions relating to variation, patterns, behaviour etc. (e.g., see reviews by Underwood 1993c; Underwood and Atkinson 1993; Castilla 2000; Chapman 2000; Underwood 2000; Underwood et al. 2000).

Indicators of environmental health

The diversity and species composition of benthic invertebrate communities can give a good indication of "health", and the condition of invertebrate communities in a range of marine habitats have been proposed as indicators of habitat quality under the Environmental Indicators for *State of the Environment Reporting* recently developed on behalf of Environment Australia (Ward et al. 1998a). Benthic invertebrates have also been used to examine long-term changes in Port Phillip Bay (Harris et al. 1996; Wilson et al. 1998).

State of the Environment Reporting

The Australian State of the Environment Reports (State of the Environment Advisory Council 1996; Australian State of the Environment Committee 2001) have a section on coasts and oceans. Due to the bulk of our report having been written prior to the 2001 document, we refer to the 1996 version through most of this document. Ward et al. (Ward et al. 1998a) discuss the marine environmental indicators for the national state of the environment reporting.

There is an enormous (but to date, largely unrealised) potential for the use of invertebrates in assisting conservation planners to help select areas for protection. Not only are they very diverse, but many species have small geographic ranges. They also occupy smaller and more numerous ecological niches in all dimensions (space, time, etc.)

than most vertebrates (Collins and Wells 1983). Despite these attributes, invertebrates have traditionally been largely ignored or bypassed in attempts to select areas for conservation. The recent bioregional assessment in Australia (IMCRA Technical Group 1998) used virtually no invertebrate data to define bioregions, relying instead on surrogates such as geology, substrate type, fish faunas, plants, water temperature and so forth. In contrast, the Great Barrier Reef Marine Park World Heritage Area has been analysed and mapped to show 64 bioregions (30 reef bioregions and 34 non-reef bioregions) with eight additional areas still to be classified. This was achieved using much of the available biological (including data on marine invertebrates), physical and geological data as well as expert knowledge (Day et al. in press). This process has revealed that some bioregions are not protected within the existing park. This data are being used to review the whole zoning of the region and, in tandem with other considerations (such as cultural, economic and social factors), will result in a revised zoning plan that will ensure that the bioregions identified are adequately conserved. It is anticipated that the new zoning plans will be ready by early 2003 for approval by the Minister and Parliament. Unfortunately, however, this amount of data would not currently be available for many or most marine areas under Australian jurisdiction.

Moverley and Hirst (1999) recently reviewed the use of benthic macrofauna in assessing estuarine health. Such studies utilizing marine invertebrates in environmental monitoring are common. A few examples are: Bilyard (1987) recommended the use of benthic infaunal invertebrates in marine pollution monitoring studies because they are sedentary and respond to pollutant stresses; Sanchez-Jerez and Ramos Espla (1996) showed that the structure of isopod and amphipod communities (as well as fish) is an efficient tool for monitoring the impacts of bottom trawling on seagrass meadows in the Mediterranean; and the Sydney "Oyster Watch" program (e.g., Scanes 1996) monitored trace metal and organochlorine concentrations in Sydney's coastal waters. Mussels (e.g., Muñoz-Barbosa et al. 2000 and references therein) are also commonly used for bioaccumulation studies.

3.2.5 Economic and medical reasons

Marine invertebrates contribute substantially to the economy and human welfare, both directly and indirectly, but their potential has hardly been realised, nor is their value generally recognised. (Council of Europe 1987; CONCOM 1989; Beattie 1994; Yen and Butcher 1997; Benkendorff 1999b). The direct economic benefits of commercial food species (such as abalone, scallops, oysters, prawns, crayfish, squid etc.), whether wild-harvested or produced through aquaculture, are obvious enough, as is the use of a wide range of invertebrates as a resource for individuals for food and bait. Other commercially exploited taxa with direct economic benefits include pearl oysters and black corals. Indirect contributions to the economy by marine invertebrates include their critical role in the food chains of exploited invertebrates and fishes, their role in the maintenance of marine ecosystems, including fisheries productivity, maintenance of water quality, and the role of coral reefs in buffering shorelines against wave and storm damage. Other indirect economic benefits include provision of food and shelter for juvenile stages of commercial fish species, regulation of climate, storage and cycling of essential nutrients, and the absorption, breakdown and dispersal of organic wastes, pesticides, air pollutants and water pollutants (Burgman and Lindenmayer 1998). However, these service values tend only to be highlighted when ecosystems become degraded to the point that the economics of restoration become measurable in dollar terms (Burgman and Lindenmayer 1998). The role of invertebrates in maintaining healthy marine environments is important not only for the exploitive industries but for the recreational and tourist industries.

While many species have been utilised sustainably, economic exploitation has had conservation implications for some invertebrate species. For instance, overexploitation of trochus shell, giant clams, some abalone, scallops etc. in some areas has led to closure of fisheries or severe restrictions on harvesting (see Section 6.3).

Many marine invertebrates are likely to have features that can be translated into important economic benefits. These range from the development of new adhesives and other industrial products, to research on nerve function and the synthesis of anti-coagulants, antibacterial agents, anti-cancer drugs etc. in human medicine (Norton 1986; Wilson 1987; Gaill et al. 1991; Roberts 1992; Beattie 1994; Carté 1996; Volkman 1999; Benkendorff 1999b, 2000).

Some of the major direct economic benefits of marine invertebrates are discussed below.

Marine invertebrates as food

The Australian Aborigines living on the coast relied heavily on marine invertebrates as a source of protein, as seen in extensive deposits of shell middens. The use of molluscs as a food source by Aborigines can be traced back through shell middens for up to some 35,000 years (Meehan 1982; Thomson et al. 1987). Thomson et al. (1987) suggested that in coastal areas molluscs probably made up 25% of the diet. These authors noted that the impact of the Aboriginal people on mollusc populations is difficult to determine but that in all likelihood local populations could well have been over exploited and local extinctions may have occurred, especially when combined with stresses caused by environmental fluctuations.

The two major invertebrate groups that currently contribute most to human diet are crustaceans and molluscs. Most of the exploited species are marine. Some have been developed into major commercial industries, while others are captured non-commercially. Exploited species include molluscs such as scallops, mussels, abalone, squid and octopus; crustaceans such as prawns, lobsters, and crabs; as well as holothurian echinoderms (sea cucumbers – *trepang* or *bêche de mer*) and echinoid (sea urchin or sea egg) roe. Molluscs and crustaceans are widely collected by recreational fishers, while other invertebrate species, such as beachworms and bloodworms (polychaetes) and cunjevoi (ascidians) are commonly exploited for use as bait (see Section 6.10.3).

Invertebrate species are among the most valuable fisheries, accounting for a high proportion of income relative to the quantity harvested. For instance, in 1998-99 crustaceans and molluscs made up only 37% by weight but 67% by value of Australian commercial fisheries (ABARE 1999). Individual high-value groups included prawns (accounting for 20% of the total dollar value), rock lobster (20%), abalone (9%), and other molluscs (mainly pearl oysters; 11%) (see Figure 3.1). Many of these products fetch very high prices in overseas markets, and around 80% of this total value comes from exports. In 1997 Australia was ranked 20th in the world in terms of the value of its marine export fisheries, despite the low productivity of the waters and low weight of catch (41st in terms of tonnage of exports and 54th in terms of total production)[29], because a higher-than-average proportion of the fisheries are high-value invertebrate species.

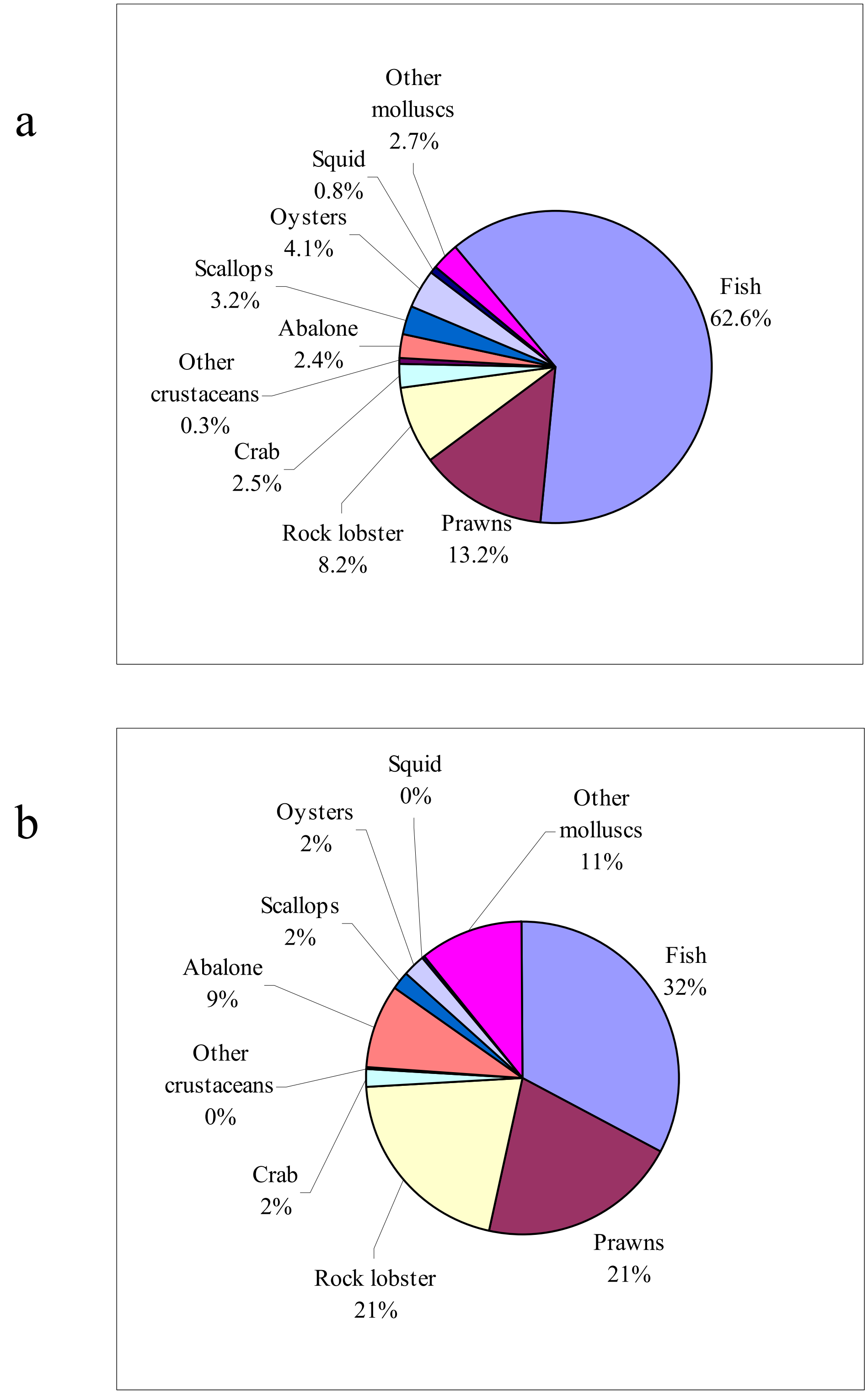
The worldwide value of, and reliance upon, invertebrate fisheries is likely to increase in the future with the so-called "fishing down" of marine food webs, brought about by the collapse of many traditional commercial fisheries and increasing utilisation of lower trophic level organisms. This trend is illustrated by the decline from 1950 to 1994 of the mean trophic level of the species groups reported in the Food and Agricultural Organisation (FAO) global fisheries statistics, which reflects a gradual transition in landings from long-lived, high trophic level, piscivorous bottom fish toward short-lived, low trophic level invertebrates and planktivorous pelagic fish (Pauly et al. 1998). In addition, aquaculture production is rising both in Australia and overseas (FAO Fisheries Department 1997), with many invertebrates lending themselves better to aquaculture methods than most fishes.

Biotechnological and pharmaceutical potential

Some marine invertebrates have developed, during the course of millions of years of evolution, complex chemical signals for everything from the regulation of spawning and larval settlement to defence against bacteria and predators. The screening of marine invertebrates for novel compounds and bioactive substances is a rapidly expanding area of biotechnological and pharmaceutical research (see Volkman 1999 for a recent review of Australian research). Such compounds have a variety of potential applications, ranging from new drugs to "natural" sunscreen creams or antifouling paints for boats and other man-made structures.

[29] Based on 1997 figures. Source: FAO fisheries statistics database, http://apps.fao.org/lim500/nph-wrap.pl?Fishery.Primary&Domain=SUA

Figure 3.1: (a) Quantity (t) of Australian fisheries production 1998-99, by group; (b) Value ($'000) of Australian fisheries production 1998-99, by group (source of data: ABARE 1999).



Scientists and pharmaceutical companies are dedicating increasing resources to *bioprospecting* (searching for potential drugs and other compounds) in living organisms. In 1998 the biomedical industry spent more than $US2.5 billion searching for potentially useful enzymes (Moore and Zeidner 1999), but the biotechnological potential of marine animals has barely been touched (Carté 1996; Moore and Zeidner 1999; Benkendorff 1999b, 2000). Moore and Zeidler (1999) highlighted the so-called “extremophiles” – microorganisms with the ability to thrive in extreme environments such as hydrothermal vents – as holding promise for genetically based medications and industrial chemicals and processes, since they have unique enzymes (“extremozymes”) which enable them to function in these inhospitable environments. However, there are also a large number of invertebrates from several phyla adapted to these conditions and they too could be the source of a variety of useful compounds.

There are many potential uses of chemicals obtained from marine organisms other than their application in human health. For instance, natural antifouling compounds would have enormous practical value given the serious pollution problems caused by some of the chemicals currently in use (e.g., TBT). Antifouling compounds could also have other commercial applications such as prevention of fouling on fish nets, aquaculture facilities, cultured pearl oysters (currently cleaned by hand), etc. Many sponges have allelochemicals which appear to inhibit the settlement of ascidian larvae (Davis et al. 1991). The Centre for Marine Biofouling and Bio-Innovation (CMBB) at the University of New South Wales was established to investigate chemical signalling between marine organisms, with the aims of (1) understanding how the ecology and behaviour of marine organisms are mediated by chemical signals, and (2) applying this knowledge to the development of novel, environmentally benign, biotechnologies including inhibition of the settlement and growth of marine organisms on artificial surfaces (antifouling). Other potential biological resources from marine invertebrates include adhesives from deep-sea annelids (Gaill et al. 1991).

Bioprospecting

The Australian Institute of Marine Science (AIMS) has a Marine Bioproducts section which is looking at bioactive substances from marine organisms with the aim of developing novel commercial and biomedical applications. Their work includes screening marine biota for compounds – resulting in a collection which currently covers 12 000 samples from 1700 collection sites within the EEZ (C. Wolff pers. comm.) – and related research on the physiological and biochemical adaptive responses of marine organisms to environmental stresses. One commercial application has resulted from the discovery that many marine organisms biosynthesise or diet-accumulate UV-absorbing compounds known as mycosporine-like amino acids, which protect them by acting as a natural sunscreen with considerable commercial application. Griffith University in partnership with the Queensland Centre for Biodiversity, Queensland Museum have similarly prospected the marine invertebrate faunas of tropical Australia for natural therapeutic chemical substances of potential pharmaceutical benefit. In seven years nearly 4,000 species of sessile marine invertebrate taxa (mainly sponges, octocorals and ascidians) have been sampled with a large proportion already recognised as new to science, and several compounds currently under testing as potential drug candidates (Hooper et al.

1998). Irrespective of any eventual commercial successes there are already substantial benefits to knowledge of marine invertebrate biodiversity (Hooper et al. in press).

There are also potential uses of cone shell peptides in medicine[30], and some bryozoans (Bugulidae and Catenicellidae) produce cytotoxic, antibiotic, or antiviral secondary metabolites of actual or potential interest to medicine (Newman 1996; Prinsep and Morris 1996).

Other medical applications
Scientists at the University of Technology, Sydney are working on a prosthetic eye design using coral and corals have also have been suggested as a replacement for bone (in bone grafts etc.). There are numerous other substances or materials derived from marine invertebrates that could have considerable application – for example adhesives (for use in the sealing of wounds in internal organs), or entirely new structural materials (Benkendorff 1999b).

Prosthetic eye design using coral: Coral will be used as the template for a porous ceramic coating that will enable muscles and other surrounding tissues to attach to an implanted artificial eye. Current artificial eyes fail to deliver natural movement and can cause sagging of the lids due to the unsupported weight of the ball. But since the 1970s it has been known that if a biocompatible ceramic material could be developed with the appropriate surface pattern of interconnected pores, hard and soft tissue will grow into it. Some Australian corals have the right pore size and connectivity (Ben-Nissan 1999).

Other industrial applications
In the area of structural engineering, the ultrastructure of the shells of deep-sea molluscs has provided valuable information for the design of novel man-made materials that combine rigidity with some degree of flexibility (Webb et al. 1991; Beattie 1994). The ceramics industry has also been analysing mollusc shells and radulae for the control of crystallisation processes, especially where there are specific and complex additives that must be incorporated into the final product (for example, the radula may be hardened with iron oxides incorporated into the final structure to make an extremely hard surface). This area of applied research already has its own journal, called 'biomimetics'.

Tourism revenue
Tourism is another important economic benefit gained from marine invertebrates. For example, the Great Barrier Reef, the structure and much of the diversity of which is formed by corals and other invertebrates, experiences more than 1.5 million visitor-days a year and is estimated to generate over $1 billion annually in tourism revenue[31]. Invertebrates can be attractions in their natural environment, such as the intertidal animals on rocky reefs or colourful or unusual invertebrates seen by recreational divers (see also Section 3.2.7).

3.2.6 Education

[30] See http://grimwade.biochem.unimelb.edu.au/cone/index1.html for details of this research
[31] http://www.gbrmpa.gov.au/corp_site/key_issues/tourism/index.html

The importance of direct experience of complex, natural ecosystems, and the animals that they contain, for developing curiosity, observational and critical skills and for stimulating the mind to formulate questions and develop new ideas was emphasised by Vermeij (1998). Although invertebrates are readily accessible and easy to use in biological and environmental education, they are under-utilised (Meehan 1995; Yen and Butcher 1997; Kinnear 1999). They can also be used in craft activities (e.g., shell craft). Many children have a fascination with invertebrates and positive interactions with invertebrates could be important in determining the future attitudes of these children towards conservation and the environment (e.g., Yen 1993; Yen and Butcher 1997; Kinnear 1999). Kinnear (1999) discussed how familiarity with organisms is an important factor in how children decide whether they have ecological roles and should be conserved. Invertebrates are a logical replacement for the increasingly difficult use of vertebrates in biological or environmental education. The strand line offers excellent opportunities for school projects involving non-destructive biodiversity sampling and natural history education, particularly with the empty shells of marine molluscs, the empty tests of echinoids, dead sponges, etc., all of which can be readily found in beach drift. Marine invertebrates are also used widely in university teaching, for instance in zoology and physiology courses.

3.2.7 Recreation / Aesthetics

Most of Australia's population lives adjacent to the coast, and coastal activities are very important for recreation. Many people derive great enjoyment from hobbies that revolve around the collection or observation of marine invertebrates. These range from snorkelling and scuba-diving (with coral and rocky reefs being particularly popular), including underwater photography, to bait or shell collecting, aquarium keeping or just casual observation in intertidal habitats.

Many invertebrates are of great aesthetic value and are sources of inspiration (Council of Europe 1987). They can be conserved for their beauty, symbolic value or intrinsic interest (e.g., McMichael 1982; Wilson 1987; Yen and Butcher 1997). Invertebrates are used either directly as ornamentation in jewellery, souvenirs and as decorative collections, or indirectly as the basis of decorative designs (e.g., Fitter 1986). Marine invertebrates in the jewellery and curio trade include many shelled molluscs, corals, echinoderms, crustaceans etc. Certain marine invertebrates have long been targeted, because of their aesthetic qualities, for use in jewellery or as adornments. Particularly prized have been the shells of certain marine molluscs (e.g., *Trochus,* cameos) and the pearls and nacreous shells of pearl oysters, and black coral, as well as precious corals often growing in deep-water.

Some people collect invertebrates as a hobby, some for their aesthetic value, and others have a more scientific interest. It has been an essential element in the careers of many biologists (Yen and Butcher 1997), and the collections made by amateur collectors can provide an invaluable source of information for scientists.

Small scale collecting (collecting specimens for personal use) generally does no appreciable harm to the population. The aesthetic appreciation of invertebrates can go beyond appreciating dead specimens and the observation and photography of these animals in their natural environments is now a popular pastime, as is maintaining them in aquaria. Hermit crabs have been popular "pets" for several years and their supply represents a small industry. Growth in the hobby of aquarium keeping is leading to increased trade in certain live marine organisms, including invertebrates, particularly corals (Green and Shirley 1999) and anemones (see Section 6.10.3) leading to some conservation concerns about collecting methods (see Section 6.10.3) and the potential risk of the introduction of exotics (see Section 6.5).

3.2.8 Cultural significance

Marine invertebrates were of major significance as a source of food to coastal Aboriginals and Torres Strait Islanders, but also played a part as objects of cultural appreciation and social or religious symbols. In Australia, marine invertebrates played a relatively small part in Aboriginal art, with shells, crabs etc. sometimes being featured.

Shells and some other invertebrates were widely used as decorations by many people of the Indo-Pacific and as currency in some countries. Some are used for their supposed aphrodisiac properties, or for medicinal purposes in Asia.

3.2.9 Ethical reasons

We demonstrate above that there are many economic and other utilitarian reasons for focusing on the conservation of marine invertebrates. However, there is also a strong ethical argument, based on the inherent "right of existence" of all living organisms. While humans are currently the dominant species on earth, the enormous variety of organisms with which we share the planet is equally the product of billions of years of evolution. For this reason alone, it can be argued, each organism can be seen as entirely "deserving" of a chance to continue its existence, regardless of its immediate value or otherwise to human society. While this right of species to exist is often cited as an ethical rationalisation for conservation, the moral, legal or religious justification of this seemingly reasonable view is difficult to find (Yen and Butcher 1997). However, it can be reasonably argued that we have a moral obligation to conserve species and their habitats for (the use of) future generations (e.g., Greenslade 1985; Office of the Chief Scientist 1992; Burgman and Lindenmayer 1998).

Perspectives such as cultural, spiritual and aesthetic values are utilitarian because they place the emphasis on the preferences and needs of people, rather than the intrinsic values of species and communities (Burgman and Lindenmayer 1998) (see also Section 3.2).

Burgman and Lindenmayer (1998) discussed how the ecocentric ethic recognises that all species have long evolutionary histories and are inter-related in their life processes. In this view, the biotic community as a whole is the object of concern. The World Charter for Nature (UN 1982) recognises this ethic by stating that humankind is part of nature, and that every form of life is unique and warrants respect regardless of its worth to human beings. In contrast, the biocentric ethic argues for the value of all *individuals* - all are worthy of consideration. This ethic demands that humans treat the natural environment, and the individuals of all species within it, with a "hands off" policy.

The IUCN's Working Group on Ethics and Conservation developed an ethical basis for the conservation of nature (McNeely et al. 1990), composed of the following points (among others):

- The world is an interdependent whole made up of natural and human communities.
- All life depends on the functioning of natural systems to ensure the supply of energy and nutrients.
- The ecological limits within which we must work give direction to how human affairs can sustain the environment.
- All species have an inherent right to exist.
- The ecological processes that maintain the integrity of the biosphere are to be maintained.
- Sustainability is the basic principle of all social and economic development. This principle will enable utilitarian values of nature to be equitably distributed and sustained for future generations.

A different, though related, issue involves the ethical use of marine invertebrates (or biodiversity in general). Research on living organisms generally requires approval from an ethics committee from the institution in question; yet invertebrates are generally not given the same consideration nor accorded the same degree of protection as vertebrates. Nor are uses other than research subject to the same controls. The ethical aspects of research in marine areas were discussed by Hutchings (1998b).

3.3 Extent of the problem

3.3.1 Are marine invertebrates really at little risk of extinction?

There is a widely held view that marine invertebrates are less vulnerable to extinction than their non-marine counterparts. This is due to the belief that the oceans are too vast, and their inhabitants too prolific and widely distributed, for humans to ever extinguish species. However, many of these assumptions are now being challenged (Gould 1991; Carlton 1993; Norse 1993; Culotta 1994; Malakoff 1997; Carlton 1999; Hooper et al. in press). One reason that marine invertebrates are widely considered to be less vulnerable to extinction is that they generally are much more widely distributed than terrestrial species, and show a tendency both in the fossil record and modern times for lower extinction rates (McKinney 1997). However, it should be noted that a great number of marine invertebrates do not leave fossils so we have no data on their extinction rates over time. Species with restricted ranges are also less likely to be found in the fossil record simply because there is a much less likely chance of them being fossilised. The risk of extinction varies enormously both among and within different marine invertebrate taxa[32].

The notion that most marine invertebrates have large geographic ranges resulting from larval dispersal by ocean currents, rendering them less vulnerable to extinction, is incorrect[33] for the following reasons:

- Many have direct development (i.e. there is no swimming larva), some brooding their young to an advanced stage. The implications of limited dispersal are dealt with in more detail in Section 4.4.3.
- Even among those that have planktotrophic development, few are demonstrated to have wide dispersal (e.g., Scheltema 1986) while some that do show differentiation (e.g., Staton and Rice 1999). While many marine organisms have larvae that swim for days or weeks in the laboratory and are thus assumed to disperse widely, they frequently settle much more quickly than this when given a suitable substratum[34]. Patterns may also differ from place to place[35].
- Several studies have shown that unexpectedly high levels of genetic structure over regional scales occur in some taxa with planktonic larvae, including solitary corals (Hellberg 1996), limpets (Johnson and Black 1982, 1984a; Johnson and Black 1984b), mussels (Tracy et al. 1975), oysters (Karl and Avise 1992) and abalone (Huang et al. 2000b). Even congenors with planktonic larvae can differ markedly in regional levels of genetic structuring (Kyle and Boulding 2000).
- Some – especially those without planktotrophic larvae – may have narrower ranges than commonly believed due to isolation by habitat requirements or geographic boundaries. Ocean flow fields, such as those often observed in association with biogeographic boundaries, have the potential to constrain a species' geographic range irrespective of habitat availability (Gaylord and Gaines 2000).
- Some "species" previously assumed to have wide ranges have proved to consist of several distinct taxa, due either to inadequate taxonomic resolution, or because they are comprised of allopatric sibling species only distinguishable genetically. For example, the polychaete *Terebellides stroemii* was reported as occurring throughout Australian waters and worldwide. A detailed examination of the Australian material revealed four new species, and *T. stroemii* in fact does not occur in Australian waters (Hutchings and Peart 2000).
- Planktonic larvae alone may not prevent genetic isolation and speciation from occurring in marine invertebrates (see Section 3.3.1). For example, Staton and Rice

[32] We discuss those biological, ecological and life history traits which characterise the most vulnerable species in Section 4.4.

[33] There is also evidence that some species do have effective dispersal and high levels of gene flow, at least on regional scales (e.g., Benzie and Williams 1992; Williams and Benzie 1993; Uthicke and Benzie 2000a).

[34] There are numerous such examples in the literature. For instance, when Knowlton & Keller (1986) tested rates of larval settlement in a shrimp that lives symbiotically with anemones they found the larvae, despite being capable of swimming long distances, settled very close to the parents. Similarly, Olson (1985) showed that while ascidian larvae were capable of swimming for up to two hours, they generally settled in 10-15 minutes when given a suitable substratum.

[35] For example, Miller (1998) used genetic analysis to examine larval dispersal in black corals of SW New Zealand. At one site, results suggested the larvae settle very close to parent colonies although this was not the case at the other two study sites.

(1999) found genetic differentiation, corresponding geographically with a faunal boundary noted for other species, in sipunculans (*Apionsoma misakianum* species-complex), despite these having teleplanic larval dispersal.

The specific characteristics that make some marine invertebrate taxa vulnerable to extinction are discussed in detail in Section 4.4.

3.3.2 How many marine invertebrates are threatened or extinct?

We do not know the rates or scales on which invertebrate extinctions are occurring because of the very poor data that exist for most marine species. Thus, the real crisis may be hidden due to the dearth of systematists, naturalists and biogeographers capable of documenting marine life (Carlton 1993). The sea is not immune to extinctions, as the fossil record will attest (e.g., Jablonski 1994), and they are most likely to occur in particular habitats such as estuaries, coral reefs, intertidal shores and continental shelves and slopes (Carlton et al. 1999).

Species associated with endangered vertebrates are similarly threatened. An amphipod that was associated exclusively with the now extinct Stellers Sea Cow is now also extinct (Carlton et al. 1999). There are currently many species of threatened, endangered and recently extinct marine vertebrates recognised worldwide (e.g., 27 threatened and 2 recently extinct marine mammals; IUCN 1996). The extinct taxa undoubtedly had invertebrate parasites, some of which were presumably host specific and would have become extinct along with their host.

In the absence of even an approximate estimate of the number of species of marine invertebrates in Australian waters it is impossible to determine how many species are vulnerable, threatened, endangered or have become extinct in the last 200 years. In the marine environment worldwide there are few recorded historical or recent extinctions among marine and estuarine invertebrates (Carlton 1993; Carlton et al. 1999) and, for most phyla, there are none recorded. The first marine invertebrate extinction was documented as recently as 1991 (Carlton et al. 1991). However, this does not mean that extinctions are not occurring (Carlton 1993; Carlton et al. 1999) as we have no detailed information for most groups about even the composition of the living fauna, let alone more detailed distributional information, and historical records are few. For some better-known groups (hard corals, echinoderms, larger shelled molluscs, decapod crustaceans) there is better historical data, but much of the available data resides in museum collections and has not been analysed. On the available scant data, there are no known recent extinctions of marine invertebrates in Australian waters.

Carlton (1993) suggests that recent extinctions may have been overlooked for the following reasons:

- Hundreds of species have not been reported since their descriptions in the 18th and 19th centuries;
- Species may have become extinct before they have been discovered; and

- Decline in support for studies on systematics, biogeography and natural history has resulted in too few skilled workers.

Invertebrate populations can decline without being noticed, even in well-studied areas. For example, the decline of a native mussel from California was masked by the invasion of a morphologically similar sibling species, the European blue mussel (Geller 1999).

Literature reports, museum records and observations by individuals currently provide the best evidence of changes in distribution[36]. However, the extent of the marine habitat and the difficulty of assessing the fauna, especially small-sized animals, makes definite statements about absence difficult or even impossible. For this reason, reporting possible extinctions is rare and can lead to retractions[37].

Human exploitation may also result in local depletion or even extinction (see Section 6.3), but these examples generally attract some attention. Modification of habitat is another factor that might be implicated in at least local extinctions[38]. Although reporting complete extinctions of marine invertebrate species is still a rare event, the disappearance of small, poorly-known inconspicuous species with limited range, may be a more common event than currently suspected and such species may well be disappearing before they are even discovered, let alone studied.

McKinney (1998) compared fossil and modern terrestrial and marine extinction rates to test the idea that marine organisms are less prone to extinction. While his evidence supports the predominant view that they are less vulnerable, there were a number of caveats. For example, local extinctions (e.g., in coral reefs) may be very important, and marine extinctions will ultimately result in more fundamental biodiversity loss because there are more phyla and other higher taxa. In addition, the marine environment in the past has not been subjected to the same degree or extent of impact as it is now experiencing. While coastlines and the seabed seem vast, human impacts are increasing worldwide, especially in coastal areas, such that there are now few unaffected areas. These impacts are increasing the likelihood of local (and, ultimately, total) extinction of

[36] For example, some species of macro-sponges, well represented in earlier collections, appear to have disappeared from Sydney Harbour and Port Phillip Bay (see Section 4.5.1). Of more than 1300 species of molluscs from Sydney Harbour in the Australian Museum collections, more than 40 are only known from this harbour (I. Loch, pers. comm.) and many of these have never been recorded alive (i.e., only known from empty shells) suggesting the possibility that some of these taxa may be extinct. These observations support the idea that many local or even total extinctions of smaller, less conspicuous marine invertebrate taxa (such as those associated with the sponges) may have occurred and simply gone unrecorded. If the probable disappearance of such macrofauna from two such accessible locations has gone largely unremarked upon, there is little likelihood that the disappearance of smaller, less conspicuous invertebrates or those from less accessible locations would attract any attention at all.

[37] For example, Glynn & de Weerdt (1991) reported the probable extinction of a hydrocoral in the Eastern Pacific resulting from the worldwide coral bleaching events of the 1980s. However, Glynn & Feingold (1992) later reported the discovery of five live colonies in Panama in 1992.

[38] Cognetti & Curini-Galletti (1993) record that an interstitial platyhelminth belonging to a sibling species complex was exclusively known from two bays in the east coast of Corfu (Greece), where it lived intertidally in mixed silty sediments. In 1991, large amounts of fine sand were dumped on the beach in one of these bays to improve it for recreational purposes, resulting in the total disappearance of that bay's population.

some taxa, there being little to suggest that they will be buffered from the impacts of pollution, overexploitation, loss of habitat etc. (see Chapter 6).

3.4 Impediments to marine invertebrate conservation

The promotion of invertebrate conservation and its implementation has many hurdles to overcome. There is inadequate attention paid to conservation of any sort, but the seemingly innate human bias towards higher vertebrates and our own habitat has led to a disproportionate amount of conservation effort being devoted to terrestrial ecosystems and to vertebrates rather than to the "other 99%" of animal life. There is a similar bias in research and education, some university biology departments, for example, virtually omitting invertebrates from their courses. These problems can be regarded as both a cause and a consequence of the lack of understanding of the diversity and values of marine invertebrates and their inadequate treatment by educators (see also Chapter 9).

Not only do we have to penetrate an alien environment to observe marine life, much of the fauna is out of sight because it is benthic and infaunal. Our knowledge of these faunas, particularly in deeper water, is restricted to examining tiny fragments of habitat obtained by dredges or grabs and is completely unknown to the average person, and very poorly known even by marine biologists. Some of these issues are discussed briefly below.

3.4.1 Lack of knowledge

The lack of knowledge about marine invertebrates varies from group to group (see Chapter 2). Generally, the larger, shallow-water taxa are named, and generally something is known, at least in vague terms, about their biology. However, many groups, especially the smaller-sized taxa and the deeper water taxa, are very poorly known. Often there are not even adequate or useful collections, many species are not described and many groups do not have the experts available to work on them (see Thomas 1997, below). In addition, and in part due to the taxonomic impediment, there is little basic biological / ecological information available for most organisms, which makes informed conservation decisions difficult if not impossible (see Chapter 7). Without such data, basic management questions for particular taxa such as "how big is a population?", "has it declined?", "over what scales does dispersal operate?" cannot be answered. Yet, such information is needed to provide, for example, an indication of how large or close together marine reserves need to be to be self-sustaining.

The magnitude of our ignorance of the biota – and the difficulty of solving problems of access – is illustrated by efforts to understand even one of the better-known marine invertebrate groups, the squids. Several hundred species, several commercially fished, occur worldwide, and millions of tonnes are extracted annually, especially by Japan, Taiwan and Korea. Virtually nothing is known about their habits or biology, their role in ocean ecosystems, or their position in the food chain (Earle 1991). The 20 m long giant squid, *Architeuthis*, is known from fragments taken from the stomachs of sperm whales and a few individuals washed ashore or caught in nets, but none have been caught alive.

It is unlikely that an animal of this size would escape notice in any terrestrial habitat (Earle 1991).

As in non-marine ecosystems, there is a lack of adequate baseline data to enable an understanding of how populations, ecosystems or habitats have changed. It is important to realise that impacts have been taking place over hundreds of years, not just the last few decades (e.g., by fishing; transport of non-indigenous species through shipping; Carlton 1998) so that areas used as "controls" will probably have been subject to impacts for decades, and be already irrevocably altered from their "original" or "natural" state (if such a concept is supportable) (e.g., Carlton 1998).

3.4.2 Bias in emphasis and interest

There is a focus of research and conservation efforts towards big animals rather than small ones (in part reflected by the vertebrate/invertebrate bias), and towards terrestrial rather than marine environments[39]. When establishing research and conservation priorities there is thus an inherent bias, resulting both from the available knowledge base and from the interests of the majority of the proponents, reviewers, managers and politicians involved in decision making.

A related issue is that while we can access and easily see and touch things in the terrestrial environment, in the marine realm, only a very small part is accessible – for most people this is the intertidal zone. Below about 30 m, even scientific exploration has to rely on expensive remote sampling devices. This means that most marine habitats and creatures are not familiar to people and therefore of little or no interest or concern (see also Section 3.4.3).

Earle (1991) noted "In the sea as on land, it is important to "think small" when considering the diversity of life. Many people, even biologists, tend to ignore microbeasts, perhaps reflecting another of our human biases. Humans are giants, among the upper 5% in terms of size. Yet the biochemical action that shapes much of the biological, physical and chemical nature of the planet is accomplished by creatures at the other end of the size scale".

The estimated total number of species of vertebrates is 40,000; species of plants is 300,000; and species of marine invertebrates is 1 million or more[40] and all invertebrates up to about 30 million. Other estimates (e.g., Stork 1999) have come up with figures of

[39] These biases are deeply entrenched in conservation agendas. For example, a Steering Committee set up to explore the merits of launching a Millenium Assessment of the World's Ecosystems (Ayensu et al. 1999), along the lines of the Global Biodiversity Assessment and the assessments of the Intergovernmental Panel on Climate Change, makes virtually no mention of marine ecosystems, focusing purely on terrestrial and freshwater systems and their associated goods and services (food, timber, clean water, soil generation, maintenance of air quality etc.).
[40] There are estimates of marine invertebrates of up to 10 million species (Grassle and Maciolek 1992).

about 10-15 million in total. The number of workers involved in each major group is inversely related to the number of taxa[41].

3.4.3 Public disinterest

The vast majority of marine invertebrates lack the public appeal of higher vertebrates[42]. For example, many common intertidal taxa do not even have common names, being largely unknown outside specialist circles. Aside from there being little biological knowledge of these groups in the general population, there is also little understanding of "why marine invertebrates matter". Part of this disinterest undoubtedly stems from the fact that the marine realm is largely "out of sight, out of mind", and most people have little common experience of them (e.g., Ray and Grassle 1991). However, it probably also reflects the long-standing emphasis, on the part of governments, the scientific community, conservation agencies and educational institutions, on terrestrial environments.

Most people are familiar with terrestrial habitats but few have experienced the wonders of a coral reef or kelp forest except via books, film or TV. In addition, while it is easy to capture images of rain forests being cut down and to collect data to quantify the magnitude of habitat destruction on land, it is more difficult to study and document the processes and degradation of marine habitats, including coral reefs (Richmond 1994). Capturing and publicising such images (as, for example on news items relating to coral bleaching) would probably significantly raise the level of interest and support. Superficially, changes to marine ecosystems are less noticeable than terrestrial ones because oceans still look like oceans even when their contents are removed (Carlton 1998).

3.4.4 Resources

While Australia has recently prepared a Marine Science and Technology Plan (Section 8.3.5) to complement the Oceans Policy (Section 8.3.4), this has a strong focus on geological mapping (e.g., to assist mineral exploration). Other fundamental research, such as exploration for the purposes of increasing biological knowledge, is also advocated but no additional funding has been allocated for its implementation.

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[41] The great disparity of attention received by different groups at the basic and critical level of taxonomic understanding was illustrated by May (1994). Roughly one-third of taxonomists work on plants, while the remaining two-thirds split roughly equally between invertebrates and vertebrates. Thus, for every n taxonomists working on vertebrates, there are $0.1n$ taxonomists investigating plants and $0.01n$ taxonomists specialising in invertebrates. Most invertebrate taxonomists study insects, so there is an even greater disparity within the current taxonomic workforce that specialise in marine invertebrates.

[42] Analysis of the Queensland Museum's Reference Centre's records (Czechura 1994) showed that vertebrates were most popular (54.2% of enquires, cf. 28.3% for invertebrates). Marine molluscs generated the largest number of queries of any marine invertebrate group, but this was less than 0.5% of the total. If this pattern is general, and we see no reason to suspect that it is not, it is of considerable concern. Public support is a vital part of successful conservation, and it is expressed through political canvassing, financial support, practical assistance and so on.

Marine research is generally poorly funded in Australia (Marine Science and Technology Plan Working Group 1999) and of the small proportion allocated to marine biology, little is directed to invertebrate studies other than to coral reef research or commercial species. Thus, the lack of funding and resources relates not only specifically to invertebrate research and conservation efforts, but also more generally to marine science, research, exploration and technology. This lack of investment will have significant scientific and economic effects (Moore and Zeidner 1999). Australia's small commitment to marine research is highlighted by the fact that there is only a single research vessel – the *Franklin* – which is capable of working to 3000 m, plus an exploratory Fisheries vessel, *Southern Surveyor*. The availability of ship time to non-CSIRO personnel and projects is limited and only a very small proportion of their time is involved in projects concerning marine invertebrates.

The situation in Australia contrasts with that in other parts of the developed world, including the US, Europe and Japan. While the US has traditionally been the principal nation with interests and capabilities in exploring the oceans, Japan has now taken the global lead, having some of the world's most capable research vessels. In Japan there is broad popular and political support for the ocean sciences and their budget has grown at 15% per year during the past few years (Moore and Zeidner 1999).

Economic and political imperatives have meant that a considerable proportion of the funds available for marine biological and ecological research has been directed to vertebrates (especially marine mammals and commercial fishes) and those invertebrates that are either commercially important (e.g., rock lobsters, abalone) or the focus of extensive media coverage and popular attention (e.g., crown-of-thorns starfish, corals). While these are undoubtedly valid subjects for research, the scarcity of research funding means that the great majority of marine invertebrates, being neither generally known, nor popular, nor economically exploitable (other than as possible sources of useful chemical compounds), receive little attention at all.

General fauna surveys (as opposed to taxon-specific biological or ecological studies) have mainly been undertaken as part of the assessment process (e.g., EIS) for proposed large-scale developments, such as the third runway in Botany Bay (Wilson 1998) or the assessment of the Naval facility in Jervis Bay (Fuentes et al. 1992; Ward and Jacoby 1992; Williams et al. 1993; CSIRO 1994 etc.). Baseline studies for their own sake, especially those covering a relatively large area or carried out over a longer time period (e.g., the Port Phillip Bay survey 1969-1971 – Wilson 1998), require a considerable financial commitment.

3.4.5 Practical problems for research

Until recently, access to all parts of the marine realm has been very limited, and even now only a small proportion can be effectively reached by scientific observers, or even by remotely deployed instruments (Earle 1991). Submersibles are routinely used in various parts of the world in biological studies but Australia has no such facility. While revolutionary new techniques for mapping underwater terrain have been developed, we are still largely ignorant of much of the marine fauna. Standard sampling of benthos involves the use of a grab, dredge or benthic sledge. This has been likened to dropping a similar sampling device from an aeroplane into a forest and trying to interpret the fauna from the resulting sample (Earle 1991). Only a minute proportion of the world's ocean floor has been sampled to date.

Marine stations offer research facilities for scientists and students. New Zealand, with a smaller population than Sydney, has four marine stations with permanent staff and excellent facilities available to researchers. These stations have been the focus of a large body of work and have contributed considerably to the body of knowledge about New Zealand marine ecosystems and biota. Apart from several stations in Queensland (five on the GBR and one on North Stradbroke Island), the approach in Australia has been different. The major marine facilities (AIMS, Townsville (Queensland) and, the yet to be built, recently funded joint facility between AIMS and ANU in Darwin (the Arufu-Timor Research Facility - ATRF); and CSIRO Marine Research, Hobart (Tasmania), Cleveland (Queensland) and Marmion (Western Australia)) are dedicated scientific institutes and not set up as generally accessible marine stations. A new National Marine Science Centre at Coffs Harbour is jointly operated by two local universities and will serve as a major regional facility. The only other research facilities in temperate Australia are dedicated to fisheries research and a few small, mostly unmanned, teaching laboratories.

The lack of access to ships that can undertake offshore sampling has long been a serious impediment to Australian marine scientists - especially those not attached to a Federal Government organisation. While there is an obvious and urgent need for additional oceanographic ships, expensive ship time could often be better used. For example, a small change in CSIRO's sampling strategy (e.g., using smaller mesh size on at least some of the gear) on the seamounts south of Tasmania could have hugely enhanced our knowledge of the marine fauna. Similarly, in the past the state museums have sent people on geological cruises or fisheries research vessels, and with no extra sampling involved, a great deal of valuable material has been obtained that would have otherwise been discarded. Such value adding to costly cruises could markedly increase our knowledge of marine invertebrates without a huge extra expenditure.

3.5 Main issues and recommended actions

Issues

There is a general lack of interest in marine conservation because the marine realm is largely "out of sight, out of mind" for most people.

- The lack of interest also reflects the long-standing emphasis, on the part of governments, the scientific community, conservation agencies and educational institutions, on terrestrial environments.
- Funding, both for basic research and for conservation, is heavily biased towards higher vertebrates. Even among the marine invertebrates, certain taxa (i.e. commercial species and those important in gaining tourist revenue, such as corals) receive much more attention and funding than others.

- Because the great majority of marine invertebrates are poorly known and rarely seen, there is little understanding of why marine invertebrates matter.

There are many reasons why marine invertebrates are important. These include cultural and ethical reasons and, additionally:

  - Marine invertebrates contribute considerably to our economy - through the harvesting of commercial species (prawns, abalone etc.), as food and shelter for commercial fish species, as a source of pharmaceutical products (the potential of which has hardly been realised), through other commercial and medical applications, through tourism (e.g. coral reefs), and through their ecological services in maintaining the health of marine ecosystems.
  - Marine invertebrates already have an important role in educational activities which could be substantially enhanced.
  - Many people utilise marine invertebrates directly or indirectly in recreational pursuits.

Scientifically there is a serious taxonomic impediment and little biological and ecological knowledge of most groups. This is a result of:

  - Funding bias towards vertebrates and terrestrial ecosystems;
  - The lack of relevant expertise; and
  - The lack of facilities such as marine stations and research vessels, which is a serious hindrance to scientists, especially those not in Commonwealth agencies.

Recommended actions

Increase the conservation of marine invertebrates by:

- Development of programs and strategies to increase knowledge about, and interest in, marine invertebrates and their conservation by both government agencies and the general public.

Increase funding (and hence the level of activity), for marine research related to basic knowledge, conservation and management.

- Research efforts on marine invertebrates would be enhanced by developing mechanisms for maximising access to, and the use of, facilities such as research vessels and research stations and providing new facilities in as many bioregions as possible.

CHAPTER 4 – THE TAXON APPROACH TO CONSERVATION

The taxon approach to conservation involves the focusing of conservation efforts on individual taxa identified as being in need of targeted measures for protection and / or recovery. We have used the phrase "the taxon approach" rather than "the species approach" because, although this approach to conservation has traditionally been concerned with species, it can also be used for other recognisable taxon groups or entities at any level (e.g., family, genus, subspecies or even population) identified as in need of conservation[43].

While it is generally recognised that "biodiversity" encompasses living diversity at all levels from genes to systems, in practice the vast majority of targeted conservation efforts have, until very recently, focused on individual species (or sometimes subspecies or genera). Thus, the primary goal of threatened species conservation legislation and action has been to prevent the extinction of rare and endangered species. Under this approach, given the criteria employed under most legislation (see Chapter 8), considerable data are needed (e.g., on population numbers, range, temporal declines or range contraction, threats and so on) in order to estimate the conservation status of each species. This information is then used to classify species as "rare", "vulnerable", "endangered", "critically endangered" etc., and appropriate efforts directed to the protection and recovery of remaining populations. More recently, some legislation at both the Commonwealth and State levels, in addition to considering listing at the species level, also allows for the listing of threatened populations and communities (see Chapter 8).

The development of the single taxon approach to conservation, in response to the realisation that existing measures (such as conservation reserves) were not sufficient for some endangered species, has been ad hoc and uncoordinated. Efforts to date have been overwhelmingly biased towards threatened flora and larger vertebrates, particularly terrestrial species (the main marine exceptions being some high-profile marine mammals and seabirds). Such animals generally lend themselves well to this type of approach, being readily identifiable in the field, relatively easy to monitor and collect data for, and charismatic, attracting considerable public attention, support and sympathy. However, it has been suggested that this approach is inefficient and inadequate for addressing the conservation of a broader range of taxa, including small, uncharismatic but nevertheless important invertebrate species (e.g., Ehrlich 1992; Tear et al. 1995; New 1995a, 1995b; Reed and Clunie 1997; Yen and Butcher 1997; Ellis 1998; Kitching 1999).

The pros and cons of the taxon approach as it applies to marine invertebrate conservation are discussed in Section 4.1. On a global level, the role of the IUCN (World Conservation Union), their criteria for status evaluation and the listing of threatened species, and the relevance of these for marine invertebrates, are discussed in Section 4.2. In section 4.3 the application of the taxon approach is discussed – including status evaluation, listing of

[43] The same approach could also conceivably be used to target non-systematic groups (e.g., all the burrowing crabs living in mangrove habitats), although these would more usually be dealt with under the habitat or "systems" approach (see Chapter 5).

threatened species, and recovery plans – in Australia, as well as some of the difficulties with current criteria and listing of marine invertebrates. Given the difficulties involved in assessing the conservation status of most individual marine invertebrate species, tools to assist in the identification of potentially threatened taxa may be necessary. These could be based, for example, on characteristics they possess which render them more vulnerable to extinction (see Section 4.4). Some examples of Australian marine invertebrates of conservation concern are given in Section 4.5, while the final part (Section 4.6) evaluates management options for threatened marine invertebrate species, and some recommendations for action are made.

4.1 Arguments for and against the taxon approach

The arguments for and against use of the taxon approach for conservation of marine animals have been discussed by a number of authors (e.g., Norse 1993; Jones and Kaly 1995).

The **advantages** of this approach – focusing on individual species – include:

- It can be used for selected 'flagship' taxa (New 1991; Yen and Butcher 1997), which can focus community attention, enlist public support and enhance knowledge.
- Resources targeted at the conservation of one individual species generally have major carry-on effects for other, associated taxa. It can lead to the identification, protection and appropriate management of habitat that might otherwise not have been protected. Such habitat protection invariably assists the conservation of other species also found there.
- This approach caters for narrow-range taxa, i.e. those species restricted to very small areas that might not otherwise be included in conservation areas that have been determined using other criteria or approaches (such as an ecosystem/bioregionalisation approach, general diversity, other threatened taxa, availability of "unused" habitat, etc.).
- Identifying threatened species provides a means of measuring the decline and loss of biodiversity, and thus the effectiveness of conservation programs.

The **disadvantages** of focusing on individual species include:

- Lack of knowledge: Even for the better-studied groups of marine invertebrates, too little is known about the majority of species to make confident assessments of conservation status or the degree of threat[44].

[44] The situation is much worse for the smaller or more obscure groups, although the lack of knowledge is not confined to those that are numerically insignificant or rarely encountered. For many of the more obscure, the deep-water, and the meiofaunal groups, we have no idea of how many or which species occur in Australia, let alone an understanding of their distributions, abundances, endemicity or conservation status (see Chapter 2). Since only a proportion of marine invertebrate species are named, a still smaller fraction studied biologically or ecologically and only a very few close to being well understood, concentrating on species will never be an efficient or adequate approach for the vast majority of taxa.

- Practical difficulties: Due to sampling and observation problems, it can be difficult to assess the conservation status of many individual marine invertebrate species, especially small and sublittoral taxa, and infaunal species.
- Cost: One of the primary criticisms of the single species approach to conservation is that it is expensive (e.g., New 1995a; Reed and Clunie 1997; Yen and Butcher 1997). Considerable resources are required to support the processes of status evaluation, listing, and preparation and implementation of recovery plans, as evidenced by the high costs of recent action plans for various Australian vertebrate groups (see Yen and Butcher 1997)[45].
- Reactive rather than preventative: Woinarski and Fisher (1999) suggested that the focus of legislation in Australia and elsewhere is reactive rather than preventative and argued that it has performed poorly at preventing more species and communities from becoming threatened[46].
- May encourage illicit trade: Since one of the factors determining the monetary value of collected and traded species (e.g. marine shells) is their rarity, the listing of a species as threatened may actually be counterproductive, particularly if policing is inadequate.
- Does not take account of ecological interactions: The taxon approach does not take into account the complexities of species interactions, especially of commensals, parasites and so forth[47]. The importance of conserving *ecological function* must thus also be considered (see Section 3.2.1).
- Appropriateness of the “species” as the basic unit of conservation: Current research is changing our assumptions about species boundaries and the importance of within-species variation[48].

The reality is that the single-species (or taxon) approach to conservation was originally developed to address the particular conservation needs of highly visible, obviously endangered mega-vertebrates, and it is perhaps inappropriate to think of it as the only – or

[45] There are currently far fewer known, let alone listed, threatened species of marine invertebrates in Australia than threatened plants and vertebrates. However, with increased knowledge, the number of known threatened marine invertebrate species could be increased substantially. There are unlikely to be sufficient resources available to allow the traditional approach – i.e. development and implementation of single species-specific recovery plans – to be utilised for large numbers of listed invertebrate species.

[46] They pointed out that “inherent in a focus on endangered species is the concern that attention is concentrated on the expensive task of propping up the terminally ill at the expense of preventative management that may relatively cheaply reverse downward trends for a larger number of species and communities in decline but not yet destined for extinction”.

[47] In natural systems, species do not exist in isolation, but as part of a web involving numerous other taxa; however, in most cases our knowledge of species and ecosystems is not sufficient to allow us to factor these critical interactions into the conservation or recovery plans of threatened taxa.

[48] Many groups of invertebrates once believed to be relatively well understood by specialists are now proving to be far more speciose than earlier supposed (e.g., New 1999). Many “species” actually prove to be complexes of two or more sibling species (Wilson 1983b; Knowlton 1986; Solé-Cava and Thorpe 1986; Buss and Yund 1989; Solé-Cava et al. 1991; Stanhope et al. 1992; reviewed by Knowlton 1993; e.g., Yeatman and Benzie 1993). In addition, the evidence from molecular genetics suggests the existence of strong geographical and historical population structure in many invertebrates (e.g., Avise 1992; Bucklin et al. 1996) Thus, many widely distributed marine species must be considered as a series of genetically distinct allopatric populations as, for example, with the Pink Shrimp, *Pandalus borealis* (Kartavtsev et al. 1991) that potentially need to be given separate consideration in conservation strategies.

even the primary – tool available for the conservation of a broader range of species. In the case of marine invertebrates, the process of status evaluation can be important for problem definition (see “advantages”, point 4), and efforts directed at single species may be valuable for a few selected “flagship” taxa. However, due to the difficulties in applying this approach to the wider range of taxa in need of conservation, particularly those poorly known, inconspicuous and unlikely to gain much public interest, sympathy, or funding, means that a range of approaches are needed. Many authors have emphasised the need to shift the focus from threatened species or species-level taxa to protection of threatened communities (e.g., Reed and Clunie 1997), ecosystems and seascapes (Franklin 1992; Tear et al. 1995; Ray 1996; Bowen 1997b), conservation of ecological processes (Walker 1992; Ray 1996), and control or mitigation of threatening processes (Commonwealth of Australia 1996 etc.; Reed and Clunie 1997).

4.2 Status evaluation and threatened species lists

Yen and Butcher (1997) pointed out that there is a considerable difference between aiming for the conservation of all biological diversity and the conservation of threatened taxa. While one involves an attempt to conserve all life forms, regardless of threat, the other involves setting priorities based on the degree of threat. In prioritising threatened taxa, the terms “endangered”, “threatened”, “vulnerable”, and “rare” are the most commonly used categories, but they vary in definition and applicability. Hutchings and Ponder (1999) recently reviewed the appropriateness of the existing criteria for invertebrates in general. Here we briefly review how these terms are commonly used and assess the appropriateness of these terms, as defined, for marine invertebrates.

Conservation categories convey the conservation status, which is determined by “scientific assessment”. The intention is that this describes the degree of threat of extinction that is faced by a species. Typically, assignments are based on experience and intuition rather than on quantitative analyses. Threat has a time dimension but time lines can rarely be specified precisely. Instead, phrases such as “high risk” and “one or two decades” are used (Todd and Burgman 1998).

4.2.1 International status evaluation: IUCN categories

The International Union for the Conservation of Nature (IUCN; now known as the World Conservation Union) is often regarded as the umbrella organisation for the world’s conservation agencies and non-governmental organisations (Gimenez-Dixon 1998). The Species Survival Commission (SSC) is one of six Commissions of the IUCN. One of its goals is to assess the conservation status and threats to species worldwide. It maintains a Red List of Threatened Species, categorised according to degree of threat, as well as Red Data Books that provide more information (abundance, distribution, conservation status and other supporting data) on which the listing of the threatened species is based. The system was initiated in 1963, with the first Red Data Book produced in 1966 and the first Red List in 1986.

The IUCN provides objective definitions (which are revised from time to time) for each threat category and these are the most commonly used (sometimes in modified form) by conservation agencies. The IUCN categories of threat are a scientific assessment, and do not in themselves set priorities for action.

The first classification for levels of threat, which was applied widely up until the 1990s, consisted of six fairly loosely defined categories: Extinct, Endangered, Vulnerable, Rare, Indeterminate and Insufficiently known. Sometimes a category Commercially Threatened was also used (Gimenez-Dixon 1998). These were originally developed to draw attention to species of concern, mainly larger terrestrial animals and plants, although a Red Data Book on invertebrates was produced in 1982 (Wells et al. 1982). The categories were comparative and qualitative; risk levels or time horizons were rarely specified and the evaluation of conservation status was often based solely on the experience and intuition of experts in the groups concerned (e.g., Burgman and Lindenmayer 1998). These qualitative assessments gained wide acceptance because they were simple and required only modest data. However, they lacked explicit guidelines for assigning taxa to categories of risk (Mace and Lande 1991) and the heavy reliance on expert opinion led to a lack of consistency resulting in conflicting opinions (Todd and Burgman 1998).

In the early 1990s the IUCN initiated a review process to develop new criteria that were more objective, systematic and transparent. New categories (Extinct, Extinct in the Wild, Critically Endangered, Endangered, Vulnerable, Lower Risk (Conservation Dependent, Near Threatened, or Least Concern), Data Deficient, and Not Evaluated), with specific criteria based on decision rules related to range, population size, and population history, were adopted in 1994 (IUCN 1994). These criteria had their first major application in 1996 with the production of the *1996 IUCN Red List of Threatened Animals*.[49]

Application of the criteria highlighted some difficulties in relation to assessments of harvested species, long-lived species, and some small and very narrowly distributed plants and animals. Further reviews are currently being undertaken which will examine these problems, as well as look at the time periods over which declines are measured, and the application of threat criteria to managed species (Mace 1999). There have been a number of workshops on the relevance of the criteria for assessing marine taxa (fish – Hudson and Mace 1996; and invertebrates in general – Hutchings and Ponder 1999; marine issues in general – Mace 1999).

The workshop on general marine issues (Mace 1999) considered whether there are fundamental differences between marine and terrestrial species, i.e. whether the existing criteria were appropriate for assessments of marine species and how they could be modified to become more appropriate or relevant. The differences between marine and terrestrial species were identified as being related mainly to scale, interconnectedness and

[49] The 1996 Red List is available on the web at http://www.wcmc.org.uk/data/database/rl_anml_combo.html, or in interactive form, as the World Conservation Monitoring Centre's Animals Database, at http://iucn.org/themes/ssc/96anrl/contents.htm). A list of the current criteria can be found at http://iucn.org/themes/ssc/redlists/criteria.htm.

dispersal, as well as poor detectability and difficulties with obtaining measurements in the oceans. Some of the specific issues addressed included:

- ***Using the criteria for harvested and managed species.*** Whether any distinction should be made between declines resulting directly from a managed harvesting program and those that result from an uncontrolled threatening process, and if they can in fact be separated from each other.
- ***Measuring range sizes and the values for thresholds.*** Whether existing threshold values and guidelines for application are applicable for marine species unconstrained by land mass boundaries and often with very large (sometimes global) ranges. Because population sizes are so hard to assess in the oceans, range sizes become all important in listing marine species. Also, many marine species occupy three-dimensional spaces and the range area thresholds in the existing criteria are two-dimensional measures. It was felt that some criteria – such as the 'typically less than 100 km^2' annotation under criterion D2 ("Vulnerable") – were very small for marine species and meant that many were excluded from listing even though this may have been appropriate.
- ***Population decline and extinction risk.*** A species can be listed as threatened on the basis of a decline in population, range or habitat, measured in generation lengths to account for variation in life histories among species. However, the existing threshold decline rates during the last ten years or three generations (whichever is longer) may not take account of the variable potential for recovery among species. Another problem is how to measure and detect decline in species that show very wide natural fluctuations. For such species it was felt that the current threshold decline rates are probably too low, and therefore include too many species, especially for harvested and managed species.
- ***Population viability.*** Assessments of populations to determine their viability, or Population Viability Analysis (PVA), are an important aspect of conservation biology (e.g., Burgman and Possingham 2000), but with many pitfalls and traps for unwary players.
- ***Accounting for life history variation in marine species.*** The criteria attempt to account for variation in life histories among species by assessing population size using only the number of mature (i.e. breeding) individuals. This definition, however, may not adequately incorporate the full range of life histories seen in the marine environment and the workshop proposed amendments to the definitions of 'mature individuals' and 'generation length' to take account of some of the unusual life history characteristics of marine species (e.g., variation in recruitment, sex changes, clones, sessile species, inter-dependence between species (e.g., symbioses), and generation length).

A draft of the recommendations and proposed changes to the categories, criteria and guidelines has been produced (IUCN/SSC Criteria Review Working Group 1999). The appropriateness of IUCN and similar criteria to (terrestrial) invertebrates were also assessed in a workshop held in Sydney in 1997(Hutchings and Ponder 1999)[50].

[50] This workshop's primary objective was to assess the appropriateness of the criteria (based on IUCN) employed under the NSW *Threatened Species Conservation Act 1995*, and to identify problems with applying these and similar criteria to invertebrates. While this workshop was concerned specifically with

4.2.2 Other criteria

An alternative approach to setting conservation priorities, based on point-scoring, was developed by Millsap et al. (1990). This was based on the allocation of points to a taxon for each of a number of demographic, life-history and management variables. Point scoring systems are attractive because they provide greater resolution among taxa than do rule-based approaches, and are transparent and repeatable. However, they can be sensitive to the somewhat arbitrary weights assigned to the variables, and many of the variables are not independent (Todd and Burgman 1998).

Todd and Burgman (1998) argued that a common flaw of all of the methods for the assessment of conservation status is the fact that they ignore uncertainty in the data. They proposed methods for interpreting different types of uncertainty (including probabilistic uncertainty and reasonable bounds for expert judgement) and producing a range of plausible outcomes, rather than a single outcome such as would result from a point scoring approach.

4.3 The threatened species approach in Australia

There are two important processes involved in the taxon approach to conservation. The first involves the process of determining that a taxon is threatened (status evaluation and listing), and the second involves determining what remedial actions are required for recovery (the recovery plan).

4.3.1 Criteria and the listing process

In Australia, approaches to, and legislation for, listing threatened species and dealing with their management and recovery vary widely among the different jurisdictions (States, Territories and Commonwealth) (see Chapter 8) [51].. Most have enacted specific threatened species legislation with provisions for the listing of threatened taxa and appropriate actions for management and recovery. The categories of threat, criteria employed for listing, geographical and taxonomic scope, and procedures for listing and management vary with jurisdiction. Neither the contemporary IUCN scheme nor the

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terrestrial invertebrates (the marine fauna in NSW being covered under separate legislation that was, at the time of the workshop, not enacted), many of the issues raised are of relevance for marine invertebrates also. The greatest failing of the IUCN criteria with regards to invertebrates was seen as their reliance on set thresholds (numbers) in populations and details of geographic extent. The enormous variation among plants, vertebrates and invertebrates in such biological attributes as size and ecology make it inherently difficult to find universally applicable thresholds. For many invertebrates, there can be difficulties assessing effective population size or the number of individuals, and population numbers may differ through the life cycle of a species or fluctuate widely from year to year. The assessment of population numbers, decline etc. is difficult enough for terrestrial vertebrates but virtually impossible, and certainly impractical for most marine invertebrates (e.g., Mace 1999) (see also 4.3.3).

[51] An exception is South Australia, which currently has no dedicated threatened species legislation, although wildlife protection is covered by provisions under other Acts such as fisheries and national parks. The legislation dealing with threatened species is detailed in Section 8.2.

points systems developed by Millsap et al. (1990) are used by any agency in their original form, but they have been adapted for local use (Todd and Burgman 1998)[52].
The lack of a consistent, coordinated national approach to the protection of threatened taxa in Australia is inefficient and detrimental (Hutchings and Ponder 1999). The opportunity for coordination was offered by ANZECC – a now disbanded forum for member governments to exchange information and develop coordinated policies on national and international environmental and conservation issues. Such coordination is not reflected in the disparate legislative approaches taken by each jurisdiction. State-Commonwealth coordination is afforded in the legislation of some jurisdictions, such as NSW[53].

Currently, very few marine invertebrates are listed under any Australian threatened species legislation (although a number are afforded some level of protection through other means[54]). This is due at least partly to a perception, to some extent justified, that marine invertebrates are generally at less risk because of the ability of many to undertake larval dispersal and the enormous size of the oceans (see Section 1.1.2). It is also due to a lack of knowledge (see above) and because listing marine invertebrates has only recently become possible in most (not yet all) States and Territories (see Chapter 8). With better data, or the resources to evaluate available data (including unpublished information), it will be possible to document other threatened taxa or those already extinct. However, reasonable published data exist for only a few species, mainly those that are commercially valuable.

The IUCN Red List contains many Australian invertebrates nominated by Australian biologists and assessed using IUCN criteria, however, these lists appear to have been largely ignored by Australian conservation agencies and their advisory committees (Hutchings and Ponder 1999), Tasmania being the only exception (Yen and Butcher 1997). These taxa should be considered for listing by the relevant states, territories and the Commonwealth (Hutchings and Ponder 1999).

[52] For instance, Schedule One of the Commonwealth *Environment Protection and Biodiversity Conservation Act* (EPBC Act) *1999* (formerly of the *Endangered Species Protection Act 1992*) lists species classified as extinct, extinct in the wild, critically endangered, endangered, vulnerable, or conservation dependent, the definitions based loosely on the qualitative IUCN criteria. ANZECC (the Australian and New Zealand Environment and Conservation Council) previously (see Section 8.2.1) maintained Threatened Australian Fauna (and Flora) lists, with qualitative classifications of threat based on the IUCN system.

[53] Through provisions that any species included in the Commonwealth lists which occur in the State automatically become nominated and must be considered by the Scientific Committee (this is equally true for marine species listed under the *Fisheries Management Act 1994* as for terrestrial species). However, this is not the case for all States and Territories, and does not work in reverse, i.e. all State-nominated species do not automatically become nominated for Commonwealth lists. The recently enacted EPBC Act represents an attempt to improve coordination in this regard.

[54] For instance, all scleractinian corals and giant clams are protected both nationally and internationally by CITES legislation (see Section 9.1), and the collection of many species of marine invertebrates is controlled in various states by permit systems, especially in marine parks, or by bag limits, seasonal closures etc.

In all cases, legislation and management efforts have traditionally focused primarily on the terrestrial vertebrate fauna and vascular plants, and the listing of threatened invertebrates is a relatively recent phenomenon.

Listing of threatened invertebrates

In South Australia, invertebrates are not even legally recognised as "animals", and are not, therefore, eligible for listing. Even in other jurisdictions, the few listed invertebrates are insects (mainly butterflies) and terrestrial or freshwater molluscs, with very few marine invertebrate species currently found on the schedules of any Australian legislation. In New South Wales, for instance, full protection has only recently been extended to marine invertebrates or plants (*Fisheries Management Amendment Act 1997*). There have been almost no attempts by any jurisdiction to systematically evaluate the conservation status of their marine invertebrate faunas. The exception is Victoria, where the Department of Natural Resources and Environment recently commissioned an assessment of Victorian marine invertebrate species of conservation concern (O'Hara and Barmby 2000), although this was restricted (for reasons of data availability) to the major groups of molluscs, echinoderms and decapod crustaceans. The aim of this report was to prepare a list of marine invertebrate species of conservation concern in Victoria, including any believed to be in need of nomination for listing under the *Flora and Fauna Guarantee Act 1988*, and to identify vulnerable habitats and so forth. The authors used several criteria to identify potentially threatened species, including a small extent of occurrence (i.e. geographic range), small area of occupancy within Victoria, specificity to vulnerable habitats, unsustainable exploitation (non-commercial species only) and low abundance. Several marine invertebrate species[55] were identified as in need of nomination for listing under the FFG Act (O'Hara and Barmby 2000).

Understandably, given the magnitude of the "unknown", few authors have attempted to prepare lists of Australian invertebrate taxa believed to be under threat. Those which do exist deal almost exclusively with terrestrial and freshwater species (summarised by Yen and Butcher 1997), use various definitions of threat criteria, and are often vague about, or do not include, the data on which the assessments are based (Yen and Butcher 1997).

Yen and Butcher (1997) recommended that an "insufficiently known" category be used (equivalent to "data deficient"), enabling species suspected to be at risk, but which lack adequate quantitative data to assign them with certainty to another category, to be listed. Yen and Butcher (1997) pointed out that this would enable a "two list system" to be used which would help to ensure that species would not be listed on the basis of insufficient information. Such a system is used under the US Endangered Species Act and in Victoria: the first list being threatened species listed under the Act, the second including taxa of conservation concern, but not listed under the Act. While we can see the advantage of such a recommendation, as the alternative for most marine invertebrate taxa

[55] **Crustacea** - *Halicarcinus* sp. (undescribed), *Athanopsis australis*, *Eucalliax tooradin*, *Michelea microphylla*
Echinodermata – *Amphiura triscacantha*, *Ophiocomina australis*, *Apsolidium densum*, *Apsolidium handrecki*, *Pentocnus bursatus*, *Thyone nigra*
Mollusca – *Bassethulia glypta*

would be, in all probability, not listing them, realistically taxa listed under such a category would be unlikely to receive proper attention or priority for management.

4.3.2 Recovery plans and plans of management

All the Australian legislation requires that certain actions be taken for listed species. This usually involves the preparation of Recovery Plans and Action Plans. However, while the preparation and implementation of such plans may be an option for a few high-profile species, they are probably not a practical, let alone cost-effective, option for the great majority of taxa. Tear et al. (1995) looked at all recovery plans approved as of August 1991 under the U.S. Endangered Species Act. They found considerable taxonomic bias (mainly terrestrial and vertebrate), an overall lack of detailed biological information and considerable delays in implementation with an average of at least five years between each step in the recovery plan process (from listing to recovery plan approval and subsequent revision). They found that inadequate funding for long-term ecological studies is a major contributing factor to the scarcity of available information and results in a selective focus on economically important or high-profile species.

These authors (Tear et al. 1995) pointed out that evidence to support the criticism that the species approach (as used in the US ESA) is inefficient includes:

- The backlog of unlisted candidate species (approx. 950 species in Category 1 and 2944 in Category 2 in 1991);
- The long average time period between listing and recovery-plan approval (6.4 years average in 1988); and
- The fact that there are far fewer recovery plans than listed species (282 / 531 in 1988).

Clearly, if legislation requires production of a Recovery Plan, monitoring the success of its development and implementation needs to be undertaken as a critical component of this process.

Recovery plans for threatened species do not necessarily have to be for single species.In Victoria, for instance, they are increasingly being prepared for guilds of threatened species and for threatened ecological communities, and under the new Commonwealth *Environment Protection and Biodiversity Conservation Act* there is provision for the preparation of bioregional plans, covering various threatened habitats and suites of species. Such an approach seems more sensible given the increasingly apparent need to extend legislative protection to all taxa and more in line with other approaches such as conservation of habitats and ecosystems.

4.3.3 Difficulties with invertebrate lists and current criteria for status evaluation

The small numbers of marine invertebrates listed as threatened by IUCN is due to a number of interrelated factors, and these same reasons also explain, in part, why few marine invertebrates have been listed by the Australian Commonwealth, or the States and

Territories (see below). Legislative issues, such as the non-inclusion of invertebrates in definitions, have been another factor in Australia, especially in the recent past.

The workshops on the applicability of the IUCN criteria to marine organisms (Mace 1999) and invertebrates (Hutchings and Ponder 1999) point out that the established criteria are difficult or impossible to apply to most marine invertebrates because of problems such as the following:

- Assessment of numbers and extent of widely scattered marine populations is difficult;
- There is little or no knowledge of intraspecific genetic structure for the great majority of taxa[56];
- Populations of many marine invertebrates naturally fluctuate widely (e.g., seasonally and/or annually), making assessment based on some estimate of population numbers largely meaningless;
- There is little knowledge of the numbers of individuals necessary for a viable population (i.e., *critical population size*), a factor that will vary considerably with the taxon; and
- The success of many populations may depend more on larval survival and recruitment, about which virtually nothing is known for the vast majority of taxa. Recruitment of populations may also occur, at least in part, through long distance dispersal.

Critical population size

The application of the concept of *critical population size*, including "minimum viable population", to IUCN and similar criteria uses values developed for terrestrial organisms, in particular, vertebrates. While theoretically also relevant to many marine organisms, the broad range of life history strategies seen in marine invertebrates (e.g., asexual versus sexual reproduction, widely different breeding strategies and fecundity, etc.) mean that there are clearly exceptions that would require modification of the concept. For example, how should clonal organisms be treated? – should they be counted as different individuals? In many cases population *densities* could be more critical than population *size*, especially in relation to sedentary or sessile organisms, where population density can be critical for ensuring fertilisation success (e.g., Peterson and Summerson 1992; Malakoff 1997) (see Section 6.3.1).

Jones and Kaly (1995) argue that, due to lack of knowledge, a different approach to listing threatened marine species is needed. An alternative, though subjective, approach is to recognise the biological or ecological characteristics of species at least potentially threatened, and develop management strategies as a precautionary measure (Jones and Kaly 1995) (see below).

[56] If there is marked intraspecific genetic structure the established criteria for assessing threatened status may not be applicable.

4.4 Characteristics of marine invertebrates likely to be at risk

In the absence of complete data for any particular taxon, we can still identify groups of marine invertebrates likely to be more at risk than others, based on characteristics which theory or experience have shown tend to make taxa more vulnerable to population decline or extinction. These characteristics can be used to identify taxa (species or higher ranks) so they can be considered in management strategies, and precautionary protective measures can in turn be adopted.

Identification of the characteristics that render certain groups of marine invertebrates more vulnerable to decline or extinction has been attempted by a number of authors, including Dye et al. (1994), Jones and Kaly (1995), Stone et al. (1996), McKinney (1997), Ponder and Grayson (1998) and Roberts and Hawkins (1999). For instance, Dye et al. (1994) produced a list of the characteristics that render marine species in general (not just marine invertebrates) vulnerable to extinction. A modified version of this list, from Roberts and Hawkins (1999), is reproduced in Table 4.1. These characteristics are generated not only from theoretical population biology and common sense, but also from examination of empirical examples[57].

This approach enables a more practical assessment but clearly not all the above characteristics will be able to be scored for many invertebrates because, in practice, some information is usually unavailable (e.g., details on reproduction). However, many of the above characteristics can be scored for a great number of taxa by inference with regard to studied related taxa, and available natural history observations on the taxon in question. Thus, the use of some biological and ecological characteristics can be used to estimate probable extinction risk using actual or inferred information[58].

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[57] For instance, Carlton (1993) examined the characteristics of several marine invertebrates that appear to have, or could, become extinct and noted that these included a restricted geographic distribution, restricted habitat, and/or limited dispersal abilities. Clark (1994) showed how rare ascoglossan molluscs (seaslugs) in Florida have characteristics that might increase their extinction risk, such as non-planktotrophic development (i.e., poor dispersal ability), fragmented habitats, low density, and/or small populations.

[58] Jones & Kaly (1995) listed various characteristics of marine organisms which may make them vulnerable, such as restricted geographic range, low fecundity, habitat specialists, subject to overexploitation, restricted breeding sites, subject to mass mortality/catastrophic declines, susceptible to environmental stress, etc. The last three factors are, however, generally difficult to assess without some detailed and/or specific knowledge of the taxon concerned. In particular, susceptibility to other kinds of localised threatening processes (e.g., pollution, dredging, secondary affects of trawling etc. - see Chapter 6) cannot be readily assessed in general terms and would usually have to be considered separately.
Ponder & Grayson (1998) were able to use criteria relating to distribution, development, accessibility and value to assess the vulnerability of a large number of marine molluscs, in large part by extrapolation using known information from related taxa.

Table 4.1: Characteristics that render marine invertebrates (and other marine species) vulnerable to extirpation and extinction (reproduced from Roberts and Hawkins 1999)

[a] Semelparity - reproducing only once in a lifetime. [b] Iteroparity - reproducing multiple times in a lifetime.
[c] Protandry is when an organism has a functional male stage followed by a functional female stage
[d] Allee effects occur when a reduction in population density has significant impacts on the ability of the organism to reproduce.

Characteristics	Vulnerability	
	High	**Low**
Population turnover		
Longevity	Long	Short
Growth rate	Slow	Fast
Natural mortality rate	Low	High
Production biomass	Low	High
Reproduction		
Reproductive effort	Low	High
Reproductive frequency	Semelparity[a]	Iteroparity[b]
Age or size at sexual maturity	Old or large	Young or small
Sexual dimorphism	Large difference in size between sexes	Does not occur
Sex change	Occurs (protandry[c] in particular)	Does not occur
Spawning	In aggregations at predictable locations	Not in aggregations
Allee effects[d] at reproduction	Strong	Weak
Capacity for recovery		
Regeneration from fragments	Does not occur	Occurs
Dispersal	Short distance	Long-distance
Competitive ability	Poor	Good
Colonizing ability	Poor	Good
Adult mobility	Low	High
Recruitment by larval settlement	Irregular and / or low level	Frequent and intense
Allee effects[d] at settlement	Strong	Weak
Range and distribution		
Horizontal distribution	Nearshore	Offshore
Vertical depth range	Narrow	Broad
Geographic range	Small	Large
Patchiness of population within range	High	Low
Habitat specificity	High	Low
Habitat vulnerability to destruction by people	High	Low
Commonness and / or rarity	Rare	Abundant
Trophic level	High	Low

We consider the following characteristics to be of importance in determining whether a species is potentially vulnerable. Most should be able to be assessed for the majority of macroinvertebrates:

- Habitat readily accessible (if collected for food/ornaments etc. or commercially harvested etc.);
- Visible (if of large size and edible, or otherwise of value, on surface in daytime, contrasting colour to background etc.);
- Valuable (i.e. in terms of food, specimens, ornaments etc.);
- Restricted distribution or a few small, highly fragmented populations;
- Direct development;
- Low fecundity;
- Slow to reach reproductive maturity;
- Close association with threatened taxa (as parasites or commensals, food source etc.) or threatened habitat; and
- Rare.

Some of these characteristics are further discussed below.

4.4.1 Restricted geographic range

The significance of restricted geographic range is recognised in the IUCN criterion D2 where any taxon with a range of less than 100 km^2 is considered vulnerable. If a taxon is limited in distribution to a small area or areas, there is more chance of extinction through stochastic events or localised disturbance (e.g., an oil spill, reclamation etc.). In addition, narrow range taxa are often limited in distribution by other characteristics, such as a low capacity for dispersal or specialised habitat preferences, which would preclude them from successfully dispersing from or recolonising an area in the event of such a disturbance.

Examples

Examples of marine invertebrate taxa with extremely restricted ranges include the starfish *Marginaster littoralis*, which has a known range of 1 ha of intertidal zone in the Derwent River estuary, Tasmania (Edgar et al. 1991), four hermatypic (reef-building) coral species endemic to Houtman Abrolhos, WA (Veron 1993), and gastropods (Gofas 1992) and other taxa (Koslow and Gowlett-Holmes 1998) restricted to seamounts. Perhaps one of the most striking examples is that of *Smeagol*, a small blind slug restricted to the upper littoral of very small areas of gravel or cobble beaches in SE Australia and New Zealand (Tillier and Ponder 1992). These gravels are themselves very restricted habitats in southeastern Australia and, as far as is known, the different species of these slugs all have very small distributions, one just a few metres in extent. Such habitats are very likely to be heavily impacted by oil spills should they occur. Two species are known only from Phillip Island, Victoria (the site of a recent oil spill), one from either end of the outer coast.

While restricted geographic range is often seen as one of the most important contributing factors to a species' vulnerability and potential extirpation, the converse argument – that species with large ranges are less likely to become extinct – does not necessarily always hold true. For instance, Vermeij (1993) examined a number of recently extinct marine species and demonstrated that not all had small ranges. However, only one of these species was an invertebrate, the rest being marine birds and mammals. It is unclear whether any widely distributed marine invertebrate taxa would be likely to be threatened in the same way as marine vertebrates, which are generally large, high trophic level consumers with relatively low reproductive rates.

4.4.2 Habitat specialisation

Habitat specialisation by a species can result in vulnerability if the habitat itself is threatened. The importance of this characteristic is indicated by the fact that the central element in four apparent recent extinctions of marine gastropods was a vulnerable, extinguishable habitat (Carlton 1993)[59].

It is also important to consider the fact that the habitat requirements of a species may differ significantly at different points in its life-history; one or more of these life phases may be specialised for a particular habitat with limited availability. This can result in a demographic "bottleneck" and lead to vulnerability – to a greater or lesser extent – of the species as a whole[60]. There are many examples (Pawlik 1992) of larvae requiring specific types of habitat or chemical stimuli for successful settlement to occur, such influences potentially affecting population size and structure even if there are no obvious factors affecting the adults.

Parasitism and commensalism are two life history strategies that often involve considerable specialisation for a particular "habitat", i.e. an associate or host species. For example, many trematodes (and others) have intermediate host species or vectors (e.g.,

[59] For example, the first marine invertebrate to be documented as having become extinct in recent times – the eelgrass limpet of the east coast of North America - was a habitat specialist on seagrass (Carlton et al. 1991). Another habitat specialist, the limpet *Siphonaria compressa*, has been cited as South Africa's most critically endangered marine mollusc, being restricted to the blades of the eelgrass *Zostera capensis* and found in a single lagoon on the Atlantic Cape coast of South Africa (Herbert 1999). In Japan, 465 species of macroinvertebrates have been identified as being restricted to saltmarsh and estuarine environments now threatened by habitat loss. Of these, 389 species (83.7%) are designated as threatened and a few are probably already extinct (WWF Japan 1996). Examples of Australian marine invertebrate taxa with specialised habitat requirements include *Smeagol* (see Section 4.4.1) which is restricted to upper littoral clean gravel, a rather rare and very discontinuous habitat in southeastern Australia, and *Zoila* and *Umbilia* cowries associated with large cup sponges, a habitat highly vulnerable to, and severely threatened by, commercial trawling (Ponder and Grayson 1998). Likewise, the polychaete *Nereis posidoniae* is restricted to blades of seagrass (Hutchings and Rainer 1979).

[60] For example, the American Lobster (*Homarus americanus*) has a requirement for cobble substratum, a shelter-providing habitat of restricted distribution, during a particular phase of its life history (Wahle and Steneck 1991). Similarly, blue crabs (*Callinectes sapidus*), an estuarine and coastal species in the US, have habitat requirements ranging from high-salinity environments along the coast to low-salinity conditions in near-riverine portions of the estuary, and from oyster bars and sand bottoms to vegetated habitats such as saltmarshes and seagrass beds (Engel and Thayer 1998). Because the life cycles and growth of these and other crustaceans are relatively complex, they are vulnerable to chemical and physical alterations to their habitat, as well as negative impacts to the food webs that support them, in different ways during their different life stages (Engel and Thayer 1998).

snails, birds, fish etc). Parasites on rare or threatened species may in turn become vulnerable, although little consideration is given to the conservation of this "hidden" component of biodiversity. In fact, parasites are generally seen as an additional threat to the health of vulnerable populations.

Some taxa may be potentially at risk because they have a narrow ecological profile, or are currently living near the limits of their environmental tolerance. For example, many tropical organisms such as corals are currently living near their upper thermal limit (e.g., Hatcher et al. 1989 – see Section 6.7), hence there may be little margin for adaptation to environmental change. A similar view applies to populations on the extremes of that species' range, as these populations are under strong selection pressure.

4.4.3 Limited dispersal

The prevailing wisdom seems to be that marine invertebrates that brood their young or otherwise produce young via direct development, rather than having widely dispersed planktonic larvae, are at greater risk of extinction. Part of the reason for this reproductive mode being a risk factor is that it is often associated with low fecundity and a smaller geographic range. It may also be linked to whether reproduction is predominantly sexual or asexual. In ascidians, for example, it has been inferred that species with obligate sexual reproduction have populations strongly interconnected by larval dispersal, while those which rely on asexual reproduction have more highly differentiated local populations (Ayre 1990; Ayre et al. 1997b)[61]. There are, however, many exceptions to this pattern.

Taxa restricted to geographically or ecologically isolated habitats may gain a selective advantage from limited dispersal. For example, Gofas (1992) argued that gastropods on seamounts have developed strategies for limited dispersal because they gain a selective advantage by retaining the larvae within range of the seamount habitat rather than losing them into the wider ocean where they would have little chance of encountering suitable habitat for settlement (i.e. another seamount). This strategy has resulted in a higher incidence of endemicity than that found amongst species with pelagic larvae. While evolutionarily successful, endemics are of course more vulnerable to extirpation (e.g., if the seamount habitat is subjected to human or other disturbance). While the reproductive strategies of gastropod and bivalve molluscs are relatively easy to infer from the nature of their retained larval shell, little detail is known of the life histories of many other invertebrates. While some of these can be inferred by comparison with what is known of closely related taxa, such inference can be problematic as some taxa show considerable variation, even within genera or among phenotypes within a species. Such differences in

[61] Ayre (1990) reviewed the use of genetic data to infer the mode of reproduction and extent and directionality of dispersal for a range of Australian temperate invertebrates. This was followed by further studies on ascidians (e.g., Ayre et al. 1997b) and recent work on holothurians by Uthicke and Benzie (2000a) that has shown that gene flow occurs between isolated populations although whether this pattern reflects present day dispersal or dispersal which occurred in the past is unknown.

life-history traits among closely related species are commonly orders of magnitude greater than differences between closely related terrestrial animals[62].

4.4.4 Reproductive vulnerability

Reproductive attributes associated with vulnerable marine invertebrates include low fecundity or highly variable reproductive output[63]. When coupled with the effects of reduced population density on reproductive output (see below), this can lead to progressive contraction and fragmentation of a species' range. The importance of this effect suggests that for many marine invertebrate species, population density may be a much more appropriate indicator of vulnerability than population size, which is the criterion (developed from studies on terrestrial vertebrates) that has traditionally been applied.

Reduced population density

Since many invertebrates are broadcast spawners, fertilization success is highly dependent on population density. Low population densities can lead to recruitment failure because excessive dilution of propagules greatly reduces the probability that the gametes will meet and be successfully fertilised in the water column (this is known as the "Allee effect" or "cost of rarity"; Allee et al. 1949; Odum 1959). Non-broadcast spawners can be affected as well, if individuals are unable to find mates or interact in some other essential way. Low population densities can be caused by natural or quasi-natural events, such as cyclones or disease, or as a result of human activities such as over-harvesting. Fishing can cause a reduction in populations of non-target as well as target species, for instance through bycatch, or benthic habitat damage from trawling. This effect has helped speed the extinction of giant clams from several regions of the Indo-Pacific (Wells 1997a; Roberts and Hawkins 1999), but probably the best-documented example is the Californian White Abalone (see Section 6.3.2). Similarly, Peterson and Summerson (1992) found that the slow recovery of bay scallops after a red tide suggested they are recruitment limited at low density by the mesoscale abundance of spawning adults. This resulted in lasting effects of the red tide propagated through "basin-scale population inertia".

4.4.5 Overexploitation

Many examples exist of species that have suffered dramatic population declines as a consequence of being over-harvested by humans, whether through casual collecting, recreational harvesting for food or ornaments, or commercial fisheries. There are many such examples (see Section 6.3). However, the fact that a population is exploited is not,

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[62] For example, the intertidal genus *Patiriella* includes the greatest diversity of larval types and life histories known for any extant starfish group, with some species producing vast numbers of swimming larvae and others morphologically very similar brooding non-feeding, direct developing offspring (Hart et al. 1997; Byrne et al. 1999).

[63] For example, studies by the NSW Fisheries Research Institute in Jervis Bay, as part of the Jervis Bay Marine Ecology Study (Fuentes et al. 1992) demonstrated that scallops in this area show great variation in recruitment from year to year. Similarly, giant clams and some tropical reef corals only recruit successfully at long and unpredictable intervals (Kojis and Quinn 1994; Wells 1997a).

per se, an indication that it is threatened – some exploited species have high rates of reproduction and with careful management can be sustainably harvested even at relatively high levels. However, it is of concern where exploitation leads to decline of the species, is unchecked or the target species has other (biological or ecological) characteristics associated with vulnerability[64]. Given that many exploited marine invertebrate taxa show one or more of the characteristics associated with vulnerability (see Table 4.1), catch, effort and population parameters of targeted species need to be closely monitored to ensure that exploitation does not lead to a crash in numbers through reproductive failure or any other mechanism.

Another key factor in the risk posed by exploitation is the accessibility of target populations, whether to casual collectors, those in search of food or specimens for the shell or aquarium trades, or commercial fishers. Factors that determine accessibility include distance from population centres, road access, depth of water, presence of commercial fish stocks, and so forth. The advent of SCUBA has greatly increased the accessibility of sublittoral habitats. Areas accessed by trawling, scallop dredging etc. are also impacted frequently.

It has long been argued that many marine populations are protected from the normal risks associated with exploitation by the vastness of the oceans and the existence of de-facto “refuges” such as deeper parts of the ocean unreachable by standard fishing gear, or substrates unsuitable for trawling. These refuges can theoretically provide the seed stock for recolonisation of the more accessible, and heavily exploited, habitats. However, habitats in these inaccessible areas may be substantially different from those found in shallower zones, and many species may not extend into such areas. Moreover, continued improvements in technology and the increasing sophistication of fishing and navigation technology mean that an ever declining number of areas remain protected from disturbance or exploitation by their inaccessibility. For instance, previously isolated deep reefs, abyssal areas, and seamounts are now increasingly subject to fishing disturbance. Fisheries and collecting can also have a profound indirect impact on non-target species that share the same habitat (see Section 6.10.1).

4.4.6 Rarity

There have been many attempts to define what is meant by rarity. Species have been considered rare because they have a few individuals, occupy a small range, are marginal populations of widespread species, are seldom seen, etc. (reviewed by Gaston 1994). Chapman (1999) identified three main reasons for studying spatial and temporal patterns of abundance and population dynamics of rare species, i.e. those that appear to persist with natural small abundances or limited range and not those whose populations are in decline (either naturally or due to anthropogenic influence):

- It is often assumed that rare species are more threatened with extinction than common species (Soulé 1986; Simberloff 1988). There is, therefore, concern about the capability of populations of rare species to persist in increasingly disturbed habitats.

[64] For example, many gastropod molluscs exploited for the shell trade have low fecundity coupled with direct development (Ponder and Grayson 1998).

- Due to perceived loss of global biodiversity in recent years, declining populations of common species are recognized as being potentially under more threat than are populations of naturally rare species (Caughley 1994; Gaston 1994). Understanding what allows populations of naturally rare species to persist may help identify processes that may slow the decline of common species.
- Most species are probably rare according to one definition or another (Gaston 1994), but most intensive ecological studies have been done on more common species. There is therefore increasing emphasis on studying rare species and developing ecological theories about causes and consequences of rarity (reviewed by Gaston 1994).

Chapman (1999) reviewed theories about rarity with respect to intertidal and shallow subtidal invertebrates, and analysed how well such theories apply. The general conclusion was that the lack of quantitative data on abundances, ranges, habitat-requirements, dispersal and connectedness among populations of marine invertebrates means that their status as rare species cannot really be assessed appropriately. It is also unlikely that, without extensive sampling programs and considerable expense, adequate data could be obtained for these small, often cryptic animals, which typically have very patchy, variable and unpredictable patterns of distribution and abundance, and include species with many different life-histories and represent many phyla. Chapman (1999) therefore suggested that future research on rare organisms in marine habitats should build upon the long and successful history of experimental marine studies to test specific hypotheses about processes influencing rarity in the field. She further suggested that such studies would not only add a new dimension to our current understanding of rarity, but would also provide badly needed data on the status of rare marine invertebrates.

4.5 Some examples of threatened or potentially threatened Australian marine invertebrates

Few marine invertebrates have actually been listed as threatened under Commonwealth or State legislation (see Section 4.2 and Chapter 8). In part, this is because it is very difficult to identify species as threatened without adequate data on basic biology, ecology, distribution, and abundance, and such data is not obtainable for most Australian marine invertebrates. The exceptions are some commercially exploited species and a very small number of “flagship” species (e.g., *Patiriella vivipara* - see below) highlighted as warranting conservation concern. Even these relatively better-known organisms have been inadequately researched and lack the comprehensive data required for effective fisheries management or conservation decisions. Some species, although not listed in any category under threatened species legislation, are protected in other ways, for example through fisheries regulations and restrictions on harvesting (a de-facto assumption of vulnerable status). However, blanket regulation, such as the ban placed on the collection of all macroinvertebrates from many rocky shores in some states such as New South Wales, avoid the necessity to deal with individual taxa.

This section is not intended to be a comprehensive list of threatened Australian marine invertebrate taxa due to the paucity of information available for most marine invertebrate species, making it impossible to properly assess their conservation status. Such a list does not yet exist anywhere in the literature, nor is one likely to in the immediate future. Instead, this section contains notes on a number of invertebrate species (or higher taxa) identified, in the literature or by our correspondents, as potentially threatened or otherwise of conservation significance. As suggested by Yen and Butcher (1997), there is a need for invertebrate expert panels to be established, at the Commonwealth level, to assess conservation issues relating to marine (and other) invertebrates.

The notes for each of the examples below identify the taxon, and where possible, the habitat, distribution and why there is concern (e.g., rarity, population decline, vulnerable habitat, overexploited; plus a brief summary of threats). Some of those listed are harvested species, due in part to the greater availability of data, and thus there is some overlap with other sections (e.g., Section 6.3).

4.5.1 Porifera

Many epibenthic sponges are extremely vulnerable to physical damage and removal by trawling equipment, and form an important component of the bycatch of many trawl fisheries. Sponge abundance appears to have been very adversely affected by commercial trawling, and various large sponges that were once common on the continental shelves and/or in bays around Australia are now rare or absent from these areas. These include the cup sponges, many of which had interesting faunas associated with them (e.g., some large cowries).

One species that has been identified as threatened (J. Hooper pers. comm.) is *Poerion neptuni* (also known as *Cliona patera*), or Neptune's Cup Sponge. This species has recently been found off Cape York, in the Gulf of Carpentaria, and the Arafura and Timor Seas. It was thought to have become extinct in the early 1900's, having not been recorded from the South China Sea (where it was obtained by trawlers) since this time. Recently some populations have been discovered on muddy bottoms in the Gulf, off Darwin, Cape York and the Sahul Shelf, but they are uncommon and very susceptible to benthic trawling. Their status is thus threatened, and populations may be small and vulnerable to extinction without a trawling exclusion zone (J. Hooper pers. comm.).

Similarly, a number of macrosponges (i.e., erect sponges, up to 0.75 m) appear to have vanished from Sydney Harbour and Port Phillip Bay (although they may still occur in other locations). These areas have been extensively collected in recent times (e.g., Shelf Benthic Survey- off Sydney Heads by the Australian Museum; Port Phillip Bay Survey; Roche Reserve Trust Collections) and several species of the genera *Dendrilla*, *Dictyodendrilla* and *Taonura* well represented in earlier collections, are not found in these areas today. These sponges are large, obvious and would be noticed if present (P. Bergquist pers. comm.).

4.5.2 Cnidaria

Many octocorals (alcyonarians or "soft corals") are common in inter-reefal areas, and are severely affected by trawling, being long-lived and slow growing (e.g., Hutchings 1990). No species of Scleractinia (stony corals) are known to be particularly threatened (J. Veron pers. comm.). However, amongst the few species of hard corals present in temperate areas, it seems that those present in Port Jackson and Botany Bay have declined in population, presumably due to habitat changes as a result of dredging and/or changes in water quality (P. Hutchings pers. observ.).

In general jellyfish (Hydrozoan, Scyphozoan and Cubozoan medusae) and ctenophores are of little conservation concern, being on the whole widely distributed and abundant, and there are apparently no known threatened taxa in these groups in the Australian region (L. Gershwin, P. Cornelius pers. comm.). However, at least one Australian jellyfish may require conservation consideration. *Catostylus mosaicus* has been targeted as a commercially viable, harvestable and exportable resource (Pitt and Kingsford 2000b, a), but populations within individual estuaries appear to be separate stock units (Pitt and Kingsford 2000a).

Worldwide, habitat destruction might conceivably affect local 'races' of jellyfish such as those that occur in the Palau Island marine lakes. For instance, local varieties of the largely sedentary *Cassiopea*, recorded on some islands in various parts of the world, may be at risk if their favoured seagrass lagoons are polluted, even though this taxon apparently can swim fairly long distances and there may be few unique and geographically restricted populations (P. Cornelius pers. comm.). Such populations have not been identified in Australia.

4.5.3 Polychaeta

No Australian species of polychaete are currently known to be threatened. The giant Australian beachworms *Australonuphis teres* and *A. parateres* are collected by the thousands by recreational fishermen and professional collectors. However, the only method of collection presently used and sanctioned involves hand collecting and this does not appear to pose a threat to these species (see Section 6.10.3). The blood worm *Marphysa* n.sp (Hutchings and Karageorgopoulos in press) is commercially collected in Moreton Bay and occurs in seagrass beds. Collecting these worms damages an ecologically important habitat unless great care is taken.

4.5.4 Mollusca

A general overview of molluscan conservation (both terrestrial and marine) by Ponder (1998b), noted that there are few immediately obvious problems with the conservation of marine molluscs. Many species have large ranges and planktonic larvae and are unlikely to become threatened, although local extinction through habitat destruction, or possibly over collecting, may be possible. Massive and indiscriminate shell gathering by local people in areas such as the Philippines has resulted in local denudation of shores (Wells

1982; Gomez et al. 1994) and this can be exacerbated by concurrent habitat destruction (Ponder 1998b).

There have been relatively few attempts to list vulnerable marine molluscs. Starmühlner (1985) reviewed rare and endangered marine molluscs that had become threatened through overexploitation or pollution. Fry and Robinson (1986) listed many Australian marine molluscs as vulnerable but without supporting evidence, and although their list has been treated as authoritative (Jones and Kaly 1995), many of the taxa listed are widely distributed and in no way threatened (Ponder 1998b). Ponder and Grayson (1998) listed species considered vulnerable as a result of the shell trade.

In Australia, various mollusc species are subject to commercial exploitation. Many intertidal species are collected recreationally for food, particularly near population centres and intense local collecting can lead to local extinctions[65]. However, massive over-collecting is generally not a problem in Australia where local regulations usually prohibit such activities and the commercial return would be inadequate for the effort that would have to be expended. Abalone, species of the family Haliotidae, are perhaps the best known and currently the most valuable of exploited molluscs (see Section 6.3.2). The green snail (*Turbo marmoratus*), found in tropical Australian waters, is sometimes regarded as commercially threatened (Ponder 1998b). Some other marine molluscs are sometimes regarded as commercially threatened, including the pearl oysters *Pinctada maxima* and *P. margaritifera* (Ponder 1998b). However, these and several other commercially exploited molluscan species occurring in tropical Australian waters have wide Indo-West Pacific distributions[66]. Illegal fishing for giant clams (Tridacnidae) and *Trochus* shell can be a problem in some areas, and fishing operations for these taxa in the past have reportedly depleted numbers severely in many areas[67]. The Giant Clam *Tridacna gigas* and the Southern Giant Clam *Tridacna derasa* were listed on Appendix II of CITES in 1983 due to severe depletion of populations, and the remainder of the family were listed in 1985 because of their similarity in appearance to the threatened species (Sant 1995)[68]. Scallops are particularly vulnerable to overexploitation (see Section 6.3.2) and fishing for them by dredging is very damaging to the environment (Section 6.10.2). Most commercial Australian scallops are Australian endemics.

Some gastropods may be potentially vulnerable because of their interest to shell collectors (see also Section 6.10.3). For instance, species belonging to families such as the Cypraeidae (cowries), Volutidae (volutes), Conidae (cones) and Muricidae (muricids), are prized by serious shell collectors. These taxa are often difficult to obtain, usually reasonably large, often colourful, and some are found in inaccessible areas. Some of these taxa may be potentially threatened by collecting, especially those that have direct

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[65] For example, *Anadara trapezia*, a mud "cockle", was common in estuaries in the Sydney area (e.g., Careel Bay, Pittwater, Hutchings 1974) but is now rare or absent in most localities.
[66] For example, *Trochus niloticus* (Nash 1985; Goldman 1994).
[67] Local extinctions of some tridacnid species have been reported in parts of their range (Wells et al. 1982; Heslinga et al. 1984; Munro 1989; Munro 1993).
[68] While the large *Tridacna* species are certainly very vulnerable, this is probably not the case with the smaller species, namely *Tridacna maxima* and *T. crocea*, both of which can be extremely common, are not exploited in many areas (including Australia), and have very wide geographic ranges (Ponder 1998b).

development, low fecundity, are restricted to small geographic areas and/or are accessible to SCUBA or intertidal collectors (Ponder and Grayson 1998; Ponder 1998b). Others, such as a few large predatory species (e.g., *Charonia tritonis* and *Cassis cornuta*), may be vulnerable locally to collecting (both commercial and recreational), being conspicuous and desirable as ornaments. These latter two examples are tropical taxa that are, however, also relatively common, widespread and of little commercial value.

Many molluscan taxa are closely associated with potentially threatened habitats or host taxa, the examples being far too numerous to list here. There are some large gastropod families exclusively dependent on other groups of invertebrates, these including Eulimidae (on echinoderms), Magilidae (hard and soft corals), Cerithiopsidae and Triphoridae (sponges), and Mitridae (sipunculans).

Some marine molluscs are known to have very small ranges and are potentially at risk, e.g., *Smeagol* (Section 4.4.1) and some (mostly undescribed) species from seamounts, such as those off eastern Australia and, presumably, southern Tasmania. A number of species of saltmarsh or estuarine gastropods and bivalves have been listed as threatened by WWF Japan (1996) due to critical habitat loss.

Some species may be at risk as a result of the impacts of non-indigenous species (Section 6.5). The numbers of beach washed specimens of *Gazameda gunni*, a turritellid gastropod found in SE Australia, appear to have declined precipitously in habitats occupied by the confamilial introduced species *Maoricolpus roseus* in Tasmania (G. Edgar pers. comm.).

4.5.5 Crustacea

For the majority of crustaceans, there is insufficient information to determine whether any species are threatened. Several crustacean species are subject to commercial exploitation, including crayfish, prawns and crabs as well as a few other taxa such as the Balmain Bug (*Ibacus peronii* and the morphologically similar "Smooth Bug", *Ibacus* sp.). Most of these fisheries are regulated and populations may be reduced but they are unlikely to be threatened with extinction. However, the development of deep-water fisheries for certain crustaceans such as deep-water shrimps ("Scampi" – *Metanephrops* sp.) and giant crabs (*Pseudocarcinus gigas*) (Section 6.3.2: King Crabs) may pose a threat, as these tend to be slow-growing and commercial fishing pressure can quickly decimate populations (P. Davie pers. comm.) without allowing adequate recovery periods. Thus, although the risk of extinction for these species is probably low, good management practices are important.

Amphipods are brooders, often with restricted distributions. Physiologically, amphipods are very sensitive to hydrocarbons and thus very susceptible to oil pollution (J. Lowry and P. Berents pers. comm.). A family of amphipods, the Urohaustoriidae, live in sediment and are very dependent on grain size. Thus, sandy bottoms of varying sediment types are not faunistically the same. No known species of Australian marine isopod is known to have such a limited distribution that it is of concern, although many are known only from type localities, probably indicating lack of data (G. Poore pers. comm.). Others are parasitic and therefore linked to the density and abundance of host species. Various groups of barnacles are also associated with other animals so that they are dependent on the health and survival of those organisms[69].

While many species of decapods have a wide distribution, some such as the Moreton Bay Fiddler Crab, *Uca longidigitum*, have a restricted distribution, this latter species being confined to the Bay. Many species of decapods are commercially fished and it is essential that these fisheries are well controlled and that the biology of the target species is investigated to ensure that they are fished in an ecologically sustainable fashion. In addition, many decapods are included in the bycatch from trawling and not enough is known about their biology to know if any of these species are threatened at either local or regional scales, and the impact that any population declines may have on community structure (P. Davie, pers. comm.).

4.5.6 Bryozoa

No individual species of bryozoans are known to be threatened, although bryozoans associated with specific habitats such as seagrass epiphytes or the very rich and largely unstudied faunas associated with coral reefs are dependent on the health of those ecosystems. Other critical habitats include seamounts, seagrass beds, and submerged vertical rock walls, while soft bottom species could be severely affected by commercial trawling (D. Gordon pers. comm.). The impact of introduced species cannot be assessed because there are no baseline studies (P. Cook, P. Bock pers. comm.) although a recent paper documents the known introduced taxa (Campbell in press).

4.5.7 Echinodermata

In Tasmania, several species of endemic starfish (seastars) have been highlighted as being of conservation concern because of their small populations with restricted geographic ranges. These include *Marginaster littoralis* (found in the Derwent estuary; the sole population of which is located beside an oil terminal), *Smilasterias tasmaniae, Patiriella vivipara,* and *Pachycentrotus bajulas*. All of these are found only in southeastern Tasmania and have small populations, and are therefore potentially subject to stochastic risk (G. Edgar pers. comm.).

One of these species – *Patiriella vivipara* – is one of very few marine invertebrates to have been listed as threatened under any Australian legislation[70]. Another endemic

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[69] For example, the Pyrgomatidae are associated with tropical hard corals; Archaeobalanidae with sponges and gorgonians; while the tropical Chelonibiidae, Coronulidae, and Platylepadidae are epizoic on turtles, whales, and dolphins or dugongs respectively (D. Jones pers. comm). However, little taxonomic work has been done on any of these barnacles and virtually nothing is known regarding their biology or ecology (D. Jones pers. comm).

[70] Listed in July 1998 under the *Tasmanian Threatened Species Protection Act* 1995 (Schedule 3, Part 1 – Endangered flora and fauna – Extant). *P. vivipara* is a small (maximum radius 15 mm), orange-yellow endemic asterinid with a highly restricted distribution, being only known from four locations (Pittwater, Roches Beach, Eagle Hawk Neck and on the southern shore of Fortescue Bay) in southeastern Tasmania. It inhabits the intertidal zone, being found under rocks or in rock pools. It is one of only a few species of

Tasmanian seastar with a restricted distribution, *Smilasterias tasmaniae*[71], is listed as rare[72].

Some echinoderms are commercially exploited and potentially susceptible to local depletion through over-harvesting. These include sea urchins (Echinoidea) and sea cucumbers (Holothurioidea) (see Section 6.3.2: Holothurians)[73].

4.5.8 Tunicata: Ascidiacea

Given the current state of knowledge it is not possible to assess whether there are any threatened species of ascidians (A. Davis pers. comm.). Current genetic evidence indicates that populations of solitary ascidians are strongly connected and should therefore reinvade habitat easily following disturbance (Ayre et al. 1997b), although some species respond to very specific settlement cues (Davis 1996; A. Davis pers. comm.), and larval settlement could be affected by changes in substrate etc. (P. Mather pers. comm.). In contrast, colonial ascidians exhibit philopatry[74], with clear evidence of population subdivision, and their ability to invade new or formerly disturbed sites will be limited; thus, marine protected areas may not function well for these species (A. Davis pers. comm.).

4.5.9 Cephalochordata

Cephalochordates (lancelets or amphioxus) were once found in bays as well as on the continental shelf, but apparently no longer occur in bays (for example, *Epigonichthys bassanus* is probably extinct in NSW, although it occurs elsewhere in Australia) (B. Richardson pers. comm.). The reasons for this are unknown but it may be due to siltation

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seastar to exhibit viviparous reproduction (giving birth to juveniles). The limited distribution and low dispersive life history of this species make it vulnerable to local perturbation, and populations appear to have declined since the early 1970s and 1980s when the initial surveys were taken (Prestedge 1998; Prestedge 2001). These declines appear to have resulted from a variety of factors, including water quality decline associated with the discharge of sewage effluent and stormwater runoff, a necrotic disease, decline in the carpets of the small mussels *Brachidontes rostratus* and *Xenostrobus pulex* in which the seastars sheltered, and the impacts of non-indigenous species such as the Pacific seastar *Asterias amurensis* (Prestedge 1998; M. Byrne, L. Marsh pers. comm.). The distribution, biology and life history parameters of *P. vivipara* are described in detail by Prestedge (1998; Prestedge 2001).

[71] This species was "rediscovered" in 1994 having previously not been seen since the 1960s (Parks and Wildlife Service 1998). F. Rowe (pers. comm.) has suggested that up to 10-20 such southeastern Australian species of very small size and restricted distributions could fall into the same category requiring similar protection.

[72] http://www.dpiwe.tas.gov.au/inter.nsf/WebPages/RLIG-5447GS?open

[73] L. Vail (pers. comm.) has suggested that there is reason for considerable concern about the conservation status of the main species (*Holothuria nobilis* and *Holothuria scabra*) targeted for commercial exploitation. Worldwide, trepang fisheries are renowned for their "boom and bust" cycles due to over-exploitation (Hamel et al. 2001). These animals are very susceptible to overexploitation since they are mainly harvested in shallow water and are generally exposed and thus easy to find. They are also apparently long-lived (possibly over a decade), and seemingly have low recruitment (L. Vail pers. comm.), although recent studies have shown that *H. nobilis* has high gene flow between populations along the Great Barrier Reef (Uthicke and Benzie 2000a). It is important that more information be obtained about the general biology of the commercially important species so that sustainable catch limits can be imposed.

[74] The tendency to return to, or to stay in, a home area.

interfering with their filtration, or possibly bottom trawling (B. Richardson pers. comm.). Provided shallow water habitats are not critical to their life cycle (which is unknown), there probably are no nationally threatened species (B. Richardson pers. comm.).

4.5.10 Other phyla

Too little is known to assess the status of species of the minor phyla, or even more abundant phyla such as Platyhelminthes, Nemertea, or Nematoda. The following examples, taken from responses to questionnaires, illustrate the fact that lack of information is the main limiting factor:

- Platyhelminthes – "Conservation issues are related to habitats. At present we hardly know what is there. It is not practical to signal particular taxa or habitats as endangered or threatened. Habitat destruction (including loss of hosts for symbiotic species) is the same for everything" (L. Cannon pers. comm.).
- Nemertea – "Our knowledge of the phylum Nemertea, both within Australian waters and worldwide, is just too insufficient for any valued assessment to be made" (R. Gibson pers. comm.).

4.6 Management options for species conservation

A broad range of measures are likely to be necessary for the management and recovery of threatened species, depending on the reasons identified for the species' decline.

Protection of a species' habitat is often a key first step. Whether the targeted taxon is in a reserve or not, habitat management may be necessary to ensure long term survival of the species (Yen and Butcher 1997). Some of the threatened species legislation requires identification of critical habitat for threatened species, and its protection and/or management. In reality, to undertake such a task for even a subset of marine invertebrates would be daunting. Fortunately, the habitats of many marine invertebrates largely coincide, or at least substantially overlap, so some major generalisations are possible. Readily identifiable habitats (see Section 5.3.2) such as seagrasses, coral reefs, mangroves, intertidal rocky shores etc. have many marine invertebrate taxa restricted largely or completely to those habitats. Options for the conservation of habitats and systems, including Marine Protected Areas, are discussed in Chapter 5.

Control of threatening processes (e.g., introduced pests, pollution, recreational fishing, etc), is undoubtedly the most critical component of threatened species management. Without this, protection of habitat will be ineffective. The major threatening processes in the marine environment are discussed at length in Chapter 6, as are options and recommended actions for their management. These are important not only for threatened species but for the conservation of marine biodiversity in general. Indeed, many of the recommendations to alleviate threatening processes will be directly relevant to threatened species. For instance, for species primarily threatened (directly or indirectly) by overexploitation, some form of fisheries management will be necessary.

Captive breeding and reintroductions should be seen as the last resort in conservation. It is a very expensive option and the chances of successful reintroduction to the wild are limited. This option is discussed in detail elsewhere (e.g., Yen and Butcher 1997) and we do not consider it a viable conservation option for most marine invertebrates except in exceptional circumstances (e.g., where a high profile species, or one of considerable phylogenetic or economic importance, is involved). Successful captive breeding of giant clams has taken place at Orpheus Island and the resultant stock used to reseed fished-out areas (J. Lucas pers comm.). Another aspect of captive breeding involves specimens being bred for scientific research, live displays, trade, and aquaculture. Yen and Butcher (1997) suggest collectors could obtain perfect specimens though such projects where there is a market that is willing to pay. This could, perhaps, include rare species, or those difficult to obtain in the wild, or those otherwise protected or regulated. However, it could be difficult to distinguish between those legitimately bred in captivity and those illegally collected from the wild.

4.7 Main issues and recommended actions

Issues

Strategies for conserving and listing terrestrial vertebrates (and to a lesser extent invertebrates), are not necessarily transferable to marine invertebrates.

The taxon approach for conserving marine invertebrates is generally neither a practical nor cost effective strategy for the great majority of taxa. The adoption of habitat-based conservation strategies are the most effective for the conservation of most marine invertebrates.

Taxon-based conservation is a useful approach in some circumstances, such as for:

- However, if, in so doing, the focus is restricted mainly to readily identifiable, high profile habitats, a large proportion of marine invertebrate diversity will be neglected.
- Taxa harvested (including by collectors) or threatened indirectly by other exploitative activities (these can be managed by specific controls on numbers taken or methods and/or effort employed);
- Taxa that have narrow geographic ranges (once identified, specific measures can be implemented);
- Taxa that live in highly specialised environments threatened by specific, manageable, threatening processes (where targeted reduction in, or cessation of, impact(s) may be possible);
- Conspicuous taxa with a high, or potentially high, public profile (flagship taxa) or are critical for ecosystem maintenance and/or function (keystone taxa).

To enable more effective taxon-based conservation, studies on the biology, ecology, taxonomy and distribution of marine invertebrates must be encouraged and facilitated, especially in coastal and estuarine habitats where the threats are greatest. Basic information such as faunal composition of particular areas must be obtained, as well as information essential for the effective management of particular taxa, such as reproduction and life history, feeding, population structure, distribution and habitat requirements.

- Museum collections can potentially provide good information on the distribution and habitat preferences of marine invertebrate taxa, as well as helping to identify narrow range taxa that may be at risk

Currently there is no coordinated threatened species listing and management. Commonwealth, State and Territory agencies should use the *Environment Protection and Biodiversity Conservation Act 1999* (EPBC Act), with the goal of moving towards uniform threatened taxon legislation and a single national threatened taxon list.

There is a need for expert advice on various aspects of marine invertebrate conservation and particular taxonomic groups. Currrenly most agencies lack ready access to such information.

Recommended actions

That the criteria used to assess threatened taxa be modified, where necessary, for the assessment of marine invertebrates.

Use of taxon-based conservation actions only where justifiable and necessary.

- Following Yen and Butcher (1997), a category of “insufficiently known” should be used to enable listing of those taxa suspected to be at risk, but which lack adequate quantitative data to assign them with certainty to another category (pending reassessment of the criteria). These should not necessarily be afforded the same status as those that can be assigned to other threat categories with more certainty.
- IUCN-listed Australian taxa should be assessed and considered for listing by the relevant agencies.

Develop programs to gather essential basic information such as faunal composition, biological, distributional and ecological data.

- Encourage biological and ecological studies on marine invertebrates in relevant institutions and universities.
- Databasing of museum collections should be facilitated to provide a national on line facility.

Use habitat-based conservation as the preferred stategy for the great majority of marine invertebrates.

- Ensure that as wide a range of habitat types and bioregions are represented as possible.

State and Territory agencies should coordinate, through the *Environment Protection and Biodiversity Conservation Act 1999* (EPBC Act), a uniform listing of threatened species and habitats, and threatening processes.

- Develop uniform legislation for Australia.

A marine invertebrate expert panel or panels should be established to recommend appropriate conservation strategies:

- For habitat-based conservation strategies.
- For major taxonomic groups, especially those in which taxa have been identified as being vulnerable or at risk.

CHAPTER 5 – THE SYSTEMS APPROACH TO CONSERVATION

Because of the virtual impossibility of dealing with all marine invertebrates on a species basis, there is a need to focus on protecting the *systems* of which invertebrates are a part, at a variety of scales, from small (e.g., assemblages, communities, habitats) to large (e.g., bioregions). In this chapter we have chosen to combine all of these scales in order to discuss the "systems approach" to conservation, since many of these concepts (e.g., ecosystem, community, habitat) are vague and somewhat interrelated, and are consequently difficult to distinguish even in theoretical terms, let alone from a practical point of view.

As discussed in Chapter 4, to attempt to determine the critical habitat of even a subset of marine invertebrates would be a daunting task. Fortunately, the habitats of many marine invertebrates largely coincide, or at least substantially overlap, so some major generalisations are possible. Readily identifiable habitats such as seagrasses, coral reefs, mangroves, intertidal rocky shores etc. have many invertebrate taxa restricted largely or completely to these habitats. On a larger scale, the effects of latitude and other major biophysical gradients such as depth and substratum can produce large-scale patterns in species distributions and abundances. Given the inadequacy of current knowledge on marine species, the "systems" approach to conservation, usually with the goal of conserving a representative sample of areas covering the entire spectrum of habitat and bioregional types, has been important in the conservation of marine invertebrates.

In this chapter, we discuss the pros and cons of the systems-based approach to conservation (Section 5.1), and the commonly used definitions for each type or level of system (assemblages, communities, habitats, etc.; Section 5.2). The major marine environments and habitats are then outlined, with an emphasis on their importance for marine invertebrates and specific conservation issues relating to each (Section 5.3). We then discuss some other factors that should be considered in the "systems" approach to conservation, including the importance of scale, heterogeneity, transitional zones and linkages, and the need to conserve the ecological function of systems, as well as their individual component parts (Section 5.4). Finally, we discuss some of the common approaches to systems conservation, ranging from the listing of threatened habitat to restoration of degraded habitat and declaration of Marine Protected Areas (Section 5.5), and recommend management options (Section 5.6).

The term ***ecosystem*** is used throughout this document. It can be used to describe the interdependency of species in the living world (i.e., in assemblages, communities and so forth) with one another and with the abiotic environment. It also encompasses the concept of energy flow via food webs and the cycling of nutrients. These principles can be applied to all scales. O'Neill (2001) discussed the ecosystem concept in detail, including its history and its limitations.

5.1 Arguments for and against the systems approach

As discussed in the previous section, it has long been recognised that there are too many invertebrate taxa for a single species approach to invertebrate conservation to be successful for most taxa. There are many practical reasons why this is so. Funding in particular (and resources generally) is limited and therefore prohibitive of this approach. Another important issue is the fact that there is insufficient knowledge about most species-group taxa to determine conservation status or support a nomination for listing, with a large percentage of the fauna currently undescribed and likely to remain so for many decades. It is therefore important to direct resources towards other approaches, such as the conservation of habitats and ecosystems and the mitigation of threatening processes (e.g., Franklin 1992; Tear et al. 1995; Commonwealth of Australia 1996; Ray 1996; Reed and Clunie 1997; Gray 1997a; Bowen 1997b). The advantages and disadvantages of this approach are outlined below (see also pros and cons of taxon approach in Chapter 4).

Advantages

The advantages of focusing on systems include:

- Species in nature do not live in isolation and it may not be appropriate to consider their conservation in isolation from the systems of which they form a part. Systems, particularly those at smaller scales (e.g., communities), may provide a more **ecologically sensible** basis for conservation efforts. This approach also allows for the maintenance of interrelationships between species.
- It enables action even in the absence of complete data. Processes and habitats, with their constituent species, can be conserved even if they are poorly known (Franklin 1992). The systems approach (for example, habitat or site protection) is a surrogate or de-facto approach designed to conserve as many constituent species as possible – including unknown or poorly known species – under the one "umbrella" (e.g., Butcher et al. 1994; Zacharias and Roff 2000).
- It enables a focus on *prevention* of degradation and biodiversity loss, for instance through protection of relatively pristine habitats, rather than a *reactive* expenditure of resources on species that have already suffered possibly irreversible declines.
- Since some of the processes that threaten marine invertebrates (e.g., habitat destruction, trawling, dredging, over-harvesting) are geographically localised in impact, the conservation of particular areas or systems can be a means of protecting their constituent species from certain key threatening processes.

Disadvantages

The disadvantages of focusing on systems include:

- The assumption that "habitat" is a meaningful surrogate for species diversity is poorly tested, especially in the marine environment. It can be difficult to provide an acceptable, practical definition of what constitutes the "system" in question, and what

measures are required for its protection. Natural systems tend to be dynamic and "open", and species composition, abundances, and habitat boundaries all change over time. This, combined with the relatively more inflexible nature of legislative protection, mean that in practice "systems" conservation virtually always equates to protection of sites or areas.
- There is considerable debate over the most appropriate criteria (species richness, endemicity, habitat type, etc.) for prioritising the conservation significance of different areas. Areas selected because of high species diversity may contain many common and widely distributed species of relatively little conservation significance, while areas selected as supporting high endemicity may be relatively species poor. While there are a variety of useful tools for dealing with this issue, including the concepts of complementarity, replaceability etc. (see also Section 5.5.3). However, these approaches have been largely developed for terrestrial systems and are untried in marine environments (G. Chapman pers. comm.).
- This approach does not cater for *narrow-range taxa* if their entire range falls outside the network of protected areas.
- Habitats for some taxa may not be easily identified and thus may be missed in a habitat-based approach.
- The niche parameters of the individual taxa in any defined "habitat" will differ so that species contained within any arbitrarily defined habitat will be variously accommodated depending on the degree of overlap.
- Many taxa are mobile – i.e. individual organisms may not necessarily confine themselves to a particular site, or even to a particular habitat or community.
    - Species may occupy different habitats during their life cycle.
- In itself, habitat or site-based protection will be insufficient if threatening processes (e.g., pollution) continue to operate.

Narrow range taxa

If the range of a threatened species falls outside protected areas, protecting the kind of habitat that they occupy elsewhere has no conservation value for that particular taxon. Approaches such as ensuring representative protected areas within the areas identified by the bioregionalisation of Australia will only partly overcome this particular problem. The identification of such narrow range taxa is clearly dependent on available distributional data and taxonomic knowledge. For known taxa, much of the relevant information is in museum collections and these data can be used to identify areas that may be important for narrow range endemics (Ponder 1999; Ponder et al. 2001a).

5.2 Definitions – types of systems

In this section, we provide a brief description and definition of each of the different types of commonly recognised "system". Such terms are frequently found in the conservation literature and in legislation, but it is important to note than none of these can be defined in a fixed way. Many of the definitions are imprecise, and dependent on scale; not only the geographic scale, but also the size of the organisms under consideration. For instance, parasitic "communities" may be regarded as those found in individual hosts, while the hosts could themselves be considered as belonging to several different communities at different scales. A further complication is that some definitions (e.g., the differences between an assemblage and a community) are dependent on a knowledge of species relationships that, for most organisms, is non-existent.

5.2.1 Assemblages and communities

An *assemblage* is considered a more or less random aggregation of species that happen to inhabit the same space. While such associations may exist in nature as subsets of taxa within particular habitats, their practical recognition would be very difficult. In contrast, a *community* is a local set of functionally interdependent species, composed of species living together at some locality that relate to each other ecologically and are interdependent to some degree (Yen and Butcher 1997). In reality, such distinctions are probably artificial, and any set of organisms occurring together in nature would probably contain species that interacted or showed some degree of interdependence.

Each species in a community (or assemblage) occupies an *ecological niche*[75] and thus has a particular functional and trophic role.

Species are linked through a variety of interactions including food webs (trophic pathways) (e.g., Raffaelli 2000), through which energy passes from primary producers (e.g., plants) through herbivores to carnivores.

Community structuring
Structuring or regulating forces in communities include both physical or physiological factors (e.g., desiccation, unsuitable temperatures, salinity, wave action), as well as biological processes including interspecific competition, predation, disturbance and recruitment (Menge and Sutherland 1987). The importance of these processes has been well documented and modelled (reviewed by Estes and Peterson 2000) for marine hard-bottom (particularly intertidal) communities (e.g., Underwood 1984; Fairweather 1985, 1987; Underwood and Chapman 1998b; Menge 2000; Underwood 2000; Underwood et al. 2000) and coral reef communities (e.g., Tanner et al. 1994; Tanner et al. 1996 for coral reef dynamics at Heron Island). They have been less well documented for soft-bottom communities (Estes and Peterson 2000), but Seitz (1998) described important regulating forces in soft-sediment systems, outlined existing models of community regulation and revised a model of community regulation to incorporate soft-sediment systems.

Redundancy
A long-standing, but important question in ecology is whether ecological communities are ‘closed’ (exclusive associations of closely interdependent and co-evolving species – a true community by the above definition) or ‘open’ (a haphazard combination of species inhabiting a region that happen to tolerate current conditions, or “assemblage”) (Jackson 1994b). Such questions are important in the context of conservation since they have a bearing on the debate over whether all species found in a community are necessary for the adequate functioning of the community (as might be expected if the system is

[75] The ecological conditions of that species, including its functional relationships in the community.

'closed'), or whether there is species redundancy (which is more likely if the system is 'open'). These questions are central to understanding not only how communities develop and persist, but also the biological consequences of environmental changes (including global climate change). To answer such questions, we need to know to what extent species abundance is limited by population interactions and resource specialisation, as opposed to fluctuations in environmental conditions, the availability of colonists, history, and chance. Not surprisingly, the answer appears to vary depending on the type of system in question (Jackson 1994b). For most Australian marine habitats we have little data to answer these questions.

Delineation

While the concept of 'community' is understandable in theory, there are considerable practical difficulties in delineating communities in real environments, particularly for conservation purposes. This is because of:

- Their dynamic nature;
- The fact that they are rarely discrete 'units' but form part of larger systems; and
- The scale problem referred to above.

Begon et al. (1996b) treat community as a "level of organisation", and accept that it embodies many scales. A community is not an entity, around which boundaries can be drawn, except to the extent that continuous scale variations are sometimes "nearly decomparable" (Allen and Starr 1982) at particular points (discontinuities). The fact that the boundaries chosen when defining a "community" are arbitrary should be accepted and not seen as a disincentive to act (A. Butler pers. comm.).

5.2.2 Habitats

The usual definition of habitat is the space that an organism, group of organisms, or species occupies, including the physical and environmental elements that affect them (Yen and Butcher 1997).

Habitats as continuums

Habitats are generally not discrete units, but parts of an extremely variable natural continuum (Hawkesworth and Kalin-Arroyo 1995). Despite this, the term is often used in such a way as to suggest that there *are* physically definable entities or units. Because habitats are usually part of a continuum, their definition is subjective, as illustrated by attempts to define 'estuaries' (see 5.3.1). Temporal as well as spatial boundaries may be fuzzy and there is the additional problem of scale. For example, what might be regarded as the 'microhabitat' of one species could be called the habitat of another, smaller, taxon.

In the marine environment, it can be useful to think of habitats as representing intersecting points between multiple environmental (physical, chemical and biological) gradients. These include gradients of:

- Exposure to the atmosphere (from supralittoral to intertidal to completely immersed);
- Exposure to wave action (open vs sheltered littoral and sublittoral habitats);

- Salinity (from brackish habitats where freshwater mixes with seawater to marine or even hypersaline, e.g., in enclosed coastal lagoons with little freshwater inflow);
- Physical substratum - from fine (mud) through intermediate (sand) to coarse (gravel to boulders) sediments to rock;
- Biological "substratum" (e.g., algae, seagrass, sponges, bryozoans, coral skeletons, etc.);
- Topography;
- Temperature (sub-Antarctic → temperate → tropical); and
- Depth (coastal→ shelf→ abyssal).

The number of these gradients and the complexity of their interactions make a simple classification of habitat "types" impossible. Despite these difficulties, some (e.g., Zacharias and Roff 2000) have advocated using abiotic components of the ecosystem to use as surrogates for the identification and monitoring of the biotic (i.e., community) components.

Some habitat components are interrelated, thus not all combinations are possible. For example, sediment type can be affected by exposure to currents or wave action (e.g., coarser sediments occur where there is stronger wave action) and the presence of marine vegetation; reef-building corals occur mainly where there are hard substrates for larval settlement; and saltmarshes tend to replace mangroves in cooler regions. See also *ecotones* below.

Marine habitat definitions
In contrast to the terrestrial ecosystems where 'habitats' are often based on plant communities, many marine habitats are defined by substrata (i.e. sediments or rocks) or by other, non-plant, organisms (e.g., corals, sponges). Indeed, in many marine habitats, it is invertebrates themselves, particularly sessile and epifaunal (emergent) taxa[76], that provide most of the three-dimensional structure and spatial heterogeneity comprising "habitat" for other organisms. The relatively lesser importance of habitat-forming plants is because they are restricted to shallow water where light can penetrate for photosynthesis.

5.2.3 Ecotones

Ecotones are transitional zones that occur as part of the natural gradation between habitats. Ray (1991), in a discussion of coastal-zone biodiversity patterns, recognised four basic types of gradients that produce the boundaries between habitats: physiographic, biogenic, climatic, and physiochemical. Physiographic processes and attributes include tides, waves and surges, winds, salinity, and turbidity. Each of these can be framed in a hierarchical context, from larger to smaller temporal and spatial scales. Along biogenic coasts, living organisms, such as corals, mangroves, stromatolites and marshes, produce gradients. These organisms interact with physical features to form

[76] While coral reefs are an obvious example, there are many others, including other coelenterates (soft corals, gorgonians, hydroids, etc.), sponges, bryozoans, tunicates, and sessile molluscs (e.g., oysters, mussels).

habitats for a large number of additional organisms. Climatic gradients are especially important at larger scales. Latitudinal zonation exists along coasts and in the coastal ocean, but the biota of these zones tends to be much more tightly coupled to climatic change than is the case for terrestrial systems. Finally, physiochemical gradients, such as salinity, may exert important controls on the distributions of assemblages of estuarine organisms. The boundaries of these ecotones may change over time; for example, after heavy rainfall the freshwater/seawater boundaries in an estuary will change and mask any tidal influence.

5.2.4 Bioregions

Considerable emphasis has been placed in recent times on classifying Australia's coastal waters into bioregions, under the assumption that these represent areas in which the fauna is more similar to each other than to that of other areas. The extent to which any particular bioregional scheme corresponds with biogeographic boundaries used to delineate *faunal provinces* (thus reflecting the distribution patterns of invertebrates) is largely untested. It will remain so for most invertebrate groups until more information on their systematics and distribution becomes available.

Historically, broad bioregions (or faunal provinces) have been identified as a result of biogeographic studies (see Wilson and Allen 1987 for review), but bioregions are now being used for the purposes of choosing "representative" systems of protected areas. However, in most cases the delineation of these "bioregions" is based on variables other than the invertebrate fauna (e.g., geology, substrate, fish faunas, plants, water temperature etc.), and occasionally on a small subset of better-known large and conspicuous invertebrates (e.g., molluscs and echinoderms). The biogeography of the Australian marine invertebrate fauna, and attempts to define bioregions for Australian waters, are discussed in Section 2.4.2.

5.3 Key types of marine systems – features and conservation considerations

Australia has an enormous diversity of coastal and marine environments and habitats within each bioregion, resulting, in part, from the range of latitudes spanned (tropical to Antarctic), its long coastline (one of the longest of any country in the world) and the huge area of marine waters under its jurisdiction. Around the extensive coastline occur a wide range of habitats and biological communities ranging from the *tropical north* dominated by coral reefs, mangrove forests and other estuarine habitats, sand and mud flats to the *temperate south* where rocky shores, sandy beaches, algal reefs, sand and mud flats, and coastal saltmarshes, seagrass beds and kelp forests are found. As well as this latitudinal variation in coastal systems, there are also the much less understood mid-water, outer-shelf and deep-water habitats. Australia's marine environments also include external territories in the Indian Ocean, South Pacific Ocean, Southern Ocean and Antarctica.

For the purpose of this discussion we have divided marine environments into:

- Those of the coastal zone – i.e. estuaries and inlets, which occur at the transition between marine and riverine (freshwater) ecosystems, and open coasts, which form the transition between the marine and terrestrial ecosystems; and
- Oceanic, or purely marine environments (including the seabed and open oceans).

Obviously, these categories are very broad and encompass a wide variety of habitats; moreover, they do not reflect the relative importance (in terms of scale) of each category. In reality, coastal and estuarine habitats form only a tiny proportion of all marine environments and by far the largest area and volume is occupied by purely marine, oceanic habitats. However, the coastal zone is of particular conservation concern because it is by far the most impacted by human activities. Since each category has some distinctive characteristics which place particular stresses or restrictions on the biota, and is subject to different suites of threats, we discuss each of these broad environmental types before moving on to a discussion of key habitats, including soft sediments, rocky substrata, vegetated areas, biogenic substrata, the water column, and so forth.

The temperate marine ecosystems of Australia are principally dominated by rocky coasts and swell-dominated sandy shores, which for most part face the full force of the Southern Ocean. As such, they experience some of the highest wave energies in Australia (Edyvane 1996). Rocky shores are more widespread in temperate than tropical marine environments (particularly in the Great Australian Bight and Tasmania), occupying approximately 30% of the coastline compared with about 11% in tropical areas. However, it is sandy beaches that really dominate the coast of temperate Australia, occupying approximately 50-70% of the temperate coastline, particularly along the eastern and western coast of Australia (Fairweather and Quinn 1995).

5.3.1 Major marine environments and the threats to them

Historically, exploitation of the marine environment (including fisheries, mining, oil and gas, waste disposal, etc.) has largely been confined, for logistical and other reasons, to the coastal zone and continental shelf. However, pressures on current sources (of resources) and sinks (for wastes) mean that the utilisation of less accessible parts of the ocean is becoming increasingly attractive, both economically and politically. Consequently, impacts on oceanic environments are already increasing, and are likely to increase further in the future. These and a number of other important issues relating to oceanic environments are discussed below under the relevant sub-headings.

Unfortunately, the lack of knowledge and understanding of the diversity and complexity of oceanic environments and their faunas, translates into a lack of concern for any potential impacts upon them by activities such as mining, waste dumping, aquaculture, trawling, dredging, etc. The preconception that the oceans are vast and homogeneous, with faunas composed of widely distributed taxa, leads to the (often wrong) assumption that impacts are localised and insignificant. While there may be concern for potential impacts on some vertebrates (e.g., marine mammals, turtles, sharks), there is little serious consideration of the potential for impacts on invertebrates.

Coastal zone

Terrestrial, atmospheric and marine systems interact in the coastal zone. The "coastal zone" is variously defined and can encompass coastal (terrestrial) plains to the continental shelves. We have adopted a narrower definition, restricting this discussion to habitats forming the boundary between freshwater and marine or terrestrial and marine systems and the closely-adjacent marine zone.

While Australia has recognised the importance of the coastal zone and its vulnerability, with many inquiries having been undertaken at various levels of government (e.g. the Coastal Zone Inquiry – RAC 1993), little constructive action has been taken. In part, this is due to the very large number of agencies with responsibility for managing various aspects of the coastal zone and little co-ordination between them. This is particularly true of transitional habitats.

The International Geosphere-Biosphere Programme has proposed a global coastal research program, stating "The coastal zone, where land, air and sea meet, is a region of high physical energy that is heavily exploited by man. It is also a zone particularly vulnerable to global change" (IGBP 1990). The US Department of Energy recognised that "continental margins are vast factories synthesising about 50% of the organic carbon that is fixed in the world's oceans" (DOE 1990). These statements emphasise the importance of the coastal zone, which is quite out of proportion to its extent.

Exploitation and threats

The coastal zone is the most impacted part of the marine environment by human activity. Most exploitation (e.g., fishing, mining), pollution and development (including reclaimation and other forms of coastal modification) occur in this zone. It is also the area visited by people and utilised for recreation and tourism.

Conservation issues in coastal environments

Management and reduction of the impacts noted above is probably the most significant and immediate conservation issue in the marine environment. While coastal habitats are largely in public ownership, current reserves are not comprehensive, adequate or representative. Leases, licences and similar arrangements cover a range of commercial activities, such as aquaculture, fishing and shipping channel maintenance.

Transitional habitats

Transitional marine-terrestrial habitats and marine-freshwater habitats are usually ignored because they do not fall into standard aquatic/terrestrial or marine/non-marine categories under which people are trained and legislative and managerial responsibilities are divided (Ray 1991; Richardson et al. 1999), although each has its own characteristics and fauna.

Estuaries, inlets, sheltered bays and coastal lagoons

Freshwater inputs from catchments flow into sheltered bays, inlets and estuaries and mix to varying extents with seawater. These locations have biophysical characteristics that differ from either riverine or oceanic areas, with salinity ranging from virtually fresh water to fully marine and with lower wave energy than the open coast. They also have restricted water exchange patterns and their floors are generally covered in soft sediments (DNRE 1997). Typical habitats in these systems include sand flats, mud flats, seagrass beds, saltmarsh and mangroves. Larger embayments also contain sandy beaches, rocky reefs and islands. The boundaries of these habitats may change seasonally in areas with marked seasonal rainfall.

Key considerations

- Many species are restricted to these environments. For instance, the fauna in the upper reaches of estuaries are tolerant of changes in salinity, and typically less speciose than the rich faunas found in the more marine parts of inlets (e.g., Sydney Harbour Dakin 1952; Hutchings and Saenger 1987).
- Many commercially important species of estuarine fish and prawns breed in oceanic or coastal waters and enter estuaries and coastal lagoons from the ocean as larvae and juveniles.

The **invertebrate fauna** of these environments are particularly vulnerable to human disturbance for a number of reasons, including:

- Most major Australian cities are centred on bays, inlets and estuaries, due to their historical dependence on shipping access. Consequently, these environments are subject to a range of direct anthropogenic influences and impacts, including pollution, dredging, shipping activity, development, aquaculture, etc.
- They receive (to varying degrees) freshwater input, making them vulnerable to processes occurring in the entire catchment, including catchment disturbance and modification, inappropriate land use, siltation, agricultural runoff, and alteration to natural river flows (e.g., through damming or pumping for irrigation or urban water supply).
- Many species are restricted to these environments, and may therefore be vulnerable due to limited recruitment possibilities, particularly as habitats become fragmented.

Key threats in estuaries, bays and coastal lagoons

Key forms of human disturbance in estuaries, bays and coastal lagoons include:

- Pollution resulting from sewage discharge and runoff from unsewered areas, industrial discharges, stormwater, shipping accidents, etc. (Section 6.6).
- Terrestrial runoff - inadequate catchment management and inappropriate land uses can result in increased rates of erosion and sediment transport, while increased levels of nutrients from sewage and agricultural practices can lead to eutrophication (Section 6.6.1).
- Reduction in freshwater input as a result of the diversion of water for agricultural purposes or damming for drinking water or irrigation. The increased salinity levels, decreased natural flushing, and encroachment of the saltwater wedge further upstream can lead to loss of estuarine/brackish water wetlands, and change the distribution of the biota. Loneragan and Bunn (1999) discussed the implications of river flows for estuarine fisheries, including crustaceans.

- The hydrological regime has been altered in many estuaries by the creation of man-made entrances to the sea through barriers (see above).
- Degradation of habitats adjacent to urban areas (e.g., mangroves and saltmarsh, seagrass beds) from poor water quality and accumulation of rubbish.
- Dredging of waterways to allow passage of ships or to obtain material for construction or beach enhancement, and discharge of dredge spoils – this can change wave patterns and patterns of sediment deposition and erosion (see Chapter 6 and 6.13.2).
- Impacts from shipping activities, including pollution from toxic antifouling paints, oil spills, ship cleaning operations, disposal of ballast water, spread of non-indigenous taxa, etc. (Section 6.12).
- Other environmental impacts from human activities, including fishing and harvesting (especially destructive forms such as scallop dredging, trawling, and suction pumps used to collect bait); aquaculture; recreational use (boating, diving etc), land reclamation, and construction of buildings, wharves, etc. (see Chapter 6).

See Chapter 6 for further discussion of all these (and other) impacts.

In particular, vulnerable, but highly productive habitats such as seagrasses and mangroves have been frequently modified or destroyed by reclamation or clearing (e.g., for the construction of port facilities) (see Section 6.4.2) or smothered or degraded through sedimentation (see Section 6.6.1) or altered hydrological processes. Soft sediment habitats in the larger inlets, bays and estuaries are typically extensively disturbed by dredging (for scallops), fish and prawn trawling, etc. (see Section 6.10.2) and localised areas are also affected by dredging and spoil disposal for navigation, beach nourishment and material for construction and reclamation (Chapter 6). The contamination of estuaries, inlets and coastal lagoons from urban runoff, septic systems, and other urban, industrial and rural pollution is commonplace and leads to eutrophication, dinoflagellate blooms and other damaging effects (see Section 6.6). Pollution problems have led to a reduction in the areas where oysters can be grown for human consumption.

Estuaries

Estuaries occur at the boundary between marine, freshwater and terrestrial ecosystems. They are the most heavily utilized and modified of all coastal habitats. Thus, while the species of intertidal animals inhabiting estuaries are amongst the most tolerant of environmental change, they are also the most threatened of littoral assemblages (Smith 1997a). Saenger (1995) suggested that estuaries in eastern and southern Australia generally have lower water quality and greater catchment clearance, and a higher percentage face real threats to their conservation status, compared to northern Australian estuaries. In part, this is related to population density and urban development (see Section 6.16).

Definition

There is no accepted (or legal) definition for “estuary” in Australia, although two definitions are widely used (Moverley and Hirst 1999):

- Where a tidal river mouth meets the sea (Day 1981; Ketchum 1983), excluding river mouths with closed bars (not tidal).
- Coastal indentations with restricted connections to the ocean at least intermittently open (Cameron and Pritchard 1963; Day et al. 1989). This definition includes all estuaries encompassed by the first definition, as well as coastal lagoons, fjords and (depending on shape) some gulfs, sounds and inlets.

Moverley and Hirst (1999) consider the second definition more appropriate for the Australian environment, where there are many closed bars, coastal lagoons and rivers with intermittent freshwater discharge. However, this broader definition is in some ways less useful, because tidal systems have different characteristics to those largely cut off from the sea. There can also be problems defining the upstream limit of an estuary. Moverley and Hirst (1999) recognised various classes of estuary (classical, marine, homogeneous, layered, riverine) depending on the levels of seawater and freshwater input; these were all shown to have different macrobenthic communities.

Resources

An inventory of Australian estuaries and enclosed marine waters was provided by Bucher and Saenger (1989) and Saenger (1995). NSW alone has about 130 large and medium sized estuaries and embayments (NSW Fisheries 1998a). The National Land and Water Resources Audit (NLWRA)[77], a project funded through the Commonwealth Government's Natural Heritage Trust at a cost of $29.4 million over a four-year period, includes an audit of estuaries. Edgar et al. (1999; Edgar et al. 2000) described the distribution of macro-invertebrates (and fishes) in Tasmanian estuaries.

Habitats and fauna

The range of habitats within a large estuary is wider than other aquatic habitats, being subject to three sets of variables: substratum type (from rocky shore to soft mud), extent of tidal exposure, and salinity (Smith 1997a). Considerable environmental fluctuations are typical of many Australian estuaries due to erratic freshwater input.

This environmental unpredictability selects for taxa that can live and reproduce over a wide range of environmental conditions, typically resulting in low species diversity in these habitats (Moverley and Hirst 1999). Functional diversity is usually high (Moverley and Hirst 1999) even though taxonomic diversity may not be. These factors, plus high productivity, result in exceptionally high densities of animals at some sites.

Habitat/community types include saltmarsh, mangroves, seagrass, mud and sand flats, sheltered beaches, rocky reefs and shores, sublittoral fine sediments, etc. Polychaetes, molluscs and crustaceans dominate these environments, with echinoderms and other groups common in more saline areas. The faunal composition of such estuarine habitats varies according to physical, biological and anthropogenic factors. Unvegetated estuarine habitats, such as shallow mud flats, sand flats and deeper soft substrata, are usually the most common habitat in estuaries. While these may appear to be unproductive and are

[77] http://www.nlwra.gov.au/

often ignored at the expense of vegetated habitats, they support a very diverse and rich burrowing infauna and surface epifauna, which provide a significant food source for many fish species (NSW Fisheries 1998a).

While there have been many studies on the benthic invertebrate communities of Australian estuaries, much still remains to be done. In particular, there have been very few studies carried out in tropical Australia, where there are not even partial faunal inventories for estuarine animals in many areas (Hutchings 1999a). Hutchings (1999a) reviewed the taxonomic knowledge of estuarine invertebrates in Australia, the types of estuaries, the history of research on estuarine benthic fauna, the composition of the fauna and the characteristic fauna of different types of estuarine habitats.

Inlets and sheltered bays

Bays (with narrow entrances) and inlets with low freshwater input contain water of near normal to normal to very high salinity combined with a rich diversity of habitats and sheltered conditions. Such habitats are relatively uncommon in Australia and include Sydney Harbour, Botany Bay and Jervis Bay in NSW, Westernport and Port Phillip Bay in Victoria, Cockburn Sound and Shark Bay in Western Australia, and Spencer Gulf and Gulf St Vincent in South Australia. Shark Bay and Shellbourne Bay in north Queensland (although this has a large freshwater input during the wet season) are the only large bays in northern Australia that would qualify for this type of habitat. In all probability, there is at least a low level of endemism in some of these locations.

Knowledge base

The amount of information available on the invertebrate fauna of these environments varies from place to place. For example, despite the fact that Australia's largest city is located on Sydney Harbour, there has only recently been a study of the Sydney Port area (Australian Museum Business Services 2002), the previous one produced by Whitelegge over 100 years ago (Whitelegge 1889). In marked contrast, a detailed study of the benthic invertebrates of Port Phillip Bay has been carried out (Poore et al. 1975) and there have been some studies on some faunal groups in Westernport (e.g., Coleman et al. 1978). The benthic fauna of Moreton Bay, Queensland, has been investigated in a number of studies, for example by Stephenson and Cook (1979) and Stephenson (1980), and summarized by Davie and Hooper (1997).

Impacts

Anthropogenic impacts on these bays are the result of their heavy use as ports and shipping, as well as their utilisation for fishing, aquaculture, and recreation (i.e., recreational diving, boating, and other water sports). The combination of habitat disturbance (e.g., from scallop dredging) and shipping has led to the introduction of over 90 introduced marine plants and animals to Port Phillip Bay alone (DNRE 1997; Wilson et al. 1998; Hewitt et al. 1999), some of which have become pest species. A recent introduction, the Northern Pacific Seastar *Asterias amurensis*, is now present in very high densities (Hutchings 1999b).

Coastal lagoons

Saline coastal lagoons have entrances to the sea which intermittently open and close. They are estuaries that became separated from the ocean when sediments in the entrance area were moved and redistributed by wind, wave and tidal forces to form a barrier. Factors affecting the hydrodynamic features of these lagoons are the coastal location and exposure; entrance location and width; climate; size and shape; the size and nature of their catchments; as well as the frequency of flooding and number and size of the artificial and natural openings to the sea, the tidal range, and the extent of percolation to the sea (NSW Fisheries 1998a).

Unlike tidal estuaries, salinity changes typically occur slowly, over weeks or months, rather than twice a day (tidally). They are also influenced markedly by rainfall and dry periods, the latter often resulting in salinities higher than normal seawater.

In NSW, about 45% of major estuaries are intermittently opening coastal lagoons (NSW Fisheries 1998a). Formation and breakdown of a barrier may occur frequently (e.g., 5-6 times per year in Dee Why Lagoon) or infrequently (e.g., 2-3 times per century in Lake Conjola). Around half the NSW coastal lagoons have their entrances artificially breached from time to time.

Habitats and fauna

Intermittently opening coastal lagoons have unique biological features. They do not generally support mangrove communities but usually have an abundance of reeds and seagrass. The fish and invertebrate recruitment processes for coastal lagoons are complex and partially controlled by when and for how long the entrance is open (NSW Fisheries 1998a).

Whereas the coastal lagoons of southwestern Australia have received considerable attention (e.g., Hodgkin and Lenanton 1984; Hedge and Kriwoken 2000), those elsewhere are much more poorly studied for invertebrates. Important macro-fauna include molluscs, crustaceans and polychaetes, although they tend not to support the high productivity of edible molluscs seen in permanently open estuaries. Fauna changes over time as salinity fluctuates from being hypersaline to hyposaline.

Impacts

Many previously intermittently opening estuaries now have permanently modified entrances with rock groynes and training walls kept open with periodic dredging programs. Because they are often only intermittently open to tidal flushing, water quality deterioration is a major issue in coastal lagoons, with the build-up of effluents leading to eutrophication and blooms of cyanobacteria, algae etc. Some are partially filled through reclamation.

Open coasts

Open coastal habitats include rocky reefs and platforms, and cobble, gravel and sandy beaches. They are characterised by high wave energy and normal salinity.

Key considerations

- The intertidal fauna must to be able to withstand (or adapt behaviourally to) periodic immersion and desiccation, and exposure to radiation, wind, wave action, predators, etc.
- All the marine invertebrate phyla are present in these habitats, but the macroinvertebrate fauna in temperate waters is dominated – in terms of species numbers and abundance – by molluscs, crustaceans and polychaetes. Echinoderms, ascidians, coelenterates and sponges are also often conspicuous and ecologically important elements. In tropical waters coral reefs are supported and can dominate, although soft corals can also be important especially in inshore waters.

Key threats in open coastal environments

Key forms of human disturbance in open coast environments include:

- Recreational activity – rocky and sandy coasts are the focus for a great deal of recreational activity, including fishing and harvesting, bait collecting, diving and swimming, especially around urban centres.
- Development – many areas are heavily developed or in the process of being developed for residential, tourist and other purposes.
- Pollution – coasts close to urban centres are affected by contamination from urban sewage and other residential and industrial waste, stormwater, oil spills, etc.
- Coastal works and hydrological modification – coastal works such as beach works, dredging, destruction of wetlands etc. can affect hydrological processes and / or sediment transport along the coast and result, for example, beach erosion or sediment deposition in new areas.

Oceanic

Oceanic habitats include the water column (from surface layers to depths of 11,000 m or more in the deep-ocean trenches) and the seafloor (extending from the subtidal to the abyssal zones, and including the continental shelves and slopes, abyssal plains and mid-ocean ridges, and seafloor features such as plateaus, banks, and seamounts).

Light readily penetrates the upper 200 m of the oceans – the euphotic zone – where the major part of the oceans' primary production takes place. Below this lies the twilight zone or mesopelagic zone where there is some light penetration to about 1000 m. Beyond this lie the completely dark bathypelagic (1000-4000 m) and abyssopelagic (>4000 m) zones, the regions many consider as the true deep-sea (Gage and Tyler 1991).

The biota of these oceanic habitats generally has a more constant environment in terms of temperature and salinity than that of the coastal zone. In the deep-sea in particular, conditions are relatively constant, although deep-sea faunas must cope with enormous pressure, lack of light and low temperature. Resource (e.g., nutrient) limitations are also an issue in many cases because of the distance from nutrient sources, such as terrestrial runoff or (for the deep-sea) primary production in the upper euphotic zone.

Knowledge base
Australian oceanic organisms are, on the whole, very poorly known in comparison to those that occur in shallow coastal areas, due to the difficulties involved in sampling (especially of the deep-sea) and the vast area under Australian jurisdiction. Even from a geological and geomorphological perspective[78], much of the seabed within Australia's vast jurisdiction is poorly known – for instance, only 4% has accurate and reliable bathymetric maps. The Australian Geological Survey Organization (AGSO), in collaboration with some international agencies, is currently using swath-mapping techniques to provide bathymetric and tectonic data in deep-water areas, including detailed mapping of the seafloor along the eastern Australian coast using multibeam sonar. CSIRO, in collaboration with ACSO and the National Oceans Office, is investigating the use of multibeam sonar in conjunction with other sampling devices, to efficiently map habitats and sample the fauna in deep-waters[79]. These types of data are needed for the regional planning required by Australia's Ocean Policy.

Even in the more accessible parts of the oceans, such as the continental shelf and pelagic waters, the invertebrate fauna is very poorly known. This is even more the case for the deep-sea and offshore islands, seamounts and banks, large areas of which are still completely unsampled. Unfortunately, there has been very little systematic collecting (and even less reporting) of invertebrate faunas from offshore islands, seamounts and banks. Even in the few sampled areas, very little is known about the taxonomy, biology and ecology of the animals found there, or their environment.

Some sampling has been carried out, for example on seamounts, and where this material is lodged in museums (but not otherwise reported on), data should be retrievable, even if many of the species involved are still unidentified or undescribed. Since the cost of such sampling (which includes ship time) is considerable, making the best use of available data should be an important first step and would enable better informed planning of future cruises. The initial process must involve production of an inventory of the sampling that has been done, where material is deposited, and whether it has been sorted to family or genus level. Better use of samples (i.e. making all material and data, regardless of whether it is of direct interest to those involved in the sampling, available to other interested parties) also needs to be facilitated.

However, inventories, even if they were comprehensive, would not give an understanding of ecological dynamics, although the technology now exists for repeated visits to precise points on the seafloor, for automated recording, etc.

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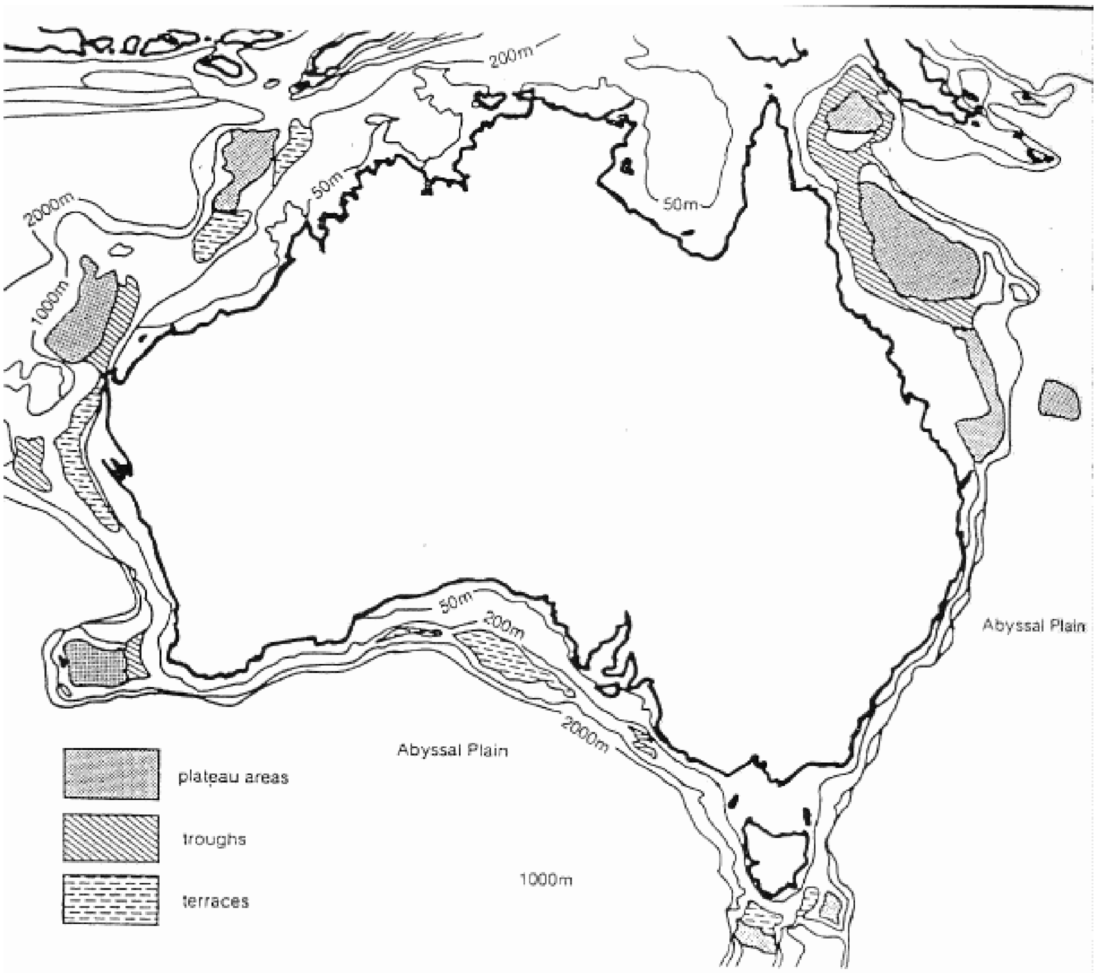
[78] An introduction to (global) marine geology and geomorphology can be found in King (1975).
[79] AGSO: http://www.agso.gov.au/marine/prjmain.html#seabed. CSIRO Marine Research: http://www.marine.csiro.au/voyage/index.html.

Figure 5.1: Major features of the Australian seabed (from Bunt 1987; Poore 1995b)



Pelagic

The upper layers of the ocean provide a home for planktonic organisms (small plants and animals which primarily float or drift with the currents). Some float on the surface (**neuston**), while others (the **nekton**) are pelagic and actively swim, such as some crustaceans, squid, etc.; as well as many fishes and marine mammals. This important community is highly productive, being based on photosynthetic planktonic organisms. It is also economically very important, as the basis of many important fisheries. The **plankton** is comprised of organisms that have poor or no powers of locomotion and drift with water movements.

Most pelagic organisms travel great distances, either by swimming or by being transported in ocean currents, and therefore tend to have wide distributions. While some fish, marine turtles and mammals with slow growth rates and / or low rates of reproduction may be threatened by over-exploitation or accidental capture in ocean fisheries, this is unlikely to be the case for any pelagic invertebrates. Concern over the over-harvesting of krill in the Antarctic region, for example, has focused on the potential threat to predators that utilise this food source, rather than the effect on populations of the species itself (e.g., Everson and de la Mare 1996).

Neuston

Invertebrates that float on the surface include the siphonophores (e.g., *Physalia* – bluebottles, and *Velella* – By-the-wind Sailor), *Janthina* (Violet Snail) and *Glaucus* (Sea Lizard, a nudibranch slug). It appears that many of these species are truly cosmopolitan, having been recorded from most world oceans. There are also the sea-skaters (*Halobates* spp.) – one of very few groups of truly marine insects (Cheng 1976; Matthews and Queale 1997) – although these skate on the surface of the water and are therefore not part of the pelagic community.

Plankton and Nekton

Most oceanic organisms drift in the plankton, although many of these organisms are capable of some movement and many undergo impressive diurnal vertical migrations. Obligate (lifelong) pelagic invertebrates include crustaceans (e.g., krill and many types of minute crustaceans that form an important component of the plankton, such as copepods), cnidarian medusae (jellyfish), some species of polychaetes, salps, etc. Many other invertebrates have a pelagic stage in their life history, usually the larvae. Because of the greater probability that planktonic organisms will be dispersed between ocean basins, endemism between basins is rare for species, and extremely rare for genera (Longhurst and Pauly 1987). Thus, a large proportion of Australia's tropical planktonic species are shared with the wider Indo-Pacific region (or beyond), while many temperate and Southern Ocean species have circumpolar distributions (see Fig. 4.4 in Longhurst and Pauly 1987).

Actively swimming pelagic invertebrates (**nekton**) include the cephalopods (squid) and heteropod and some pteropod gastropods. The division between plankton and nekton with the smaller actively swimming invertebrates becomes blurred - for example many crustaceans swim, although most not strongly and some small pteropods swim rather weakly and are better considered to be part of the plankton.

A general introduction to Australian plankton is given by Kingsford (1995). Tropical planktonic communities are discussed at some length by Longhurst and Pauly (1987). Globally, copepods are the dominant component of the zooplankton, though the greater diversity of tropical plankton means that copepods are much less dominant in tropical plankton than in high latitudes. Scheltema (Scheltema et al. 1998) reviewed Australian pelagic and planktonic molluscs. Other important taxa include chaetognaths, appendicularians, ostracods, polychaetes, thaliaceans, siphonophores, etc. Diversity is reduced in estuarine and coastal lagoon plankton (Longhurst and Pauly 1987).

There have been many studies of Antarctic and sub-Antarctic zooplankton (a few recent examples including Razouls 1994; Newman et al. 2000; Beaumont et al. 2001). Limited work has been carried out on Australian zooplankton communities, some examples being given in the following text. Nyan Taw and Ritz (1978; Taw and Ritz 1979) worked on the zooplankton of the Derwent River in Tasmania, and related the species composition to the hydrology of the estuary. Ritz and his co-workers have studied intensively mysid populations around Tasmania, in terms of productivity and their importance as food resources for fisheries (Ritz and Hosie 1982a; Ritz and Hosie 1982b) and for seabirds (Bishop et al. 1983). Edgar (1991) reported on the plankton in Bathurst Harbour, SW Tasmania. There have been relatively few recent studies on zooplankton (other than fish larvae and single-taxon studies) in the northern half of Australia, examples including Greenwood (1980), Sorokin (Sorokin 1992; 1995) and Rothlisberg (1994). Gaughan (Gaughan and C. 1994; 1997) has studied zooplankton in south Western Australia.

Impacts

The pelagic zone is very productive and important economically. While human impact in this zone remains relatively low, oil spills and other forms of pollution can have some effects, although these are mostly localised. Over-exploitation by fisheries can presumably change the structure of communities, which may have cascading effects. These interactions, however, are likely to be highly complex and may be difficult to detect without appropriate sampling and monitoring programs. The most determined effort to understand and manage such cascading effects is occurring under CCAMLR (Convention on the Conservation of Antarctic Marine Living Resources 1980; see Chapter 8, Table 8.1).

Continental shelf

The continental shelf slopes gradually from the coastal zone to about 200 m depth – the *shelf break* area – beyond which the sea floor descends rapidly as the continental slope. The Australian continental shelf surrounds the entire continent and varies in width from around 15 km to 400 km (in the Timor Sea), with a total area of approximately 2.5 million square kilometres (Poore 1995b).

Environmental conditions

It is covered with a variety of sediments, often biogenic in origin (e.g., coarse calcareous sands derived from the remains of bryozoans, molluscs and foraminiferans, or carbonates derived from corals) with terrigenous sands or muds typically occurring near the coast or in the vicinity of major rivers. Some of the largest modern cool-water shelf accumulations of carbonate sediments in the world are found in southern Australia (particularly in the Ottway region in the southeast and the Great Australian Bight) (Edyvane 1996). The variation in surface sediments on the Australian continental shelf, geographically and with depth, has been reviewed by Bunt (1987) and was summarised by Poore (1995b).

Marine invertebrate fauna and biogeographic patterns

While the geology of surface sediments (which is important in determining the composition of biological communities) has been studied superficially, knowledge of the environment and biology of the continental shelf and slope is patchy (Poore 1995b). The only three areas investigated intensively are the Northwest shelf, the Great Barrier Reef lagoon and Coral Sea, and Bass Strait and the southeastern shelf (and slope). Almost nothing is known of other areas. In the few places where detailed work has been undertaken (and even there, research has been confined to the shelf and upper slope, mostly down to little more than 1000m), studies have revealed a rich and diverse fauna, much of which is undescribed. In southern waters (such as the Great Australian Bight), a combination of low terrestrial-based sedimentation and cold upwelling ocean water has resulted in luxuriant growths of bryozoans and coralline algae, together with sponges, molluscs, asteroids and foraminiferans (Edyvane 1996).

On the Northwest Shelf, a multidisciplinary study of the demersal fish and invertebrate communities was initiated by CSIRO in 1982 with the aim of investigating and managing the finfish and crustacean fisheries (e.g., Ward and Rainer 1988; Rainer 1991). The epibenthos of this region is composed of sponges, gorgonians, soft corals and sea pens and is scattered over areas of rippled bare sand. 308 species of demersal crustaceans – mostly crabs but also some penaeid prawns and carid shrimps – were collected from 47 trawl and sled samples at 40m and 80m depth (Ward and Rainer 1988). The infauna is extremely species rich and is dominated by polychaete worms (Rainer 1991). The number of taxa from the Northwest Shelf is higher than that caught in more intense surveys in the western Atlantic (Poore 1995b).

A variety of studies have been carried out in different parts of the Great Barrier Reef lagoon and Coral Sea. In the Gulf of Carpentaria, Rainer and Munro (1982) and Rainer (1984) found that depth played a major part in explaining the distribution of fishes and cephalopods. On the central Great Barrier Reef (near Townsville), Birtles and Arnold (1983; 1988) examined patterns in the distribution of soft substrate epibenthos. They found that the most diverse taxa – echinoderms (103 species) and molluscs (196 species) – were divided into an inshore community on muddy sediments and an offshore community on sandy sediments at greater depths. In trawls in inter-reef areas on the continental shelf and slope of the northern and southern GBR, Cannon et al. (1987) recorded about 700 species of fishes and invertebrates.

In southern Australian waters, biological surveys have been carried out in Bass Strait (e.g., the Museum Victoria's Bass Strait survey, 1979-1984; Wilson and B. 1987; Poore 1995b) and on the southeastern Australian continental slope (see also Section 2.3.1).

In the Bass Strait 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 have resulted from this survey and many more will appear over the next decades (Poore 1995b). In the one quantitative component of the survey, grab samples from 1.2 m^2 of benthos in a small area of eastern Victoria turned up 353 species of invertebrates, about half of them crustaceans, the rest polychaetes and molluscs (Parry et al. 1990). None of the four sites sampled had more than 45% of the total number of species collected, suggesting that greater sampling effort would reveal more species (Poore 1995b). In depths of 11-51 m along a small section of the Victorian coast, Coleman et al. (1997) found an extremely species-rich fauna, with 60 258 individuals and 803 species obtained from a total sample area of 10.4 m^2. Few species were highly abundant and 51% of species collected were apparently undescribed. Compared with data from similar areas in other parts of the world, species diversity for this area of southeastern Australia is exceptionally high (Parry et al. 1990; Poore and Wilson 1993; Coleman et al. 1997).

Surprisingly, there is little information on the biology of the continental shelf off Sydney, Australia's largest city, apart from a report edited by Jones (1977) on ecological investigations prior to the construction of the deep-water sewage outfalls. There are also some consultant's reports produced for an Environmental Impact Assessment on a dredging proposal off Providential Head and Cape Banks (Geomarine Pty. Ltd 1992; The Ecology Lab Pty. Ltd 1993). The report edited by Jones (1977) concentrated on soft-bottom communities in the area around the proposed outfall off Malabar, and rocky-bottom communities at Long Reef, and North Head. Soft-bottom communities were dominated by molluscs (215 species), with solitary corals, crustaceans and polychaetes abundant at some sites; other common taxa included sipunculans, echinoderms and ascidians. Rocky-bottom communities were dominated (to differing degrees at different locations) by coelenterates, mainly turfing hydroids and anenomes, and erect and encrusting sponges, with minor elements including ascidians, molluscs, polychaetes, echinoderms etc.

Impacts
The greatest anthropogenic impact on the continental shelf is from trawling, with much of the fauna of the Australian shelf, like continental shelves in many other parts of the world, having been severely impacted from trawl fishing activities (see Section 6.10.2). The effects of offshore disposal of sewage (through "deep-water" ocean outfalls) and dumping of waste or dredge spoils can also be important in some areas (Chapter 6), these impacts being mainly concentrated on the inner part of the Shelf. Impacts from increased sedimentation due to land clearing, development and agriculture often extend beyond the coastal zone onto the Shelf.

Continental slope to abyssal depths (the "Deep-sea")

From the seaward edge of the continental shelf, the *shelf break*, the seabed slopes steeply downward as the **continental slope**, which is often interrupted with terraces and canyons. Extending beyond the slope is the *continental rise* that generally has a smoother topography than the slope. By a depth of about 4000 m, the seabed has levelled off to a wide expanse of relatively flat **abyssal plain** that extends gently to a depth of about 6000 m.

These parts of the seafloor cover a much larger area than the shallow shelf zone. For instance, the *bathyal zone* (depths between about 200-3000 m) occupies 17.8% of the world's oceans (Zezina 1997), and the abyssal and hadal zones (below 3000 m) cover about two thirds (Vinogradova 1997), compared with only 7.3% for the continental shelves (Zezina 1997). Within the Legal Continental Shelf area of Australia, some 80% is deeper than 200 m, 70% deeper than 1000 m, and 65% deeper than 2000 m, with the deepest part at about 5500 m (Newton 1999). Thus, the majority of the EEZ and its legal shelf extensions is actually deep-sea environment and yet we know virtually nothing about this habitat.

While terms such as "the deep-sea" and "deep-water fauna" are commonly used, there is no standard definition for what these actually comprise. Here we treat the deep-sea as encompassing everything beyond the edge of the continental shelf (following, for example, Zezina 1997; Newton 1999), down to the abyssal depths, which may reach more than 11,000 m (e.g., in the Marianas Trench). This area includes seamounts and underwater rises, although we have treated these features separately in subsequent sections.

Continental slope
The Australian slope is continuous with that of Indonesia and New Guinea, and in parts is much steeper than continental slopes elsewhere, commonly with gradients of up to 40°. The slope is intersected by canyon systems, especially in eastern and western Bass Strait where very steep gradients and rocky outcroppings occur (Poore 1995b). It is further interrupted by several terraces, major plateaus and troughs best developed on the northeastern and western margins of the mainland and on the southern Tasmanian coast, with minor terraces also in the Great Australian Bight (Poore 1995b; see Figure 5.1). The slope, at least in southeastern Australia, is a complex hydrological region with sediments of biogenic origin derived largely from the nearby shelf (Poore et al. 1994).

Abyssal plain
The abyssal plain is a vast, largely featureless expanse of soft muddy material derived from the remains of planktonic organisms that lived in the euphotic zone. The abyssal plains are separated by a couplet of *mid-ocean ridges*, an immense underwater mountain range that stretches for 64,000 km and covers nearly one-quarter of the earth's surface, forming a significant part of the deep-sea floor (Newton 1999).

Environmental conditions
Environmental conditions in the deep-sea are generally rather constant (but see below), typically lacking sharp gradients and temperatures are low (usually 4 to -1˚ C; except at hydrothermal vents). Light is absent and hydrostatic pressure increases with depth (1 atmosphere for every 10 m). Substrates are mainly soft mud, and food resources are extremely limited. An interesting and diverse fauna – including representatives of practically all the major classes of marine invertebrates – has become adapted to live under these conditions. Deep-sea organisms tend to have low metabolic rates, resulting in slow growth and longevity (Newton 1999).

There is less constancy in the deep-sea than was once assumed. For example, although salinity, dissolved oxygen and temperature are generally stable over large areas, the "uniformity" of this environment is broken by slow currents (<10 $cm.s^{-1}$) which result from differences in water temperature. Large and more powerful eddies can also give rise to 'benthic storms' that can last for weeks and disturb and transport large volumes of sediment, thus affecting the structure of the benthic community (Gage and Tyler 1991). Generally food supply is unpredictable and most feeding strategies are probably opportunistic and generalist. The main food supply is phyto-detritus, or marine snow, from the euphotic zone (see Newton 1999 for a detailed summary). There is also evidence of variation in this fallout which can potentially affect the feeding behaviour and recruitment patterns of deep-sea organisms (Tyler 1988). Sampling by Russian research vessels from depths of 3000 to 6000 m in the Pacific and Indian Oceans showed that it is possible to define eutrophic and oligotrophic areas in these oceans. The eutrophic areas correspond to regions of high productivity in surface waters and extend over the peripheral and equatorial parts of the oceans, and the sediments in these areas contain enough digestible organic matter to allow deposit-feeders to predominate. Oligotrophic areas are restricted to the open parts of the oceans, outside the equatorial belt, where sedimentation rates are low, organic matter is scarce and suspension-feeders predominate, although in small numbers (Sokolova 1997).

Invertebrate fauna

The deep-sea invertebrate fauna was once regarded as depauperate, but in the last few decades it has been found to contain a high diversity at all taxonomic levels. Much of this diversity appears to have evolved *in situ* (Wilson and Hessler 1987; Grassle 1991; Grassle and Maciolek 1992; Blake 1994; Blake and Grassle 1994; Blake and Hilbig 1994), although the extent of deep-sea diversity is still the subject of considerable debate (e.g., Gray 1997b; see also Section 1.2.2). Sampling with epibenthic sleds (Sanders 1968) demonstrated that the total *density* and *biomass* of the soft-sediment benthos declines with increasing water depth, probably due to the greater separation between the energy source at the surface and the benthic fauna, and from the generally greater distances from coastal zones with their higher productivity (Peterson and Wells 1998). Despite this, species *diversity* may, in some areas of the world's basins, actually increase with depth. This may be due to:

- The greater physical stability of the environment, which has allowed greater diversification by evolution of more narrowly specialised forms;
- Greater disturbance by predators, preventing competitive exclusion and thus maintaining high species diversity (Dayton and Hessler 1972); and
- The great area of deep-sea habitat worldwide, which has led to evolution of many species by chance alone (Peterson and Wells 1998).

Some authors have argued that species diversity in the deep-sea is very high, with estimates in the order of 10^7 species (Grassle and Maciolek 1992) or more (Poore and Wilson 1993), although these figures have been disputed by others (e.g., May 1992a; Gray 1997b). May (1992a), for instance, argued that the total number of deep-sea species was unlikely to exceed 5×10^5.

The errant deep-sea macrofauna is dominated by echinoderms, especially ophiuroids (brittle stars), and to a lesser extent decapod crustaceans. For instance, a recently discovered scavenging community was composed of previously unknown, giant-sized amphipods (Gage and Tyler 1991). Similar communities of giant isopods also exist off eastern Australia (J. Lowry, pers. comm.). Most deep-sea isopods are tiny and shallow burrowers in soft sediments or epibenthic swimmers (Hessler and Thistle 1975; Hessler et al. 1979). The errant macrofauna also includes fish, polychaetes (bristleworms), echiurans, hemichordates, pycnogonids (sea spiders), etc. (Gage and Tyler 1991). The sessile macrofauna consists of hexactinellid sponges (replacing, although not completely, the shallow water Demospongiae and Calcarea), cnidarians (all groups), barnacles, bryozoans, brachiopods, benthic tunicates and so forth (Gage and Tyler 1991). With some exceptions (e.g., the giant amphipods and isopods), there is a trend towards miniaturisation and body frailty in the deep-sea (Gage and Tyler 1991), most organisms being minute.

Biogeographic patterns

Previously, the supposed global uniformity of the basic deep-sea environment and lack of isolating barriers led to a widespread belief that deep-sea taxa had cosmopolitan distributions. This was overturned by extensive global sampling by Russian workers who found that only 15% of species occur in more than one ocean basin and only 4% are found in all of them (Gage and Tyler 1991). The reduced or absent planktonic dispersal typical of deep-sea animals presumably hinder gene flow and favour restricted distributions, the latter enhanced by selection induced by environmental structure and gradients, different environmental tolerances and dispersal abilities and other different biological attributes (Wilson and Hessler 1987). Deep-sea benthic macroinvertebrates occurring below 3000 m depth include some species with a broad cosmopolitan geographical distribution and others with apparently more limited, sometimes local, ranges (although whether these patterns reflect true absences or simply sampling zeros is, as in most of the sea, unknown). Those that occupy a wide range of depths (eurybathic species) tend to be more widely distributed. By contrast, those restricted in depth range (stenobathic), and groups with a high proportion of truly abyssal species, show a high level of endemism, and 95% of hadal (deeper than 6000 m) occur only in a single trench or a group of adjacent trenches (Vinogradova 1997).

Geographical and bathymetric (depth-related) patterns in deep-sea biodiversity and faunal distributions have been reviewed by authors such as Zezina (1997) for the bathyal zone, Vinogradova (1997) for the abyssal zone, Alongi (1992) for the western South Pacific, Svavarsson et al. (1990) for Norwegian and Greenland seas; and Rex et al. (1993; Rex et al. 1997) for global and latitudinal patterns. For instance, Zezina (1997) suggested that the bathyal zone (continental slope and rise) may have acted as a reserve of species for recolonisation of the shelves and the abyss following past extinction events, as it contains many relict species, including some of the most primitive extant members of their groups. Some groups of bivalves and gastropods, as well as the stalked crinoids and brachiopods, attain their maximum diversity in the bathyal zone (Zezina 1997). Bathymetric patterns in deep-sea benthic communities (bathyal to abyssal depths) in the western South Pacific (Solomon and Coral Seas) were investigated by Alongi (1992). This worker found that densities of meiofauna and macro-infauna were low at most stations compared to communities of other regions at comparable depths, and declined significantly with ocean depth, although many taxa and total macro-infaunal biomass did not. There is, as yet, very little information about such patterns in Australia.

Hydrothermal vents

Hydrothermal vents are extremely important and interesting deep-sea habitats, although none have been found in Australia's oceanic territory. They occur nearby, however, in the Manus Back Arc (PNG), off the Solomons and New Zealand. Macquarie Island lies on a mid-oceanic ridge but to date, no vent habitats have been reported from this area. These habitats, associated with deep-sea ridges, are found mostly between 2500 – 3000 m, and have only been discovered in the last few decades. They output hot water with high levels of hydrogen sulphide that forms the metabolic food source for chemosynthetic bacteria. These bacteria are utilised by suspension and detritus feeders or are symbionts in some animals. There is a large diversity of unique animals associated with these vents, many very abundant and often large (Grassle 1986; Tunnicliffe 1992). The biogeography of the deep-sea hydrothermal vent fauna was reviewed by Tunnicliffe et al. (1998).

Research and information needs

Deep-sea communities are poorly known due to the difficulties (and cost) involved in studying them. This is particularly true in the Australian region, mainly because the necessary facilities are either very expensive (oceanographic research vessels) or not available (submersibles). The few deep-sea surveys conducted in the Australian area routinely turn up a great number of undescribed species. For example, the exploratory RV *Franklin* cruise in 1989 (Australian Museum), in dredge sampling of the upper bathyal zone (120-1600m) in the northwest Tasman Sea, the southern Coral Sea, and along the east coast of Australia, found many new taxa in various phyla. This included 11 species of deep-sea crabs, five of them new, collected from seamounts in the Tasman and southern Coral Seas (Richer de Forges 1993; see also Section 2.3.3). Sampling on the southeastern Australian slope between 200 and 3000 m has also turned up a rich fauna (Poore 1995b). For example, of the isopod crustaceans (the best studied taxon in this area), 359 species belonging to 36 families had been described as of 1995, only 10% of which could be identified to known species (Poore 1995b). The results were similar for ostracod crustaceans of which more than 90 (mostly new) species were discovered (Kornicker and Poore 1996b). The aplacophoran molluscs are concentrated on the slope and numerous undescribed species have been found in this region (Scheltema 1990). In some of these surveys, the relatively coarse mesh used has resulted in most of the fauna (a large proportion of which is minute) not being sampled, suggesting that a very large number of smaller new species are yet to be collected. In addition, the method used to sample is often a dredge so that the infaunal taxa are inadequately sampled.

Threats

Threats to deep-sea faunas include mining, waste disposal and fishing. Of particular concern are the consequences of unrestricted commercial exploitation of the non-sustainable fish and invertebrate populations on the upper part of the continental slope and on seamounts, some of which lie outside the limits of the EEZ. This is especially true for seamounts and ridges because these areas are especially vulnerable due to their small size. Historically, exploitation usually precedes scientific exploration and assessment in all marine environments.

Because exploitation of the deep-sea floor is very expensive, it is the least exploited marine ecosystem. Nonetheless, activities such as deep-sea fisheries, mining and waste disposal all have the potential to impact deep-sea communities. Deep-sea organisms typically display characteristics such as slow growth, extreme longevity, delayed age of maturation, and low adult mortality. In addition, many important habitat-forming taxa are colonial organisms with fragile bodies. These features are characteristic of systems with low productivity and turnover (Levin et al. 1991), but make them extremely vulnerable to exploitation and disturbance.

At present, deep-sea fisheries are limited and mainly exploratory. Factors that have (historically) limited the development of deep-sea fisheries include the low population densities of most species and the cost of harvesting. While technological improvements and market demand have the potential to overcome such limitations, most deep-sea fishes and invertebrates, being generally long-lived and slow growing, are probably inherently unsuitable for intensive exploitation. The difficulty achieving sustainable exploitation of such species is evidenced by the decimation of Orange Roughy populations in southern Australia (Koslow et al. 1997). Deep-sea invertebrates recently exploited in Australian waters include the King or Giant Crab (*Pseudocarcinus gigas*), an endemic of southern Australia (see Section 6.3.2), and deep-water shrimps or "Scampi" (*Metanephrops*). The risk of extinction for these species is probably low, but concern has been expressed about the potential of commercial fishing pressure to quickly decimate populations (P. Davie pers. comm.). The bryozoan-rich shelf-break habitat occupied by *P. gigas* may also be sensitive to the impact of demersal trawling (Caton et al. 1998).

Mining: There is a growing interest in the industrial potential of thermophilic bacteria found at hydrothermal vents and there has been interest in mining manganese nodules and other minerals, as well as hydrocarbons, from the deep-sea floor (Gage and Tyler 1991; Newton 1999). Mining results in the disturbance of bottom sediments, loss of habitat and smothering (see Section 6.13.2). Lissner et al. (1991) noted that the recolonisation of deep-water hard-substrate communities following oil and gas development would take several years, because the taxa involved (vase sponges and anemones) were slow-growing and long-lived, and because of the uncertainties of long-range recruitment.

***Waste disposal*:**The remoteness of the deep-sea, both physically and psychologically, has made it attractive for the disposal of waste, especially that unsuitable for disposal on land or coastal waters (Gage and Tyler 1991; Newton 1999). Dilution factors, low cost, lack of a global policy to reduce waste and growing constraints on waste dumping in terrestrial and coastal areas, mean that deep-sea dumping is likely to increase, despite an international convention[80]. In Europe and North America, the proximal deep-sea is already being heavily used as a dump for dredge spoil, sewage sludge, and pharmaceutical, industrial and low level radioactive wastes (Newton 1999). This is currently not very significant in Australia, though some does occur, particularly dumping of dredge spoil.

[80] The 1975 *Convention on the Prevention of Marine Pollution from Dumping of Wastes and other Matter* (or the *London Dumping Convention*), which involves 57 countries, including Australia.

Plateaus, banks and rises

Major plateaus, rising from the continental slope to between 900 and 2000 m from the sea surface, are best developed on the northeastern and western margins of the mainland and south of Tasmania (Poore 1995b). Many have cays, reefs and even small islands associated with them. Areas shallower than 150 m or so were probably dry during the last glacial period.

Some examples
The Coral Sea Plateau is a large mid-ocean plateau, about 500 m below the sea surface, which is separated from the Australian continental shelf by deep ocean troughs. A number of shallow shelves, less than 200 m below the surface, are scattered across this plateau, including the Lihou Bank, the Coringa Bank and the Herald Cays. Sandy islets and cays sit on top of most of these shelves, surrounded by patches of reef up to 300 ha in size and only 1-4 m below the sea surface (Environment Australia 1996a). These environments differ from those of the continental shelf in being isolated from terrigenous input, and thus, presumably have lower levels of nutrients, etc. The surrounding waters have very high water clarity, and the marine invertebrate fauna is quite different from that of the Great Barrier Reef, with fewer hard and soft corals and many more sponges (Environment Australia 1996b).

The Western Australian and Northern Territory museums have undertaken some surveys of the shoals and reefs off Western Australia (which are mostly located on the shelf or are fringing reefs) for Environment Australia. For instance, Berry (1986a) described the formation of the Rowley Shoals, Scott Reef and Seringapatam Reef and the physical environment of these reefs, and provided species lists and comments on selected invertebrate groups (decapod crustaceans, corals, molluscs and echinoderms). The results of marine faunal surveys of Ashmore Reef and Cartier Island were described by Berry (1993). The marine flora and fauna of the Abrohlos Islands were the subject of a recent workshop (Wells 1997b).

Since 1994, AIMS WA has been undertaking regular surveys at Scott Reef (Done et al. 1994; Heyward et al. 1997a; Heyward et al. 1997b).

Continental islands

Continental islands are surrounded by a continental shelf that is usually bordered by a very steep slope that falls into abyssal depths. Such islands may or may not be lagoonal (inside a coral reef).

Environments and fauna
Their environments can extend through many of the types of habitats encountered in coastal areas (mangroves, hard and soft shores, coral reefs, etc.) but differ in that they are very restricted in area and therefore potentially more vulnerable to disturbance. All of these environments will have rich invertebrate faunas, and they often have many endemics. They may also have unusual communities, especially if they are marginal to

major biogeographic regions. Sadly, however, there has been little sampling of the offshore habitats associated with these islands, and the results of the little sampling that has been done have in large part not been published. However, those that do exist suggest that endemism is generally reasonably high. For example, an extensive program in deeper waters surrounding New Caledonia has revealed a rich fauna with high apparent endemism (e.g., Crosnier and Bouchet 1991; Bouchet 1995).

Examples in the Australian region include Lord Howe Island, Norfolk Island, the Cocos (Keeling) Islands, and Christmas Island. Lord Howe Island has the most southerly coral reef in the world and several endemic marine invertebrates (Harriott et al. 1993). The only attempt to document the endemism in shelf faunas across several invertebrate taxa was done recently for Lord Howe Island, based on dredged material from four stations on the shelf between 44 and 73 m in 1976, as well as a sample from 2738 m and another from Ball's Pyramid (Ponder et al. 2000). In all, 232 invertebrate species (207 of which were molluscs) were located from these samples in the Australian Museum collections. Of the 160 sufficiently well known to ascertain their distribution, 14 (i.e. 8.75%) are assumed endemics, as they are not known from elsewhere. Norfolk Island also has some endemics, and has an interesting fauna with similarities to a few other oceanic islands, such as the Kermadec Islands and even Easter Island (Rehder 1980). The isolated Cocos (Keeling) Islands in the Indian Ocean have a typical Indian Ocean fauna (Maes 1967 lists the littoral mollusc fauna; Woodroffe 1994). Christmas Island is close to Java in Indonesia, presumably sharing much the same fauna, although, from an Australian viewpoint this has many differences from other Australian tropical faunas and some unusual elements. The echinoderms of Norfolk (and Lord Howe) Islands have received a considerable amount of attention (McKnight 1968, 1976, 1977; Rowe and Albertson 1987; Rowe 1989).

Threats
Because of their small area, the threats to the marine faunas of continental islands are similar to those of sea mounts.

Seamounts

Seamounts are a unique environment, forming "islands" of relatively shallow substratum rising from abyssal depths. They are typically the peaks of extinct underwater volcanoes that may have originally risen above sea level, but are now eroded so that the top of the seamount lies below the surface of the sea. Their existence in the world's oceans, particularly in the Pacific, has only become widely known to the scientific community in the last 50 years (Rogers 1994).

Australian examples
There is a chain of seamounts in the Tasman Sea along the western side of the Lord Howe Rise off the eastern coast of Australia that commence at 21°S (Chesterfield Islands) and end at about 38°S (Quilty 1993). Terrestrial members of this group include Lord Howe Island and Balls Pyramid. Additional seamounts lie on the southeast slope of Tasmania, the Tasman seamount east of Tasmania, and on the south Tasman Rise south

of Tasmania. The tops of the seamounts in the Coral and Tasman Seas range from about 200-500 m in depth, while others such as Middleton and Elizabeth Reef are younger and shallower, forming cays. Muirfield Seamount, south of the Cocos (Keeling) Islands in the Indian Ocean, rises to some 18m below sea level.

Habitats and fauna

Seamounts usually have very steep sides and a flattened top. Their topography creates a distinct environment, with strong currents and, consequently, little sediment. Individual seamounts or seamount regions often have apparently unique benthic communities, although the presence of many rare and endemic species (found only on one seamount, for example) may be largely a sampling artefact. These communities are often dominated by colonial organisms such as corals and other filter feeders (Genin et al. 1986; Koslow and Gowlett-Holmes 1998), the patterns of distribution being related to current conditions (Genin et al. 1986). The huge distances that often separate them make these seamounts or guyots islands of benthic fauna surrounded by abyssal depths (Richer de Forges 1993). The biology of seamounts has been discussed by Rogers (1994).

There are few studies that have looked at species composition and turnover on seamounts. Leal and Bouchet (1991) studied the distribution patterns and dispersal of gastropods along an Atlantic seamount chain off the eastern coast of Brazil. They found that the number of species decreases significantly from the continent towards easternmost localities. These results suggested that both planktotrophs and lecithotrophs can be effectively dispersed, probably by passive larval transport, in an “island-hopping” pattern across the relatively small distances (100-250 km) that separate the summits in this chain. Gofas (1992) studied the gastropod fauna of seamounts near Portugal, which were found to show high endemicity in small taxa that have non-planktotrophic larvae. Some appear to have evolved from taxa with planktonic larvae into species with direct development, an advantage in minimising larval wastage on such small “islands”.

In Australian waters, the areas explored include the northwestern Tasman Sea, the southern Coral Sea, the seamounts of the Lord Howe Rise (along the east coast of Australia), and the Tasmanian seamounts south of Tasmania. Surveys of the benthic fauna of the seamounts of the Lord Howe Rise[81] were undertaken by an Australian Museum cruise on RV *Franklin* in 1989 (Lowry 1989). Some survey work on the submarine banks and guyots (Gifford, Capel, Kelso, Argo) lying north of Elizabeth and Middleton Reefs and scattered south of the New Caledonia on the Norfolk ridge and the Loyalty ridge has also been carried out in recent years (Pichon 1995; Richer de Forges et al. 2000). Much of the biological data from these surveys remains unpublished, but specimens from some of these cruises are lodged in museum collections. The little that has been worked up from these surveys has revealed the existence of previously unknown marine species (e.g., five new species of decapod crabs; Richer de Forges 1992, 1993). Seamounts may act as a refugium for archaic deep-sea fauna, such as “living fossil” Mesozoic sponge fauna associated with old sea levels (John Hooper pers. comm.). Other

[81] Often previously referred to as the Tasman Seamounts or Tasmantid Guyots, on the basis that they occur in the Tasman Sea, but we have avoided this usage because of possible confusion with the Tasmanian Seamounts (south of Tasmania).

living fossils include the sea lily, *Gymnocrinus richeri*, previously presumed extinct for 140 million years and discovered on the seamounts south of New Caledonia (Ameziane-Cominardi et al. 1987; Bourseau et al. 1987). Sphinctozoan calcareous sponges, known since the Palaeozoic (ca. 300 million years ago), were thought to have disappeared ca. 70 million years ago, but were also rediscovered on these seamounts (Pichon 1995). Such relict faunas are potentially invaluable in chemistry, genetics and biological discoveries (J. Hooper pers. comm.). Richer de Forges et al. (2000), in a discussion of diversity and endemism in the benthic seamount fauna of the southwest Pacific, stressed endemicity and the high species richness of the sea mounts of the Tasman Sea.

The CSIRO was contracted to carry out a study on the fauna of the seamount communities off southern Tasmania and the impacts of trawling, after an Interim Protection Zone (370 km^2) was declared in 1995 to allow them time to investigate the fauna and the impacts of trawling on it. These surveys demonstrated that the seamounts supported unique communities, dominated by filter feeders (particularly hard and soft corals, hydroids, sponges and filter-feeding echinoderms), and characterised by high species richness (262 invertebrate species recognised from a single cruise), with many species (24-43%) new to science and many (16-33%) apparently restricted to the seamount environment (Koslow and Gowlett-Holmes 1998; Koslow et al. 2001). Some of the seamount taxa show low rates of recruitment and considerable longevity; for example, unpublished data suggest that bamboo corals (Isididae) from these south Tasmanian seamounts may live to about 100 years (Koslow and Gowlett-Holmes 1998). Subsequently, the IPZ has been declared a Marine Protected Area (the Tasmanian Seamounts Marine Protected Area) and a management plan should come into force in mid 2002.

The CSIRO has also recently surveyed Muirfield Seamount, off Cocos Island (in the Australian EEZ), but this was found to be depauperate (A. Butler pers. comm.).

Threats

The faunas of seamounts are very localised and highly vulnerable; the smaller the area of habitat, the greater the vulnerability. The CSIRO study on the effects of trawling on the seamounts off southern Tasmania (Koslow and Gowlett-Holmes 1998) indicated that these communities are extremely vulnerable to the impacts of trawling (Koslow et al. 2001). For instance, on the most heavily fished seamounts, the reef aggregate had been mostly removed from the slopes or turned to rubble, and the benthic biomass was 83% less (and the number of species per sample 59% fewer) than from lightly fished or unfished seamounts (Koslow and Gowlett-Holmes 1998). Following the recommendations of this report, approximately 20% of the seamounts in the region (covering an area of 370 km^2) were declared as a Commonwealth marine reserve (the Tasmanian Seamounts Marine Reserve). Fishing and petroleum / mineral exploration are prohibited in the Highly Protected Zone, which extends from a depth of 500 m to 100 m below the sea-bed (Hill 1999).

5.3.2 Key habitat types

In this section, we briefly discuss some of the main habitat types, particularly those that have attracted attention as being of conservation concern. For the most part, the habitats discussed below occur in the coastal zone where human impacts are most obvious. These are based on vegetation (seagrasses, saltmarshes, mangroves, algae); geological substrata (hard substrata, soft sediments); or are habitats provided by colonial invertebrates (coral reefs, other invertebrates), or other biogenic substrates. We discuss above other types of "habitats" (in the broader sense – equivalent to "place") such as restricted mainland habitats, seamounts and oceanic islands, or hydrothermal vents and seeps.

"Plant"-based habitats

The major types of plant communities found in the intertidal and shallow sublittoral zones along the Australian coast include saltmarsh, mangroves, seagrasses and (although not strictly plants), various algal communities. These coastal wetlands exhibit high productivity (e.g., Keough and Jenkins 1995) and are widely used as nursery grounds (e.g., Jernakoff 1987; Robertson and Duke 1987). However, their location in sheltered bays and estuaries has meant that large areas of these habitats have been destroyed or heavily impacted (McComb and Lake 1988), so that maintenance of these habitats is a critical issue in Australia and elsewhere. Conservation issues in Australian wetlands (including marine and estuarine wetlands) are discussed, State by State, in McComb and Lake (1988).

We outline the main features of each of the major vegetated habitat types and their fauna and also briefly discuss for each their values and the specific threats and other conservation issues relevant to that habitat.

Mangroves

Mangroves are widely recognised as complex, important and productive marine ecosystems. Overviews of mangrove habitats are provided by Tomlinson (1986), Hutchings and Saenger (1987), Teas (1983), Chapman and Underwood (1995) and Harty (1997, for NSW and Victorian mangroves). Remote sensing data on mangrove surfaces have been compiled into a "World mangrove atlas" (Spalding et al. 1996). Mangroves comprise many different unrelated kinds of trees and shrubs, from several different families, that grow in the intertidal zone, generally on sheltered estuarine shores (King 1981; Hutchings and Saenger 1987). These species are characterised by adaptations to loose, wet soils, saline habitats, and periodic tidal submergence (Hatcher et al. 1989). They are typically found in the quiet, muddy areas of sheltered tropical and subtropical coasts, although they can also grow on clean sand or even on rocky shores. Many species of mangroves possess aerial breathing roots (pneumatophores) that help them tolerate low oxygen levels in the sediment. Mangrove forests are also rich in organic matter, much of which is derived from fallen mangrove leaves, bark and wood.

Distribution

Mangroves occupy far larger areas in the tropics where they are often contiguous, more diverse and structurally complex, whereas in the temperate zone they typically occur in small isolated stands. In higher latitudes they are progressively replaced by saltmarshes (Hatcher et al. 1989), so there are no mangroves south of 38.5ºS in Australia and only one species south of 34ºS. The coastal habitats of northern Australia are dominated by sheltered tidal mangrove environments; for instance, the most recent survey of the Australian coast showed there were approximately 11,230 km^2 in tropical and subtropical Qld, WA and the NT (including 4119 km^2 – or approximately 48% of the coastal habitat – in the Northern Territory). This compares with 330 km^2 (5% of coastal habitat) in southern temperate Australia (Galloway 1982). Aside from the fact that mangrove species' distributions are limited by temperature and the occurrence of frosts, the temperate marine environments of Australia are generally characterised by a lack of the necessary conditions – e.g., mud and tidal deltas – for development of mangrove habitats. The exceptions are some large sheltered embayments such as Westernport, Spencer Gulf, and Gulf St Vincent. Within temperate regions, the largest areas of mangrove forest occur in South Australia (approximately 210 km^2), largely within the sheltered gulfs of Spencer Gulf and Gulf St Vincent (Galloway 1982). In NSW, the total area was 108.1 km^2 when it was last surveyed in 1981-1984 (West et al. 1985).

Values

Mangroves are important habitats for the following reasons (modified from NSW Fisheries 1998a):

- They provide important nursery areas for fish and invertebrates, including commercial species such as banana prawns, mud crabs and bait prawns (Staples 1980; Staples et al. 1985; Staples and Vance 1986; Robertson and Duke 1987; Robertson and Alongi 1995). This is due to the following factors:
    - Their structural complexity; trunks, fallen wood, aerial roots (including pneumatophores), crab burrows etc., all provide feeding opportunities and shelter;
    - Decomposing materials (leaves, wood, bark etc.) from the mangroves provide the basis of detrital food chains essential for many invertebrates (which in turn provide food for fishes) (Lee 1999, and references therein).
- They act as filters by trapping sediments and contaminants, and absorbing nutrients (Robertson and Alongi 1995). The mangroves (roots, pneumatophores, trunks and seedlings) slow water movement causing sediment to settle. Mangroves thus help to protect other marine habitats from sedimentation. They have also been shown to convert excess nutrients into extra growth (Robertson and Alongi 1995).
- They stabilise shorelines and riverbanks, protecting them (and the communities behind them) from erosion.

In addition, some elements of the mangrove invertebrate fauna are probably essential for the health of the mangroves, particularly crabs and teredinid molluscs (shipworms) (which facilitate the breakdown of detritus by shredding and consuming leaves and ingesting wood, and aerate the soil through burrowing; e.g., Robertson and Daniel 1989; Robertson 1991; Smith et al. 1991; see also Section 3.2.1).

However, Baran (1999) noted that while mangroves are frequently claimed to be important for coastal resources such as fish and shrimp, and the ecological background of these relationships is in some cases well documented, rigorously quantified relationships are surprisingly few. Some exceptions, for invertebrate fisheries, include:

- Turner (1977) for the relationship between hectares of vegetated estuary (Northeast Gulf of Mexico and Lousiana) and annual shrimp yield; r^2=0.69;
- Martosubroto and Naamin (1977) for the relationship between mangrove area (Southeast Asia) and shrimp production; r^2=0.79;
- Paw and Chua (1989) for the relationship between log10 of mangrove area (Philippines) and log10 of penaeid shrimp catch; r^2=0.66;
- Staples et al. (1985) for the relationship between length of mangrove shoreline (Australia) and banana prawn catch; r^2=0.58.

Nonetheless, these are correlations only and the causal link has not been established experimentally. Nor has it been proven that mangrove is the causal factor, rather than other associated factors such as extensive shallow seas, intertidal area, tidal creeks or length of coastline (Baran 1999). Landings of prawns in Northern Australia are related to levels of flooding which bring down organic matter from the wetlands acting as food supplies for prawns (Staples et al. 1985; Staples and Vance 1986; Staples et al. 1995).

Fauna

The fauna of mangroves is derived both from the terrestrial and marine environments, although most of the aquatic and infaunal elements are derived from the sea (Hutchings and Saenger 1987). In temperate regions, few mangrove specialists occur among the marine invertebrates, but these increase with decreasing latitude (Hutchings and Recher 1982). Crustaceans are conspicuous elements of the mangrove forests, particularly species of fiddler crabs (*Uca*), marsh crabs (*Sesarma*) and stalk-eyed crabs (*Macrophthalmus*) (Jones and Morgan 1994). Many molluscs are also conspicuous, especially large creeping snails of the families Potamididae (*Telescopium*, *Terebralia*, *Cerithidea*) and Batillariidae (*Batillaria*, *Pyrazus*). In addition to this specialised intertidal fauna, the creeks and lagoons in the mangroves also have a characteristic estuarine subtidal fauna dominated by polychaetes, crustaceans and molluscs (Hutchings and Saenger 1987). Colonial animals such as sponges and ascidians tend to be restricted to the seaward margins of the mangroves, except where tidal creeks flow through them (Hutchings and Saenger 1987).

Although microfauna, such as nematodes, are abundant in mangroves, their presence has been very poorly documented in Australia (Dacraemes and Coomans 1978; Alongi 1987). The meiofauna of the temperate mangrove system of Fullerton Cove, Hunter River, was documented by Hodda and Nicholas (1985) and Coull (1999) discussed their role in estuaries.

Conservation issues

The present-day distribution of mangroves results from a combination of historical and modern processes (Saenger 1998). The boundaries of mangrove areas are naturally dynamic in response to changes in patterns of sediment deposition and other factors. However, anthropogenic processes such as habitat modification and destruction are now having a substantial impact (Saenger 1998). In many tropical areas in SE Asia, mangrove habitats have been seriously depleted through clearance for aquaculture and for firewood; a figure of 1% per year was given by Ong (1982) for Malaysia, and can probably be taken as a conservative estimate of destruction of mangroves in the Asia-Pacific region as a whole (Ong 1995). Clearance for aquaculture has not occurred to the same extent in Australia, although they are used for oyster farming (which can involve habitat modification), as well as commercial and recreational crab trapping and recreational oyster collection. In some parts of Australia, declines in mangroves have been due to clearing or "reclamation" and changes in water flow attributable to adjacent developments (e.g., urban development, agriculture, port and marina development etc.). In some other areas mangroves appear to be increasing due to colonisation of sediment build-up (NSW Fisheries 1998a). This can result from land clearing and increased sediment transport in catchments and is one of the factors attributed to saltmarsh decline in Homebush Bay, Sydney (Burchett and Pulkownik 1996; Burchett et al. 1998; Burchett et al. 2000; Hutchings in press).

Another important cause of death of mangrove trees involves changes to the hydrological regime, including reductions in freshwater flow following the construction of dams or diversion for irrigation, and impoundment due to the construction of level banks and roads (Hatcher et al. 1989). Damage to mangroves can also occur locally where tidal exchange has been impeded by foreshore structures such as floodgates, culverts and levee banks, these blocking the dispersal of mangrove propagules and rendering upstream water quality unsuitable for healthy growth (NSW Fisheries 1998b).
Other anthropogenic threats to mangrove habitats include increasing demand for foreshore land and uninterrupted views; drainage and flood mitigation works; further deterioration in estuarine water quality; oil spills; insect (especially mosquito) control programs, species introductions, and rubbish dumping. There is also some concern for the potential impact of climate change. If relatively rapid sea level rise did occur, there would be little likelihood of saving mangroves whose landward margins have been developed (Ong 1995).

While precise figures on current areas and losses of mangroves (and other vegetated habitats) are difficult to obtain, some areas have recorded increases in mangroves (Hyland and Butler 1988), but there have also been major losses, especially around coastal cities such as Cairns, Brisbane, Newcastle and Sydney (e.g., Robertson and Alongi 1995). Of particular concern is the rapidly increasing coastal population in Queensland, which has and will continue to put pressure on mangrove habitats. Large areas of mangroves around Darwin, near Cairns, and Gladstone and in southeast Queensland are currently under threat from a variety of development projects (Hyland and Butler 1988; Robertson and Alongi 1995), with large areas already cleared in Darwin Harbour for port development.

The effects of such losses on the invertebrates and fishes, many of which use these habitats as nursery and feeding grounds, are difficult to establish, in part due to the poor data available (e.g. on fish landings, and fish landings do not necessarily reflect where the fish were caught). Anecdotal evidence from fishermen suggests that fish, at least, have become smaller and fewer (P.A.H., pers. comm.).

Information, research and management
Process-oriented research on mangroves has lagged behind that on coral reefs, and did not really get underway until the late 1960s. Until this time, mangroves were viewed in most countries as wastelands, suitable only for "reclamation", despite the fact that they were being successfully managed for timber and charcoal production in some places, such as Malaysia (Hatcher et al. 1989).

Protection of mangrove habitat is required through State legislation (including fisheries legislation) and by local councils through planning controls. However, further public education is needed regarding the values of mangroves in order for mangrove protection to gain the political and public support it needs if these habitats are to remain viable. In addition, proper planning is impeded by the fact that detailed mapping of the existing areas of mangroves has not been done around much of the Australian coast.

Saltmarshes

Marshlands have been described as among the most productive systems in the world (e.g., Adam 1990; Mitsch and Gosselink 1993; Begon et al. 1996a). Saltmarsh plants are found at the terrestrial-marine interface on flat areas on coastlines wherever conditions are so sheltered that wave action cannot physically erode the sediment and associated plants (Adam 1990). Such conditions occur, for example, on the landward side of mangroves.

While saltmarsh communities[82] are not subject to daily tidal inundation, they are flooded by larger tides, and the salt-tolerant plants are adapted to occasional inundation and semi-permanent pools of brackish water are sometimes common. They are vegetated by herbs, grasses or low shrubs, and are distinguished from mangroves by the absence of tall woody plants.

Distribution
In cooler areas (where mean winter temperatures drop below 16°), they extend into habitats that in warmer regions would be occupied by mangroves (Chapman 1977), but they co-occur with mangroves in NSW, Qld, NT and northern WA. Recent studies show that the saltmarsh habitat dynamically interacts with mangrove habitat, which in some areas replaces the seaward edge of the saltmarsh (Burchett and Pulkownik 1996; Burchett et al. 1998; NSW Fisheries 1998a; Burchett et al. 2000; Saintilan and Williams 2000; Hutchings 2001, in press).

Values
Saltmarshes act as a buffer and filtration system for sediments and nutrients. They are also highly productive components of the marine food web, as well as important feeding

[82] See Morrisey (1995) for an overview of saltmarsh ecology in temperate Australia.

grounds for birds (Parker 1995). During tidal inundation, they are used by nektonic crustaceans, such as prawns and portunid crabs (recently reviewed by Connolly 1999).

Fauna
Studies on their fauna have dealt with individual species or with the roles of animals in processing materials and energy in the saltmarsh ecosystem. Such studies have shown that grazing invertebrates do not necessarily feed directly on the saltmarsh plants but many feed on fungi growing on decaying vegetation (e.g., Graça et al. 2000). Conspicuous saltmarsh invertebrates include amphipods, isopods, some species of crabs and gastropod molluscs such as the primitive ellobiid and amphibolid pulmonates.

As pointed out by Richardson et al. (1998), different saltmarsh vegetation patterns are not necessarily a good surrogate for estimating spatial patterns of invertebrate diversity. In their study, they compared the crustacean and molluscan assemblages in a series of Tasmanian saltmarshes with the plant communities, in order to identify the main environmental factors controlling variation and to identify geographical groupings of marshes within the region. Using the animal data (in ordinations) led to grouping of the sites by the degree of submersion, whereas vegetation data grouped by the salinity of the substratum. Thus, conservation of saltmarsh communities based solely on representative plant communities will not necessarily provide adequate conservation for invertebrates.

Conservation issues
The main threats to saltmarshes include destruction or degradation of the habitat from agriculture, reclamation and development (roads, housing etc.). Photogrammetric surveys show that they are steadily declining in SE Australia due to invasion of mangroves and development (Saintilan and Williams 2000). Land 'reclamation' has had a major impact on saltmarshes, including filling for the construction of ports, marinas, canal estates, and urban and industrial sites (Adam 1995; State of the Environment Advisory Council 1996). Coastal development and reclamation have led, for example, to the loss of about 21% of the saltmarsh habitat from Moreton Bay in Queensland (State of the Environment Advisory Council 1996). Australia-wide the losses are not so great, but most have been concentrated in the southeast. This region, while having the smallest initial area, had the highest biodiversity and incidence of endemic species, so that losses are considered significant at both regional and national scales (Adam 1995). Much larger areas have been degraded, but data on the extent and impacts are not readily available. Degradation can occur as a result of rubbish dumping, off-road vehicles, invasion by weeds and non-indigenous species, pollution, spraying and drainage for mosquito and sandfly control, and heavy grazing by cattle (Adam 1995; State of the Environment Advisory Council 1996). Uncontrolled recreational use (e.g., trail bike riding, bait collecting) can also be highly destructive.

The extent and distribution of both mangroves and saltmarsh habitats will ultimately be affected by global warming (Hutchings 2001), with any rise in sea level having the potential to cause significant contractions in saltmarshes (Adam 1995; Zann 1995). However, as with other vegetated habitats, monitoring such changes will be difficult without adequate base-line data relating to the current distribution and extent of these habitats.

Seagrasses

Seagrasses[83] are all flowering plants belonging to several genera, including *Posidonia*, *Amphibolis*, *Heterozostera* and *Posidonia,* the latter commonly forming extensive "meadows" below the low tide mark. Seagrass beds are most prominent and widespread along coastlines of limited riverine runoff and relatively oligotrophic waters and are excluded from areas of high turbidity (Dennison et al. 1993). Seagrass ecosystems are highly productive at all trophic levels and turnover of populations is generally great due to high consumption rates (Keough and Jenkins 1995). They are known to support diverse communities of invertebrates.

Distribution
Australia has the largest seagrass meadows and the largest number of seagrass species in the world (Kuo and McComb 1989). Seagrass beds occur around the entire coast of Australia in shallow, protected marine lagoons, the mouths of some estuaries, and in sheltered bays and inlets. Of the 51,000 km^2 of seagrass meadows in Australia, the largest meadows occur in tropical Australia (Mukai 1993), including Torres Strait (17,500 km^2 or 35.3%) and Shark Bay (13,000 km^2 or 25.5%). In Queensland alone, at least another 4000km^2 are expected to be recorded from areas still under study (Lee Long et al. 2000). It appears that tropical seagrass beds are generally highly variable in abundance and distribution (McKenzie et al. 1998), although seagrass habitats in Moreton Bay have been stable over 25 years (Poiner 1985). Within temperate waters, seagrass meadows are largest and most diverse along the southern coast of Western Australia (9,000 km^2 or 17.6%) and Spencer Gulf and Gulf St Vincent in South Australia (5,000 km^2 or 9.8%) (Keough and Jenkins 1995; Poiner and Peterken 1995). In southeastern Australia, in contrast, seagrass abundance (and diversity) is low, and the high-energy coastline restricts seagrass to estuaries and protected bays. For instance, seagrass occupies approximately 500 km^2 (1.0%) in coastal Tasmanian waters, 150 km^2 (0.3%) in the waters of New South Wales and 100 km^2 (0.2%) in Victorian waters. The status of seagrasses in Australia was reviewed by Kirkman (1997) and Butler (1999).

Values
Seagrasses fix substantial quantities of carbon during growth, although very little of this production is utilised directly by herbivores (Lepoint et al. 2000)). Most carbon fixed by seagrasses enters into detrital food chains of the lagoonal and nearshore seafloor, through shedding of leaves (Peterson and Wells 1998). In addition to the direct importance of seagrass to detrital food chains, Peterson and Wells (1998) list several reasons why seagrass plants are important habitats for invertebrates:

- They provide a firm surface in otherwise soft and mobile sediments. Their leaf surfaces are colonised by numerous microalgae grazed by abundant small gastropods

[83] For an introduction to seagrass ecology see Keough and Jenkins (1995).

and crustaceans and other micrograzers(Jernakoff 1996). These, in turn, provide food for mobile predators such as fishes, crabs, lobsters, and octopuses.

- Seagrass blades are also inhabited by attached invertebrates such as hydroids, sponges, certain polychaetes, bivalves and bryozoans, which themselves harbour and provide food for other animals.
- The subsurface roots and rhizomes of the grasses affect the penetrability of the sea floor by digging and burrowing predators, such as benthic fishes and crustaceans, and thus provide a partial refuge for infaunal invertebrates. Seagrasses also enhance the abundances of many other soft-bottom species.
- Abundance of invertebrate species may also be enhanced by the indirect effects of seagrass blades, which project into the water column and moderate currents, enhancing deposition of organic-rich fine particles, detritus, and even larvae.

Seagrasses have considerable value as nursery, feeding and shelter areas for dugong, turtle and many commercial fish and invertebrate species (e.g., Jernakoff 1987; Poiner et al. 1987; Poiner and Peterken 1995; Acosta 1999; Aragones and Marsh 2000; Nagelkerken et al. 2001; Peterson and Heck 2001) and hence help in maintaining coastal fisheries. They also contribute organic matter, baffle water currents causing them to drop sediment loads (thus maintaining water quality) and help prevent erosion (Connolly 1995; NSW Fisheries 1998a).

Seagrass communities are amongst the better-studied marine ecosystems in Australia and elsewhere. Their importance as habitat and nurseries has been assessed in a number of studies that have compared the biodiversity and biomass of seagrass beds with that of unvegetated sediments (e.g., Hutchings et al. (1991, Princess Royal Harbour, WA); Arrivillaga and Baltz (1999, Guatemala's Atlantic coast); and Bird and Jenkins (1999, Port Phillip Bay, Victoria). Acosta (1999) found that Caribbean spiny lobsters were significantly more abundant on coral islands surrounded by seagrass, and more juveniles were present compared with islands surrounded by rubble.

The reasons for the greater abundance, biomass and/or diversity of fauna associated with seagrass compared with unvegetated areas have been experimentally assessed in a number of studies (e.g., Bell and Westoby 1986a, 1986b; Bell et al. 1988; Connolly 1995).

The detached blades of seagrasses also wash onto shores and form an important habitat for high-tidal invertebrates, the decomposing litter being a significant food source for detritivores (e.g., Kirkman and Kendrick 1997) as well as habitat for strand line invertebrates such as seaweed flies (Blanche 1992).

Conservation issues

Much has been written about the importance of, and threats to, seagrasses. Destruction of seagrasses can result from natural causes (e.g., storm damage, destruction from predation; Maciá and Lirman 1999), but these can be exacerbated by human factors such as clearing in catchments. The major anthropogenic threats include clearing, reclamation, pollution, and sedimentation. Many studies have documented the sensitivity of seagrasses to reduced light levels resulting from turbidity (e.g., Fitzpatrick and Kirkman 1995), smothering from sediment (e.g., Preen et al. 1995) (see also Section 6.6.1) or algal overgrowth (e.g., Hauxwell et al. 2001). In Queensland, large-scale destruction of beds has occurred on various occasions after floods and cyclones (e.g., Preen et al. 1995) but recovery has normally taken place within 10 years (Birch and Birch 1984). This contrasts with temperate seagrasses that seem to recover much more slowly. Nearshore meadows have been very susceptible to losses from impacts associated with coastal development and catchment run-off (Hyland et al. 1989).

Walker and McComb (1992) described seagrass degradation in Australian waters and summarised human-induced declines in seagrasses from 11 sets of locations around Australia. Similarly, Kirkman (1997) reviewed the status of seagrass habitat state-by-state and listed 15 major areas of seagrass loss (table reproduced in State of the Environment Advisory Council 1996, and in Section 6.4.2, Table 6.2). Lee Long et al. (2000) summarised the major issues for seagrass conservation and management in Queensland.

The loss of seagrass habitat is of particular concern given that seagrasses generally do not readily colonise areas from which they have been eliminated (e.g., NSW Fisheries 1998a; Inglis 2000), although dispersal ability varies from species to species. For instance, species with small, poorly dispersed fruits (e.g., *Halophila* and *Halodule*) are more likely to form persistent seed reserves and be rapid colonisers of disturbances within established beds, whereas genera with large buoyant fruits (e.g., *Thalassia*, *Posidonia*) rarely recruit within existing beds of conspecifics. Attempts to re-establish seagrasses have frequently been unsuccessful. For instance, attempts to transplant *Amphibolis griffithii* in West Australia found that the small transplanted units of seagrass could not be adequately anchored for any extended period of time – particularly in areas of any wave action – and therefore failed to develop the necessary root matt (Paling et al. 2000). Processes which destroy the below-ground as well as the above-ground biomass (e.g., exposure of rhizomes through reduction in sediment level; dredging; grazing) can result in delayed recovery since sexual recruitment and seedling growth tend to be much slower than asexual regrowth (Maciá and Lirman 1999).

While the implications of seagrass loss for species such as dugongs have received most attention (e.g., Preen and Marsh 1995), the seagrass-associated invertebrate fauna will also be affected by the loss of habitat, shelter, and productivity. For instance, the epifauna (particularly that which is obligate on seagrass leaves) may decline or disappear altogether after loss or degradation of seagrass (Jernakoff 1996).

Information and research

Butler and Jernakoff (1999) reviewed knowledge of seagrasses in Australia and discussed development of a Research and Development plan.

Algal communities, including kelp forests

Algae provide a range of very important habitats for invertebrates, forming diverse communities in the intertidal and shallow sublittoral zones, particularly on hard substrata (mainly rocky shores and coral reefs). Algae are not only highly diverse taxonomically but show considerable structural diversity, with different communities including coralline algal mats (e.g., corallines, *Halimeda*), other kinds of turfing algae (reds or browns), and other macroalgae[84] or "seaweeds"[85] including branching algae and kelp "forests". These or similar "functional form" groupings of algae has been used extensively in the literature (see Padilla and Allen 2000). Southern Australia has the highest levels of species richness and endemism of marine algae in the world (Phillips 2001).

Distribution

Macroalgal communities, composed of the larger red, brown and green algae, occur mainly in temperate southern Australian waters, where they comprise a major component of the shallow-water reef communities. The 5500-km coastline from the southwestern part of Western Australia to the New South Wales/Victorian border has a high diversity of macroalgae, particularly brown algae (Phaeophyta) and red algae (Rhodophyta), with 558 species recorded on the southern coast of Western Australia, 1151 species in South Australia and Victoria, and 398 species around New South Wales and Lord Howe Island (State of the Environment Advisory Council 1996). The biology and distribution of Australian macroalgae have been reviewed by Clayton and King (1990) and those of southern Australia by Womersley (1984; 1987; 1994). Sanderson (1997) also reviewed macroalgal assemblages, particularly biogeographic patterns, for temperate Australian bioregions as part of a series of State of the Environment technical papers. Millar and Kraft (1993; 1994a; 1994b) catalogued the red, brown and green (respectively) algae of NSW, including the rich algal flora of Lord Howe Island. Also in NSW, Underwood and Chapman (1998a) and Chapman and Underwood (1998) examined variation in algal assemblages on wave-exposed rocky shores.

Fauna

The structure and complexity of algal communities determine the types of invertebrates found associated with them. Like higher plants, algae have chemical defences and so invertebrates that graze directly on the algae become adapted to feeding on particular kinds[86]. Algal mats or turf (including corallines) and the intertidal branching algae in pools and in the shallow sublittoral have their own communities of invertebrates different from those found on seagrasses. These faunas consist of a huge variety of small or minute invertebrates including polychaetes, crustaceans, molluscs, echinoderms etc. Other than casual observation, little is known in detail of these faunal communities in Australia. Studies by Smith et al. (1996b) on the macrofaunal communities of *Ecklonia radiata* holdfasts have revealed a diverse and dynamic benthic community associated with this habitat. Some studies have been undertaken recently on the fauna of intertidal algal turf (Kelaher 2000; Townsend 2000).

[84] Macroalgae (the larger multi-celled species of algae generally referred to as seaweed) have a variety of growth forms and appearances ranging from a tall leathery thallus several metres in length (kelp), to dense accumulations of fronds or filaments or encrusting mats.

[85] Hinde (1995) provides an overview of the fauna and ecology of seaweeds and other algae in temperate Australia, while Kennelly (1995) reviews the ecology of kelp forests and their fauna.

[86] E.g., Steinberg (1995) studied interactions between the echinoid *Holopneustes purpurescens* and its host kelp *Ecklonia radiata*; and Poore (1999) discussed host plant use by a marine amphipod.

Kelp forests are important in cooler waters. While there is a considerable amount of information on the ecology of Californian kelp forests (e.g., Cowen et al. 1982; Coyer 1984; Dayton 1985) [87], much less is known about those of temperate Australia, recent reviews being given by Kennelly (1995) and Steinberg (1999). Kennelly and Underwood (1985; Kennelly and Underwood 1992) have studied invertebrate communities associated with kelp forests in NSW, while Edgar (1983) studied the ecology of animal communities associated with algal communities in southeast Tasmania. Edgar found that the dominance and animal density were related to levels of food supply. The animal diversity depended on the weight of sampled algae but was also strongly influenced by wave exposure and habitat complexity (i.e., number of microhabitats).

Invertebrates associated with intertidal algal communities on rocky shores and subtidal rocky reefs have been studied to some extent, particularly in relation to the effects of grazers on algal biomass and community structure. The most frequently studied groups of grazers are gastropods (summary of work included in general review by Underwood 2000) and sea urchins (e.g., Jones and Andrew 1990; Andrew 1991; 1993 for temperate Australia), the latter being important in maintaining patches free of algae (so-called "urchin barrens") that can be settled by other species.

Drift algae can also be an important habitat for fauna (e.g., Smith 2002), sometimes with its own faunal elements. It is also an important means of distributing small invertebrates.

In addition to their role in providing habitat, algae form a major source of detrital input into shallow water marine systems and can contribute a considerable proportion of the calcareous material that makes up coral reefs (Hatcher and Larkum 1983).

Conservation issues

The effects of human activities on algal communities have been less well studied and are consequently less well understood than those on some other marine communities, such as seagrasses and coral reefs. The most serious potential pressures are probably those that affect the habitat-forming species - particularly the large algae (State of the Environment Advisory Council 1996). Such threats include:

- Discharges and other activities that affect water quality[88].
- Commercial fishing – for instance, trawling may directly remove plants;
- Harvesting of grazing animals – the removal of herbivorous animals, such as fish, rock lobster or abalone, may alter the community balance and lead to a shift in the relative abundance of algae;
- Algal harvesting – this is an important industry in some other parts of the world, and some occurs in the Bass Strait islands, but this small-scale industry as currently practiced is apparently sustainable and probably does not pose a threat.

[87] see also http://www.biology.ucsc.edu/people/raimondi/readdie/index.html

[88] Victorian and overseas studies claim that sewage outfalls reduce canopy-forming brown algae (State of the Environment Advisory Council 1996; Archambault et al. 2001) and change the structure of intertidal communities which can be reversed when the sewage is removed).

Although generally not well documented, in some areas, substantial declines have occurred in the distribution of macroalgal communities. For instance, studies (Sanderson 1990; Sanderson 1997) of the distribution of *Macrocystis pyrifera*, an ecologically important species in Tasmanian waters, showed that stands of this species were significantly reduced from levels determined in the early 1950s (Cribb 1954). While changes in oceanographic currents were considered to be the principal cause, other possible factors included (Sanderson 1997):

- Disturbance to the substrate from dredging (resulting in the silting up of inshore reef areas);
- Increasing sediment load in coastal waters as a result of land clearing and wood-chipping;
- Increase in boat traffic which cuts off growing fronds;
- Over-fishing of rock lobster which are believed to feed on sea urchins, which in turn feed on *M. pyrifer*a;
- Commercial harvesting of *M. pyrifera* in the early 1970s; and / or
- The recent introduction of *Undaria pinnatifida* (a Japanese seaweed thought to have been introduced via ballast water) that occupies a similar ecological niche to *M. pyrifera* and is thus a potential competitor.

Introduced algae can pose a serious threat by replacement of native species (for instance by smothering native algae and seagrasses), and by providing less suitable habitat for native-algal adapted invertebrate faunas[89]. A number of macroalgal species have been introduced to Australian waters, two of the most significant being *Undaria pinnatifida* in Tasmania and *Caulerpa filiformis* on the New South Wales coast. Both blanket the rocky reef bottom for large areas, limiting colonisation by native algae (Sanderson 1997). Members of the Caulerpaceae (Chlorophyta) can be particularly invasive. For instance, Davis et al. (1997) documented the effects of a rapid invasion of a sponge-dominated deep reef in Botany Bay by *Caulerpa scalpelliformis*, a species indigenous to Australia but not to this locality. A recent paper by Jousson et al. (2000), has shown that the strains of *Caulerpa taxifolia* found in Eastern Australian waters may be the possible original source of this invasive species which has now been reported from California, but additional genetic studies need to be carried out.

Sediments

The main types of habitats formed by sediments in marine coastal systems are outlined below, along with a brief overview of their characteristic macroinvertebrate faunas. All these sediments also contain (largely unknown) minute invertebrates living in the interstices between the sediments, the *meiofauna*. Overall, summaries of the main research needs and conservation issues are also provided.

The meiofauna is composed of microscopic interstitial animals that inhabit the spaces between sand grains (the meiobenthic realm), and many of them are very unusual and specialised. The majority of recognised animal phyla have meiofaunal representatives,

[89] For example, the relacement of native green algae in Britain by introduced species has led to declines in specialist (host specific) suctorial ascoglossan seaslugs (Trowbridge and Todd 2001).

with five phyla (Gnathostomulida, Kinorhyncha, Loricifera, Gastrotricha, Tardigrada) being found exclusively in these habitats. Studies on Australian meiofauna are few and most of the fauna remains unsampled. Elsewhere meiofaunal studies continually find many new animals that exhibit interesting and unusual adaptations. Introductory texts on the meiofauna include Higgins and Thiel (1988), Giere (1993) and Thrush (1988) while Coull (1999 and references therein), includes a summary of what is known about Australian meiofauna.

Beaches

Sandy shores occupy approximately 50-70% of the temperate Australian coastline, particularly along the eastern and western coast of Australia (Fairweather and Quinn 1995; Edyvane 1996).

The intertidal sandy beach is a physically rigorous environment because of wave action, shifting sediments and aerial exposure. Attached plants are typically absent due to the mobile nature of the sand. Jones and Short (1995) provide an overview of the physical features and the ecology of sandy beaches in temperate Australia.

Fauna

The invertebrate fauna on Australian sandy beaches is not well studied although Dexter (1992) investigated the role of exposure and latitude on community structure. This fauna, while not rich, is highly specialised and typically found only in this type of habitat. It includes predators (such as certain crabs and snails) and infaunal detritus and suspension feeders (such as bivalves, polychaetes, crustaceans, holothurians, etc.).

Three potential food sources make the intertidal beach a food-rich environment; zooplankton (brought by the substantial flow of water), various microalgae such as diatoms (since the beach provides a well-illuminated substratum for their growth), and detritus, especially seagrasses and algae deposited by waves (Peterson and Wells 1998). This latter food source supports a specialised strandline fauna, particularly small crustaceans (amphipods and isopods), these often being found in large numbers (see below). Physical and ecological factors such as wave action, depth of water, sediment grain size and predation determine the distribution of the animals across and along a beach (Dexter 1984, 1985, 1992; James et al. 1995; James and Fairweather 1996; McLachlan et al. 1996) and some taxa migrate with the tidal cycle (Hacking 1996). The nematode fauna has also been investigated on a sandy beach in eastern Australia (Nicholas and Hodda 1999).

Species diversity and abundance increases on more sheltered beaches, many of the species found in these habitats apparently not being able to survive on the most exposed sandy beaches (Dexter 1992). Dexter (1992) also found that the number of species per beach was highest for cold temperate beaches and decreased gradually towards the tropics, with high numbers in the tropics observed only on very protected beaches. There were no reported studies at latitudes higher than 60°S, since most Antarctic shorelines are rock and gravel, and any sandy shores freeze in winter and at best support a temporary summer fauna (Dahl 1952).

Bivalve molluscs form a major part of the biomass on sandy beaches in some parts of the world and are often commercially harvested (e.g., the bivalve pipis, Donacidae, in NSW). However, in general there are few commercially harvested species on exposed sandy beaches in Australia, and none at all in some areas. The ghost crabs (Ocypodidae) and mole crabs (Hippidae) are characteristic inhabitants of tropical and subtropical sandy shores (Jones and Morgan 1994). Several studies have been carried out on sandy beach fauna in New South Wales (e.g., Dexter 1984; Hacking 1998).

Beachwashed drift algae, seagrassand other detritus provide an important habitat (including shelter and a food source) for strandline fauna, some of which may be confined to this narrow, vulnerable zone. Above the strandline, the "supralittoral zone" grades into a fully terrestrial habitat and this too has unique assemblages of animals adapted to these environments on all sorts of shores including rocky coasts. Amphipods are one of the most common and abundant detritivoral groups found in strand line litter and their feeding affects the turnover of detritus and biomass and facilitates nutrient cycling (Harrison 1977; Bartodzieg 1992). Amphipod abundance in Thomson Bay, Rottnest Island, was studied by Fong (1999) and, there was a low diversity of amphipods at this location (five species), one of these accounted for nearly 95% of total amphipod abundance. Richardson et al. (1997) showed that there is a discrete fauna of translittoral amphipods and argued for the need to reserve transitional habitats. Shores in a state of "wilderness" (i.e. unimpacted) are particularly important as few are now left with native pioneer vegetation and transition to native terrestrial vegetation undisturbed. Such areas are important for conservation and to enable research on sea/land transitional habitats. "Wilderness beaches" are particularly important for the conservation of strandline fauna (A. Richardson pers. comm.).

Beach washed drift algae, seagrass and other detritus are frequently removed through beach cleaning (see Section 6.14) and algal harvesting. The ecological significance of drifting and stranded algae and seagrasses in Australia, and the effects of harvesting, were reviewed by Kirkman and Kendrick (1997).

Tidal mudflats and sandflats

In sheltered waters (e.g., in estuaries, sheltered bays or inlets), "beaches" and intertidal flats consist of sandy to muddy substrates containing much nutrient-rich organic material. Although these habitats often appear unproductive and may be ignored at the expense of vegetated habitats, they support a very diverse and rich burrowing infauna and surface epifauna which is a significant source of food for fish (Inglis 1995; NSW Fisheries 1998a; Potter and Hyndes 1999).

Fauna

These rich invertebrate communities are dominated by molluscs, polychaetes and crustaceans, but also include many other infaunal groups as well as echinoderms, anemones, etc. The crustaceans include soldier crabs (Mictyridae), sentinel crabs, hermit crabs and pistol shrimps (Alpheidae) (Jones and Morgan 1994).

Salt flats

Salt flats are extensively developed in northwestern Australia as a result of the arid climatic conditions. These largely bare areas, which often occur to the landward side of mangroves, are hypersaline due to salt deposits left from evaporating seawater. In some cases, the resulting deposits are actually harvested for human consumption. Salt flats occur at or above the highest tide levels and are immersed only rarely at very high tides or during rain. They are essentially the unvegetated equivalents of saltmarshes.

Fauna
Organisms on these flats can tolerate wide variations in salinity and extreme exposure to the sun and air. Their most conspicuous invertebrate inhabitants include air-breathing crustaceans (e.g., fiddler crabs, terrestrial hermit crabs etc.; Jones and Morgan 1994).

Unvegetated subtidal sediments (sand and mud)

Subtidal sand plains – shallow sand plains that extend from the surf zone to about 20 m depth – are shallow enough for wave energy and water currents to disturb the bottom, sometimes shifting large amounts of sediment and ensuring the maintenance of a sandy substratum. Many suspension feeders are found in these environments and zonation of species is produced by the gradient in wave intensity and intense competition for space (Peterson and Wells 1998). Predation can also be intense, the main predators including fish, starfish, gastropods, octopuses and crustaceans.

Fauna
While typically receiving far less attention than the adjacent (and far more spectacular) coral reef communities, soft-bottom inter-reefal areas are also a critical habitat, and possess diverse and abundant infauna and epifauna (e.g., Birtles and Arnold 1983; Cannon et al. 1987). For instance, 1 km^2 of inter-reefal habitat on the Great Barrier Reef contained around 220 species of macro-invertebrates (Hooper 1988). Such soft-sediment benthic communities are susceptible to anthropogenic disturbance, particularly from trawling (Hutchings 1990; Thrush et al. 1998; Thrush et al. 2001), from increased sediment load due to human disturbance (Brown and Rakocinski 2000; Frouin 2000), from chemical pollutants (Rakocinski et al. 2000) or euthrophication (Heip 1995).

Soft bottom habitats in temperate areas can likewise support very high diversity. For instance, a quantitative survey of the macrobenthos of Western Port, Victoria (Coleman et al. 1978) obtained more than 19 600 individual invertebrates comprising 572 species. The fauna was dominated by polychaetes, crustaceans and molluscs, which provided respectively 54.1%, 31.7% and 6.6% of the individuals collected and 35.7%, 47.7% and 10.3% of the species identified. Other studies on the diversity of soft sediment communities from different parts of Australia are summarised in Chapter 2.

Shallow soft bottom communities in the sub-Antarctic must contend with major disturbance from severe storms and even iceberg impacts, with such impacts temporarily removing nearly all the fauna (Peck et al. 1999).

Information and research - sediments in general

Fairweather and Quinn (1995) and Constable (1999) reviewed the status of ecological research on soft shores. Both noted that soft sediment environments have been far less well studied than other habitat types; for instance, Fairweather and Quinn (1995) identified unvegetated soft bottoms as the least studied benthic habitat in Australia, while Constable (1999) pointed out that the ecology of these systems has been given much less attention than rocky shores. Estes (2000) reviewed the general accomplishments of ecological research on seafloor systems.

Constable (1999) reviewed the ecology of benthic macroinvertebrates in soft sediment environments and the progress towards quantitative models and predictions, noting that most models have focused on the dynamics of exploited populations. He summarised work on a number of different areas, including:

- Recruitment;
- Interactions with the environment – e.g., tolerances to the physical environment, structure and stabilization of the substratum, bioturbation;
- Biological interactions;
- Extrinsic factors (stressors) influencing soft-sediment habitats;
- Availability of patches, etc.

In addition to displaying an often-surprising degree of spatial and temporal heterogeneity (Morrisey et al. 1992a; Morrisey 1992b; see also Section 5.4.2), soft-sediment habitats can be dynamic environments, and the substratum can be completely removed or replaced by new layers of sediment. Such changes may be exacerbated by anthropogenic impacts such as dredging or increased rates of sedimentation resulting from land clearing.

Conservation issues - sediments in general

Threats to the invertebrates of exposed sandy beach habitats are not as great as in some other habitat types. Harvesting of larger bivalves may require the imposition of bag limits to ensure population viability but harvesting beach worms for bait is said to be sustainable using current methods (see Section 6.10.3), although no detailed ecological studies have been conducted to support this finding. Harvesting of bivalves on more sheltered shores is often logistically easier, and hence more efficient, and may require careful management.

The use of suction pumps to catch saltwater yabbies (*Callianassa, Upogebia*) on sheltered flats or in seagrass beds is damaging, causing disturbance and smothering. Controls on what equipment may be used or where pumping may be carried out are in place in only some States.

Epifaunal soft bottom communities are particularly vulnerable to the impacts of trawling. In protected localities, such as sheltered bays and estuaries, the whole community may be subject to impacts associated with urban and industrial development, such as pollution and catchment runoff. Fairweather and Quinn (1995) pointed out that many soft shores are grossly polluted (e.g., urbanised estuarine mudflats) or regularly disturbed by recreational activities.

Hard substrates

The main types of habitats formed by hard substrates in marine coastal systems are outlined below, along with a brief overview of their characteristic macroinvertebrate faunas and specific research needs and conservation issues.

Rocky shores

Rocky shores[90] are more widespread in temperate marine environments (particularly in the Great Australian Bight and Tasmania) than in the tropics, occupying approximately 30% of the coastline (cf. approximately 11% in tropical areas) (Edyvane 1996). Rocky shores are most predominant in areas where wave energy is high to moderate, although rocky shores also occur in sheltered estuaries. Man-made structures such as seawalls, breakwaters, etc. provide somewhat similar, but typically much less diverse, habitats. These are discussed under a separate heading below.

Habitats and fauna

Stable rocky shores of bedrock and large boulders support a great variety of marine invertebrates often in large numbers, including many species of molluscs, barnacles, crabs and other crustaceans, polychaetes, ascidians, echinoderms, etc. The rock not only provides hard surfaces for attachment, but also provides many permanently moist and/or shaded microhabitats in pools, cracks and crevices, as well as loose rocks and rubble beneath which many invertebrates shelter. The type of rock, its surface texture and even its mineral composition (Bavestrello et al. 2000) can be important in determining the faunal composition.

Attached algae and aggregations of invertebrates (barnacles, mussels, serpulid worms with their distinctive calcareous tubes etc.) provide shelter for numerous small invertebrates (molluscs, crustaceans, polychaetes, echinoderms, etc.).

Intertidal habitats are subject to alternate periods of fully marine and aerial conditions during the tidal cycle. Consequently, there are very striking gradients in the physical environments from the bottom to the top of the shore that tend to be correlated with decreasing sizes and diversity of the organisms. This, together with varying degrees of competitive interactions and predation pressures, often results in the most abundant and conspicuous intertidal organisms exhibiting “zonation”. Although the description of shores based on these patterns of zonation was common in early papers on intertidal

[90] See Underwood and Chapman (1995) for an excellent introduction to rocky shore habitats in temperate Australia.

ecology (e.g., Dakin 1952; Bennett 1987), it is now seen as neither a useful, nor a realistic concept for understanding the ecology of intertidal organisms (e.g., Underwood 1998).

Many organisms on rocky shores disperse with planktonic propagules (typically larvae), and their numbers are therefore very variable in time and space. As a result, they live in a complex, dynamic and variable world (Underwood et al. 1983; Underwood and Fairweather 1989).

An enormous and significant body of work has been done on the ecology of rocky shores, particularly in southeastern Australia, and there is a correspondingly enormous literature on the subject (Some recent reviews include Fairweather and Quinn 1995; Underwood and Chapman 1995; Andrew 1999; Estes and Peterson 2000; Menge 2000; Underwood 2000; Knox 2001).

A few studies have been carried out on rocky shores in the Antarctic and sub-Antarctic (e.g., Smith 2000). Faunas in these areas must contend with severe storms and ice, with ice impacts holding communities at early successional stages (McCook and Chapman 1991; Peck et al. 1999).

Conservation issues

The most significant threats to these habitats are likely to be (depending on the location) human predation, trampling, siltation and pollution (Crowe et al. 2000). Underwood (1993b) described the effects of exploitation of species on rocky shores of NSW, and discussed management options (see also Section 6.10.3). Disturbance by fossickers can also have a significant impact in areas subjected to high levels of visitation. The major damage is from overturning rocks on reefs and exposing the invertebrates on the undersides to predation and desiccation (Keough et al. 1993; Keough and Quinn 1998). Some of this damage could possibly be prevented by better education - such as encouraging people to return rocks to their original position.

Introduced species (such as the alga *Caulerpa filiformis* in NSW) can also pose a threat by outcompeting native biota. Within estuaries, the introduced Pacific oyster (*Crassostrea gigas*) can modify hard substrate habitats in areas where native rock oysters (*Saccostrea glomerata*) do not occur. Pacific oysters are now the dominant intertidal invertebrate in the Derwent Estuary in Tasmania, where previously no rock oysters were found, and they are also displacing the slower-growing *S. glomerata* in many areas of NSW. They also breed slightly earlier so their spat settle and reduce the available space for the later settling spat of the Sydney Rock oyster (Holliday 1987).

Subtidal rocky reefs

Subtidal hard substrates include both natural rocky reefs[91] and man-made habitats such as the subtidal part of breakwaters. The former occur seaward of most headlands and rock

[91] See Butler (1995) for an introduction.

platforms and are found in deeper waters offshore, around islands and in estuaries. Artificial substrates are discussed in the following section.

Habitats and fauna
Invertebrate diversity in subtidal rocky reefs is usually high, typically with many hundreds of species. Many commercially and/or recreationally important invertebrates (as well as fish) depend on these habitats for some or all of their life cycle (e.g., eastern rock lobster, abalone). Thus, these habitats are ecologically important for many invertebrate taxa and fishes, and are important refuges and feeding areas for fishes.

The nature of subtidal rocky reef habitat changes with depth, water movement, degree of shading and latitude. These reefs support a mosaic of kelp beds, foliose turfing algae and, in some areas, "barrens" dominated by thin encrusting coralline algae and maintained by grazing sea urchins (Jones and Andrew 1990; Steinberg 1995; NSW Fisheries 1998b; Steinberg and Kendrick 1999). The physical structure and habitat complexity of these habitats is enhanced by macroalgae (e.g., rich growths of large brown algae such as *Ecklonia*, *Sargassum*, *Macrocystis*, *Durvillea* and *Cystophora*, which in their most developed state, form kelp "forests" – see under "Plant"-based habitats above), as well as sponges, ascidians and other sessile invertebrates. Deeper reefs (i.e. those below about 20 m) and well-shaded areas of shallow reef (e.g., under overhanging ledges) have insufficient light for much algal growth and are dominated by sessile invertebrates such as sponges, ascidians, bryozoans and cnidarians (Zann 1996; Roberts 1996a, 1996b; Glasby 2000; Glasby and Connell 2001). Subtropical and tropical subtidal rocky reefs are dominated by hard and soft corals, sponges, and other colonial animals.

There are several reviews of the ecology of Australian subtidal rocky reefs available (Butler 1995; Keough and Butler 1995; Andrew 1999; Butler and Connolly 1999).

Submerged cave habitats provide a different type of environment, being dark and usually more sheltered. They can have rich growths of colonial animals and some taxa are found in these dark habitats that are otherwise buried deep in rubble or some may be unique to these habitats (e.g., Hart et al. 1985).

Information and Research
Research on subtidal hard substrata, including dominant assemblages, types of animals, processes and threats, has been reviewed by Keough and Butler (1995) and the accomplishments of such ecological research were reviewed by Estes (2000). In general, the conservation status of rocky reefs is poorly known, as a result of limited taxonomic expertise, difficulties (including the high cost) associated with underwater research, and lack of recognition of their value (Zann 1996; NSW Fisheries 1998b).

Just how poorly these habitats are known is illustrated by Harriott et al. (1999) who investigated subtidal rocky reef communities in northern New South Wales from the Queensland border (28˚S) to the southern extent of extensive coral communities in coastal Australia (31˚S). The survey produced 28 new records for coral species, increasing the species richness of hermatypic corals known for the northern NSW region (excluding Solitary Islands) from 14 to 43.

Conservation issues
Submerged rocky reefs are likely to be less affected by human interference than most other habitats because they are relatively inaccessible, and generally unsuitable for destructive commercial fishing methods (particularly trawling). However, near the major urban centres reefs can be locally affected by over-harvesting, sewage pollution from ocean outfalls, sedimentation, deterioration in water quality and introduced species such as the Northern Pacific Seastar (*Asterias amurensis*) (see Section 6.5.2). Boat anchors can damage rocky reef habitat by smashing corals and other sessile growths and significant damage is likely at popular angling and dive sites. SCUBA divers may cause local damage either through direct contact via fins and other equipment, or through the action of exhaled air bubbles on cave ceilings and other overhanging sections (NSW Fisheries 1998b). These areas are increasingly being used by SCUBA divers and snorkellers for recreational purposes, offering diverse and abundant marine life as well as interesting topographical features such as cliffs ("dropoffs") and caves. They are also used by commercial fishers (Blacklip Abalone, rock lobsters, sea urchins, turban shells); spearfishers (cuttlefish, octopuses, as well as fishes) and collectors (mainly various molluscs, but also gorgonians etc.).

Gross changes caused by erosion or sedimentation can be easily indicated by the physical extent of submerged rock habitat. There can be difficulties in assessing changes in community composition and other features due to the presence of patchiness on a variety of scales, although this can be overcome by proper sampling design (e.g., Underwood and Chapman 1998b; Chapman 2000; Underwood 2000).

Artificial hard surfaces

Artificial substrates create, to varying extents, habitat analogous to natural intertidal and subtidal rocky reefs. They include such structures as bridge and wharf pylons, oyster leases, breakwaters, pontoons, purpose-built artificial reefs, shipwrecks, dumped materials such as car bodies, and even the hulls of moored boats. These habitats occur around the entire coastline, but particularly near major urban centres and within bays and estuaries (NSW Fisheries 1998b, pp.153-162).

Fauna
Artificial hard surfaces may be colonised by a wide variety of sessile invertebrates and algae, although they are not equivalent to natural rocky reef communities (Glasby 1997; Connell and Glasby 1999; Glasby and Connell 1999; Connell 2001; Glasby and Connell 2001). The composition of the substrate will effect species composition and abundance (Anderson and Underwood 1994; Holm et al. 1997), as well the orientation and position of surfaces (Glasby and Connell 2001).

Lindegarth (2001) recorded considerable variation in soft bottom communities associated with and without pontoons and that impacts of pontoons occur at some sites and not others, suggesting that there may be intrinsic differences among sites or that pontoon maintenance varies.

Intertidal anaerobic habitats

Naturally occurring intertidal anaerobic habitats are very localised and unusual when found in fully marine situations. While anaerobic conditions are sometimes seen as undesirable and often result directly or indirectly from are a consequence of anthropogenic pollution, some anaerobic habitats are natural and contain suites of species dependent on these environmental conditions.

Fauna
A number of molluscs live only under large rocks embedded in partially anaerobic sediments in the intertidal zone in which ferruginous bacteria thrive. These habitats are very sensitive to disturbance and are extremely threatened in some areas. Usually even the act of dislodging a rock to locate suitable habitat will alter the flow regime beneath the rock rendering it unsuitable for the anaerobic community. Such habitats are found in Australia and have interesting but largely unworked unknown faunas (WFP pers. observ.). Some species recently found in the Seto Inland Sea of Japan are, as far as is known, restricted to these anaerobic habitats under rocks in one small system of intertidal pools only a few metres in extent (Fukuda et al. 2000). Similarly, a suite of mollusc species that was once common in similar habitats in the extensive embayment of the Lake Worth area of in Florida (including Palm Beach), has been severely restricted by shore modification through "development" and is now apparently found only on a single small island that is itself threatened with development (WFP pers. observ.).

Anchihaline caves

Anchihaline caves occur on the coast in many parts of the world. They are distinguished by having a mixture of freshwater and seawater and often contain highly endemic faunas.

Fauna
Strange and highly endemic faunas (mostly crustaceans) are being discovered in these habitats in various parts of the world (e.g., Vonk and Stock 1987; Huys 1988; Sket 1988; Stock and Iliffe 1990; Stock and Vonk 1990; Jaume and Garcia 1993; Jaume 1995; Jaume and Boxshall 1995; Huys 1996; Jaume and Boxshall 1996; Sket 1996; Jaume and Boxshall 1997; Jaume et al. 1998; Ohtsuka et al. 1999). Dr W. Humphries' recent work in habitats of this type in NW Australia has discovered a very unusal and unique crustacean fauna (e.g.,Humphries 1999) but the fauna of these habitats is unknown for the rest of Australia.

Biogenic substrata

Many colonial invertebrates, including bryozoans, sponges, soft and hard corals, oysters, mussels, ascidians, tube worms, etc., are habitat-forming and provide food and shelter for numerous other invertebrate species. Some of these associated species have symbiotic or commensal relationships while others simply seek shelter or protection. While all of these habitats are very important, most form a component of other systems (rocky shores, subtidal reefs etc.). It is only with coral reefs that invertebrates have become the dominant feature of the environment; hence, only coral reefs are given further treatment here.

Biogenic substrata in the deep-sea, such as wood, seagrass stems, cephalopod beaks, whale bones, elasmobranch egg cases etc., provide habitat for a small but highly specialised invertebrate fauna (certain crustaceans, limpets, etc.) typically restricted to just one of these types of substrata.

Coral reefs

Coral reefs are widely recognised as one of the most diverse ecosystems on earth, perhaps paralleling the position of the rainforests in the terrestrial environment (Knowlton 2001). Coral reefs are the largest biological structures in the sea, growing as huge, "wave-resistant piles of calcareous skeletal debris, the accumulated deposits of a thin veneer of living organisms" (Jackson 1991). Structurally, they are composed primarily of scleractinian corals and coralline algae, but support a huge diversity of associated fishes and invertebrates. Coral reefs have been extensively reviewed by Wells (1988), Dubinsky (1990), Veron (1995b; Veron 1999), Wallace (1999a), Birkeland (1997), and others. The status of, and threats to, coral reefs worldwide in 1998 was reviewed by Wilkinson (1998; 1999).

Types of reef

Coral reefs vary enormously in their pattern of growth and mode of formation, and various schemes have been devised to classify this variation. Categories include *fringing reefs* (those lying immediately adjacent to volcanic islands or continents), the generally elongate, narrow *linear, ribbon* or *wall reefs* (aligned along the edge of the continental shelf), and oval *platform reefs* (scattered on the continental shelf). *Coral atolls* are scattered throughout the Indo-West Pacific region, and are formed as a result of the growth of reef-building corals upon submerged volcanic peaks. Within each reef, there are a variety of zones, e.g., *reef slope*, *reef crest* and *reef flat*, a *lagoon* which is often absent from small reefs, and an *intertidal shore* if an island is present (Catterall 1998).

Distribution and coral diversity

At least 50% of the world's coral reefs occur in the central Indo-Pacific (Veron 1986, 1995a; Veron 1999). Reefs in this area have about 600 coral species, compared with fewer than 50 in the tropical eastern Pacific and eastern Atlantic provinces, and perhaps 100 species in the tropical western Atlantic (Jackson 1991; Veron 1995a; Veron 1995b; Veron 1999; Bellwood and Hughes 2001). While many coral species are widespread in the Indo-Pacific, Knowlton and Jackson (1994) point out that new taxonomic approaches are showing the existence of sibling species; therefore some taxa may not be as widespread or display such wide ecological tolerances as previously assumed.

In Australia, coral reefs occur off the eastern, northern and western coastlines. The most extensive and best-known system is the Great Barrier Reef (GBR), which stretches for more than 2000 km along the east coast. It includes more than 24,000 individual reefs that collectively occupy about 9% of the continental shelf waters in this area (Talbot and Steene 1984). Veron (1995a) recognised seven distinctive groups of reef in Australia: high latitude eastern Australia; GBR; Coral Sea; northern Australia; Cocos Atoll and Christmas Is; Northwest shelf; and coastal WA, each with a distinctive coral fauna. For example, the GBR is the most speciose, with fewer species found at the higher latitude reefs of the Solitaries and the Abrolhos, although these do include some endemics. In other places, such as Rottnest Island, there are a few species but these do not form reefs and, in some of the northern areas (e.g., Darwin Harbour), there are species that can tolerate high sediment loads (Wolstenholme et al. in press).

Associated fauna

Australia's coral reefs are a centre of high diversity for many marine organisms, including fishes, molluscs (e.g., Wells 1990; Ponder and Loch 1997; Catterall 1998), echinoderms (Rowe and Gates 1995), polyclad flatworms (Newman and Cannon 1994), polychaetes (no published figures; Grassle 1973; PAH pers. obs.), soft corals (Fabricius and Alderslade 2001; Fabricius and De'ath 2001) and crustaceans. This diversity is due in part to the great variety of microhabitats that they offer, including both hard and soft substrates. The invertebrate fauna is very heterogeneous in species composition, such that very small differences in geomorphology of the reefs typically translate to quite large differences in species composition (Hooper et al. in press; J. Hooper pers. comm.).

Microhabitats available to the myriad invertebrates that live associated with reefs comprise different combinations of sand, coral rock, algae and live coral substrata. Live corals support relatively few other invertebrates because the nematocysts in the tentacles of the living coral polyps enable them to feed on molluscan larvae and repel potential predators. Nonetheless, some animals feed on live hard and soft corals (the Crown-of-Thorns starfish (*Acanthaster plancii*) being a famous example) or live inside the living coral skeletons. Others live in association with live coral and may take advantage of the protection offered from predators. Areas of dead coral within or adjacent to reefs often support large numbers of invertebrates.

In terms of biomass and primary productivity, sponges are the second-largest component of coral reefs. However, most species are cryptic or ultra-cryptic, and there are many new species (J. Hooper pers. comm.). Most of the species in several mollusc families occur in coral reefs of the Indo-west Pacific region and contribute to the characteristic nature of these reefs. Characteristic molluscs include the gastropod families Strombidae, Cypraeidae and Conidae and the bivalve family Tridacnidae. Many other frequently encountered families also share a significant number of species with other environments (Catterall 1998).

Coral reefs are dynamic systems characterised by a delicate balance between reef growth and reef destruction, the latter primarily caused by bioerosion of coral substrate by grazing and boring organisms (Hutchings 1986; Pari et al. 1998). Excessive bioerosion weakens the structure of the reef framework and makes it more susceptible to damage by cyclones and storms (Bythell et al. 2000), El Niño events (Tudhope et al. 2001), and by coral predators such as the Crown-of-Thorns starfish (see Section 6.8.2). Experimental studies on rates of bioerosion in the Indo-Pacific have been undertaken on the GBR (One Tree Island – Kiene 1985; and Britomart Reef in the central GBR – Sammarco et al. 1987; Sammarco and Risk 1990; Lizard Island – Hutchings et al. 1992; Kiene and Hutchings 1994a, 1994b) and in French Polynesia (Pari et al. 1998; Pari et al. in press).

Information and Research
Over the past couple of decades, coral reefs have become extremely popular with divers and researchers, because of their ecological and economic importance and the range of threats they have faced (see below). The GBR is one of the better-studied reef systems in the world with long-term monitoring (of coral cover, Crown-of-Thorns, etc.) having been carried out since 1985 (Sweatman et al. 1998 and AIMS website- provides up to date information re the Crown-of-Thorns starfish), and there are now several permanent research stations. However, most of the large amount of research on coral reefs – worldwide as well as in Australia – has concentrated on a relatively small number of reefs and mainly on accessible, emergent reefs. There has been little research on non-depositional coral communities and subsurface shelf reefs, pinnacle reefs and deep reefs (Eakin et al. 1997). In addition, "most long-term data on non-commercial species seems to be restricted to a few 'fashionable' organisms such as corals, reef fish and Crown-of-Thorns starfish" (C. Wolff pers. comm.). Other reef invertebrates, as well as inter-reefal habitats (which cover a far greater area than actual reefs and are very important ecologically) have been far less studied. The inter-reefal habitats were included in the proposed Representative Areas Program for the GBR, where they were classified into numerous sub-habitats. We hope that representatives of each of these will be protected.

Conservation issues
The status of coral reefs, threats to them, and their management, have been discussed at length by many reviewers (e.g., Hatcher et al. 1989; Grigg and Dollar 1990; Hughes 1993; Sebens 1994; Veron 1995a; Ormond and Douglas 1996; Eakin et al. 1997; Wilkinson 1998; Hughes and Connell 1999; Risk 1999; Wilkinson 1999; Goreau et al. 2000; Koop et al. 2001). These threats range from natural (cyclones, rising temperature, changes in salinity, black band and white band diseases, predation by echinoderms, molluscs and fishes, and competition from other invertebrates) to anthropogenic (pollution, dredging, mining, boating and tourism, sedimentation from cleared catchments, overfishing, physical removal, construction of tourist complexes, etc., and coral bleaching). Many of these are discussed under the appropriate headings in Chapter 6 (Threatening Processes).

It is generally recognised (see above references and Chapter 6) that anthropogenic disturbances are at the centre of the recent declines seen in coral reefs worldwide. Human populations mostly live in coastal areas and the associated urbanisation, agricultural activity and the loss of important coastal habitats (e.g., forests, coastal wetlands) have increasingly threatened coral reefs and associated marine systems. The influences of natural stresses on reef systems are less well understood than the direct human impacts (Eakin et al. 1997). In particular, the dramatic effects of the 1982-83 El Niño in the eastern Pacific and recent coral bleaching throughout tropical areas have raised concern over the impacts of climate change on corals (e.g., Hoegh-Guldberg 1999; Reaser et al. 2000; see Section 6.7.3).

The status of coral reefs around the world has been reviewed on a number of occasions, including by Jameson et al. (1995), Eakin et al. (1997) and Wilkinson (1998). The first of these, the State of the Reefs: Regional and Global Perspectives was a report produced for the International Coral Reef Initiative[92], and evaluated environments, threats and opportunities for improved management. While there is general agreement amongst coral reef research workers that coral reefs around the world are deteriorating and declining, it is currently difficult to determine in detail the extent of threats, rates of decline and the consequences in terms of global resources and impacts on dependent communities[93]. Habitat size correlates well with species richness of corals (Knowlton 2001) so reduction in size of world reefs could result in reduced diversity. Other factors determining coral diversity are also important, including productivity, disturbance, biotic interactions and regional and/or historical effects (Cornell and Karlson 2000).

In Australia, the status of the Great Barrier Reef – including the extent of coral bleaching and crown-of-thorns outbreaks – is monitored by GBRMPA and AIMS and was the subject of a recent State of the Reef report (Wachenfeld et al. 1998). Since then, massive bleaching of inshore reefs has occurred on the GBR during 2001/2002[94]. By world standards, most parts of the Reef are in good condition although subject to localised disturbance (for instance, some inshore areas have been badly affected by poor water quality and sedimentation resulting from catchment development). Oceanic atoll reefs near the continental shelf edge of northwest Australia have been subjected to traditional harvesting by Indonesians and, more recently, commercial exploitation (fisheries and tourism). The region is also currently being subjected to oil and gas exploration. These remote reefs are well isolated from typical terrigenous influences, but are likely to be subject to increasing human activities. Long-term baseline monitoring of benthos and reef fish is being carried out at three of these reefs (North Scott, South Scott and Seringapatam Reefs) (Heyward et al. 1997a; Heyward et al. 1997b).

While information and monitoring programs are essential in obtaining data on the status of coral reefs, little progress will be made in their conservation without commitment by all levels of government to reduce major impacts (including reduction of greenhouse emissions).

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[92] The International Coral Reef Initiative began in 1994 with an international partnership of eight countries (including Australia), three UN organisations, inter-governmental organisations, multilateral development banks and NGOs. This program has sponsored various workshops and a program where the status of individual reefs is being continually monitored (Wilkinson 1998).

[93] For example, largely human-induced changes on fringing reefs in Moorea resulted in half the mollusc species disappearing over a 23 year period while numbers on the largely unaffected barrier reefs remained stable (Augustin et al. 1999).

[94] http://www.gbrmpa.gov.au/

Assessments of reef status are usually made using indices such as the percentage of live coral cover or coral mortality, but Edinger and Risk (2000) suggested that a system based on coral morphology could be better at predicting areas of "conservation value"[95]. This method involves assigning "conservation classes" (CCs) of 1, 2, 3 or 4 to reef sites dominated by massive and submassive corals (CC 1), foliose or branching non-*Acropora* corals (CC 2), *Acropora* corals (CC 3), or approximately equal mixes of these three (CC 4). Edinger and Risk (2000) applied this method to 15 Indonesian coral reefs and found that it was a reliable predictor of coral species richness, habitat complexity, and the occurrence of rare coral species.

5.4 Other considerations in the system approach to conservation

5.4.1 The importance of scale

Selection of the appropriate spatial and temporal scale for conservation, experimentation and so forth is a very important issue and has generated much literature. Constable (1999, p. 464) noted that ecological research is usually undertaken at smaller spatial and temporal scales than those required by the problems being tackled. Different ecological processes operate at different scales and some are independent of scale (Aronson 1998).

More on scale

The most appropriate scale depends very much upon the issues being examined and the organisms involved; for instance, the question of "how big is big enough?" for Marine Protected Areas (MPAs) will very much depend on what habitats (e.g., seamount vs mangroves) or organisms (e.g., polychaetes vs whales) are being targeted for protection.

The identification of scale- and habitat-dependent ecological patterns is central to management efforts aimed at predicting the response of organisms to the increasing threat of habitat fragmentation (Eggleston et al. 1999). In an experiment using artificial habitat plots of different types and sizes, Eggleston et al. (1999) showed that invertebrate macrofauna responded to habitat patchiness in a complex manner that varied according to habitat type, experimental site, species, taxon, functional group and animal body size. They found a disproportionate reduction in faunal diversity (for small species) in small versus large patches.

Spatial (especially topographical) heterogeneity (see 5.4.2) can influence community structure through a variety of processes (see below) but the *observational scale* can also influence perceptions of community patterns and processes. Blanchard and Bourget (1999) studied the effects of three scales of spatial heterogeneity on benthic community species richness, diversity, total biomass and biomass of dominant taxa. Large-scale heterogeneity (looking at a 1: 20 000 map) had no influence; medium-scale heterogeneity (linear coasts and headlands) had only a minor influence, while small-scale heterogeneity had a major influence. Similarly, Thrush (1991) indicated that spatial scales of field

[95] It should be noted that "conservation value" is a human construct that depends entirely on where we choose to place value. The concept has no objective or ecological meaning.

sampling or experimentation are important influences on data interpretation. This is particularly true for soft sediments, which provide a three-dimensional habitat for macrobenthic organisms, although the analysis of spatial patterns of organisms living in soft sediments is difficult due to much of the fauna being hidden (buried) and the need to disrupt the habitat to assess patterns.

5.4.2 Structural complexity and spatial heterogeneity

Structural complexity and *spatial heterogeneity* in the environment are both important considerations in both conservation initiatives and ecological studies. Habitat structural complexity can facilitate the provision of refuges, as well as offering a much greater diversity of food. Spatial heterogeneity adds habitat diversity and this has important implications for invertebrates. The importance of structural complexity and spatial heterogeneity in the environment, and the consequences for the dynamics and diversity of populations and assemblages has been recognised for many years (reviews in Pickett and White 1985; Kolasa and Pickett 1991; Thrush 1991; Morrisey et al. 1993; Constable 1999; Beck 2000; Hovel and Lipcius 2001; Hooper et al. in press). Some issues relating to temporal variation are discussed in Section 7.7.1.

Structural complexity can vary considerably depending on the biotic or abiotic factors providing the complexity. Corals, sponges or other large emergent colonial animals and algae or marine plants are the usual biotic contributors to structural complexity. This may be at many different scales (Ritchie and Olff 1999). Complex habitats providing shelter and food for minute animals may appear essentially uniform to larger organisms.

Spatial heterogeneity may be pronounced or subtle. For instance, Constable (1999) noted that even in soft substrates, which appear to the casual observer to be rather uniform habitats, the existence of heterogeneity (patches) is important – as it is in rocky habitats – and has important consequences for population dynamics (see also Pickett and White 1985; Kolasa and Pickett 1991; Thrush 1991; Morrisey et al. 1993 for reviews). Kendall and Widdicombe (1999), for example, investigated small-scale patterns in the structure of macrofaunal assemblages of shallow soft sediments in shallow water and found that samples separated by more than 50 m could be significantly different from each other, largely due to changes in the pattern of dominance among the most abundant species (particularly polychaetes). Similarly, Hutchings and Jacoby (1994) found a considerable amount of heterogeneity in the benthic infaunal communities of Jervis Bay. Interactions between algae and animals (e.g., fishes, urchins) assist in maintaining the patchiness on temperate subtidal reefs (Jones and Andrew 1990).

Disturbance can be viewed as a force that structures communities (Skilleter 1995 see also for overview) - for example in the deep-sea (e.g., Dayton and Hessler 1972); intertidal hard shores (e.g., Dayton 1971) and soft-sediments (e.g., Hall and Harding 1997; Cowie et al. 2000). Disturbances can be natural or anthropogenic and much more common in some systems than others. For example, a long-term study on the temporal patterns in hard and soft coral assemblages and macroalgae on the GBR from 1992-1997 (Ninio et al. 2000) found that trends in coverage of these assemblages were usually consistent among clusters of adjacent or nearby reefs. This consistency probably reflected the spatial scales of episodic disturbances (e.g., Crown-of-Thorns (Section 6.8.2), or cyclones (e.g., Bythell et al. 2000)).

Implications for conservation

Different marine taxa appear to exhibit different gradients in biodiversity. Development of a consensus model, if possible, is vital to the design of effective conservation strategies and marine protected areas that protect/ contain truly representative biodiversity. This is in contrast to models based on umbrella taxa, such as large charismatic species, often used inappropriately as surrogates for real species diversity (Mace et al. 2000). Even studies of major groups may not serve as surrogates (Kerr 1997 and references therein). There is a need for a diverse range of taxa to be considered in determining priority areas for conservation.. For example, recent studies on the biodiversity of sponges (Hooper et al. in press) has shown that there are recognisable "biodiversity hotspots" for these animals. However, a conservation strategy designed to protect these areas may not be adequate for other groups.

5.4.3 Linkages between systems

There is strong interconnectivity between marine ecosystems, all being linked through the water column (which allows exchange of larvae, nutrients, organic material, pollutants and other materials) and through boundaries between adjacent, interdependent systems. In turn, marine systems interact with coastal terrestrial and freshwater systems.

Conservation implications

Exchanges of water and biota occur between onshore and offshore areas and along coasts (Fairweather and Quinn 1995). For example, the larvae of marine species can be transported from one system to another, which can be a key element in maintaining ‘downstream’ systems (including those in marine parks), and in allowing re-colonisation after natural or human induced disturbance. Mangrove-lined shores prevent coastal erosion and consequent silting of adjacent ecosystems. They also serve as nurseries for some juveniles of marine species that live in other habitats as adults. Coral reefs protect shores from violent wave action. Runoff or windblown dust from terrestrial environments provide nutrients, energy and sediment but can also degrade marine systems. Droughts (e.g., Attrill and Power 2000) can greatly impact estuarine systems and the animals in them by causing massive fluctuations in salinity. Thus, damage to one part of the system (marine or non-marine) may affect other parts.

These linkages mean that the conservation of marine habitat may be ineffective if surrounding areas continue to be degraded. For instance, the protection of insular habitats like coral reefs may be ineffective if related habitats like seagrass meadows and mangroves are left unprotected and if terrestrial catchments are not properly managed (Hutchings and Haynes 2000 and references therein). Robertson and Alongi (1995) described how the seagrass meadows on which juvenile tiger prawns depend often occur immediately seaward of mangrove forests in the wet tropics of NE Australia. Thus, removal of or damage to estuarine mangroves is likely to result in increased sediment smothering the seagrass. Since many marine species are not confined to one locality but utilise various habitat types throughout their life, conservation strategies must also consider protecting areas with heterogeneous habitats likely to meet the changing habitat requirements in complex life cycles.

The importance of linkages between marine ecosystems is clearly illustrated by disasters like the *Exxon Valdez* oil spill (e.g., Paine et al. 1996; Flaherty 1999), where oil-covered beaches and dead animals were found up to 640 km from the site of the spill.

5.4.4 Conserving the ecological function of marine systems

Most approaches to system conservation ignore ecosystem function, treating the system merely as a collection of individuals or taxa present at a particular point in space or time. However, the diversity of living things is intimately related to the function and stability of communities and ecosystems (e.g.,Loreau 2000) because plants and animals do not operate in nature as single organisms (Hawkesworth and Mound 1991).

Interactions and ecosystem function

A "system" can be defined as including the complex network of interactions, dependencies and inter-dependencies that link individual organisms. The existence of each species depends on a variety of other species, and fundamentally on micro-organisms as the basis of food pyramids (Price 1988). Systems are thus rarely discrete or self-contained units, being part of a larger continuum and reliant on interactions with, and/or inputs from, other systems (see further discussion below). As such, it would seem critical that the integrity of ecosystem function be maintained by conserving vital ecosystem processes, and indeed Walker (1992) argued that this is the only way to avoid or minimise species loss long-term. However, the ecosystem function of biodiversity is rarely addressed, although it has concerned the International Council of Scientific Unions' (ICSU) Scientific Committee on Problems of the Environment (di Castri and Younès 1990). It is, of course, difficult to investigate – more so in the ocean than in (at least some) terrestrial ecosystems.

In general terms, the importance of species interactions, including predator-prey relationships, commensal and symbiotic interactions, should be given greater consideration during all conservation decisions. Particular actions, for example fishing targeting a keystone predator or herbivore, may have cascading effects on the composition, health and function of the community (e.g., Lindberg et al. 1998; Steneck 1998; Walters et al. 1999; Pinnegar et al. 2000). Such effects can be mitigated by (for example) taking a precautionary approach, such as that taken to the harvesting of krill (on which many other species depend). However, such approaches are not commonly attempted.

The recognition of the need to manage the complexities of the ecosystem in developments and resource utilisation has lead to the concept of *ecological engineers*[96].

[96] This concept has ecological principals encompassed within engineering practices (e.g., Schulze 1996).

Biodiversity and ecosystem function in marine invertebrate communities

As discussed earlier (Chapter 1), invertebrates are key elements in all marine systems and have a wide variety of functional and trophic roles; as such their conservation is essential to ensure the functioning of the ecosystem as a whole, whether or not the invertebrates themselves are the target of conservation efforts.

Marine invertebrates, like other animals, form a wide range of relationships including as predator or prey, as hosts or their commensals (e.g., Morton 1988), symbionts or parasites, or simply by providing shelter. Neighbouring organisms need not always be competitors for space. A number of sessile organisms form "defence associations" in which undefended species derive benefit from their close proximity to a defended neighbour - a famous example being anemone fish. Many invertebrate larvae select surfaces at settlement likely to enhance their subsequent survival. Davis (1996) describes an association among subtidal ascidians where recruitment of one species is facilitated by the other.

If biodiversity represents a form of "biological insurance" against the loss or poor performance of particular species then communities with larger numbers of species should be more stable (or at least predictable), although there is a counter argument that the existence of a larger number of linkages may make it more, rather than less, difficult to predict changes. The acceptance of this hypothesis has important ecological, management and economic implications given how many diverse natural ecosystems have been replaced by, or degraded to, much less diverse systems. The idea that biodiversity provides increased ecosystem "reliability" has been experimentally tested by Naeem and Li (1997; Naeem and Li 1998) using microbial microcosms with varying number of species per functional group to see whether larger numbers of species provide a more constant level of performance over a given unit of time than smaller numbers of species. Their results suggested that "redundancy (in the sense of having multiple species per functional group) is a valuable commodity, and that the provision of adequate redundancy may be one reason for preserving biodiversity". However, the adequacy of their experimental design has been challenged by Wardle (1998).

The large number of fundamentally different life forms may confer high ecological complexity to marine and coastal ecosystems (Ray 1991). However, it remains to be seen to what extent biological diversity relates to ecological complexity. In particular, comparisons among different systems can be made difficult by each individual system's history ("the larger temporal and spatial scales within which the community is imbedded"). Furthermore, a prediction of functional change within an ecosystem after reduction in species diversity depends on knowledge of how that system's diversity was generated and is maintained (Ray 1991). While there are few studies addressing this issue in natural marine systems, Eakin et al. (1997) suggested that coral reefs with low diversity are subject to more dramatic changes than high-diversity reefs.

While probably theoretically sound, focusing conservation efforts towards the maintenance of ecological processes has considerable practical difficulties. For example, it is difficult to know, in the absence of better data, how to actually ensure that ecological processes are maintained, except indirectly (by conserving as much biodiversity as possible, through protection of entire assemblages and habitats as well as individual species, and minimisation of deleterious impacts). One approach recognises species redundancy (see Section 3.2.1 and below) and focuses on species or functional groups (guilds) that play a particularly vital role in their community – the so-called "keystone taxa" although, again, there has been considerable debate over whether such taxa can be so easily defined.

The notions of species redundancy, keystone taxa and guilds (functional groups)
A profound problem for conservation is that there is very little information about the relationship between species diversity and ecological function, although it is generally considered that the redundancy in diverse systems can be important in maintaining function in the face of disturbances. Much of the available evidence suggests that species-rich systems "perform" better than those less diverse (Snelgrove et al. 1997; Duarte 2000; Loreau 2000; Petchey 2000; Snelgrove et al. 2000; Hughes and Petchey 2001; Austen et al. 2002). For instance, Schläpfer et al. (1999) surveyed scientists as to their opinion on the functional role of biodiversity in ecosystems and the material services provided by ecosystems. They generally responded that (1) ecosystem process rates are strongly correlated with biological diversity, and (2) these same processes are important for the delivery of human defined "ecosystem services" by natural systems. These authors also found that "due to the presence of better performing individual species or complementary resource use of functionally different species", species-rich experimental ecosystems have outperformed less diverse systems in about half of nearly 100 reported field and laboratory trials (Schläpfer et al. 1999). However, this is still a very controversial issue that continues to generate much debate. What is certain is that we still do not know in detail how the biosphere as a whole operates or even understand the functioning of localised ecosystems, let alone the role of individual species in ecosystem function (Schulze and Mooney 1994). This makes it difficult, if not impossible, to predict the consequences of loss of biodiversity for ecosystem integrity and function. Models (e.g., Loreau 2000; Petchey 2000) show that communities that are more diverse have more resilience than those that are species-poor. As an illustration of the uncertainties that exist, Ray (1996) gave five case studies involving discontinuities (where ecosystems absorb stress over a long period of time and then change) and synergisms (unpredictable consequences of two or more factors operating together).

Species redundancy**:** The concept of species redundancy arose from the idea that the functional role of some species is duplicated by others so their loss in terms of ecosystem function is not important. Arguments about the extent of "species redundancy" in communities and ecosystems are discussed by Lawton and Brown (1994). The questions they address are:

- How much species redundancy is built into ecological processes?
- To what extent are patterns of biological diversity important in determining the behaviour of ecological systems?

Lawton and Brown (1994) argue that there are at least two extreme possibilities: the first is that all species contribute to the integrity of the biosphere in a small but significant way, and there are limits to how many can be lost before the system collapses[97]. The alternative view is that species richness is irrelevant; so long as the biomass of primary producers, consumers, decomposers, etc. is maintained, and that these ecological processes will continue to function with very few species.

The fossil record suggests that most of the ecological systems over the last several hundred million years have been much less species-rich than extant assemblages, both in the sea (Sepkoski et al. 1981) and on land. These assemblages suffered occasional mass extinctions, the largest of which, in the late Permian, may have eliminated 95-96% of species, 78-84% of genera and 54% of families (Erwin 1989). While there is no evidence suggesting that the major life-support systems of the planet worked in substantially different ways due to these extinctions, the current rates of extinction are several orders of magnitude faster than anything that occurred in the fossil record.

Major patterns of energy flow and biomass in extant ecological assemblages seem to be broadly insensitive to numbers of species in terrestrial systems – for instance, many highly productive systems, such as saltmarshes and agricultural tropical grasslands, are not particularly rich in species. Broadly similar conclusions appear valid for freshwater and marine ecosystems (e.g., Genin 1986; Sprules and Munawar 1986). Data on food webs shows that they share a common structure, irrespective of the number of species in the web, in terms of the average proportion of top, intermediate and basal species, the ratio of predator to prey species, the proportions of trophic linkages between different levels, the number of trophic levels, etc. (Pimm et al. 1991). On the other hand, very recent work suggests that the apparent similarity of function at different scales (so-called “scale invariance”) may be an artefact of poor data, particularly the failure to study and document the structure of very large food webs with sufficient rigour (Lawton and Brown 1994).

Walker (1992) argued that the best way to minimize species loss is to maintain the integrity of ecosystem function and that it is therefore important to identify the taxa significant to ecosystem functioning. This involves establishing the extent of species redundancy in the biological composition of ecosystems. Walker (1992) suggested an approach based on the use of functional groups of organisms defined according to ecosystem processes. Berlow (1999) argued that the loss or removal of individual species could cause dramatic changes in communities. He points out that experiments show that in many communities only a few species will have such strong effects, whereas most will have weak effects (due to small *per capita* effects and/or low abundance). Berlow argued that extinction of these “weak interactors" could significantly alter natural communities because they play important stabilizing roles. He demonstrated that some “weak interactors" may also be important by magnifying spatio-temporal variation in community structure, and that although their effects may seem weak when averaged over broad scales, they could be strong in more local contexts. These and other results

[97] This latter view is Erlich & Erlich's (1981) ‘rivet hypothesis’, an analogy drawn with the loss of rivets from a plane.

challenge the assumption that research and management should focus solely on species that exhibit strong mean impacts on community structure (such as keystone species).

Keystone taxa: The concept of "keystone species" was first proposed by Paine (1966; Paine 1969), who used the term to describe predators in marine communities whose activity and abundance determined "the integrity of the community and its unaltered persistence through time, that is, stability" (Paine 1969). Keystone species have since been described as those whose effect is disproportionally large relative to their abundance (Power et al. 1996) or those that have "a disproportionate effect on the persistence of all other species (Bond 1994). Experimental removal of a keystone species should result in the loss of some species and their replacement by others. Community changes may be due both to direct effects of the keystone species on other species, for example its prey, or to indirect effects such as the competitive exclusion of weak competitors by a species usually kept in check by its predator.

The keystone species hypothesis was reviewed by Mills et al. (1993), who discussed its importance in conservation and the difficulties in using the concept. Marine examples of keystone predators include fishes, large crustaceans, starfish (Paine et al. 1985), gastropods, and humans (e.g., Lindberg et al. 1998; Castilla 1999). The major practical difficulty is the lack of precise information (especially given the almost complete lack of biological information about the majority of marine invertebrates). The adoption of this approach would have to be based on extremely superficial information for a tiny proportion of the species in any given system given our current state of knowledge of marine ecosystems. There are other difficulties with defining keystone species *a priori* (Lawton and Brown 1994), and the fact that a species is keystone does not necessarily mean that it is vulnerable. Sometimes small, unlikely looking species, not only the dominant ones, can have massive effects on the ecosystem and we have no idea of what proportion of species in any ecosystem are keystone species (Lawton and Brown 1994). In addition, it is not even clear by what mechanism keystone species might be expected to exert their effects. In some cases it is trophic (e.g., by eating other potentially dominant members of the community) but in others it may be by nutrient cycling or ecosystem "engineering". The main message in the keystone concept is probably that all species are not equal (Lawton and Brown 1994).

Guilds – functional groups: The concept of a functional group[98] is similar to that of a guild[99], but in a functional group the emphasis is on the end result of the group's actions. Although functional groups are an important aspect of community ecology, and although using this concept allows many taxa to be considered at once, its use may obscure individual community members' responses to ecosystem structure (and changes to that structure). It is a trade-off between considering individuals or a collective.

5.5 Information needs and management options for systems conservation

[98] A functional group is a group of species that perform a similar function.
[99] A group of species that use a similar resource in a similar way.

Information needs for systems conservation were discussed by Eakin et al. (1997), and although they focused specifically on coral reefs, many of their points are just as relevant – if not more so – for other marine systems (which on the whole are less well known than the relatively well-studied coral reefs). These authors stressed that information is still needed at global, regional and national levels in a variety of areas, including:

- The extent, distributions and variability of these systems;
- The degree and nature of human and natural stresses and impacts;
- The degree and nature of human dependence on these systems;
- The expected values for indicators to assess system health;
- The expected response of each type of system to increasing levels of stress; and
- The degree to which the health of these systems can be accurately assessed through *in situ* and remotely sensed monitoring approaches.

As with many aspects of conservation, there are insufficient data to apply some of the conservation approaches that appear to be sound logically or theoretically. Concepts such as keystone species, ecological services and definitions of communities, habitats, etc. may have a reasonable theoretical basis, but in practical terms, are not useable in most conservation decision-making due to the complexities in natural systems and the lack of detailed data. However, modern ecological studies have come a long way towards dealing with many of these issues so, over time, much useful information is likely to become available.

A *site-based approach* to conservation may help circumvent this problem if it is coupled with a reasoned strategy to include examples of major habitat types and at a scale appropriate to allow for invertebrate species turnover. Another useful approach is *integrated system management*.

Site-based approach

Particular *sites* may be conserved through the declaration of Marine Protected Areas (MPAs) (see below, Section 5.5.3), while particularly sensitive *habitat types* (e.g., mangroves, seagrass; wetlands under RAMSAR) may be protected if they are listed under relevant legislation, with approval required for any proposal to damage or clear that habitat in any location. In addition, activities that cause generic damage (e.g., land clearing, trawling) can be regulated or mitigated through a variety of approaches.

Integrated system management

Given that wholesale protection is impossible, an integrated approach to management of whole systems (e.g., catchment management) is generally the preferred option. As an example, the Department of Natural Resources and Environment (1997) see the priority management responses for estuaries as to:

- Restore the health of inlets and estuaries through improved catchment management;
- Prevent the establishment and control the presence of noxious marine species;
- Reduce theft and illegal fishing methods through education and enforcement;
- Increase understanding, protection and monitoring of vulnerable habitats, particularly seagrass, mangroves and saltmarsh;
- Promote ecologically sensitive tourism;

- Ensure ecologically sustainable harvesting and management of fisheries resources;
- Plan for oil spill contingencies; and
- Minimise industrial waste and progressively improve sewage treatment.

5.5.1 Listing of threatened communities and habitats

This approach allows communities rather than single species to be listed as threatened. Problems include the difficulties in defining the "community" and incorporating the likelihood of ecological change. Commonly the community must be defined largely in terms of the place it occupies and thus becomes analogous to site-based conservation.

Threatened habitats can be declared under fisheries legislation, planning controls, etc. This approach differs from site-based conservation in that a type of habitat or community is listed as protected regardless of where it may occur.

Commonwealth legislation

Threatened communities could be listed under the Commonwealth's Endangered Species Protection Act 1992 and this was seen as a mechanism for maintaining ecological processes and safeguarding elements of the biota that have yet to be recognised taxonomically or as threatened. Despite this acknowledgement that conservation actions might be delivered by protection aimed at entire ecosystems, rather than individual species, only one (terrestrial) community was listed by the Commonwealth (Woinarski and Fisher 1999) under that legislation, partly because of jurisdictional problems. The *Environment Protection and Biodiversity Conservation Act*[100] (see also Chapter 8) enacted on 1 July 2000 has overcome some of these problems and, while several threatened communities have already been listed under the Act, none are marine.

State legislation

The details of the appropriate state legislation are given in Chapter 8. The only Australian marine community that has been listed is in Victoria (O'Hara 1995) where the rarity and vulnerability of diverse reef and seagrass flats off San Remo have led to their being listed on Schedule 2 of the Flora and Fauna Guarantee Act as the "San Remo Marine Community". This listing was initiated in response to a proposal to develop a large marina on the site.

Threatened habitats can be declared under some state legislation (see Chapter 8). In NSW, for example, seagrasses and mangroves are protected under the Fisheries Management Act (see Section 8.2.3). This is implemented through banning of certain practices, requirement to seek approval to destroy areas of these habitats, and implementation of programs to map areas of occurrence, etc.

5.5.2 Habitat management and remediation

The remediation of habitats damaged or destroyed is another potential management option, albeit an expensive and inefficient one, and one which is generally unlikely to be

[100] http://www.ea.gov.au/epbc/index.html

used except as a last resort. It would normally only be undertaken in situations where there were major benefits to be gained. A major consideration is whether the area in question is failing to recruit naturally, because, in most instances, natural recovery processes are likely to be sufficient in the medium to long term.

Apart from modification of the physical environment, such as in beach restoration following erosion, examples of marine biotic systems that have undergone some form of remediation include seagrasses (Bell et al. 1997), mangroves (Robertson and Alongi 1995) and coral reefs (Rinkevich 1995; Edwards and Clark 1998).

***Coral reef restoration*:** Edwards and Clark (1998), for instance, discussed whether coral transplantation is “a useful management tool or misguided meddling”. Transplantation of corals has been carried out to:

- Accelerate reef recovery after ship groundings, crown-of-thorns starfish or red tides, dynamite fishing or coral quarrying;
- Replace corals killed by pollutants;
- Save coral communities or locally rare species threatened by pollution, land reclamation or pier construction; and
- Generally enhance the attractiveness of underwater habitat.

This practice – which is effectively a form of ‘gardening’ of coral reefs – involves the ‘planting’ of the affected reef with asexual recruits (coral branches, colony fragments, and whole small colonies) and/or sexual recruits (laboratory or *in situ* settled planula-larvae) (Rinkevich 1995, 2000). The latter is more difficult. Whether such transplantation of corals (or any other marine organism) is likely to be effective depends on many factors, including water quality, exposure, substrate, and the other species present.

5.5.3 Site-based conservation – marine protected areas (MPAs)

Background

On land, the declaration of various categories of protected areas (nature reserves, national parks etc.) has had a long history with the first terrestrial national parks declared well over a century ago (e.g., Yellowstone National Park (US), established 1872; Royal National Park (NSW), gazetted 1879). In contrast, recognition of the need to establish protected areas to conserve biodiversity in the marine environment has developed much more recently. The need to manage and protect marine environments and their resources only really gained momentum during the 1950s and early 1960s (Kelleher and Kenchington 1992; Parker 1995; Kelleher 1999). In 1987 the IUCN, at the 4th World Wilderness Congress, passed a resolution establishing a policy framework for marine conservation, and a similar resolution was passed by the 17th General Assembly of the IUCN in February 1988 (see Parker 1995). These actions were followed by the development of international support for the establishment of marine and estuarine protected areas (MEPAs) (Kriwoken and Haward 1991). The last couple of decades have seen a rapid increase in the number of MEPAs declared worldwide. For instance, in 1970 there were 118 MEPAs in 27 nations. By 1985, some 430 MEPAs had been proclaimed by 69 nations with another 298 proposals under consideration (De Silva et al. 1986 cited in Kelleher and Kenchington 1992). Many more have been declared since[101]. A major international overview of marine protected areas, including the rationale behind them, guidelines and case studies is provided by Agardy (1997) and a comprehensive bibliography of the benefits of marine protected areas is available on the web[102].

In Australia, the Australian Committee for IUCN (ACIUCN) was established in 1983. In 1984, the ACIUCN formed a working group known as the Marine Reserves Sub-Committee (ACIUCN-MRSC) (Parker 1995). One of the early achievements of the MRSC was the publication of *Australia's marine and estuarine areas – a policy for protection* (ACIUCN 1986). Another major task was the 1991 Fenner Environment Conference – *Protection of marine and estuarine areas – a challenge for Australians*, held at the Australian Academy of Science in Canberra (Ivanovici et al. 1993). *Towards a strategy for the conservation of Australia's marine environment* was launched as an ACIUCN Occasional Paper (No. 5) in 1994 as one of the follow up activities to the Fenner Conference.

Kriwoken and Haward (1991) reviewed the history of MPA policy in Australia, including State-Commonwealth jurisdiction, the GBR etc. MPAs have been declared in Australia for over 50 years, although prior to the declaration of the Great Barrier Reef Marine Park in 1975, only small areas had been protected. Currently Australia has several of the world's largest marine protected areas, including the Macquarie Island Marine Park, Great Barrier Reef Marine Park and the Great Australian Bight Marine Park, each covering millions of hectares. However, it is difficult to be precise about the number and area of marine protected areas currently declared in Australia. Lists have differed markedly from each other, depending on how the author has chosen to define a "marine protected area" and what to include and exclude[103]. For example, Coveney (1993) recognised 173 marine parks present in Australia to September 1991 (including 26 that were greater than 18 000 ha in areas), whereas McNeill (1994) stated that 267 MPAs had been declared by 1992. The most recent Australia-wide evaluation was by Cresswell and Thomas (c. 1997). A brief evaluation of the national system of MPAs is provided below.

Purposes and design of marine protected areas

Marine Protected Areas can be declared for a variety of reasons, ranging from protection of fish stocks or recreation areas to conservation of biodiversity or provision of scientific reference areas. They are also ideal places to carry out research on habitats and populations as free as possible from human interference (e.g., Creese and Jeffs 1993).

The IUCN has defined eight categories of protected areas (IUCN 1984). Many authors have added an additional three categories, representing areas recognised under international agreements for which conservation is an objective (e.g., Amos et al. 1993). These categories are:

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[101] The IUCN maintains information about worldwide marine protected areas at http://wcpa.iucn.org/region/

[102] http://www.pac.dfo-mpo.gc.ca/oceans/mpa/MPAs%20-%20nonannotated.doc

[103] For instance, marine components of terrestrial national parks, areas of coastline where harvesting is banned and protected habitats are examples of areas potentially included or excluded from definitions of "marine protected areas".

	Category	Purpose
I	Scientific Reserve/Strict Nature Reserve	To protect nature and maintain natural processes in an undisturbed state
II	National Park	To protect natural and scenic areas of national or international significance for scientific, recreational and educational use
III	Natural Monument/Natural Landmark	To protect and preserve nationally significant features because of their special interest or unique characteristics
IV	Nature Conservation Reserve/Managed Nature Reserve/Wildlife Sanctuary	To ensure the natural conditions necessary to protect nationally significant species, groups of species, biotic communities or physical features of the environment by direct human management
V	Protected Landscape or Seascape	To maintain nationally significant natural landscapes characteristic of the harmonious interaction of humans and land while providing opportunities for recreation and tourism within the normal lifestyle and economic activity of these areas
VI	Resource Reserve (Interim Conservation Unit)	To protect the natural resources of an area for future designation and prevent or contain development activities that could affect the resource
VII	Natural Biotic Area/Anthropological Reserve	To foster the continuation of the way of life of societies living in harmony with the environment little disturbed by modern technology
VIII	Multiple Use Management Area/Managed Resource Area	To provide for the sustained production of natural resources, wildlife and recreation with the conservation of nature primarily oriented towards the support of economic activities
IX	Biosphere Reserve	To provide areas for research, monitoring, training, and demonstration as well as conservation
X	World Heritage Site	To conserve areas of natural and cultural value, those of "outstanding universal value"
XI	Wetlands of International Importance (RAMSAR)	To conserve wetland habitats[104], especially for waterfowl

In Australia there are examples of categories I, II, III, VIII and X, the best known example of VIII being the Great Barrier Reef Marine Park, much of which is actually World Heritage listed (X). Smaller examples of category VIII are Jervis Bay and Solitary Islands Marine Park (NSW) and the Solitary Islands Marine Reserve (Commonwealth). Outside parts of the GBR, examples of scientific or strict nature reserves (Category I) include Ashmore Reef, Mermaid Reef, Shark Bay, and the Tasmanian Seamounts Marine Reserve where fishing is allowed above the seamounts to certain depths.

In contrast to the terrestrial environment, where protected areas are now generally (though by no means universally) seen as incompatible with extractive or exploitative resource use, a minority of marine protected areas are declared as highly protected areas (e.g., strict nature reserves or "no-take" zones). Many still permit activities such as fishing, and many (particularly the larger areas) are, almost by necessity, managed as "multiple use marine parks", with activities in different areas controlled by means of zoning plans. In the Great Barrier Reef Marine Park (GBRMP), for example, only about 4.5% of the total area consists of no-take zones, and the majority of these areas protect reefs rather than soft-bottom inter-reefal or other habitats (GBRMPA 1999; Day et al. in

[104] Under the Convention on Wetlands of International Importance (RAMSAR), wetlands are defined as areas of marsh, fen, peatland or water, whether natural or artificial, permanent or temporary, with water that is static or flowing, fresh, brackish or salt, including areas of marine waters, the depth of which at low tide does not exceed 6m.

press and L. Fernandes pers. comm.). This percentage may well increase now that the first stage of public participation in the Marine Representatives Area Program has been announced (May 7th 2002).

Much debate has occurred in Australia (and elsewhere) over the value of large multiple use parks versus highly protected areas (e.g., several papers in Ivanovici et al. 1993). Some authors (e.g., Roberts 1997) have argued that marine reserves should be established on the principle of 'no-take', because while some beneficial effects have been measured from reserves that allow some kinds of exploitation, experience shows that 'no-take' reserves are much easier to implement and enforce. Roberts (1997) suggested that as we lack information on the effects of fishing at ecosystem scales we are unable to say *a priori* what kinds of take will not compromise their effectiveness. 'No-take' reserves can provide reference areas and large-scale experiments with which we can examine human impacts on marine ecosystems. However, much of this debate may be somewhat theoretical, as highly protected areas can be nested within Multiple Use Parks.

While declaration as a multiple-use marine park is the only realistic option for large areas that have many existing users (such as the Great Barrier Reef), smaller areas may be declared as nature reserves or similarly highly protected areas, with the primary aim of conserving biodiversity, a particular habitat or a particular topographic feature. For example, Ship Rock in Port Hacking (NSW) is only $0.02km^2$ and was designed to protect a sublittoral cliff covered in encrusting organisms. Others have been declared to protect the spawning stock or habitat of a particular species, or as harvest refugia for exploited animals such as abalone, crayfish or sea urchins (e.g., Quinn et al. 1993)[105]. In some areas, seasonal closures may be declared at certain times of the year to protect vulnerable populations (e.g., spawning aggregations, or migratory species). Other areas may be closed periodically to collecting/harvesting to allow replenishment of stocks (usually fishes) (Schmidt 1997). Balancing conservation with traditional use of resources by Aborigines and Torres Strait Islanders is another important goal.

Australia's National Representative System of Marine Protected Areas

One of the primary aims of the Interim Marine and Coastal Regionalisation for Australia (IMCRA Technical Group 1998; see Section 2.4.2) has been to facilitate the development of a representative[106] system of Marine Protected Areas around Australia, to ensure that a sample of all habitat types and bioregions are protected in some form of reserve. The initial IMCRA schemes were based on physical data such as depth and sediment type and limited biological data such as fish distributions, and their correlation with invertebrate distributional patterns and biodiversity are largely untested. The basic assumption is that if a representative set of sediment types, combined with adequate depth and latitudinal representation, are selected and conserved as MPAs, then much of the (as yet poorly known) soft bottom fauna will also be conserved.

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[105] The use of marine reserves to manage benthic fisheries was discussed by Baker et al. (1996, with a particular emphasis on the South Australian abalone fishery), and is discussed further in Section 6.10.4.

[106] For a discussion on the practical problems of the concept(s) of "representative" see Inglis (1993).

To date the declaration of marine parks in Australia (as elsewhere in the world) has been rather ad hoc and biased towards certain habitat types and geographical areas at the expense of others. For example, in 1995, 99.5% of the area in MPAs occurred in tropical waters and only 0.4% in temperate waters (Kelleher et al. 1995). Although this imbalance has been greatly improved with the recent declaration, in Commonwealth waters, of the Great Australian Bight Marine Park, Lord Howe Island Marine Park, Tasmanian Seamounts Marine Reserve and the enormous (16.2 million ha) Macquarie Island Marine Park, there is still a tendency to focus on systems such as coral reefs rather than other important, but less spectacular, habitats. Similarly, Bridgewater and Ivanovici (1993) estimated that of the 32 biogeographic regions around Australia, 21 lacked any significant protected areas. Despite the shortcomings (due to data deficiency) of the existing bioregionalisation (IMCRA), the advantage of using this as the basis for establishing a National Representative System of Marine Protected Areas is that it should encourage a more balanced selection of areas for reserves. Unfortunately, in coastal areas, where there are often large numbers of existing users, the declaration of protected areas can still be difficult and contentious. In such circumstances, site availability and community support may be as important as scientific factors or conservation 'value' in determining where protected areas can be located.

Within large geographic areas, the issue of adequate representation of different habitats within protected areas is important. The Great Barrier Reef Marine Park Authority is currently attempting to improve on its original zoning system through its own *Marine Representative Areas Program* (e.g., GBRMPA 1999; Day et al. in press).

It is difficult, on current information, to effectively evaluate the significance of "marine representative areas" for invertebrate conservation. Certainly, as described above, the current revision of zoning plans for the GBR is taking into account much of the available information on benthic communities. Conservation of benthic communities has also been a specific objective of reserves such as the Tasmanian Seamounts Marine Reserve, the Great Australian Bight Marine Park and the Macquarie Island Marine Park, but these communities remain poorly known. With the current policy of declaring multi-use parks, better information is required regarding the distribution and composition of invertebrate communities, and their ecological interactions (and not just the large, conspicuous species), to ensure that appropriate management zones are created. Many marine parks have inadequate protection against fishing and mining, both of which have the potential to seriously impact marine (particularly benthic) invertebrates; for instance, trawling is still allowed in many parks and petroleum and gas exploration are allowed in some areas. More account needs to be taken, in all these instances, of the impacts on invertebrates.

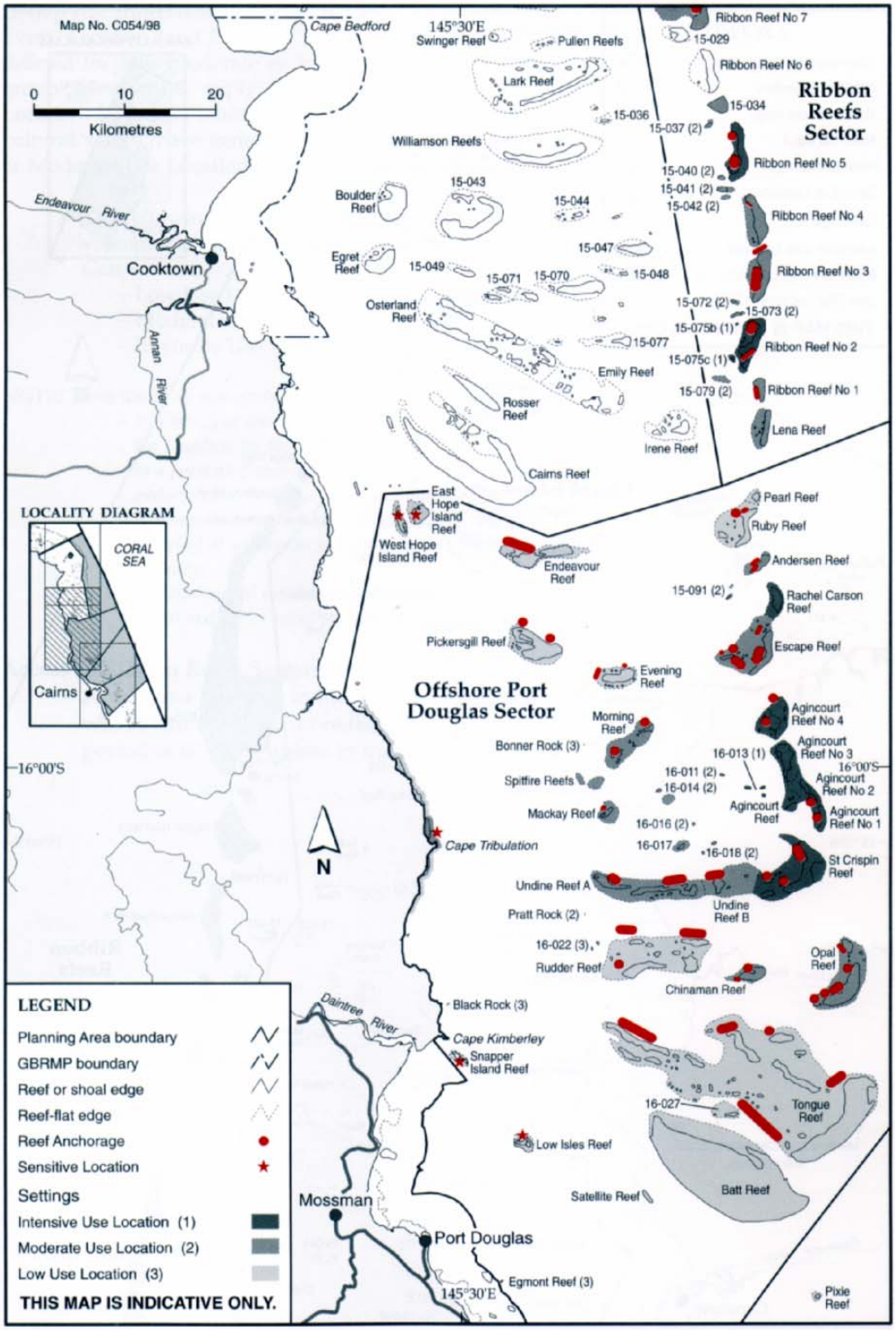
The Marine Representative Areas Program was designed principally to protect the reefal areas (although they also included a large cross-shelf transect in the far Northern Section which was closed to all forms of fishing). However, there have been considerable increases in the pressures on the reef from fishing and tourism, along with a growing realisation of the importance of inter-reefal areas and more detailed knowledge relating to current patterns and larval dispersal. In 1999, the Authority, through a series of workshops, used all the physical and biological information available to it (including expert knowledge) to develop a detailed map of habitats within the GBR marine park area. Over 77 habitats have been recognised, which represents a considerable refinement of the IMCRA maps for the region (Day et al. in press)]. The next stage in the process will involve determination of the number of replicates of each habitat type needed and their adequate latitudinal representation. Draft biological zoning maps will then need to be considered in relation to social and economic factors such as major shipping lanes, trawling grounds, indigenous activities and tourist sites. Thus the revised zoning plans released for public comment will have considered the variety of habitats available, those requiring higher levels of protection, and the key threatening processes which impact on these habitats. While it would be ideal if such a methodology could be used for the rest of Australia, the data available will vary considerably, with some areas being extremely poorly known, especially in terms of the benthic fauna.

Management of marine protected areas

As discussed in Chapter 8, the responsibility for declaration and management of marine protected areas can fall on a number of agencies, or even several agencies simultaneously. In addition, activities occurring outside the boundaries of the MPAs (e.g., elsewhere in the marine environment, or on land) often impinge on them, and the agencies responsible for these activities also need to be involved in management. With the growing recognition of the interconnectedness of marine habitats (and marine-terrestrial systems), areas adjacent to terrestrial national parks are increasingly being considered for declaration of MPAs. For example, the terrestrial national parks surrounding much of Jervis Bay should facilitate the management of the catchment area and control of pollutants into the Bay. Similarly, GBRMPA has long recognised the need to control terrestrial run off (Chapter 6). To address these problems, a more co-ordinated approach to marine park management is required, along with an avoidance of territoriality.

The declaration of a large marine park, almost by necessity, dictates that the area be multi-use. The management of any park requires the active cooperation of all users, and all users (and other members of the public) need to have the opportunity for comment during the development of a zoning plan and before the plan is finalised (Allison et al. 1998). Zoning plans can be complex (e.g., Great Barrier Reef Marine Park, see Fig. 5.2), with highly protected areas, for example, often surrounded by buffer zones (areas of intermediate protection) which are themselves surrounded by less protected areas. Different activities are allowed in each zone. Only when the public feels that they “own” the plan is self-policing likely to work. Large areas such as the GBRMP are impossible to police continually, although with onboard VMS (Vessel Monitoring Systems) it is now possible to know exactly where fishing vessels working, with this evidence apparently acceptable in court.

Figure 5.2: A map showing different management zones in one part of one sector of the Great Barrier Reef Marine Park (from GBRMPA 1998).



The Great Barrier Reef Marine Park, an area covering some 344,000 km^2, is managed as a single ecosystem through agreements between the Commonwealth and the Queensland Government relating to Marine Parks and Island National Parks. The Great Barrier Reef Marine Park model is designed for "large area" or ecosystem management. Its aim is to manage for ecologically sustainable use, including the need to balance all interests, including legitimate extractive activities, as well as to protect significant areas (Craik 1996). The model includes legislative support of the complementary management arrangements of state and federal agencies (e.g., fisheries agencies). While generally working well, disadvantages include the geographical restriction of the Great Barrier Reef Marine Park to marine areas (major impacts occur from terrestrial sources), the high level of resources required for adequate management, the difficulties of having several agencies involved, and the legislative requirements (Craik 1996). The recently released Sturgess Report (Sturgess 1999) commissioned by the Queensland Government evaluated these problems and many of these issues are now being addressed by the Authority.

Funding

Funding for marine park management is generally inadequate given their level of use and the economic and recreational benefits derived from them. For instance, Driml and Common (1995) examined the economic benefits derived from five major Australian World Heritage Areas – including the Great Barrier Reef – and the economic/ecological trade-offs associated with the management of these areas. They calculated that the total expenditure associated with visits to these five areas by tourists in 1991/92 was approximately $1.4 billion, although they considered that this was an underestimate because it did not include money spent in travelling to the areas. By way of contrast, the budgets dedicated to managing these natural areas are relatively small, amounting to less than about 5% of the tourist expenditure for such areas. The budget for the Great Barrier Reef amounted to only 2.3% of tourist expenditure (see Table 5.1).

Table 5.1: Expenditure and management costs associated with the Great Barrier Reef World Heritage Area (based on 1991/1992 values) (after Driml and Common 1995)

	Annual expenditure by tourists ($m)	**Managemen t budget ($m)**	**Budget as a percentage of tourist expenditure (%)**	**User fees collected ($m)**	**Fees as a percentage of budget (%)**
Great Barrier Reef WHA	776	18.1	2.3	0.79	4.4

This lack of funding equates to the government agencies charged with the responsibility of managing these areas being inadequately funded to cope with the rapidly increasing numbers of tourists visiting them (Driml and Common 1995). One option for providing more funding is to levy a user fee against those people visiting a given area. This is already being done in the GBR, where there is a tourist tax ("reef tax") collected by the tourist operators, but it could undoubtedly be extended to other areas.

Evaluating the effectiveness of MPAs

Given the theoretical and practical benefits that result from research into the effects of marine reserves, the number of publications that present empirical data on this topic are surprisingly small, particularly when compared with the number of reviews and desktop studies (such as this one) that make general recommendations and observations (e.g.,Jones et al. 1993; Kenchington and Bleakley 1994; McNeill 1994; Gubbay 1995; Allison et al. 1998; Tuck and Possingham 2000). Field investigations vary in quality, with studies generally compromised by ecological differences between the sites investigated, by a lack of site replication, or a lack of information about the biota and conditions prior to the declaration of the reserve(s). Because of the importance of secondary effects in marine ecosystems (e.g., Menge 1995), the implied acceptance that resources will be enhanced with the declaration of a marine reserve is questionable (Edgar and Barrett 1999).

The establishment of marine reserves represents a manipulative removal experiment at a vast spatial scale (Edgar and Barrett 1999). There are opportunities to assess the impacts of fishing (and other types of exploitation) on wild populations, by comparisons between protected and non-protected areas or by documenting temporal changes resulting from cessation of exploitation. For example, there is a need for unfished control sites if fisheries research is going to be able to determine appropriate levels of harvest pressure for maintaining sustainable fisheries and marine biodiversity (e.g., Engel and Kvitek 1998). There are few studies, however, which document the effects of such protection (Bell 1983; Russ and Alcala 1989), and most of these focus on fishes (Cole et al. 1990). A recent Australian study by Edgar and Barrett (1999) investigated the effects of marine reserves on Tasmanian reef fishes, invertebrates and algae (see also Section 6.4.1). They investigated the reef biota in four Tasmanian marine reserves and at associated unprotected reference sites over a six-year period following protection from fishing and showed that the effectiveness of marine reserves appeared to correspond with reserve size[107]. Their results also provided the first clear evidence that shallow Tasmanian reef ecosystems are overfished, and that unfished coastal ecosystems differ substantially from those where fishing occurs.

Some other examples

Other relevant studies include:

- Cole et al. (1990) investigated the possible effects of marine reserve protection on densities of large invertebrates through a series of sampling programs from 1976 - 1988 in northern New Zealand. A detailed survey in 1988 between sites inside and outside the marine reserve showed no clear patterns for sea urchins but a very striking increase in numbers of rock lobsters (*Jasus edwardsii*) within the marine reserve.
- Increases in primary (e.g., kelp forests) and secondary productivity occurred in reserves relative to unprotected areas in New Zealand (Babcock et al. 1999).

[107] The largest reserve at Maria Island (7 km coastline length) proved the most effective, with the number of fish, invertebrate and algal species, the densities of large fishes and rock lobsters, and the mean size of abalone increasing significantly within this reserve compared with external reference sites. Changes in species richness of fishes, invertebrates or algae (other than an increase in the number of larger fish) were not detected in any of the three smaller reserves.

- A study by Cole and Keuskamp (1998) compared sea urchin populations in a marine reserve with exploited areas and found that numbers decreased in the reserve, probably due to the presence of more predatory fishes.
- Lasiak (1998) compared rocky intertidal macrofaunal assemblages from "no-take" marine reserves and adjacent exploited localities in South Africa and found a clear-cut separation between them. The major differences were the lower abundance and biomass of sessile filter feeders and microalgal grazers (dependent on the primary substratum) and the greater abundance of phytal-associated species under exploited conditions. These differences were caused mainly by the removal of larger grazers for food, leading to greater domination of primary space by algae.
- In the Caribbean, declining stocks of spiny lobster and conch have been causing concern (Davis and Dodrill 1980), with nearly every adult spiny lobster removed from Florida reefs by fishing each year (Davis and Dodrill 1989). Populations in Florida reserves appeared to have sustained these fisheries (Roberts and Polunin 1993). Stockhausen et al (2000) working in the Caribbean found that the larval supply of the Caribbean spiny lobster (*Panulirus argus*) increased linearly with reserve size, but when reserve size was expressed as the fraction of the coastline protected, larval production decreased for some reserve configurations. These authors argue that the use of a simple reserve-design rule (i.e., protect 20% of a coast) would in the some cases lead to a false sense of security, thereby endangering rather than protecting exploited stocks. The optimal design of marine reserves therefore requires attention to the joint effects of larval dispersal, reserve location and reserve size on fishery yield and recruitment.
- In the Hol Chan Marine Reserve in Belize, surveys showed that densities of conch (Strombidae) were substantially higher, and individuals larger, within the reserve than in unprotected areas, while spiny lobster densities were 25 times higher inside the reserve than outside (Azueta, unpublished report, cited in Roberts and Polunin 1993).
- The numbers of *Concholepas* in Chile (Manríque and Castilla 2001) and abalone in California (Rogers-Bennett and Pearse 2001) were higher in reserves than in areas where they were harvested.
- A survey of recreational divers in Jamaica found that they preferred a wide range of the environmental attributes of protected areas for diving where fishing bans were effective (Williams and Polunin 2000).

Future marine parks

Parker (1995) undertook a survey of marine and estuarine conservation of NSW coastal waters and produced an "Atlas" of proposed marine national parks, based on an assessment of the suitability for protection of the marine or estuarine waters adjacent to over 60 national parks, nature reserves or state recreation areas. Field surveys were undertaken and information on current usage obtained from Fisheries Inspectors, NPWS personnel, local fishers, naturalists and research papers. Where appropriate, benthic habitats, including plant and animal communities, were investigated. These results, combined with existing data from the literature, formed the basis of proposed reserve boundaries. To date this report has not been acted upon. More recently, Otway (1999a) (for NSW Fisheries) identified candidate sites for declaration as aquatic reserves for the conservation of rocky intertidal communities in the Hawkesbury Shelf and Batemans Shelf Bioregions.

Reponses to a questionnaire distributed to over many experts (see Acknowledgements) as part of the consultation process for the current report identified a number of areas as being critical for marine invertebrate conservation, or otherwise of special value (e.g., high biodiversity). These include (a sample only):

- Soft-bottom inter-reefal areas in the tropics (J. Hooper pers. comm.) – these often contain the highest diversity of invertebrates and have been largely overlooked (although they are currently being considered under GBRMPA's Representative Marine Areas Program, GBRMPA 1999; Day et al. in press).
- Large enclosed bays such as Moreton Bay, Port Phillip Bay, Westernport (Coleman et al. 1978), St Vincents Gulf and Cockburn Sound – these are refuges for tropical and temperate species as well as estuarine and indigenous species (P. Mather pers. comm.), and are poorly represented in the existing system of MPAs.
- Subtidal reefs in Bathurst Channel estuary (Tasmania) – these support an unusual invertebrate community in shallow water with numerous taxa unknown elsewhere, or otherwise recorded in depths greater than 50 m. (G. Edgar pers. comm.).
- Rocky headlands in NSW and Port Phillip Heads, Victoria – contain many interesting species and need more protection than they currently have (A. Davis pers. comm.; J. Watson pers. comm.).

The primary needs are for the:
Establishment of an adequate series of large marine parks that adequately encompass the full range of marine habitats and therefore, hopefully, most of the animals found in the EEZ.

- Inclusion of tropical habitats, other than coral reefs, which are currently poorly represented. While authors such as Edyvane (1996) have (understandably) focused on the imbalance of tropical and temperate protected areas, representation of tropical habitats is very skewed by the GBRMP.
- Protection of coral reefs outside of the GBR (i.e., in the north and northwest) – these are very different to those of the GBR and there is an opportunity to protect them before they become important tourist attractions.
- Protection of small isolated areas likely to have local endemics (e.g., seamounts).
- Provision of funds for the development and implementation of management plans, which need to be flexible and incorporate new information as it becomes available.
- Recognition and control of threatening processes and downstream effects.
- More co-ordination between agencies.
- Support of further research to expand the database for the area (e.g., documenting the fauna and confirming or disproving the adequacy of the reserve system selected using surrogates such as depth, sediment, etc.).
- Development of monitoring strategies and evaluation processes to see if management practices are working and the usefulness of the reserves.

Data availability

Reserve design and effectiveness can be dramatically improved by better use of existing scientific understanding (Allison et al. 1998). To date, most reserve design and site selection processes have involved little scientific justification. They must begin to do so to increase the likelihood of attaining conservation objectives. We suggest some research priorities to fill critical information gaps in Section 7.8.

Much of the data that are potentially available are not easily accessible - e.g., the invertebrate collections in the State Museums[108] (see Section 7.2.3).

5.6 Main issues and recommended actions

Isssues

Marine invertebrate faunas are very heterogeneous in species composition and patchy in distribution, in part due to stochastic factors such as larval recruitment.

- Thus, small differences in geomorphology or other parameters (e.g., depth, wave exposure, latitude) often translate to very large differences in species composition.

The linkages between marine systems mean that the protection of apparently insular habitats (e.g., coral reefs) may be ineffective if related habitats (e.g., seagrass meadows and mangroves) are left unprotected and waterborne transport – of organisms, nutrients and pollutants – is not considered.

- The linkages between the marine and terrestrial environments mean that it is essential to consider "downstream" impacts of terrestrial activities on marine ecosystems.

Existing data (such as that available in the literature or museum collections), as well as the expertise of scientists, needs to be fully utilised in order to develop bioregionalisations that better reflect patterns of invertebrate diversity, and to enable better planning of additional surveys and cruises.

Even protected areas may deteriorate over time if anthropogenic impacts (i.e. threatening processes) are not controlled.

Public awareness programs and community involvement in the declaration of protected areas are essential to make people aware of the value of protected areas and the need for them.

Recommended actions

Because of the heterogeneity of marine ecosystems, conservation strategies must aim to preserve suites of habitats (i.e. not just single examples of each system), in order to encompass geomorphological and latitudinal diversity and ensure preservation of maximum genetic diversity.

[108] The Australian Museum has, for example, more than 50,000 records for NSW marine and estuarine molluscs in its database and many tens of thousands of records of molluscs from the GBR (a summary of the collection resources of all the State museums is provided in Appendix 2). None of these data, or most other invertebrate data, have been used in conservation planning to date.

- Conservation strategies need to incorporate the protection of areas with heterogeneous habitats important to meet the changing habitat requirements in complex life cycles.
- Management should be at the level of the catchment (e.g., Fairweather and Quinn 1995).

Given that invertebrate populations often undergo major natural changes both seasonally or inter-annually, programs need to be developed to undertake long-term monitoring of at least a subset of the taxa, and criteria need to be developed to assess what constitute “natural” versus unacceptable changes.

Develop strategies for better utilisation of existing data.

Public awareness programs need to be developed and informed community involvement in the declaration of protected areas encouraged.

CHAPTER 6 – THE THREATENING PROCESSES APPROACH TO CONSERVATION

A threatening process is a process that detrimentally affects, or may detrimentally affect, the survival, abundance, distribution or potential for evolutionary development of a native species or ecological community (Burgman and Lindenmayer 1998). This approach to conservation deals mainly with the management of human impacts on the environment responsible for the decline of biota, rather than managing a particular habitat or group of organisms. For conservation strategies of any sort to succeed, it is essential that management of threatening processes goes hand in hand with other strategies, either taxon-based or habitat based systems.

While the immediate ("proximate") threats to marine life have been highlighted and discussed in a number of documents, and are those most commonly dealt with because they are a manageable size, it should be emphasised that these are only aspects of the fundamental, underlying or root ("ultimate") factors. Norse (1993) identified these root causes as:

- There are too many people (population growth);
- We consume too much (over consumption);
- Our institutions degrade, rather than conserve biodiversity;
- We do not have the knowledge we need; and
- We do not value nature enough.

Threatening processes can be natural or anthropogenic and change the environment in ways that may jeopardise the continued existence of fauna in their natural role within their environment (Yen and Butcher 1997). While natural processes that threaten taxa or habitats are, by their very nature, part of natural evolution, these may be difficult to separate from anthropogenic changes (e.g., the current debate on global warming and sea level rise). Yen and Butcher (1997) did not consider natural impacts in their review, focusing entirely on anthropogenic threats, although they recognised that natural processes might be modified by these threats. We consider that, in some cases, the impacts of some natural changes are relevant when they are compounded by anthropogenic changes and some attempt should be made to manage them to prevent biodiversity loss. For example, habitat destruction may have reduced a particular habitat type to one small area and this may contain rare or threatened species. If storm damage destroys this last remaining piece of habitat, is this loss entirely natural as its loss is clearly the consequence of other anthropogenic threatening processes?

Most people now accept that anthropogenic changes to the earth and its atmosphere have resulted in climate changes (Section 6.7) notably in increased land and sea temperatures, increased incidence of storms and changed rainfall patterns. Several other potential threatening processes are apparently part of the natural system but which (some argue) may be partially caused, or at least exacerbated, by human activities. These include pathogens and diseases; population outbreaks; and algal or dinoflagellate blooms.

Another difficulty in assessing the effects of threats, or in determining the causes of observed declines, is that most marine ecosystems experience considerable natural spatial and temporal fluctuations in abundance, community composition etc. (see Section 7.7.1). Fluctuations may be attributed by a researcher operating in a particular place and time to some anthropogenic impact, but in fact it may be part of a natural cycle, a response to a natural process or event, or the result of a combination of factors. Without proper long-term baseline studies with controlled reference sites, there is little opportunity of actually assessing the impact. Yet, very few long-term studies of natural fluctuations of marine invertebrates have been undertaken.

While all marine ecosystems are exposed at various times to a range of 'natural' stresses ranging from predation to disease to natural 'disasters' such as storms and cyclones, or, in the sub-Antarctic, even iceberg impacts, these are essentially dynamic with a disturbance and recovery phase. Thus an ecosystem suffering a dramatic change caused by a natural disturbance should, other things being equal, eventually recover. However, human disturbances interact with natural stresses and can reduce the capacity of the system to recover from further impacts, thus producing chronic degradation.

Unfortunately, it is often difficult to distinguish between natural and anthropogenic processes when attempting to assess the causes for a decline in some systems. Natural disturbances may be physical (e.g., hurricanes, floods, earthquakes, low tides) or biological (e.g., diseases, outbreaks of predators). It is often not possible in practice to distinguish, for example, between the effects of climate change, of natural environmental variability, or of non-climatic anthropogenic alteration; effects may be interactive, or a single stress may have multiple sources (Smith and Buddemeier 1992). Natural impacts, such as cyclone damage to coral reefs (e.g., Van Woesik et al. 1995; De Vantier et al. 1996; Bythell et al. 2000) may be the result of direct impacts such as wave action or secondary effects, such as disease or from sediments carried by floodwaters.

While some threats may directly impact particular elements of the biota, especially by changing or destroying their habitat, others act indirectly (e.g., global warming, pollution). Not all threatening processes are obvious. Some may be quite subtle and insidious; for example, those causing sublethal effects that result in reduced fecundity, food supply, changes in skeletal density or gradual reduction in habitat. It should also be recognised that impacts from threatening processes may be cumulative and that several different threatening processes can act simultaneously.

Human impacts on marine environments are mainly concentrated on coastal habitats where the greatest human populations are located. However, in Australia, as elsewhere, the use of innovative technology is increasing the level of offshore mining, oil and gas exploration and deep-sea fishing; activities that could become an increasing threat to offshore marine communities.

6.1 Pros and cons of the threatening processes approach

The threatening processes approach is important with respect to management action. The management of particular activities and threats by certain agencies is, in many cases, already well established (e.g., shipping, fisheries, pollution licences, etc). Conversely, some threats are not well controlled, sometimes due to inefficient or inadequate management. As with terrestrial environments, this is often because multiple agencies are involved, for example in various aspects of managing a particular habitat, area or activity. Managing agencies may be limited in their control of threatening processes that originate outside the area or sphere of activities under their jurisdiction. In the case of the Great Barrier Reef, where the entire system is nominally under the jurisdiction of a single managing authority (the Great Barrier Reef Marine Park Authority), there are relatively few options for direct action on threats that originate outside the designated area. For instance, the poor quality of terrestrial runoff has been identified as a key threatening process (Baldwin 1990; Bell 1991; Yellowlees 1991; Brodie 1995a, 1997; Hutchings and Haynes 2000), yet this is primarily due to actions by land owners and is therefore the responsibility of local councils, catchment management authorities and individual landholders, rather than the Marine Park Authority[109]. In cases where multiple agencies are responsible for different aspects of a system or activity, one authority may approve a development that results in some loss of habitat and be unaware of other developments in other parts of the system that fall under other jurisdictions. Thus habitat can be lost progressively by 'the tyranny of small decisions'.

PROS

- It is easier to manage or place controls on particular industries or types of users identified as causing one or more significant threats than to co-ordinate all the different threats facing particular species or habitats.
- Well considered threat management can identify and focus upon processes that may result in severe consequences and threats if unmanaged. For example, strategies to minimise the importation of marine organisms in ballast water are much more likely to succeed than programs to eradicate existing marine pests. Prevention is better than cure.
- This approach allows some action to be taken without necessarily having detailed taxonomic or systems-level information.

CONS

- Multiple threats are typically operating and it is often difficult to identify the most relevant.
- Identification (or misidentification) of one obvious threat may ignore the cumulative or synergistic effects of other more subtle threatening processes.

[109] The recently enacted EPBC Act provides a potential mechanism whereby land runoff and water quality that impacts on a World Heritage Area such as the GBR could be controlled. The Commonwealth can act to improve water quality in the GBR region by controlling terrestrial runoff because it has been identified by the authority as a key threatening process. This a widespread problem because most marine coastal areas in Australia are impacted to varying degrees from coastal run off, but the EPBC Act can only be invoked where it impacts on Commonwealth waters or World Heritage Areas.

- Cost benefit, political or social considerations may be major factors and override ecological considerations (e.g., destruction of benthos by commercial trawling).
- Management problems, including lack of coordination when multiple agencies are responsible, and inter-departmental rivalry.
- There are logistical and political problems separating the threatening process from the agent of that process (e.g., coastal development is not a threatening process in itself, but because its implementation often results in a number of different threatening processes being realised, all such development is, perhaps incorrectly, considered a threat).
- There is a need for enforcement, usually by way of legislation or regulation, in order to be effective.
- Targeted management may ignore other threats (e.g., fisheries response to overexploitation is concerned with the protection of the resource, not the environment – e.g., impacts of trawling).
- Regulations must be strongly enforced to be effective. Such regulations are often focused on particular suites of taxa (e.g., edible fishes and invertebrates, and species collected for use as bait), with little attention paid to activities that impact other taxa.

The listing of specific threatening processes in legislation (such as previously in the Commonwealth's *Endangered Species Protection Act 1992*) has been problematic. This was seen as potentially the ESPA's most important contribution, but the cases initially accepted concerned four terrestrial feral pests and one pathogen. Processes involving human action have been far more contentious (Woinarski and Fisher 1999). For instance, despite widespread acceptance that vegetation clearance is one of the most important factors contributing to loss of Australia's terrestrial biodiversity (State of the Environment Advisory Council 1996), it was not listed, although it has recently been declared a key threatening process under the TSC Act in NSW. This was largely because a national plan was required and the cooperation of all states is necessary, and in this case agreement was unlikely. This criterion raised a barrier that can be used to prevent the listing of activities clearly detrimental to biodiversity (Woinarski and Fisher 1999). This situation has changed under the *Environment Protection and Biodiveristy Conservation Act 1999* (EPBC Act) and, already several "Key threatening processes have been listed[110] (see Section 8.2.2).

6.2 Threatening processes – their relation to marine-based industries and activities

Many threatening processes relevant to marine ecosystems have been identified in the literature, ranging from discussions of general, broad scale issues and impacts on the marine environment in a variety of reports and popular books, to scientific papers examining the threats facing particular taxa, areas or habitats. Threats to the marine environment have been outlined in a number of key Australian government documents including the National Strategy for the Conservation of Australia's Biological Diversity (Commonwealth of Australia 1996), Australia's Oceans Policy (Commonwealth of

[110] http://www.ea.gov.au/biodiversity/threatened/ktp/index.html

Australia 1998a, 1998b) and State of the Environment Reports (e.g., Zann 1995; Zann and Kailola 1995; Zann and Sutton 1995; State of the Environment Advisory Council 1996; Zann 1996; Australian State of the Environment Committee 2001). International publications include reviews of marine biodiversity and conservation by Norse (1993) and GESAMP (1997). For instance, the national State of the Marine Environment Report (Zann 1995) summarised major uses of the marine environment with potential impacts (recreation and tourism; fisheries; and marine transport and energy), as well as general issues and pressures (coastal modification; coastal development and sea level change; marine pollution; introduced species; and population outbreaks). In Victoria, Norman and Sant (1995), identified heavy commercial and recreational harvests of some species, destructive fishery practices, non-collecting visitation pressures, marine and coastal developments, eutrophication from sewage discharge, siltation, chemical pollutants and introduced biota as exerting pressures on marine invertebrates. From a global perspective, Norse (1993) identified five broad classes of threats to marine biodiversity (over-exploitation, physical alteration, marine pollution, introduction of alien species, and global atmospheric change), resulting from a range of human activities. GESAMP (1997) identified habitat degradation and fragmentation, climate change, UV radiation, fishing, pollution, litter, non-indigenous species and the impact of tourism as significant threats. In a review of literature relevant to the conservation of shallow tropical marine ecosystems, particularly coral reefs, mangroves and seagrass communities, Hatcher et al. (1989) included anthropogenic impacts such as sedimentation, chemical pollution, sewage pollution, thermal pollution, radioactive pollution, hydrodynamic influences, physical disturbance, extractive industries, introductions and tourism. Suchanek (1994) reviewed threats to temperate coastal marine communities and identified (1) habitat loss and degradation, (2) pollution (from numerous sources including sewage, pesticides, pulp mills, thermal effluents, polychlorinated biphenyls, heavy metals, oil and radionuclides), (3) over-exploitation, (4) species introductions, (5) global climate change, (6) misguided human perceptions and (7) legal complexities as the most significant categories of threats.

In many cases these discussions only partially separate the ***actual threatening processes*** (i.e. impacts which affect marine organisms, e.g., marine pollution, introduced species, population outbreaks) and the ***agents or activities responsible for those impacts*** (fishing, shipping, development etc.). However, it can be important to distinguish between processes and agents, as many activities result in a variety of impacts while many impacts can have a range of contributing causes. For example, direct exploitation through fishing is often considered a threatening process, but if harvest levels are sustainable, fishing is not necessarily a threat to the harvested species. On the other hand, it could result in other threats (e.g., habitat damage). Consequently, we have divided this chapter into two parts, dealing with the **processes** (the threats) and the **agents** (the causes of the threats).

Table 6.1 lists some of the direct impacts associated with each industry or activity (i.e., agent) responsible for the threats (ie., threatening processes), and illustrates the relationships between the various categories of threats and industries/activities. Most of these processes have direct and indirect, lethal or sub-lethal, effects on invertebrate communities and ecosystems in general, including changes to community structure and composition, alteration of trophic structures and food webs, etc. While some of these are discussed where appropriate under each agent, any threatening process that alters habitat, inputs of nutrients or pollutants, introduces new biota, etc., is likely to have 'knock-on' effects. For instance, exploitation of a particular species may reduce the populations of non-target species to below the density needed to sustain their populations. Exploitation may also affect community structure and competitive interactions if key predator or herbivore species are targeted. Thus some threatening processes may affect particular taxa or suites of organisms directly (e.g., harvesting, toxic pollution, etc.) while others may affect some biota indirectly through alterations to their habitat, or through changes in community processes or structure.

The potential for synergistic effects, involving two or more different threatening processes, cannot be over emphasised. Similarly, cascading effects – where one threatening process leads to another – are also important. Both synergistic and cascading effects are discussed in more detail in Section 6.9 below.

While there is good information on some of the major threatening processes and their impacts on marine invertebrates a considerable amount is unknown and can only be surmised. In many cases the only well documented examples are non-Australian, although we have included information from Australian studies where possible.

Table 6.1: The main industries and activities associated with impacts on marine invertebrate fauna and examples of the types of impact that may occur under each of the major categories of threatening processes.

Industry or Activity (agent)	**Types of activities**	**Direct exploitation**	**Physical damage or alteration (including habitat destruction)**	**Pollution**	**Non-indigenous species**	**Climate change**
Fisheries and other forms of biotic exploitation	Commercial fisheries (esp. trawling and scallop dredging)	Over-harvesting of certain target species	Damage from trawling and dredging gear to non-target organisms (injury, death, burial by sediment plumes) and habitat (flattening, scraping, loss of 3D structure); bycatch; community effects (e.g., loss of predators, alteration to food chains)	Rubbish (bait boxes and associated ties, damaged nets, rope etc.) thrown overboard; oil spills, etc.	Accidental transport of non-indigenous species in nets or attached to hull; or in water in bait wells, recirculating tanks or ballast.	
	Recreational fisheries	Over-harvesting of certain species, including those used for bait?	Trampling (intertidal areas), disturbance from bait pumping etc	Rubbish and waste disposal, bait wells etc		
	Aquarium trade	Over-harvesting	Habitat damage during collection		Accidental or deliberate release of non-native species; accidental importation and release of non-target small invertebrates on corals, algae etc.	
Aquaculture	Intertidal or estuarine (e.g., oysters?), marine fish-farms (e.g., salmon?), land-based	Harvesting of some species, e.g., wild pearl oysters, for culture	Alteration of coastal habitat for construction of ponds, cages etc. (e.g., clearance of mangroves); siltation and smothering by wastes	Excessive nutrients and creation of anaerobic conditions (esp. in sediments); antibiotics, etc	Accidental releases of cultured species or of pathogens or other taxa accidentally included in stock; mixing of genetic stock (i.e. through movement of stock, e.g., oysters); translocation of species with aquaculture gear	
Shipping/transport			Injury and death of	Oil spills, dumping of	Transport by hull	Significant

			organisms and physical removal of habitat through anchor damage, groundings and shipwrecks; loss of habitat during construction of shipping facilities, (ports, marinas etc)	waste and rubbish; antifouling chemicals	fouling and ballast water	greenhouse gas emissions
Petroleum, gas or mineral exploration and production	Submarine drilling for gas and petroleum; dredging for sand etc		Damage and habitat loss during construction of platforms and associated infrastructure; smothering by drilling wastes and mud; damage from dredging or other mining activities	Oil, toxic drilling wastes; antifouling chemicals	Fouling organisms on ships or structures or transported in ballast water	Significant greenhouse gas emissions
Recreational use and tourism	Intertidal trampling and collection; snorkelling and SCUBA-diving; recreational boating; construction of tourist facilities; beach reclamation and "improvement", etc	Recreational collecting for food, bait, ornaments etc.	Physical damage from dive impacts, reef walks, moorings and anchoring; trampling of intertidal fauna; removal of subtidal sand during dredging for beach reclamation; loss of strand-line habitat from raking to remove rubbish and beach-washed material; loss of habitat for construction of tourist facilities	Oil spills, rubbish; waste; sewage; antifouling chemicals	Fouling organisms on boat hulls, translocation of species on anchors, or collected and discarded by divers or shore fossickers	Significant greenhouse gas emissions
Waste disposal	Sewage, industrial wastes, stormwater		Habitat damage during construction of pipelines etc; habitat alteration through increased turbidity, sediment loads and nutrients	Nutrients, heavy metals, organochlorides, oil and petroleum; radioactive and solid wastes		Methane emissions
Coastal development and modification	Construction and development (resorts, ports, marinas, canal		Clearance of coastal habitats for construction of facilities; disturbance	Sediment resuspension, sediment plumes; increased waste,		Significant greenhouse gas emissions

	estates, seawalls, wharves, etc.); reclamation of wetlands, mangroves etc; beach works; dredging of estuarine waterways; sand dredging for construction or fill		to subtidal habitats from dredging; changes to hydrography (from construction of retaining walls, groynes etc) resulting in changed patterns of sediment transport and deposition and wave action	sewage, stormwater etc. resulting from developments		
Land use in catchments (agriculture, urban development, industry, vegetation clearing, dams, excavation etc.)	Urban and industrial development (e.g., construction of housing, factories, roads, bridges); agriculture; vegetation clearing; dams; excavation etc.		Changes in volume of freshwater leading to change in estuarine habitats; loss of seagrasses etc. from increased sedimentation	Erosion and sedimentation; excessive nutrients (and eutrophication) from agricultural application of fertilisers; pesticides and herbicides; acid runoff from development of acid sulphate soils; urban runoff; industrial pollution; heavy metals		Significant greenhouse gas emissions

THREATENING PROCESSES

In line with Norse (1993) we recognise five major categories of threatening processes for marine invertebrates:

1. Direct exploitation;
2. Physical damage or alteration (including habitat destruction)[111];
3. Pollution;
4. Introduction of non-indigenous species;
5. Long-term climate change.

These differ from the five processes identified by Yen and Butcher (1997) for non-marine invertebrates in that their separate categories of habitat destruction and alteration are combined, and pollution is included as an additional category.

Although each of these categories can be considered independently, they are often correlated. For example, reclamation of mangroves not only destroys habitat but results in the incidental death of the organisms living there at the time and increases the likelihood of pollution by the loss of the filtering effects of the mangroves, leading to damage of adjacent seagrass beds and saltmarsh. In addition, the same threatening process can be the result of several different human activities.

In addition to these major categories, we include some discussion of threats with complex, unknown or debated origins, including diseases, parasites, and population outbreaks of destructive or "pest" species. These have been variously attributed to natural and anthropogenic causes or to some combination of these. Even if entirely "natural" in origin, they may achieve "unnatural" significance in areas already stressed or degraded by a variety of anthropogenic activities, and thus it is important that they also be given consideration.

Most of the above categories of threatening processes can be subdivided into primary and secondary processes (see below). Some of these processes can be associated with the main maritime industries and other uses of the marine environment; others (e.g., pollution, climate change) primarily result from human impacts in the terrestrial environment.

6.3 Overexploitation

Many marine invertebrate species are harvested, commercially and recreationally, for a variety of uses including food, bait, and ornament (see Sections 3.2.7 and 6.10.3). Economically important marine invertebrates include the many species harvested as "seafood" (e.g., molluscs such as oysters, scallops, abalone, and squid; crustaceans such

[111] We have not attempted to separate physical damage to organisms (e.g., indirect effects of fishing and trawling such as injury or death of non-target organisms) with physical damage to habitat, since the two are virtually inseparable for much of the marine environment.

as lobsters, prawns, and crabs; and holothurian echinoderms – trepang or bêche de mer), as well as species harvested for bait (worms, intertidal molluscs, crustaceans and ascidians or"sea squirts"), those collected for their use in collections, jewellery or other ornament (shelled molluscs, coral, some echinoderms and crustaceans), and species collected live for the aquarium trade (corals, molluscs, anemones etc.). Direct exploitation of marine invertebrates for human food is a major industry in Australia, the total catch in 1997-98 being at least 83 Mt and worth more than A$1.3 billion (ABARE 1998; see also Section 3.2.5).

Exploitation does not by any means always constitute a threatening process, and may be sustainable provided the rate of removal does not exceed the reproductive potential of the species, and the impact is not compounded by other effects such as habitat damage. There are many documented examples of marked declines due to overexploitation of populations or entire species of marine invertebrates, a few being detailed below (Section 6.3.2). Here we discuss the various effects of overexploitation, ranging from population decline, reproductive failure and change in population structure of targeted species, to changes in community structure and trophic interactions resulting from the removal of key predator or prey species. There are also a range of accidental, indirect and secondary effects that may result from the techniques or gear used to harvest marine species. These, including the accidental capture of non-target species (bycatch), and damage by fishing gear to individuals or habitats (e.g., trawling), are discussed in Section 6.10.2.

6.3.1 Effects of exploitation on target and non-target species and communities

This section deals with the effects of overexploitation on target species – both directly induced declines and those brought about indirectly through (for example) critical reductions in population density – and the effects of this removal of biomass on the communities and ecosystems of which the exploited species form a part. Other impacts of fishing not directly related to the effects of overexploitation (e.g., habitat damage from trawl gear) are dealt with later in the section on Fisheries (Section 6.10).

Population depletion / decline

Much has been written about the status of global fisheries and the difficulties in sustaining current catches. These apply mainly to pelagic and demersal finfish, but in general terms are applicable to marine invertebrates. Traditional fisheries models predict that species are generally protected from overfishing by commercial checks – i.e. exploitation alone is unlikely to drive a species to extinction since the fishery will become uneconomic (returns not worth the effort) before the last remaining individual is taken. Huxley expressed this idea as early as 1883 in an address to the International Fisheries Exhibition in London (Roberts and Hawkins 1999: p. 243). However, such commercial checks can only apply to single species fisheries, not to multispecies fisheries where rare species continue to be captured whenever they are encountered (Roberts and Hawkins 1999). Even in single species fisheries, rarity may not protect a species from heavy fishing. In cases where species show strong schooling behaviour or aggregate around particular habitat features, capture of the last individuals might still be commercially viable. The commercial assumption also takes no account of the fact that rarity may lead to increasing demand and market value; the occurrence of overcapitalisation and specialisation by fishers (leading to the "locking in" of effort and inability to diversify); or the development of markedly improved technology for locating and catching target species (Roberts and Hawkins 1999).

Worldwide, various species of crustaceans and molluscs are at present suffering severe declines due to overfishing. For instance, the Green Snail (*Turbo marmoratus*) and both Black-lip (*Pinctada margaritifera*) and Gold-lip (*Pinctada maxima*) Pearl Oysters were classified as commercially threatened in the 1994 IUCN Red List (Groombridge 1993), although the latter two species are the focus of carefully controlled pearling operations in Australia. In Australia, the Bass Strait scallop fishery is a fraction of its former size (Zacharin 1990) and the scallop fishery has been closed in Port Phillip Bay (Victoria) and in all of NSW. The eastern rock lobster fishery has reached low levels of profitability (Parker 1995) and the abalone fishery is at a third of 1970s levels (Parker 1995), although catches have remained stable for the last five years (NSW Fisheries 1998b). Examples of species that have suffered (or have the potential to suffer) declines as a result of overexploitation are detailed in Section 6.3.2.

Reproductive failure: the Allee effect

While, in theory, overexploitation alone should not be capable of driving a species to extinction, in many cases the critical threshold is not the capture of the last remaining individual but a certain level of population density below which the population is unable to sustain successful reproduction and recruitment. The term "Allee effect" is used for reproductive failure that occurs below a certain threshold of population density due to remaining individuals being unable to find mates, or an insufficient density of gametes in the water column to enable fertilisation. Thus, "recruitment overfishing" (fishing a population to below the density required for it to remain self-sustaining) can occur before the population actually becomes rare enough for action to be taken, and the population may continue to decline even when the fishery is closed. This effect has been implicated in the local extinction of giant clams (*Tridacna gigas*) from several regions of the Indo-Pacific, including Fiji, Guam, New Caledonia and the Northern Marianas (Wells 1997a); in the failure of stocks of Queen Conch (*Strombus gigas*) to recover following closure (Stoner and Ray-Culp 2000); and the near extinction of the White Abalone (*Haliotis sorenseni*) in California (Tegner et al. 1996; see Section 6.3.2 for details).

Thus, while estimates of reproductive output for a population are typically based on body size or gamete production alone (gamete production being positively correlated with body size), these may be misleading since zygote production also depends upon fertilisation success. For instance, while the prevailing assumption is that high population density leads to reduced per capita zygote production, Levitan (1991) found, in a field study of the sea urchin *Diadema antillarum*, that increased fertilization success at high densities could compensate for decreased gamete production. Thus, small individuals at high population density had similar per capita zygote production as large individuals at low population density.

Changes to population structure and characteristics

Heavy collecting, even if sustainable, can deplete local populations or, at least, change the population structure. Where harvesting is discriminate (e.g., selective of a particular size class, through imposition of size limits, nature of gear, or hand collecting), it acts as a form of selective pressure that may potentially alter the structure of the population. For example, recently fossilised Queen Conch shells in the West Indies were found by Stager and Chen (1996) to be significantly larger than those currently being harvested, and very large individuals were proportionately fewer in living populations, possibly due to recent overfishing.

Indirect effects on communities and trophic interactions

The effects of fishing on marine ecosystems, including effects on trophic interactions resulting from predator and prey removal, species replacement, scavengers and discards, etc., were reviewed by Jennings and Kaiser (1998). These authors observe that "One of the most widely expressed concerns about the intensive and selective fishing activities of humans is that they will lead to imbalances in ecosystem function which have ramifications for non-target species". Many harvested marine invertebrate species play important ecological roles in their communities, for instance as predators, scavengers or prey species, and their loss, through overexploitation, can have significant consequences for community structure and trophic interactions.

The harvesting of prey species can indirectly impact upon their predators. In the Antarctic, developing fisheries for invertebrates such as krill and squid are managed, under the Convention on the Conservation of Antarctic Marine Living Resources 1980 (CCAMLR), with the aim of restricting impacts on dependant predatory species such as marine mammals and seabirds (e.g., Everson and de la Mare 1996; Rodhouse 1997).

Griffiths and Branch (1997), in a review of the exploitation of coastal invertebrates and seaweeds in South Africa, found that effects on community dynamics are far less well appreciated than effects on target species, though they can be equally dramatic and may carry major economic implications. For example, declines in rock lobster, sea urchins and abalone are easily understood but the fact that a complex web of biological interactions links them is not.

6.3.2 Examples - effects of exploitation on target species

In the following section we provide a brief discussion of several specific examples, from both commercial and semi-commercial or recreational fisheries, to illustrate some of the effects over exploitation can have on targeted marine invertebrate species, and the conservation and management issues that arise. These examples are not exhaustive and the issues involved are generally applicable to a number of fisheries. For a discussion of the different types of fisheries (commercial and recreational), together with their "accidental" or secondary effects and management options, see Section 6.10.

Scallops

Scallops (Pectinidae) are characterised by fluctuating recruitment and wide and patchy distribution, making it notoriously difficult to manage these fisheries. A few species of scallops are fished commercially in Australian waters but the example below relates to the fishery based on *Pecten fumatus* in the Bass Strait area.

In Bass Strait, the status of scallop stocks is still classed as "uncertain" following a long history of over-harvesting and stock depletion (Caton et al. 1998). During the 1970s, the discovery of new beds in the central Bass Strait area led to the rapid expansion of the scallop industry. The total catch peaked in 1982-83 at close to 12,000 tonnes, with the number of participating vessels tripling in two years to 231 (Young and Martin 1989; Zacharin 1990). However, the catching capacity of the scallop fleets had developed to the point where once a bed was located, fishing effectively removed it (Caton et al. 1998), making the industry unsustainable. The main beds were depleted by 1985, the last major bed – Banks Strait – was fished out during 1986, and the fishery had effectively collapsed by 1987, when Tasmanian vessels landed less than 500 tonnes (a drop of 95% in 6 years) and Victorian vessels landed 220 tonnes (a drop of 90% over the same period) (McLoughlin 1994). Closures were introduced, as surveys found severe stock depletion and a lack of recruitment, with little sign of scallop beds in some areas (McLoughlin 1994), and the fishery was divided into three management zones, a Tasmanian and Victorian zone (extending 20 nautical miles off each State's coast), and a Commonwealth-managed Central Zone. There appeared to be some improvement in 1993-94, but this may have been largely due to an extraordinary settlement event (Zacharin 1994), and there is uncertainty as to the true extent of recovery (Caton et al. 1998). For instance, in the Central Zone (Commonwealth) fishery, the catch rate increased from about 3 bags per hour dredging in 1994-95 to about 5 bags per hour in 1997, but meat yields decreased from 1047 tonnes in 1994 to 690 tonnes in 1997. Total wholesale value of the 1997 catch was A$10 million, compared with A$20 million in 1995 and 1994.

During the 20 year history of the Bass Strait fishery few, if any, commercially fished beds have supported exploitation for more than two consecutive seasons. Single recruitment events result in discrete scallop beds of single year classes quickly fished out before they attained full spawning potential (McLoughlin 1994; Caton et al. 1998). Exploited beds do not, in general, appear to regenerate or provide additional year classes of scallops in the time frame of the current fishery (McLoughlin 1994). The over-harvesting of stocks is compounded by the damage caused by the scallop dredges, with evidence from dredge trials showing that up to 50% of scallops in the dredge's path may be damaged (Zacharin 1994). Damaged scallop beds attract starfish predators and *Vibrio* infection and tend to die out if not harvested (Zacharin 1994). Damage to the substrate and benthic communities by the dredges is considerable (e.g., Harris and Ward 1999) with a study in the UK showing that the majority of the damage to the larger invertebrates is during the dredging operation itself and remains unobserved on the seabed rather than in the bycatch (Jenkins et al. 2001).

The Bass Strait Scallop Consultative Committee, designed to coordinate management approaches among the three jurisdictions, developed a five-year strategic research plan in 1998 that identified the development of an "environmentally friendly" scallop dredge as being one of the priority research areas. A new five year strategic research plan was developed this year (2002).

In NSW, commercial harvesting of *Pecten fumatus* peaked in 1970/71 but Gorman and Johnson (1972) concluded that a single recruitment event was responsible for the fishery and without further recruitment the commercial prospects were not good. Their study led to the perception of the "boom and bust" nature of the scallop fishery in NSW (Williams et al. 1993). Scallop fisheries elsewhere in Australia and the world also follow this pattern (Fuentes et al. 1992). The scallop fishing industry in Jervis Bay has been closed down since late 1991 after over-harvesting reduced stock levels to the point where fishing became uneconomic (Fuentes 1994).

In Queensland, Dredge (1988) suggested that scallops (*Amusium balloti*) have been subject to recruitment overfishing, the effective effort directed at the scallop stock having increased by a factor of 14 between 1977 and 1987, while the annual catch fell from a peak of 1220 tonnes in 1982 to 450 tonnes in 1987. *Amusium* is also fished in Western Australia (Harris et al. 1999) and work has been carried out on its aquaculture potential (Cropp 1993).

The aquaculture of *Pecten* theoretically has the potential to alleviate some of the pressure on wild populations, although it does not address the problem of the large investments already made in the commercial wild scallop fisheries sector. Currently the culture of scallops is not as well developed as that for mussels or oysters. In Tasmania, for example, it has been conducted on parts of the east coast for about ten years, but there are only two enterprises growing significant quantities of scallops, and they rely largely on collection of spat from the wild (DPIWE 1999).

Abalone

The most valuable gastropod fishery is that for abalone (*Haliotis* spp.), for which there is an extensive literature. International symposiums on abalone biology, fisheries and culture have been held in 1989 (Shepherd et al. 1992), 1994 (Shepherd et al. 1995), 1997 (Cook et al. 1998) and 2000. The first Australasian abalone symposium was held in New Zealand in 1996 (Shepherd et al. 1997).

In southern Australia, populations of the Greenlip Abalone (*H. laevigata*) have suffered severe declines due to commercial overexploitation (S. Shepherd pers. comm.). In South Australia, about 75% of fished reefs have declining catches, and 50% of fished reefs have catches which have declined by >50%. The mean percentage decline in catch of these reefs is 70%. Reports over the last five years have reiterated the serious nature of the declines, but nothing has been done (S. Shepherd pers. comm.; Shepherd and Baker 1998; Shepherd and Rodda 2001; Shepherd et al. 2001). In Victoria, Greenlip populations have not been fished since the late 1970s and currently have the status of rarity. Populations around Flinders Is., Tasmania, have declined on average by 40-50%

(Shepherd et al. 1998). Greenlip populations in WA have also declined but the declines are largely undocumented, though they can be seen by examining the catch data over 20 years (S. Shepherd pers. comm.). Aquaculture of abalone is occurring in Australia (e.g., Westaway and Norriss 1997; Southgate 2000) and elsewhere, which may ultimately reduce the pressure on wild populations.

The White Abalone: The well-documented case of the once abundant Californian White Abalone (Tegner et al. 1996; Malakoff 1997; Davis et al. 1998) shows how a highly fecund invertebrate can be fished to near extinction. Abalones were extremely abundant in California before diving and effective trap fisheries (Dayton et al. 1998). The rapid spread of modern diving technology in the mid twentieth century greatly facilitated abalone exploitation (Davis et al. 1998; Dayton et al. 1998). In the early 1970s, California divers and shore pickers gathered 4,000 to 5,000 tonnes of abalone each year, creating recreational and commercial fisheries worth more than US$20 million annually. As populations of pink (*H. corrugata*) and red (*H. rufescens*) abalone declined, divers targeted deeper reefs for the even more valuable white abalone (*H. sorenseni*), with commercial divers alone landing more than 60 tonnes in 1972. After just 7 years, adult populations were exhausted and catches dropped to near zero. There was little concern as it was believed that there would be recruitment from deeper reefs out of reach of divers. It was also believed that the high fecundity of females (producing 3.6-6.5 million eggs a year) meant that only a few survivors would be needed to restore populations. However, these beliefs are now repudiated (Davis et al. 1998). Abalones are currently so scarce that all five species fished in southern California have been closed to both sport and commercial harvest, and there is good reason to believe that the white abalone will become the first marine invertebrate known to become biologically extinct as a result of human fishing (Tegner et al. 1996). Davis et al. (1998) searched 107, 650 m^2 of white abalone habitat at 39 locations around the California Channel Islands, the species' historical centre of abundance. At 25-42 m, where mean densities in the 1970s were 2,000 to 10,000 white abalone per hectare, they found a mean density of 1.6 ± 0.5 ha^{-1} in the early 1990s. Following a 270 tonne commercial harvest in the 1970s, landings virtually ceased. No fishery-independent population assessment was made until 1992-93, and the fishery remained open until 1996. The management scheme, based on a minimum harvest size of 153 mm and a closed season during spawning, apparently failed to protect adequate spawning stock density. The population has not recovered from the harvests, and the survivors are dying of old age. Spontaneous recovery is highly unlikely, even in the absence of continued harvest. Davis et al. (1998) concluded that active management intervention would be required to prevent extinction and to restore the species to a viable status. A program of captive rearing and refugia-based management was identified that incorporated a public education component and existing governmental processes, as necessary to restore the white abalone population.

Other molluscs

Molluscs are commercially fished for food, for shells or shell products (e.g., mother of pearl) and pearls. Bivalves comprise some very important commercial fisheries, including pearl oysters, oysters, mussels, giant clams, pipis and other "clams". Some species of giant clams have become locally extinct in parts of the Indo-Pacific and overfished on the

GBR (Braley 1987). The threat of extinction of tridacnid clams was recognised by the IUCN in 1983. Widespread gastropod fisheries include *Trochus* (for meat and shell) and *Turbo* (meat) fisheries in tropical Australia and the Indo-west Pacific in general (Wright and Hill 1993; Sant 1995; Hutchings and Salvat 2000). Both *Turbo* and *Trochus* are protected species in French Polynesia but the legislation is not enforced (Hutchings and Salvat 2000), and populations have declined but are strictly controlled in tropical Australia. On the Great Barrier Reef there are currently six operators licensed to collect *Trochus*, and these take about 170 tonnes annually (GBRMPA 2000; S. Breen pers. comm.). As with abalone and Giant Clams, these large gastropods are simply picked up from the sea floor and are easily depleted. Some harvested burrowing bivalves have also been seriously depleted, such as the New Zealand Toheroa (see below).

Large-scale commercial harvesting of cephalopods is a relatively recent industry in Australia, with large-scale commercial fisheries developed worldwide only in the 1950s and 1960s and production and demand have increased steadily. There have been two well-known "failures" in squid fisheries, one species in the NW Pacific in the 1970s, and another in NW Atlantic in 1980s (Lu 1998). While ground fish landings have declined globally, cephalopod catches have increased. Lipinski et al. (1998) and Caddy and Rodhouse (1998) point out that most coastal and shelf cephalopod fisheries are likely to be fully exploited or over-exploited. Worldwide, various cephalopods are targeted in commercial fisheries, which have expanded enormously since about 1960. A large percentage of catches are made up of squids (Mangold et al. 1998). In Australia at present, squid are targeted in jig fisheries and a range of cephalopod species are caught as bycatch of trawl fisheries. At present in Australia, squid are targeted in jig fisheries and are caught as bycatch of trawl fisheries (Harris and Ward 1999).

What little management there is of Australian cephalopod fisheries, it is based on rather poor data. Little is known about their biology, population structure[112] or ecology of the commercially important species, let alone those caught incidentally. There are even undescribed species of octopus and squid in commercial landings. A five year strategic research plan for the southern squid jig fishery was released in 2002 by the Australian Fisheries Management Authority.

New Zealand Toheroa: In New Zealand, the Toheroa (*Paphies ventricosum*), an infaunal bivalve found on sand surf beaches along most of the western coast, has been heavily exploited by public and commercial (canners) harvesters since the early 20th century (Cassie 1955; Waugh and Greenway 1967). The Toheroa was once highly abundant on some beaches, but dramatic declines in intertidal populations have occurred on various occasions, being mainly attributed to over-exploitation, destruction of the beds by wave action, spatfall failures, and mass mortalities (Waugh and Greenway 1967). For example, along Ninety-mile Beach (North Island), 87km of good Toheroa beds were recorded in 1920, and, despite mass mortalities in 1930 and 1932, the number of living toheroa in 1933 was estimated as in the order of 30 million. Up to 1.5 million toheroa were being

[112] Recent allozyme studies on the population structure of the Southern Calamary (*Sepioteuthis australis*), the basis of a fishing industry, show it is comprised of several discrete stocks - whereas currently it is managed as a single stock (Triantafillos and Adams 2001).

taken annually by a canning factory. In 1939, following another mass mortality, the population was estimated as around 12 million, with 0.5 to 1 million still being taken annually for canning. In 1946 total numbers were estimated at 6 million, and by 1948 numbers had been reduced to such an extent that R. G. Rule, a cannery manager, was reportedly able to find only one living specimen on 53 km of beach (Cassie 1955). Stocks still remain on a few South Island beaches (Millar and Danette 1995).

Holothurians ("sea cucumbers" or bêche de mer)

The sea cucumber fishery in the Pacific was reviewed by Preston (1993) and Sant (1995). Concerns have been raised (e.g., Conand 1997) about the sustainability of holothurian fisheries in general and some conservation groups have raised this issue in regard to the Great Barrier Reef area. There seems to be little available information on the sustainability of the Australian fisheries, but in some other parts of the world, stocks have collapsed due to overfishing. Holothurian fisheries have a very long history and annual world catches are now around 120,000 tonnes and are valued at over US$60 million (Conand 1997; Hamel et al. 2001). While the large catches could be interpreted as a sign of sustainability, there are also suggestions that current catches are of increasingly inferior quality (Conand 1997). A dozen Indo-Pacific coral reef species constitute the major part of world catches of these export fisheries, which are nonetheless poorly documented and not well-managed (Conand 1997; Hamel et al. 2001). With the increasing market demand, there is a real risk of biological overexploitation occurring well before economic over-exploitation. For the New Caledonia fishery, customs records show a decline from 1992 that does not appear in the FAO statistics (an example of the difficulty of getting reliable data). Conand and Byrne (1993) showed a shift in species collected, from high to medium quality, due to harvesting. Previous studies by Conand (1990; 1995) have shown that the holothurian resource is very vulnerable and that the maximum sustainable yields are probably low, e.g., only a few dozen kg per hectare per year.

Nevertheless, there is a small licensed industry operating on the GBR. The main target species are Black Teatfish (*Holothuria nobilis*) (Uthicke and Benzie 2000b), Sandfish (*H. scabra*), White Teatfish (*H. fuscogilva*) and Prickly Redfish (*Thelenata ananas*) (Uthicke 1996). Many species have not yet been carefully studied, although recent work (Uthicke and Benzie 2000a) shows that there is a high level of gene flow amongst populations of *H. nobilis* of the GBR. Growth and mortality rates are still mostly conjectural and other aspects of their biology and ecology are poorly understood, including the impact from trawling (Makey and Slater 1997). Other problems regulating this fishery in Australia have resulted from illegal fishing, mainly from Indonesia and Papua New Guinea. Certainly some harvesting by Indonesians occurs on Ashmore reef (Russell and Vail 1998), where only collecting using traditional methods are allowed.

King Crabs

King or Giant Crabs (*Pseudocarcinus gigas*) are unique to southern Australia, living in depths to 600 m but mainly at 130-370 m in a "shelf-break" habitat rich in bryozoans. This rather restricted habitat is certainly sensitive to the impact of demersal trawling (Caton et al. 1998). King Crabs were an occasional bycatch of Southern Rock Lobster

(*Jasus edwardsii*) pot fishers and, more recently, demersal trawlers, but are now commonly targeted by crab potting operations. Export markets have developed and the price has reached $52/kg, making the 1996-97 catch of about 300 tonnes worth up to A$11 million.

The fishery is located in waters off southern Australia, outside the 3 nm state waters limit of Victoria, Tasmania, South Australia and Western Australia. Previously managed by the Commonwealth, this fishery is now the responsibility of the States and a draft management plan was released in 1998 (DPIWE 1998). Although many rock lobster licenses have endorsements to take crabs, only about 25 operators take about 80% of the catch.

The status of King Crab stocks is uncertain; they may be at risk from over-potting, increased effort, or habitat degradation from demersal trawling. Depletion in some areas of the fishery, increased fishing in recent years and uncertainty about the stocks, resulted in management measures being introduced in the mid-1990s. These included area and seasonal closures, a minimum size limit, prohibition on the taking of crabs carrying eggs, and prohibition of the dismembering of live crabs (Caton et al. 1998). The Tasmanian King Crab fishery is managed by a quota system, with an initial TAC (Total Allowable Catch) of around 100 tonnes per annum (about 85% of the average catch from 1997 and 1998), until there is enough research to determine a scientifically-based TAC (DPIWE 1998).

6.3.3 Information, research needs and management

It is difficult to evaluate the degree of threat posed from harvesting for most marine invertebrates because there is a need for :

- Accurate catch data (both professional and amateur);
- Knowledge of species biology and population genetics, structure and size;
- Historical data;
- Suitable reference areas for comparison of exploited with unexploited populations; and
- (in some cases) certainty of taxonomic status.

In Australia, accurate figures on commercial landings of invertebrates are notoriously difficult to ascertain. Landings at fish co-ops are probably always underestimates, as fishermen may sell some of their landings privately, and may also underestimate their landings as these figures are used for management and tax purposes. Data do not usually provide information about where individual species are caught, nor any estimate of the time or effort taken to land the catch, making accurate estimates of increases or decreases in landings per unit effort virtually impossible. In the case of some fisheries (e.g., prawns and squid), several species may be included under one heading making species-level analyses impossible. The situation is infinitely worse with amateur fishing.

Pauly (1995) has argued that we have little idea of the full extent to which fisheries have reduced populations of marine species, even high-profile non-target species like mammals, because we have no proper baseline against which to judge declines. Without control areas, it is very hard to gauge the true impact of fishing of extant populations. In addition, it is apparent that many invertebrates have large natural population fluctuations, making it difficult to determine whether these populations are "declining" due to harvesting or whether catch levels are sustainable. Better knowledge of the biology, ecology and population dynamics of such species can assist in modelling responses to, for example, environmental variables, and thus predicting stock levels and setting TACs. Such models have been developed for prawns (e.g., Rothlisberg et al. 1985; Staples et al. 1995; Wang and Die 1996) and have been moderately successful (Caton et al. 1998).

The systematics of some commercially important taxa is uncertain. This has important ramifications when dealing with translocation of breeding stocks, comparison of fisheries data etc. Some examples from commercially important bivalves illustrate this point.

- The commercially fished species of scallop (*Pecten*) may be a single species in temperate Australia or may consist of a species complex. Its relationships with a very similar commercial 'species' in New Zealand are also unresolved.
- The valuable eastern Australian rock oyster (*Saccostrea*) fishery is largely based on a species that has had a confused taxonomic history. For many years it was thought to be a separate species from the northern New Zealand rock oyster but is now thought to be conspecific. However, its relationships with similar populations in other parts of Australia and the Indo-Pacific have yet to be resolved.
- The NSW pearl oyster (*Pinctada*) has been shown to consist of two very similar species, one of which is conspecific and genetically very similar to the commercially valuable Japanese pearl oyster (Colgan and Ponder in press).

6.4 Physical damage / habitat alteration and loss

In the terrestrial environment, the issue of habitat destruction has received a great deal of attention, and is frequently considered the primary threatening process for the majority of terrestrial animals and plants (e.g., Glanznig 1995; Yen and Butcher 1997). Habitat destruction also is a serious issue in marine ecosystems (e.g., GESAMP 1997). Whilst habitat damage is most obvious in the coastal fringe, it is not confined to this zone, with fishing activities having also caused considerable modification of habitats in shallow sublittoral areas in harbours, bays and coastal areas, as well as on the continental shelf and slope (Watling and Norse 1998b; Jackson et al. 2001).

Physical damage and loss of habitat in the marine environment directly results from human activities such as trawling, dredging, mining, ship groundings, anchor damage, and (on a smaller scale) recreational uses such as reef walks and SCUBA diving. However, habitat modification can also be an indirect or secondary consequence of storms and other natural events, or changes (natural or human-induced) to water quality, hydrography, or community structure and ecological interactions. The specific effects of each relevant human activity or industry are discussed separately in sections 6.10 – 6.17. Trawling is undoubtedly the primary form of physical disturbance for soft-sediment marine benthic communities and has been likened to the clear felling of forests (Watling and Norse 1998b). In this chapter, the impacts of trawling are is discussed in section 6.10.2 and the other activities listed above are discussed in the following sections. This section briefly outlines the various forms of habitat alteration, fragmentation and loss, evidence for their effects on marine invertebrates, and the research needs relating to these as threatening processes.

6.4.1 Effects of physical damage and habitat alteration

Damage to habitat-forming organisms

As discussed in Chapter 5, a key difference between terrestrial and many marine habitats is the relative significance of plants as providers of invertebrate habitat. In the former, most habitats are comprised primarily of higher plant communities, whereas in the oceans photosynthetic life is restricted to the upper (euphotic) layer and relatively few coastal zone habitats are dominated by angiosperms (mangroves, saltmarshes and seagrasses) or macroalgae. Despite the diversity and ecological importance of these plant-dominated habitats in shallow marine ecosystems, the habitats of many marine invertebrates, even in shallow water, consist instead of substrates such as sand and rock, or assemblages of other invertebrates. Indeed, in many marine habitats it is the invertebrates, particularly those sessile and epifaunal (emergent), that provide most of the three-dimensional structure and spatial heterogeneity that comprise “habitat” for other organisms (see Section 5.3.2). Thus while the destruction of vegetated habitats such as seagrasses, saltmarshes, mangroves and algal communities (Section 5.3.2), presents a problem analogous to that of vegetation clearance on land it is a somewhat different case for the fauna of rocky substrates, soft sediments and coral reefs. In these cases, activities resulting in habitat destruction are also responsible for direct damage or death of many components of the invertebrate fauna, particularly habitat-forming taxa such as epifaunal sponges, bryozoans, corals, bivalves and tube-worms.

Modification of habitat structure

Many undisturbed marine habitats have a well-developed three-dimensional structure and heterogeneity, providing niches for a diverse array of organisms. Activities such as trawling and dredging, in particular, grossly simplify habitat structures either by uprooting, crushing and flattening epifaunal species or by disrupting sediment stratification, burrows and other structures for infaunal species (see Section 6.10.2).

Habitat fragmentation and loss

The consequences of habitat fragmentation (e.g., Young and Clarke 2000) have received a great deal of attention for terrestrial communities but have been largely ignored for marine organisms and communities because of their (perceived) greater capacities to disperse among fragments (but see Sections 3.3.1 and 4.4.3). The effects of habitat fragmentation in the marine environment are very poorly understood and are likely to vary considerably depending on the habitat, the taxon, the distances being considered and the physical parameters (e.g., coastal configuration, currents).

Many marine invertebrates have good dispersal ability, typically by way of planktonic larvae, leading some authors (e.g., Jones and Kaly 1995) to suggest that marine species are relatively unaffected by habitat fragmentation, with habitat *degradation* being more of a key issue (Bowen 1997b). However, others (e.g., Roberts and Hawkins 1999) point out that many taxa do not have planktonic larvae or high mobility as adults. Many species have direct development, often with brooding, and can disperse only metres from their parents. Others have larvae only briefly planktonic that can disperse over a few hundred metres to a few tens of kilometres. Habitat loss can greatly reduce the chances, even for those with relatively good dispersal, of finding suitable places to settle and live.

The extent of larval dispersal and the consequences of habitat fragmentation need to be understood in order to determine:

- The effect that size of remaining patches has on taxon diversity;
- Whether dispersal between patches is sufficient to allow recolonisation in the event of a catastrophic elimination of the fauna, either through ‘natural’ (e.g., a cyclone) or anthropogenic (e.g., dredging) causes; and
- Whether there is sufficient gene flow between patches to ensure retention of the full complement of genetic diversity.

All three questions are relevant to the conservation of marine biodiversity but involve differing spatial and temporal scales. For instance, Sammarco and Andrews (1988), in their *Helix* experiment on the Great Barrier Reef, found that, over evolutionary time, there was sufficient dispersal of coral larvae to ensure gene flow to reefs hundreds or possibly thousands of kilometres apart, but in the short term reefs appear to be essentially self-seeding, although local currents and weather patterns can be very influential (Hughes et al. 1999; Bellwood and Hughes 2001).

Global warming, coral bleaching and overfishing are capable of changing reef biodiversity and reducing the quality of reefs over large areas. If coral reef biodiversity is to be adequately conserved, we must protect habitats over large regional-scale areas (e.g., Bellwood and Hughes 2001).

Prediction of the response of organisms to the increasing threat of habitat fragmentation depends on scale- and habitat-dependent ecological patterns, as well as the reproductive capacity and dispersal ability of the individual faunal components. Unfortunately, there are no data for most taxa on how large or close the fragments need to be to enable effective gene flow or recolonisation. This clearly poses a problem for the effective design of marine protected areas (Section 5.5.3).

In summary, while there are relatively little experimental or observational data, the available information suggests that the potential consequences of habitat fragmentation are:

- Loss of taxonomic diversity;
- Loss of genetic diversity; and
- Greater susceptibility to localised environmental catastrophes.

Effects on taxonomic diversity

The size of remaining patches and their degree of isolation from related habitats can determine the diversity of the species supported, an effect well documented on land through island biogeographic theory (MacArthur and Wilson 1967). For instance, 75% of documented terrestrial extinctions since 1600 have occurred on islands, even though islands support a small number of species relative to continents (Calow 1998). Habitat patches may also act as "islands", with the number of species retained as a function of the size of the patch. The effect is less well documented in the marine environment. Small stands of mangroves do not develop the habitat complexity or support the entire fauna of large stands. Similarly, fragmented patches of seagrass lack the fully developed infauna associated with well-developed seagrass beds (PAH pers. obs.). The minimum size required for isolated pockets of habitat, such as wetlands within an estuary, to sustain functioning habitats is unknown, but is probably highly variable through dependence on many factors.

Indirect evidence for the importance of 'patch' size comes from a study by Edgar and Barrett (1999) on the effects of marine reserve declaration on reef fish, invertebrates and plants in Tasmania. They found that numbers, densities and mean sizes of certain species increased significantly at the largest reserve (Maria Is, 7 km coastline length) relative to external unprotected sites, whilst changes in species richness were not detected in any of the three smaller reserves (1-2 km length).

Effects on colonisation and recolonisation

Successful colonisation of a habitat by larvae is critical for the continuing existence of that taxon in that community. Many experimental studies have demonstrated the complex interactions between larvae and substrates, including highly specific "choice" by some species. These interactions can be very complex, being influenced by currents, chemical stimuli, substrate, and micro-environmental factors such as local eddies etc. (e.g., Hughes et al. 1999).

Eggleston et al. (1999) used artificial habitat plots of different types (oyster shell, seagrass and mixed habitat) and sizes to test hypotheses about macrofaunal colonisation. They found that macrofauna responded to habitat patchiness according to habitat type, experimental site, species, taxon, functional group and animal body size. For small species, there was a disproportionate reduction in faunal diversity in small compared to large patches (of oyster shell), which they felt "heightens concern over the negative impacts to biodiversity through large-scale fragmentation of (habitats such as) subtidal oyster reefs in certain regions".

Effects on retention of genetic diversity

There is a considerable amount of genetic structuring in many marine taxa (see Ayre 1995 for an introduction to the genetics of marine animals) with many studies on individual taxa (e.g., Rose-Velez and Suarez-Vasquez 1991; Brazeau and Harvell 1994; Palumbi 1994; Burnett et al. 1995; Bucklin et al. 1996; David et al. 1997; Palumbi et al. 1997; Ayre et al. 1997a; Benzie 1999; Pernet 1999; Staton and Rice 1999; Hancock 2000; Huang et al. 2000a; Kyle and Boulding 2000; Lessios et al. 2001; Triantafillos and

Adams 2001; Uthicke and Benzie 2001). Consequently, there is a risk that large-scale loss of habitat could result in the loss of genetic diversity. However, there are issues as to the relevant levels of differentiation and the practical application of these data to conservation planning (e.g., Pearman 2001). Conversely, some studies show that, for some taxa, that there is little genetic variation over large distances[113].

6.4.2 Examples – effects of habitat loss in marine ecosystems

The conservation issues associated with various marine habitats have already been summarised in Chapter 5. Some brief examples relating to two key habitat types are given below to illustrate the diverse problems associated with habitat destruction and fragmentation.

Unlike many of the highly visible changes in coastal marine ecosystems, changes to benthic habitats in the deeper sublittoral, continental shelf and slope zones are 'invisible' and hidden from public or political attention. There are several activities that impact locally on these habitats including sewage disposal from ocean outfalls (Section 6.6.1), the dumping of rubbish at sea (Section 6.6.1), shipwrecks, pollution from oil and cargo (Section 6.12), offshore oil exploration and mining (Section 6.13), etc. There are also significant impacts in particular areas from scallop dredging (Sections 6.3.2; 6.10.2). By far the most significant and widespread impacts, however, result from trawl fishing (Section 6.10.2).

Coastal wetlands such as mangroves, seagrasses and saltmarshes are highly productive and important as nursery grounds for a large range of species (e.g., Nagelkerken et al. 2001). However, their location in sheltered bays and estuaries has meant that large areas of these habitats have been destroyed or heavily impacted, so that their maintenance is a critical issue in Australia and elsewhere. Destruction or alteration of vegetated habitats occurs not only through deliberate clearing and filling (e.g., mangroves – see Section 5.3.2) or dredging, or as an unintentional secondary impact of activities such as trawling (see Section 6.10.2), but also through more subtle long-term processes such as water quality decline, siltation, disease, and changes to hydrographic and climatic regimes.

Seagrasses

Seagrasses are important habitats for many organisms (see Section 5.3.2; Butler and Jernakoff 1999). They tend to occur in sheltered areas such as estuaries, which is often where coastal development is concentrated, and many are subject to a range of associated threats including urban and industrial run off, dredging and burial, and damage from recreational activities such as bait collecting and boating. Any activities which increase water column sediments, nutrient load, and phytoplankton or epiphyte density, can lead to an attenuation of light, reduced seagrass survival and consequent loss of infauna (Hutchings et al. 1991).

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[113] E.g. Forbes (1999) on a study of the Giant Tiger Prawn, *Penaeus monodon* in he SW Indian Ocean.

Reusch and Williams (1999) describe the synergistic effects between eelgrass fragmentation and the proliferation of non-indigenous species. Reduction and disturbance of seagrass habitat may make it more susceptible to disease, which can, in extreme cases, lead to the extinction of seagrass habitats over very large areas. This occurred along the eastern seaboard of the USA in the early 1930s, and led to the extinction of a limpet (Carlton et al. 1991). The demise of such habitats are a threat for any of the seagrass-associated fauna, including a range of epiphytic species such as bryozoans, hydroids and polychaetes.

The loss of seagrass beds in a number of locations around Australia was documented by Kirkman (1997) and summarised in the 1996 State of the Environment Report (State of the Environment Advisory Council 1996) (see Table 6.2). Kirkman (1997) calculated that, of the approximately 51 000 sq km of seagrass meadow in Australia, human activities had caused the loss of about 450 sq km and natural events had caused damage to the extent of 1000 sq km in the previous ten years. One area where significant losses have occurred, and the effects upon the associated invertebrate fauna are well-documented, is Royal Princess Harbour in WA. Substantial losses of seagrass have occurred at this site over the past two decades (Walker et al. 1991). Wells et al. (1991b) found that the loss of seagrasses led to sharp declines in benthic marine invertebrate biomass (from 1937 to 340 tonnes dry weight) and production (from 279 to 55 tonnes dry weight per year).

Light deprivation and increased turbidity (either from phytoplankton growth in response to nutrient loadings, or suspended sediments) has been cited as cause of seagrass losses in many areas, including Lake Macquarie and the Clarence River in NSW, and Hervey Bay in Queensland (Table 6.2; see also Section 5.3.2).

Table 6.2: Loss of seagrass by area, with a brief description of the probable cause of the loss (after Kirkman 1997).

	Location	Area lost (ha)	Percentage loss	Probable cause
1	Cockburn Sound	3 300	79	Elevated nutrients from factories, sewage and abattoirs
2	Princess Royal Harbour	810	66	Elevated nutrients from factories and sewage
3	Oyster Harbour	720	46	Elevated nutrients from farm run-off
4	Gulf St Vincent	7 000	?	Sewage and stormwater discharge; coastal works
5	Westernport Bay	17 800	85	Siltation
6	Birch Point	397	?	?
7	Ralphs Bay	430	?	?
8	Pittwater	1201	?	?
9	Norfolk Bay	2148	?	?
10	Botany Bay	257	58	Erosion, coastal works, elevated nutrients and sea urchin grazing
11	Lake Macquarie	700	44	Increased turbidity
12	Clarence River	445	60	Increased turbidity and general decline in water quality
13	Hervey Bay	100 000	?	High turbidity from flooding of the Mary and Burum Rivers
14	Torres Strait	>10 000	?	Floods in 1991–1992
15	West Island–Limmen Bight	18 300	20*	Damage from cyclone Sandy, 1985; much of the area had recovered by 1994

*Note: This loss represented 20% of the seagrass of the Gulf of Carpentaria.

Coral reefs

Because of the huge numbers of species associated with coral reefs (see Section 5.3.2), processes that lead to death or damage of reef systems are significant for marine biodiversity. The reefs of the GBR, as with other reef systems worldwide, are suffering degradation from multiple stresses, including fishing (e.g., Wachenfeld et al. 1998), increased sea temperatures (e.g., Hoegh-Guldberg et al. 1997a; Berkelmans and Oliver 1999; see Section 6.7; Hoegh-Guldberg 1999), predation, storm damage, and pollution, including nutrient enrichment (see Section 6.6.1).

The destruction of reefs through blasting for building material or as a method of fishing, which occurs in some tropical countries, is a critical issue for coral reef conservation worldwide. While such activities as "blast fishing" are minimal and mostly illegal in Australia, they do pose a significant problem in many of the reef systems to our north.

Another destructive practice that is of little if any significance to Australian coral reefs is the use of cyanide to stun fish for the aquarium trade or kill fish. Cyanide fishing is highly destructive since, in addition to killing fish, it affects the zooxanthellae of the coral and leads to coral death (Jones and Hoegh-Guldberg 1999).

In Australia, deliberate habitat destruction in coral reefs is minor compared with less intentional impacts such as incidental damage from boating and shipping (including ship groundings with resultant loss of cargo and/or fuel), construction of ports, marinas etc., as well as degradation through bleaching, water quality decline, siltation and so forth.

Stone et al. (1996) modelled some of the important ecological processes in coral reefs – including colonisation, mortality and competition for space – and used these to predict the effects of destroying portions of the spatial landscape on reef biodiversity and species extinction. The results were compared with real data from a study of community structure at reef sites damaged by oil pollution (Loya 1976). One of the most worrying aspects of the model was its prediction that a small amount of habitat destruction could lead to considerable extinction; thus, damage to 10% of the habitat would lead to a predicted loss of about 50% of existing species. This was due to a significant proportion of species in the simulations having the combined disadvantage of low abundance and poor recolonisation ability; i.e. a high extinction risk. Field data suggests that this is commonplace in coral communities.

***Blast fishing*:** Pet-Soede et al. (1999) gives an economic analysis of blast fishing on Indonesian coral reefs, a highly destructive activity that is illegal but common and widespread in Indonesia. Blast fishing provides income and fish for a vast number of coastal fishers who claim that they have no alternative to make a living. The incomes earned are comparable to the highest obtained in conventional coastal fisheries. However, Pet-Soede et al. (1999) calculated the cost-benefit analysis at the society level, which showed a net loss after 20 years of blast fishing of US$306 800 per km^2 of coral reef where there is a high potential value of tourism and coastal protection (i.e. four times higher than total net private benefits), and US$33 900 per km^2 of coral reef where there is a low potential value. The main quantifiable costs were through loss of the coastal protection function, foregone benefits of tourism, and foregone benefits of non-destructive fisheries.

6.4.3 Information, research needs and management

Habitat alteration and destruction is probably the most serious of all the threatening processes. Its causes are numerous and are dealt with under each of the agents responsible (fishing, including trawling, Section, 6.10.2, construction of aquaculture facilities, Section 6.11.1, shipping accidents, Section 6.12.1, petroleum drilling, mineral extraction and sand dredging Section, 6.13, recreational activities, Section 6.14 and coastal development, Section 6.16).

In order to assess habitat loss or reduction we need to consider the impacts resulting from two different spatial factors:

- Reduction in, or loss of area; and
- Increased separation.

Better information is needed to assess the extent of habitat loss. The extent of the habitat of interest, in all probability, will not have been previously surveyed. At best, the data will be patchy and inadequate. The impact of the loss or reduction of particular patches of habitat and their increased separation will vary considerably amongst taxa depending on their ecology and biology (e.g., dispersal abilities; needs at different parts of life cycle; food and feeding etc.). In most instances, there is insufficient ecological and biological information to make even crude predictions of potential impacts.

Experimental studies are needed to assess the likely impacts resulting from habitat fragmentation. These particularly need to address maintenance of taxon diversity and recruitment success.

Genetic studies are needed across various invertebrates exhibiting a wide range of reproductive strategies so that a clearer indication of the spatial extent of significant genetic structuring can be obtained.

6.5 Introduced (non-indigenous) species

Human movements over the past 2000 or more years have altered the global distributions of many species dramatically and marine species are no exception. While pre-19th century distributions are often held to be 'natural', many marine invasions did occur prior to this time (Carlton 1999). Species have been introduced, both accidentally and deliberately, by many means including vessels (hull fouling, ballast, and sea chests), aquaculture, the aquarium trade, and intentional and accidental releases into the wild (Carlton 1987; Carlton 1999). Reviews on marine invertebrate introductions (Carlton 1987; Barber 1997; Carlton 1999) give examples where considerable economic damage has occurred, as well as documenting the degradation of natural ecosystems, following introductions of invasive species. It is also likely that introduced species carry with them both internal and external parasites that may not be host specific and could impact on native species (I. Whittington pers. comm.).

The worldwide interest in introduced species and the potential for these species to become pests (Mack et al. 2000), has led to predictions as to which species have the potential to become invaders, and the characteristics of these invading species (Kolar and Lodge 2001). By reviewing the literature, Kolar and Lodge have found that there are consistent patterns and statistically identifiable relationships between success in invasion transitions and characteristics of release events. They suggest these models may help natural resource managers to predict future introductions and reduce their occurrence and impact. Mack et al (2000) suggested that failure to address the issue of biotic invasions could effectively result in severe global consequences, including fishery resources in some regions, disruption of the ecological processes that supply natural services and the creation of homogenous, impoverished ecosystems composed of cosmopolitan species.

A meeting of scientists and lawyers provided decision-making guidance to policymakers, managers, scientists, and other stakeholders regarding alien marine species. The framework consists of seven basic steps: 1. Establish the nature and magnitude of the problem; 2. Set objectives; 3. Consider the full range of alternatives; 4. Determine risk; 5. Reduce risk; 6. Assess benefits versus risks; and 7. Monitor the situation. This framework can provide guidance for control efforts under the existing patchwork of national laws and could provide a foundation for international co-operation (Bax et al. 2001).

Deliberate introductions into Australian waters have occurred mainly for the purposes of aquaculture, the Pacific Oyster (*Crassostrea gigas*) being a good example. Accidental introductions into Australian waters are far more widespread and have occurred by a variety of methods, including hull fouling, ballast water, sea chests and attachment to marine debris that has floated onto our coasts. While the advent of transport by ballast water (Hutchings 1992a) is a relatively recent phenomenon, the other methods of transport have been going on for centuries.

It is often extremely difficult to distinguish between native and introduced species, particularly in countries such as Australia where the native fauna is poorly known. Nevertheless, it is clear that there are many introduced marine pests in Australian waters. Likewise, while there are still no detailed inventories of the fauna for most Australian ports, although this is improving as port surveys for international ports around Australia are being carried out, there is little doubt that the number of introductions is increasing.

Pollard and Hutchings (1990) reviewed the species that had been introduced into Australian waters and attempted to identify the likely mechanisms and timing of each introduction. Species were only listed if material had been deposited in a museum and the identifications confirmed by relevant specialists. CRIMP (CSIRO's *Centre for Research on Introduced Marine Pests*) has recently estimated that more than 250 non-indigenous marine species have now been identified in Australian waters (Thresher 1999; R. Thresher pers. comm.). Furlani (1996) presented annotated descriptions of 70 of these listed introduced species to facilitate their recognition. Increases in the number of known introductions are largely the result of the considerable effort that has been expended during this period by CRIMP and others in undertaking port surveys to detect introductions. Port Phillip Bay, Victoria is by far the best-studied area to date.

***Port Phillip Bay*:**A detailed benthic survey of Port Phillip Bay was undertaken and the results could be compared with extensive benthic surveys carried out in the late 1960's early 1970's. This survey found 165 introduced and cryptogenic species with representatives from all the major phyla. Based on the coverage obtained in this survey, the total number of non-indigenous *benthic* species alone was estimated at 300-400, with 2-3 additional species establishing themselves every year (Hewitt et al. 1999). Though most have not attained pest status and are concentrated around the active port areas, a few have become extremely numerous or widespread. For instance, the three most abundant species in muddy sediments in the Bay – the bivalves *Corbula gibba* and *Theora lubrica* and the polychaete *Euchone limnicola* – are all probable introductions from Europe, Japan, and Japan or NW America, respectively (Wilson et al. 1998). Wilson et al. (1998) discuss the changes in the benthic fauna over time, with some native species disappearing from the Bay with others being found for the first time. To have such baseline studies is unusual. In most ports where benthic studies have been undertaken to document introduced species, no previous benthic surveys have been undertaken. A subsequent survey of the epibenthic community structure in Port Phillip Bay during 1998 found that seven of the 63 epibenthic organisms collected during the survey were exotic introductions to the bay. As many of these species are widespread and abundant (35% of all individuals), their effect on the ecology of the bay is likely to be significant (Cohen et al. 2000).

Vectors for translocation of non-indigenous taxa

Non-indigenous marine invertebrates have been transported to Australia since the first sailing ships arrived here (Pollard and Hutchings 1990). Hull fouling has been responsible for most introductions historically and probably accounts for several of the most recent high profile introductions. Examples from Port Phillip Bay alone include the Japanese kelp *Undaria* and spaghetti weed *Codium fragile tomentosoides* (Hewitt et al. 1999). The Northern Pacific Seastar (*Asterias amurensis*) possibly also was introduced by hull fouling**,** probably through translocations of populations introduced elsewhere.

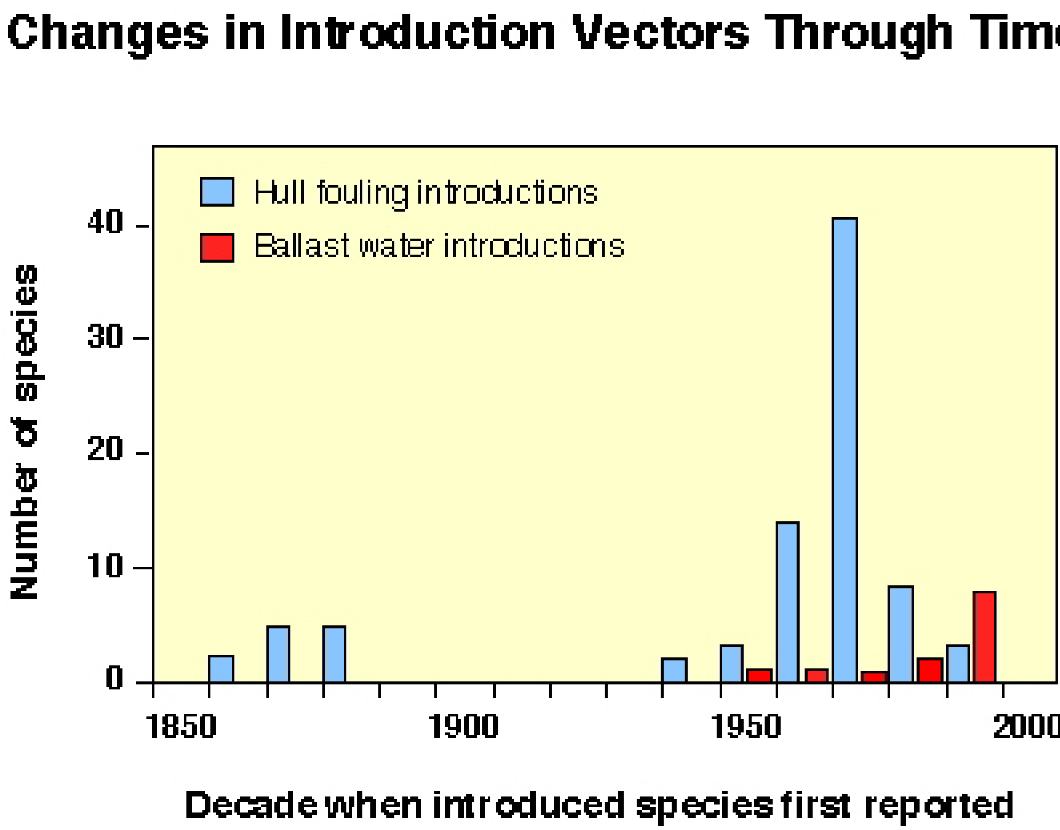
The rate of invasions appears to have increased over the last few decades with the increase in shipping, particularly from the Northwest Pacific region, and the use of ballast water (Hewitt et al. 1999). It has been estimated that the world shipping fleet transports approximately 10 billion tonnes of ballast water around the globe each year (AQIS 1999). Australia is a net exporter of raw materials (e.g., grain, iron ore, coal, etc.) transported using dedicated shipping, and the unladen vessels return with ballast water pumped on board as the cargo is emptied. Ships visiting Australian ports discharge around 150 million tonnes of ballast water annually, with an additional 30 million tonnes moved by coastal shipping from one Australian port to another (AQIS 1999). The significance of ballast water as a vector of introductions to Australian waters has increased dramatically over the last couple of decades (see Figure 6.1).

Ports authorities and relevant government organisations are working together in NSW and Victoria to develop practical ways of reducing the risk of spreading marine pests, through hull and gear fouling, assisted with funding from Commonwealth Natural Heritage Trust Coasts and Clean Seas Program (Craven 2000).

Ballast water

Ballast water is typically taken from estuarine or sheltered waters – along with associated fauna and flora – and then discharged on reaching the destination port. Individuals, larvae or cysts may survive the journey and establish in the new port. A recent desktop study by the Australian Museum (1997) reviewed the physical and biological characteristics of ballast water in relation to some widespread introduced species in Australia. They concluded that several of these species could have been transported as larvae in the ballast water tanks as the voyage was sufficiently short to allow the larvae to survive. Similarly, Carlton and Geller (1993) analysed ballast water from cargo ships in Oregon to determine the presence and abundance of non-indigenous organisms. The diversity of taxa present, especially as inconspicuous life-stages, suggested that invasions already were pervasive.

Figure 6.1: Changes in introduction vectors over time (from Hewitt et al. 1999)



Sea chests
Transport in sea chests (also known as sea inlet boxes or suction bays) is a mechanism that requires far more study than it has received. Sea chests are the spaces in a ship's hull into which water is drawn into in order to be pumped into the ballast system (Carlton 1999). A variety of studies have shown these to contain diverse fouling faunas that have been protected from antifoulants. For instance, Richards (1990) found a population of the tropical muricid snail *Thais blandfordi* on the walls of a cargo vessel that had passed between Saudi Arabia, Kenya, Malaysia, Singapore, Hong Kong, England, and Papua New Guinea. The population structure suggested they had reproduced in the sea chests, and they had survived British winter water temperatures before returning to the tropics. When found they were so abundant that they were blocking the pipes and filters of the ship's water-cooling system (cited in Carlton 1999). Similarly, McEnnulty et al. (1999) sampled four vessels participating in the 1998 Tallships Australia event and found that, in three ships which were lightly fouled overall, the fouling was concentrated in dead water areas and areas lacking antifouling, including the sea chests.

Marinas and pleasure craft
Floerl (2001) investigated recreational boats in marinas in far North Queensland. He found that 20-45% of boats examined carried small to large amounts of marine organisms on their hulls. Boats moored for long periods had hull faunas reflecting the species

present in the marina. Protective break walls that provide shelter for vessels from rough weather enclose most coastal marinas in Queensland. These enclosures restrict the flushing of water from within the marinas and appear to concentrate larvae where they can settle on boats in large numbers. A consequence of this is that rates of fouling can be up to 19 times higher than in less protected, better-flushed open marinas in other coastal locations.

If exotic species are present on the hull, these may reproduce or otherwise be dispersed (e.g., through cleaning). Floerl (2001) found that 21 vessels surveyed in the Townsville marina had visited more than 30 other marinas around Australia during the past two years. With over 250 marinas around Australia and many thousands of shipping movements, the chances of successful dispersal are high.

Aquaculture

Worldwide, aquaculture has led to the introduction of a wide range of exotic species, including seaweeds, fishes, invertebrates, parasites and pathogens (Naylor 2001). International transfers of non-native species for native species for aquaculture pose high ecological risks given the absence of strong policies in most countries.

Aquaculture is the primary reason for most deliberate introductions into Australian waters. The best examples are salmon and the Pacific oyster (*Crassostrea gigas*). In places along the east Australian coast the Pacific Oyster is replacing the native Sydney rock oyster (*Saccostrea commercialis*) to the detriment of the oyster industry (see Pollard and Hutchings 1990 for more details).

Trawling and dredging

Damage to substrates through scallop dredging and trawling may have assisted the spread of non-indigenous taxa such as the New Zealand Screw Shell (*Maoricolpus*), Northern Pacific Seastar (*Asterias amurensis*), and the Mediterranean sabellid polychaete (*Sabella spallanzanii*). *Sabella spallanzanii* only appears to colonise disturbed habitats, and the scallop industry has been implicated in its spread in Port Phillip Bay (Currie and Parry 1996).

Other vectors

Other vectors for the transport of non-indigenous taxa exist, although they are undoubtedly of lesser significance than those already discussed. For example, Gregory (1998) points out that many marine fouling organisms have long since achieved a global distribution through natural methods, such as attachment to passively drifting logs, seaweed and pumice, or active marine animals such as turtles. However, an increasing role is being played by marine debris, particularly plastics. Most plastics float, are highly durable in seawater, and are potential surrogates for natural substrata such as logs and seaweeds. Studies of beach-cast pelagic plastics have shown that they can support a varied community of encrusters and fouling epibionts as well as a diverse motile biota (Gregory 1998).

The importance of accidental releases of non-indigenous species from aquaria is unknown. There is an unconfirmed report that the northern hemisphere anemone *Aiptasia* is present in commercial aquaria in Australia. This species can reach plague proportions if uncontrolled and could become an important pest if accidentally released into marine waters (W. Rudman pers. comm.).

6.5.1 Effects of introduced species on native species, communities and ecosystems

Impacts caused by introduced taxa are not limited to fauna but also affect marine plants and algae, as well as whole communities and processes. In addition, introduced species may benefit some organisms, be detrimental to others of be neutral. Here we focus on the negative impacts.

The introduction of non-indigenous species can lead to predation or competitive and spatial interference with native species (Galil 1994). Some introduced species have obtained pest status due to their numerical and ecological dominance and/or economic impacts, and have succeeded in alienating large areas of habitat (e.g., extensive beds of the Giant Fanworm *Sabella spallanzanii* in Port Phillip Bay; large aggregations of predatory Northern Pacific Seastar, *Asterias amurensis*, in the Derwent Estuary and Port Phillip Bay). Such outbreaks clearly have a significant impact on native species, community structure and function, and probably, ecological processes.

A recently published study (Wilson et al. 1998; Hewitt et al. 1999) on the changes in benthic communities in Port Phillip Bay between 1969 and 1995 revealed that polychaetes have become more abundant than crustaceans and molluscs and that the proportion of suspension-feeding organisms has increased at the expense of deposit feeders. Introduced species (of which they identified 165 in all major plant and animal groups) and changes in nutrient input, were considered primarily responsible for these changes. However, it was difficult to separate the impacts due to introduced species and reduced nutrient loading associated with reduced discharge from Melbourne's sewage treatment works. Similarly, Currie and Parry (1999) evaluated changes in the structure of benthic communities in Port Phillip Bay over 20 years by comparing results of a survey of 86 sites in 1969-72 with a survey of 13 sites in 1991-92. The species composition of communities differed significantly between surveys. Of particular concern was the establishment and abundant spread of a further three non-indigenous species (*Sabella spallanzanii*, *Corbula gibba, Euchone limnicola*) contributing to long-term and possibly irreversible changes to the ecology of Port Phillip Bay (Currie and Parry 1999; Talman et al. 1999; Cohen et al. 2000; Currie et al. 2000; Talman and Keough 2001).

Hutchings (1999b) laments that much of the work over the last decade has simply documented the distribution and abundance of introduced species in Australian waters, with little effort at trying to ascertain the impact of introduced species on the native fauna (see note below). There are some exceptions: the impact of the Northern Pacific seastar, *Asterias amurensis*, on the benthos of the Derwent in Tasmania has been documented in a qualitative way (Grannum et al. 1996; see below). In addition, much of the concern about the impact of these species has focused on their potential economic effects (e.g., aquaculture losses, biofouling), rather than the damage to native marine communities.

6.5.2 Examples of introduced marine species

Northern Pacific Seastar - *Asterias amurensis*

This species was introduced into the Derwent from Japan and was first observed on a Hobart wharf (in front of CSIRO Marine Research!) in late 1992 by W. Zeidler. Judging from the numbers present it must have been introduced several years previously (Turner 1992) and specimens collected in 1986 were incorrectly identified as a similar native species (Buttermore et al. 1994). It is now well established in the Derwent and parts of southeast Tasmania (Byrne et al. 1997b) and occurs in large numbers in Port Phillip Bay, Victoria (Cohen et al., 2000). The impacts of this introduction have been studied by Ross et al. (in press-a; in press-b; submitted).

Asterias amurensis is a significant predator with a high fecundity of up to 19 million eggs annually per adult female (Buttermore et al. 1994). It has become the dominant invertebrate predator of benthic communities in the Derwent River (Grannum et al. 1996), feeding on a wide range of epibenthic organisms including bivalve and gastropod molluscs, barnacles, crabs and other crustaceans, worms, echinoderms and ascidians. They are also cannibalistic. As there are no baseline data on biodiversity in the areas where *A. amurensis* has established in the Derwent, it is very difficult to measure its impact on the original communities, although casual observations suggest that dense populations are having a major impact on fauna. Few studies have been undertaken on the impact of this species on infaunal communities, but changes may have occurred as the surface habitat has changed from 3-dimensions to a 2-dimensional structure (Ross et al. in press-a; Ross et al. in press-b).

In Port Phillip Bay, 4-5 specimens were collected by scallop dredges from various sites around Port Phillip Bay during 1995-1997. In January 1998, they were found on mussel ropes in Dromana in the Bay (Garnham 1998). These individuals were apparently part of a cohort found from the Port of Melbourne around to Mornington (G. Parry, MAFRI pers. comm.). During the subsequent years, *Asterias* has spread throughout the Bay and is now a major component of its epifauna (Cohen et al. 2000). How the seastars spread from Tasmania to Port Phillip Bay is debatable. Larvae may have been transported via ballast water but they are very fragile (T.O'Hara, pers. comm), or adults may have been transported in sea chests. A ship in Bass Strait was reported as having adult seastars in their sea chests (R.Thresher, pers. comm.). However, during 1995-1998, a live fish trade was operating between the Derwent (where *Asterias* is abundant) and Port Phillip Bay. It is possible that the tanks containing the live fish could also have transported seastar larvae or juveniles. More recently, *Asterias* have been found in NE Tasmania where live fish trade is carried out between this region and the Derwent (T.O'Hara, pers. comm.). To date no records have been found in nearby Westernport, and also of concern is the potential introduction of *Asterias* into the Spencer Gulf where conditions would ideally suit this species. The species is tolerant of warmer water temperature and potentially could live as far north as Sydney. So far, no effective control measures are available, the only recommendations being removal using divers or baited traps. Prevention of spread will be helped by the treatment of seawater transported from affected areas, particularly around the spawning season (Buttermore et al. 1994).

Byrne et al. (1997a; 1998) urge caution in using parasites to control introductions. A ciliate that damages *Asterias amurensis* is not host specific and could easily infect and decimate native populations of echinoderms if introduced as a mechanism to control *Asterias*.

Giant Fanworm - *Sabella spallanzanii*

Another recent introduction is the large sabellid polychaete *Sabella spallanzanii* that is native to the Mediterranean. Dense populations were first observed from Cockburn Sound, West Australia in the 1960s and then from Port Phillip Bay in the 1990s (Hutchings 1999b). More recently it has been recorded from Devonport, Tasmania and Eden on the NSW south (Talman et al. 1999). This large sabellid (80-400 mm in length) forms dense colonies and markedly modifies local water currents and rates of sediment deposition. Presumably, there are also changes to the benthic infaunal community beneath the worms, although data to substantiate this are lacking. *Sabella spallanzani* is regarded as a pest because of its capacity to foul hard surfaces including rocky reefs, the shells of commercial shellfish and a wide range of man-made structures. While there is no evidence that it can exclude native species, it is likely to compete with native suspension feeders for food and space and interfere with their recruitment. Worms readily settle on mussel grow-out lines and may reduce mussel growth by altering water flow around the lines and competing with mussels for suspended food. There are indications that less robust species of seagrass are adversely affected by the settlement of worms on their fronds (CRIMP Marine Pest Information Sheet 5). It was recently introduced to Port Phillip Bay where it is now a conspicuous component of most benthic communities (Cohen et al. 2000; Currie et al. 2000). The reproduction of the species in Port Phillip Bay has been investigated. The animals are dioecious and breed annually over over the autumn/winter period. Spawning was synchronous and coincided with falling seawater temperatures and shorter day-lengths. The females were highly fecund, with probably more than 50 000 eggs being spawned by a large female (Currie et al. 2000). The spread of *S. spallanzanii* within Port Phillip Bay has been monitored by divers on an annual basis since 1994. The most recent dive survey (1998) indicates that *S. spallanzanii* has extended its range through out the entire 2000 km ^{2} embayment, and has invaded most subtidal habitats. Quantitative estimates of *S. spallanzanii* abundances were highest on pier pylons (12.5 individuals m^2, 0.5 to 7 m depths). On sediments, estimates were highest at shallow sites (0.3 m^2, 7 m depth), but numbers declined significantly with depth (0.1 m^2, 17 to 22 m depth). *S. spallanzanii* demonstrates a clear preference for growth in sheltered, nutrient-enriched waters, so it may not spread from Port Phillip Bay into the adjacent oceanic waters of Bass Strait; however, in view of *S. spallanzanii's* current high abundance, fecundity and extended spawning periodicity, there is a high risk of future range expansions, mediated by shipping, into other temperate-water ports.

Black-Striped False Mussel – *Mytilopsis sp.*

The most recent and widely publicised introduction of a non-indigenous marine species involved the discovery in March 1999 of the Black-Striped False Mussel (*Mytilopsis sp.* [114]) in several marinas in Darwin (Russell and Hewitt 2000; Willan et al. 2000). *Mytilopsis* is a Black-Striped False Mussel that reaches an average maximum size of around 25 mm. It is native to the tropical and sub-tropical eastern Pacific but in recent decades has invaded several Indo-Pacific ports, including India (1967), Taiwan (1977), Hong Kong (1980), Japan (1974) and possibly Fiji (1949?) (R. Willan, pers. comm.). It was probably introduced to Australian waters via the hulls and internal water systems of commercial and recreational vessels. Willan (pers. comm.) suggests that there is no evidence that this species was introduced from Panama or the Caribbean, and he suggests that tropical Asia is more likely. *Mytilopsis* is taxonomically and ecologically similar to the freshwater Zebra Mussel (*Dreissena polymorpha*), which has caused significant ecological and economic impacts in Europe and North America (Ackerman et al. 2001). The impacts of the Black-Striped False Mussel are similar to the Zebra Mussel and include massive fouling of wharves, marinas, marine farms and other seawater systems (e.g., aquaculture pumping facilities, vessel ballast and cooling systems). The mussel can form dense monocultures that exclude other species leading to a substantial reduction in biodiversity in affected areas. It is extremely prolific and fecund, can tolerate water temperatures from 5 to 40° C and salinities from 0 to 50 parts per thousand (i.e. freshwater to hypersaline), and can attach to virtually any solid surface (Russell and Hewitt 2000; Willan et al. 2000).

The discovery of *Mytilopsis* in several Darwin marinas led to an immediate program of eradication involving large-scale chemical treatment of affected areas with chlorine and copper sulphate. The program appears to have been successful in eliminating the mussel, although monitoring is still continuing on boats coming into Darwin to check for new infestations, and to track boats which left Darwin and may have inadvertently transported *Mytilopsis* to other Australian ports. The species is potentially capable of colonising areas throughout northern Australia from Fremantle to Sydney, as well as the warmer parts of the Spencer and St Vincent Gulfs in South Australia (Bax 1999).

The European Shore Crab

A native of European Atlantic coastlines, this crab was probably introduced into Australia in the 1950s. Because of its long history as part of the intertidal and shallow water fauna of southern Australia and the lack of baseline studies in these habitats prior to its establishment, the impacts of *Carcinus* on native species in mainland Australia cannot be assessed with any certainty. However, impacts have been documented in other areas where it has become established, particularly along the US Atlantic coast. It is a voracious predator with a broad diet and has been implicated in the decline of native

[114] There are unresolved taxonomic problems associated with mussels referred to as Black-Striped False mussels. These uncertainties relate to both generic (*Mytilopsis* or *Congeria*) and specific names (*sallei* or *adamsi*). Until this uncertainty has been resolved, this species will be referred to as the Black-Striped Mussel, *Mytilopsis* sp. (CRIMP Marine Pest Information Sheet #11).

shellfish populations, including some commercially important species. In Tasmania, predation by *Carcinus* is a major cause of mortality in native crab and bivalve populations, including one that is basis for a modest fishery (CRIMP Marine Pest Information Sheet 4). Recent studies by Thresher et al.(2000) have been investigating the use of the parasitic castrator *Sacculina carcini* against the European shore crab *Carcinus maenas* and related species. They stress the need for detailed knowledge of the parasite's behaviour and physiological interaction with its host. In addition it is essential that the host specificity of this parasite must be well understood before any field trials are carried out because there are a number of related crabs that could potentially be affected.

Other species

Other examples of introduced species which occur in large numbers and are inferred to be having an impact on the native fauna are the Asian Mussel *Musculista senhousia* (Crooks 1998) and the European Clam *Corbula gibba* (Talman and Keough 2001) and the New Zealand Screw Shell *Maoricolpus roseus*.

Between about 1885 and 1930, an abortive attempt was made to deliberately introduce the New Zealand oyster (*Tiostrea chilensis lutaria*) to SE Tasmania. Probably through these shipments, a number of other species, including the New Zealand Screw Shell (*Maoricolpus roseus*), several other molluscs (*Chiton glaucus*, *Neilo australis*, *Ruditapes largillierti*), a starfish (*Patiriella regularis*) and a crab (*Cancer novaezelandiae*), were accidentally introduced (Dartnall 1969; Carlton 1992; Furlani 1996; R. Willan pers. comm.) and are now well established there. For instance, the New Zealand Screw Shell has now spread along the eastern continental shelf of Tasmania, where it is often the dominant element of the macrofauna, and has recently crossed Bass Strait. A few individuals have also recently been found in Sydney Harbour and Botany Bay.

Of concern are species not yet present but likely to become serious invaders. There needs to be awareness of what the likely invaders are and their likely modes of entry. The characteristics of some species make them likely suspects as invasive species. Howevr, there also needs to be caution excercised. For example, Chapman and Carlton (1991) predicted that the supposedly invasive Oriental Isopod (*Synidotea laevidorsalis*) was established in Brisbane, or Sydney, Australia. These authors later reported that their prediction was proved correct (Chapman and Carlton 1994) but Poore (1996) showed that this supposed species was actually a set of closely related taxa, two of which are found only in Australia.

Information sheets on several marine pest species, including the introduced algae, can be accessed via CRIMP's website[115].

[115] http://www.ml.csiro.au/~spinks/CRIMP/Marine%20pest%20infosheets.html

6.5.3 Information, research needs and management

Information and research

A considerable amount of time, effort and funding is being devoted to undertaking surveys of ports around Australia by CRIMP and others (Hewitt and Martin 1996; Choat and Hoedt 1999; Hewitt and Martin 2001; Hoedt et al. 2001) for the purpose of documenting introduced species. These surveys also highlight the lack of basic knowledge about native fauna.

The initiation of these port surveys has greatly increased out knowledge of the fauna of many ports- some of which had probably never been surveyed before, and if repeated over time, they have the potential to considerably add to our knowledge of the marine biodiversity of these areas. However, as conducted at present, these surveys have some major limitations including:

- Sampling is generally carried out by non-specialists and, particularly, the more cryptic elements may not be adequately sampled.
- The surveys of epifaunal organisms includes sampling, by scraping, small areas on wharf piles. This often leads to fragments of colonial animals being collected which are extremely difficult to identify even by specialists. This problem can be overcome by ensuring that the entire colony of the colonial animal is collected when it occupies part of the quadrat.
- Sorting and identification is generally undertaken by inexperienced people. Many non-indigenous species belong to genera in which native Australian species also occur and distinguishing these species often requires a specialist in that taxonomic group.
- In preparing the budget for these port surveys, adequate funds need to be allocated for expert identification because, in many groups, distinguishing between native and non-indigenous species is almost impossible because detailed taxonomic studies have not been undertaken.
- Defining the natural range of a species can be difficult, especially in northern Australia, where many species potentially have wide Indo-Pacific ranges.
- Visual and video transects are required (according to the CRIMP protocols) to assess the abundance and distribution of sessile animals along the wharf piles. These may, however, have limited usefulness in heavily used areas of ports, such as Darling Harbour and Circular Quay in Sydney Harbour, where visibility is extremely limited and decreases rapidly with depth.
- The surveys are essentially one-off qualitative surveys of certain habitats within the Port region. While providing some useful information about the presence or absence of non-indigenous species listed as belonging to the various Appendices which have been developed by CRIMP, they are:
    - Limited to commercial shipping areas. Sampling is not undertaken in areas used extensively by recreational or fishing vessels. In ports where these activities are extensive, these areas should also be sampled.

  - Limited to target taxa - the non-target components of the fauna collected during these surveys are not identified to species so that the chances of missing non-reported introduced taxa are high.
- Dealing with all the macroinvertebrates would, gradually, enable much of the fauna of each port to be documented. This would also facilitate the recognition of any additional non-indigenous species introduced into the port over subsequent years
- Vouchers of this valuable material should be lodged in a museum collection so that specimens are available to future workers for the confirmation of identifications.
- A large number of agencies have been contracted to undertake port surveys, ranging from Commonwealth to State, and private consulting companies and there are often difficulties in obtaining the resulting unpublished reports. We recommend that the reports be made publicly available through a single source, ideally CSIRO.

To date, only a few studies have assessed the impact of introduced species (Talman et al. 1999; Currie et al. 2000; Talman and Keough 2001; Ross et al. in press-a; Ross et al. in press-b), or any parasites which these introduced species may contain[116]. While it is laudable that emphasis is being given to the identification of introduced species in the Clean Sea Programme/Introduced Species, this must surely be undertaken in conjunction with studies to document the native fauna. Only then can we correctly identify introduced taxa and begin to assess their impact. For example, Buttermore et al. (1994) emphasised the difficulty in determining the impact of the Northern Pacific Seastar, *Asterias amurensis*, on the original ecology of the Derwent Estuary ecosystems (see Section 6.5.2) due to the absence of general baseline data, highlighting the urgent need for well formulated surveys to obtain baseline information about marine communities.

Thus, programs relating to the assessment of introductions should aim to:
- Undertake baseline studies on native fauna;
- Determine the effects of introduced taxa on native fauna and communities; and
- Develop control strategies.

It is often difficult to determine the natural ranges of species, and therefore distinguish between native and introduced taxa. Some species are cryptogenic, i.e. their origin is not demonstrably non-indigenous or native and therefore uncertain (Carlton 1996a). Many marine species have pelagic larvae and may naturally be widely distributed. In countries such as Australia, where the native fauna is poorly known, the lack of adequate data can make it difficult to distinguish between an introduction and a previously undescribed "endemic" species. Genera often have wide distributions, and distinguishing between native and introduced species in the same genus may only be possible by an expert. For example, species of the spionid polychaete genus *Polydora* have been transported around the world with oysters, but trying to ascertain the natural distribution of the numerous species in the genus is extremely difficult, as is distinguishing between species. The decline of a native mussel from California was masked by the invasion of a morphologically similar sibling species, the European Blue Mussel (Geller 1999) and it is likely (and to date completely unstudied) that a similar takeover is occurring in temperate

[116] CRIMP has recently begun studies on some parasites (Hewitt, pers. comm.).

Australia and New Zealand. Similarly, the introduced Pacific Oyster (*Crassostrea gigas*), which is very similar to the Commercial Rock Oyster (*Saccostrea glomerata*), went undetected in NSW for some time before being noticed, despite the commercial importance of the latter species.

Management

Currently we have no methods or protocols for the removal of non-indigenous species once they have become established in Australian waters. It is likely that an introduced species will be already well established by the time it is first recognised. Removal of marine exotics is unlikely to be a feasible option and is certainly not a cheap one. The eradication of the Black-Striped False Mussel from marinas in Darwin was achieved at a great cost (Aust $2.6 million and rising; Pyne 1999) (five times the cost of the entire CRIMP Port Phillip Bay survey), and time will tell if this species was actually removed from Australian waters.

Currently there is an eradication program to remove the Asian Green Mussel, *Perna viridis*, from Trinity Inlet in Cairns[117] (R. Willan, pers. comm.).

CRIMP is currently investigating options for biological control of the more serious pests. However, potential problems with this approach – as on land – include the time and cost associated with research, host specificity of selected biocontrol agents, and potential non-target ecological impacts. We therefore emphasise that the preferred (and most practical and efficient) approach is to prevent or minimise the possibility of introductions, rather than focusing on controlling or eradicating them once they have become established. This entails management and treatment of the key transport vectors, e.g., treatment of ballast water, use of antifouling paints, hull-cleaning protocols and controlled movement of aquaculture products. Since all treatments only minimise risk – few are 100% effective and chance introductions will still occur – we also require rapid response strategies to identify and then eradicate outbreaks as soon as possible after they occur, for example through a national invasion-response network similar to that set up to handle oil spills.

The national responsibility for quarantine and control of the importation of species rests with AQIS, the Australian Quarantine and Inspection Service. However, in 1999 – partly in response to the serious outbreak of the Black-Striped False Mussel in Darwin that year – a National Taskforce on the Prevention and Management of Marine Pest Incursions was set up by the Australian and New Zealand Environment and Conservation Council (ANZECC) and the Ministerial Council on Forestry, Fisheries and Aquaculture. The Taskforce's report, submitted in December 1999, proposed a National System for the Prevention and Management of Introduced Marine Pests that is coordinated by a single body reporting to all relevant Commonwealth and State ministers. The report also detailed recommendations for interim and longer-term action to establish the three main components of the National System, i.e.:

- For *preventing new marine pest incursions and translocations*, it recommends maintaining the national leadership role of AQIS, and further developing and

[117] Being co-ordinated by EPA (Qld) and Qld DPI (Northern Fisheries).

implementing AQIS' draft Action Plan for minimising risks from the introduction and translocation of marine pests by vessels.

- For *coordinating emergency response actions to new marine pest incursions and outbreaks*, it proposes a leadership role for the National Office of Animal and Plant Health in AFFA, and includes an Australian Emergency Marine Pest Plan for guidance on a staged response to a marine pest outbreak, with a proposed initial list of marine species that would automatically trigger a response.
- For *longer term mitigation and control measures to combat established marine pests*, it proposes that the Commonwealth explore the option of developing statutory plans to reduce, eliminate or prevent the impacts of introduced marine species on the biodiversity of Australia using provisions of the *Environment Protection and Biodiversity Conservation Act 1999*.

An update on the status of this program is available at http://www.ea.gov.au/coasts/imps/index.html.

6.6 Pollution

While it has long been assumed that the oceans are too vast, and their dispersive and degradative powers too great, for the various forms of waste disposal and pollution to have any discernible impact, there is increasing evidence that this is not the case. Many marine habitats, particularly those in estuaries and coastal zones close to human population centres, are suffering a range of impacts resulting in a deterioration of the habitats and the communities found in them (e.g., Glasby 1997; Koop and Hutchings 1997b; Connell and Glasby 1999; Glasby and Connell 1999).

Sources of marine pollution may be classified as point source or diffuse. Point sources include all forms of direct discharge into the marine environment, whether through deliberate waste disposal[118] or accidental spills, such as from sewage outfalls, paper mills, factories, abattoirs, gravel washing plants, dredging activities, mining, ships and power stations (NSW Fisheries 1998a). Diffuse sources are generated from a variety of activities on land and carried as catchment runoff to the sea via watershed streams and ground water, or from atmospheric sources through winds and rain. Terrestrial runoff can include pesticides, herbicides and fertilisers from agricultural areas; animal wastes from farms and feedlots; sediments as a result of erosion following land-clearing, development or construction work; and urban stormwater containing a range of household and garden chemicals, oils, faecal material, rubbish etc.

The causes and effects of water pollution in Australia, including the behaviour of chemicals in aquatic systems, pollution ecology, and effects of various classes of pollutants, have been reviewed by Connell (1993).

[118] See Scanes (1995) for an ecological overview relevant to temperate Australia.

6.6.1 Effects of different types of pollution

Pollution may impact **directly** upon individual organisms, causing mortality, physiological stress or reproductive impairment, either immediately (e.g., through acute toxicity or smothering) or through cumulative effects (e.g., bioaccumulation of heavy metals or organochlorines). The vulnerability of some plant species (particularly seagrasses) to certain types of pollution (e.g., siltation, turbidity, or chemical pollutants) may result in significant habitat changes or losses. **Indirect or secondary effects** can include changes in community composition and function (e.g., by selecting for pollution-tolerant species or certain trophic groups, such as filter-feeders) or effects on the health or survival of predatory species.

The major types of marine pollution, and their effects, are:

- Toxic substances (e.g., organochlorine, PCBs, heavy metals, acids, radioactive wastes etc) – can cause direct poisoning and an array of secondary or cumulative effects;
- Excessive nutrients – can result in eutrophication, leading to changes in communities, algal blooms, increased Biological Oxygen Demand (BOD);
- Sediments – siltation and sedimentation can result in smothering, substrate change, clogging of gills, reduction in light and consequent loss of vegetation;
- Oil and petroleum – spills can result in smothering of benthic organisms, and toxic effects from both the dissolved fraction of the oil and the chemicals used to clean it up;
- Rubbish – can be responsible for smothering, tangling, choking etc. of larger organisms, but may also provide useful habitat for many invertebrates.

There are also complex mixtures, such as sewage, which can contain many or all of the above components and have a range of effects.

Toxic substances

Toxic effects on marine invertebrates can be lethal or sublethal and may act through interference with metabolism, hormones, reproductive success, etc. Sublethal effects can also be cumulative. Some invertebrates will accumulate organochlorines, heavy metals etc. with implications not only for them but also for their predators and – directly or indirectly – for human consumption (e.g., Connell 1990, 1993; Engel and Thayer 1998).

Toxic chemicals find their way into the marine environment from many sources including industrial effluents (e.g., from paper mills, oil refineries, smelters, pharmaceutical firms), sewage (e.g., household chemicals and industrial sewer connections), terrestrial runoff (e.g., agricultural chemicals), and antifouling paints (see Section 6.12.4). While some types of waste – such as sewage or dredge spoils –are essentially within the assimilative capacity of the ocean and are similar in quality to materials naturally entering the ocean, industrial wastes pose a greater problem because they have no natural analogues and are less easily recycled. These may accumulate on the seabed with possible detrimental effects on the benthos (Gage and Tyler 1991).

The main classes of pollutants that have received the most attention are petroleum-derived hydrocarbons (discussed under "Oil and Petroleum", below), heavy metals and synthetic organics such as herbicides and pesticides. The latter two are of biological concern because of their toxic effects and the potential for "bioaccumulation" in the tissues of individual organisms and "biomagnification" through the food web to higher trophic levels (e.g., Pruell et al. 2000 and references therein). However, Australian studies and reviews have focused primarily on pollutant distribution and concentrations in water, sediments and biota, with the aim of determining sources of contamination (for industry regulation) or ensuring compliance with health standards for human consumption (e.g., of shellfish), rather than evaluating the effects on marine organisms or communities. Reviews of concentrations of trace metals and organochlorines in Australian waters (e.g., ANZECC 1991; Thompson et al. 1992; Batley 1995; Richardson 1995) clearly demonstrate the lack of an integrated national perspective, with most investigations undertaken in localised areas surrounding potential "hotspots" near major eastern seaboard cities. There has been scant attention to collecting "baseline" data in relatively pristine areas (Richardson 1995). A series of papers on the sources of pollutants flowing into the GBR lagoon and the consequences of these has recently been published (Hutchings and Haynes 2000).

Port Phillip Bay, Victoria has been intensively investigated over the past couple of decades (Phillips et al. 1992b). Formalised efforts to assess the presence of trace metals, organochlorines and petroleum derivatives in the water, sediments and biota of the Bay began in the early 1970s with the establishment of the Environmental Study of Port Phillip Bay. Large-scale monitoring studies from this time included studies on trace metals and organochlorines in mussels (*Mytilus "edulis"*), the data being used to trace sources of contamination in the Bay. Surveys to examine the occurrence and distribution of polychlorinated biphenyls (PCBs) found that contamination was widespread in both mussels (*M. "edulis"*) and sediments; PCBs were detected in every sample analysed (Phillips et al. 1992b). Apparent sources of contamination included run-off and leachates from a refuse tip; wastewaters from industry, and heavy shipping activity. Since 1980, data have also been collected to support the development of shellfish aquaculture and ensure compliance with health standards (Phillips et al. 1992b).

Many studies show that macrobenthic assemblages are good biotic indicators of environmental quality in that declines in species abundance and biomass occur with increasing contamination (e.g., Rakocinski et al. 2000). The limited information available on the impacts of chemical stressors on mangrove forests, seagrass meadows and coral reefs worldwide is discussed by Peters et al. (1997), with reference to ecological risk assessment and ecosystem management.

Heavy metals

Heavy metals are a major anthropogenic contaminant of estuarine and coastal waters. In the northern hemisphere where population densities, urbanisation and industrialisation are greater, there has been significant contamination of coastal waters by metals, but in Australia, potential problems are restricted to the vicinity of major coastal cities (Batley 1995). Shallow nearshore waters generally have markedly higher concentrations of metals than oceanic waters, because dispersion and dilution are less effective and currents generally follow the coastline, limiting the transport of contaminants offshore. Most metals attach to suspended particulates and ultimately accumulate in bottom sediments (Batley 1995).

All heavy metals are toxic above certain threshold concentrations, and can cause physiological stress, reduced reproductive success, and outright mortality in invertebrates and fishes (e.g., Batley 1995; Kannan et al. 1995; Peters et al. 1997; Burgman and Lindenmayer 1998; Tedengren et al. 1999; Inglis and E. 2000; King and Riddle 2001). The differing tolerances to these metals shown by different taxa can also result in changes to community composition in polluted environments. For instance, Ward and Hutchings (1996) examined the effects of trace metals on the infaunal species composition in polluted intertidal and subtidal marine sediments (including seagrass habitats) near a lead smelter in Spencer Gulf, South Australia. The polluted intertidal sediments had some of the highest metal concentrations ever recorded in marine sediments; Pb up to 5270 $\mu g\ g^{-1}$, Zn up to 16700 $\mu g\ g^{-1}$. These extremely high concentrations of heavy metals appeared to affect both the abundances and distributions of several species. For instance, fifteen species of polychaetes, five crustaceans and four molluscs found elsewhere in the study area were absent from the metal-polluted sites; these represented 26%, 20% and 17% respectively of the total number of species in each group. The patterns were much clearer in the most polluted (intertidal) sites, where multivariate techniques could detect which species were affected, than in the less polluted (subtidal) sites where only a few individual species could be unambiguously correlated to the presence of the metals. In another study on the impact of heavy metals on invertebrate communities, Stark (1998) investigated the effects of heavy metals (copper, lead and zinc) in urban runoff on intertidal soft-sediment macrofaunal assemblages in two Sydney estuaries. Polluted bays had significantly different assemblages, were generally less diverse, and were characterised by an order-of-magnitude greater abundance of capitellid, spionid and nereidid polychaetes and bivalves. Unpolluted bays had a greater abundance of crustaceans and several polychaete families, including paraonids and nephtyids, and were generally more diverse. Inglis and Kross (2000) found that sediment concentrations of heavy metals in a developed estuary near Townsville were several orders of magnitude greater than those from non-urban waterways. Faunal patterns were strongly associated with spatial patterns in the distribution of lead, copper and hydrocarbons, with the fauna at contaminated sites being dominated by cirratulid and sternaspid polychaetes and lacking filter-feeding bivalves and other molluscs common in the non-developed estuaries.

Pesticides

A range of organochlorine compounds have been used in Australia for various purposes, including herbicides (e.g., 2,4-D and 2,4,5-T) (Haynes et al. 2000), insecticides (e.g., DDT, lindane, chlordane, dieldrin, aldrin and heptachlor), fungicides (e.g., hexachlorobenzene and chlorinated phenols) and polychlorinated biphenols (PCBs) (Richardson 1995). Others, such as the dioxins and dibenzofurans, are byproducts of chlorination or combustion processes. Such chemicals enter the marine system via effluents, terrestrial runoff, coastal weed spraying, or other means. Despite the fact that they are internationally recognised as important contaminants in marine environments (Johnson and Ebert 2000), few Australian studies have investigated their local occurrence and distribution (Richardson 1995; and papers in Hutchings and Haynes 2000). The key properties of organochlorines that cause concern are their toxicity and long-term persistence (Richardson 1995). Organochlorine pesticides are no longer available for use but are still measurable in sediments as a result of their persistence. Another concern is their ability to accumulate in the fatty tissues of living organisms (Phillips 1993). Most studies on organochlorides have focused on their toxicity to and accumulation in mammals and birds and there is relatively little information about their effects on marine invertebrates. This is particularly the case in Australia, where very few studies have been carried out on the effects of organochlorines on native species. Consequently, most knowledge is based on overseas investigations. For instance, herbicides are known to be particularly detrimental to mangroves and seagrasses and adversely affect the animal-algal symbioses in corals, while pesticides interfere with chemical cues responsible for key biological processes, including reproduction and recruitment of a variety of organisms (Peters et al. 1997; Haynes et al. 2000; Prange and Dennison 2000). Other effects relevant to marine invertebrates or their habitats include sublethal effects on crabs as a result of DDT in surrounding waters, and possible effects of DDTs and PCBs on algae (Richardson 1995).

Pesticides now in use are generally short-lived organophosphate-type compounds that act quickly and are difficult to detect in the marine environment shortly after application, although they may still cause observable toxic effects. A case in point was azinphosmethyl-contaminated agricultural runoff that entered a South Carolina estuary in 1994. Although this caused significant mortality among juvenile fish and shrimp in a tidal creek, pesticide residues in the water were at or below detection limits within 24 hours of the incident (Chandler et al. 1994; cited in Engel and Thayer 1998). Some insecticides that target particular processes in insect pests may have similar impacts on marine crustaceans. For instance, the insecticide diflubenzuron (Dimilin®) mimics a juvenile arthropod growth hormone in crustaceans (Christiansen et al. 1978) because it blocks chitin synthesis (Engel and Thayer 1998) and effectively stops the moulting process in juveniles. This finding was important because diflubenzuron was proposed for use to control saltmarsh mosquitos and those same marshes were the prime nursery habitat for many crustaceans (Engel and Thayer 1998). Other investigations have shown that several pesticides, PAHs and metals can be accumulated by blue crabs and cause significant mortalities in laboratory tests, but direct correlations between body accumulation and toxicity for many contaminants is not as clear cut in field situations (Engel and Thayer 1998).

Following the restriction of the use of tri-*n*-butyl tin (TBT) in marine antifouling paints (Section 6.12.4), there has been an increase in the number of formulations containing 'booster' herbicides. Scarlett et al. (1999) recorded the occurrence of the antifouling herbicide, Irgarol 1051, within coastal-water seagrasses in 9 out of 10 sites sampled from the east coast of Queensland and within the Great Barrier Reef Marine Park. Although widely distributed throughout European coastal waters, it is not registered in Australia for use as a biocide in antifouling paints. Nonetheless, Scarlett et al. (1999) found concentrations of up to 118 ng g^{-1} wet weight leaf tissue – the highest plant tissue concentration ever recorded – at one site near the Gold Coast. Such concentrations are potentially toxic and could have consequences for herbivores, seagrass-associated fauna, and the endosymbiotic algae of corals.

Acidity

Acidity from development of acid sulphate soils can be a major problem (Cook et al. 2000a, and references therein) although the details of its impact on invertebrate communities are largely undocumented (see below). Development of these soils, which usually occur in low-lying coastal areas, can result in acid discharges when soil disturbance or drainage leads to oxidation of the pyrite in the soil. The resultant acidity of surface waters is often around pH 4 but can be as low as pH 2; in comparison, pH 6 is the minimum tolerated by most aquatic life. Acid discharges into river systems, estuaries and coastal waters have resulted in massive fish kills. Acid water also affects the health of fish and other aquatic life through damage to the skin and gills. Skin damage increases the susceptibility of fish to fungal infections that may lead to diseases such as epizootic ulcerative syndrome, also known as 'red spot'. Gill and skin damage reduce the ability of organisms to take in oxygen or regulate their intake of salts and water (Sammut and Lines-Kelly 1996). Although the effects on marine invertebrates are far less well documented than those on fish, effects on some crustaceans[119], molluscs[120] and polychaetes[121] have been recorded. Undoubtedly other taxa are similarly affected. The aluminium in acid water also is toxic to most aquatic organisms because it damages their gills and at lethal levels can suffocate them. Wilson and Hyne (1997) found acid-sulphate soil leachate to be toxic to embryos of the Sydney Rock Oyster, with concentrations as low as 3.3% soil leachate in seawater decreasing normal development of the embryos after 48 hrs exposure; aluminium appeared to be the main toxicant. Sulphuric acid can also dissolve heavy metals in the soil such as cadmium, which when washed into waterways can be absorbed by fish and other aquatic life (Sammut and Lines-Kelly 1996; Cook et al. 2000b).

Sammut and Lines-Kelly (1996) summarised the short term and long term effects of acid water on fish and fish habitat (and by extrapolation, therefore, on invertebrates and their habitat) as:

Short term:

- Fish kills, disease and destruction of eggs;
- Mass mortalities of microscopic organisms;
- Increased light penetration due to water clarity; and
- Loss of acid-sensitive crustaceans; and

Long term:

- Loss of habitat;
- Persistent iron coatings;

[119] E.g., *Penaeus monodon* (Allan and Maguire 1992).
[120] E.g., bivalves (Bamber 1987, 1990).
[121] E.g. *Nereis virens* (Batten and Bamber 1996).

- Alterations to aquatic plant communities, including invasion of acid-tolerant taxa
- Reduced spawning success due to stress and damaged and undeveloped eggs;
- Chemical migration barriers;
- Reduced food resources;
- Dominance of acid-tolerant plankton species;
- Growth abnormalities and reduced growth rates;
- Increased predation and other changes in food chain and web;
- Reduced recruitment;
- Higher water temperatures due to increased light penetration;
- Increased availability of toxic elements; and
- Reduced availability of nutrients.

There are relatively few studies on the effects of acid soil leachate on marine or estuarine invertebrate communities. The effects of acid water inflow on estuarine benthic (and fish) communities in the Richmond River, NSW, were studied by Roach (1997). He found that changes to benthic communities in the lower estuary were evident following rainfall (which resulted in acid water inflow through numerous flood mitigation drains), but the magnitude of any change varied among acid-affected sites, and when compared to changes resulting from the inflow of freshwater alone, were relatively subtle (Roach 1997). However, the picture may have been complicated by the effects of chronic acid inflow. The major activity on potential acid sulphate soils is agriculture, especially sugar cane (Hogarth 1998).

In NSW, a three-year research project is being funded by the Fisheries Research and Development Corporation (FRDC) and NSW Fisheries, with the aim of developing guidelines to better manage floodgates (particularly in acid sulphate soil catchments), in order to improve water quality, as well as passage of fish and invertebrates (F. Kroon pers. comm.).

Radioactive waste

The disposal of radioactive waste in the ocean has generated considerable and emotional debates. Whilst this is a global issue, it is not (currently) directly relevant to Australian waters. Studies of the environmental effects of oceanic disposal of radioactive materials have focused on the potential routes by which the radioactive material can affect humans. The effects on benthic organisms are considered of secondary importance, if at all (Gage and Tyler 1991). The main sources of radionuclides in the sea are weapons testing, nuclear accidents and direct disposal of low-level nuclear waste. There are still unknown factors that may affect the activity and transport of radionuclides at the dumpsite; the effect of bioturbation on radionuclide mobility and the effects of radionuclides on genetics and physiological processes, including reproduction, are still unclear (Gage and Tyler 1991).

Excessive nutrients

Agricultural development and urbanisation in the coastal areas has lead to a marked increase in nutrient loads entering estuarine and coastal waters. Nationally, nutrients entering marine waters through river discharge account for about 85% of nutrients in coastal zone waters (Brodie 1997). Land clearing, grazing and the use of agricultural fertilisers are recognised as being the primary causes of increased catchment nutrients and domestic sewage and industrial effluents can be significant near major urban centres (Brodie 1997; Koop and Hutchings 1997b). This has resulted in eutrophic conditions and increased algal blooms in many estuaries and bays around Australia (Brodie 1997).

Australian marine waters are naturally nutrient poor due to a combination of factors including the poor nutrient status of Australian soils, the small quantity of freshwater runoff, and the absence of major upwelling systems. Nutrient deficiency in turn limits primary production. Eutrophication alters the ecosystem via the following biological progression (GESAMP 1990):

- Increased primary production;
- Changes in plant species composition;
- Very dense, often toxic, blooms;
- Conditions of hypoxia (low oxygen concentrations) or anoxia (no oxygen);
- Adverse effects on fish and invertebrates; and
- Changes in structure of benthic communities.

Eutrophic waters typically support a high standing stock of attached algae or phytoplankton (Brodie 1997), the latter also producing an increase in turbidity resulting in decreased light penetration that affects benthic plants and corals.

Adverse effects on marine invertebrates can thus range from massive kills caused by the low levels of dissolved oxygen or the toxins released from dense blooms of noxious dinoflagellates, to more indirect effects resulting from habitat deterioration or shifts in community structure to favour more pollution tolerant taxa. For instance, the turbidity caused by increased growth of microscopic phytoplankton can have serious consequences for the growth and survival of seagrass beds and coral reefs, both of which rely on adequate transmitted light and therefore, relatively clear waters (Engel and Thayer 1998; Section 5.3.2).

Seagrass die-off due to eutrophication, with associated losses of fauna, has been documented in several locations (e.g., Hutchings et al. 1991; Section 6.4.2). Most light reduction is caused by increased water turbidity and an increase in the biomass of epiphytes on the seagrass leaves (Larkum 1976; Hutchings et al. 1991; State of the Environment Advisory Council 1996; Butler and Jernakoff 1999; Longstaff et al. 1999). In Princess Royal Harbour, Albany (southwestern Australia), Peterson et al. (1994) documented dramatic declines in the abundance of two species of suspension-feeding bivalves (*Katelysia scalarina* and *K. rhytiphora*) – previously dominant components of the fauna – from around 160 m^{-2} in 1983-1985 to nearly zero in 1992. In addition to the crash in adult abundances, recruitment was negligible compared to that observed in 1983-85 (Peterson et al. 1994). These declines co-occurred with eutrophication, seagrass die-off and macroalgal blooms. Although the mechanisms for the decline are unknown, it was seen as the result of degradation in the ecosystem due to water quality decline (Peterson et al. 1994). Increasing nutrients can also lead to large increases in the area of seagrass beds - for example around Green Island on the GBR where untreated sewage is discharged from the island (Woesik 1989; cited in Brodie 1997).

The effects of increased nutrients on coral reefs, which occur in oligotrophic waters, are now fairly well known (Scott and Cope 1990; Kinsey 1991b; Morton 1994; Brodie 1997; Tomascik et al. 1997; Causey in press) but the precise ways in which reefs respond to these increases is poorly understood (Brown and Howard 1985; Hatcher et al. 1989; Grigg and Dollar 1990; McCook et al. 1997; Koop et al. 2001). Only a few studies were based on existing sewage discharges on the reef (e.g., Smith et al. 1981; Grigg 1995) or actual eutrophication and pollution gradients (Tomascik and Sander 1985; Tomascik and Sanders 1987a; Tomascik and Sanders 1987b; Tomascik et al. 1997) to demonstrate changes. Most studies, however, have been confined to laboratory experiments, which give limited insights into how entire reefs respond to elevated nutrients (e.g., Hoegh-Guldberg and Smith 1989; Hunte and Wittenberg 1992; Hoegh-Guldberg 1994; Yellowlees et al. 1994).

Increased growth of algae and phytoplankton are stimulated by nutrients such as nitrogen and phosphorus. The phytoplankton can result in decreased light penetration affecting the deeper-water corals and (together with increased bacterioplankton) can also encourage filter-feeding organisms that compete for space with coral, smother it, or that result in increased bioerosion by filter-feeding organisms. For example, Pari et al. (1998) showed that the intensity of bioerosion by grazing increases dramatically when reefs are exposed to pollution. Holmes (1997) examined sponges along a eutrophication gradient in coral reefs in Barbados, and found boring clionid sponges near the most eutrophic site (41%) compared to the least eutrophic site (24%). Furthermore, the abundance of branching corals was positively related to the frequency of boring sponges, suggesting that increased bioerosion may be partly responsible for community shifts toward branching corals in polluted waters (Holmes 1997). While several studies (e.g., Wilkinson 1987; Wilkinson and Cheshire 1990 {Caribbean and Great Barrier Reef}; Meesters et al. 1991 {Curaçao and Bonaire}; Zea 1994 {Colombia}) have found evidence suggesting that sponge communities react positively to nutrient enrichment, a couple (e.g., Muricy 1989; Schroeter et al. 1993) have reported decreases, possibly due to increased levels of sedimentation blocking canals and tissues.

Elevated phosphorus concentrations can also reduce calcification and hence the density of the coral skeleton making the colony more brittle and susceptible to damage (Kinsey and Davies 1979; Rasmussen and Cuff 1990). In addition, increased phytoplankton resulting from increased nutrients may possibly result in increased survival of crown-of-thorns starfish larvae.

There are grave concerns about increasing nutrient levels on the Great Barrier Reef (e.g., Bennell 1979; Bell 1991; Kinsey 1991a; Bell and Elmetri 1995; Wachenfeld et al. 1998; Koop et al. 2001) since with only 17% of catchments adjacent to the GBR are now considered to be in a natural condition (cited in Koop et al. 2001; Gilbert in press) and input of nitrogen and phosphorus has increased about fourfold since European settlement (Moss et al. 1992; Neil and Yu 1996). While the inshore reefs are most impacted (Gabric and Bell 1993; Bell and Elmetri 1995; Brodie et al. 1997; Wachenfeld et al. 1998), the nutrients in river plumes may sometimes reach the outer reefs (Brodie 1996).

Bell and Elmetri (1995) strongly argue that the GBR lagoon is suffering from eutrophication, as indicated by increased phytoplankton density and a higher standing mass of macroalgae, though this assertion remains controversial (e.g., Walker 1991; Hughes and Connell 1999; Russ and McCook 1999)[122]. Nevertheless, algal overgrowth of corals is a recognised problem on many coral reefs (Szmant 2001). In some limited areas of the GBR region evidence of eutrophication is "indisputable" (Brodie 1997; Fabricius and De'ath 2001). However, Russ and McCook (1999), argue that the higher mass of algae on the inner reefs is not necessarily due to anthropogenic inputs of nutrients. They suggest that these areas may have fewer herbivores and that cyclones can greatly increase inshore production, probably through re-working of nutrients from sediments and increased river run-off. Brodie et al. (1997) monitored phytoplankton biomass associated with nutrient inputs in the Great Barrier Reef lagoon. They found chlorophyll levels were generally higher and more variable near the coast, but a compilation of 20+ years of data from the central GBR lagoon showed no evidence for a long-term increase. However, there were pronounced (2- to 4-fold) summer increases in chlorophyll at irregular intervals, emphasising the need for long-term monitoring studies to elucidate the natural patterns and extent of variation.

As a result of concerns about the effects of possible eutrophication of the GBR, the GBRMPA commenced an integrated research and monitoring program in 1991. One part of this was ENCORE (Enrichment of Nutrients on a Coral Reef Experiment), a large-scale *in situ* manipulative reef fertilisation experiment. In this study small patch reefs are being fertilized with nitrogen and phosphorus additions to assess the individual and combined impacts of nutrient enrichment on reef organisms (Steven and Larkum 1993). One aim was to separate the effects of elevated nutrient concentrations from other factors such as increased sedimentation, algal overgrowth and other pollutants (Koop et al. 2001). In summary, reef organisms and processes investigated in the ENCORE experiments were impacted by elevated nutrients, even at relatively low dosages. Dose level, or whether N and / or P were elevated determined particular impacts and were often species-specific. Moreover, these impacts were generally sublethal and subtle, and did not result in visible symptoms of a stressed coral reef (Koop et al. 2001).

Hughes and Connell (1999) argue that it is a mistake to attribute the current status of a coral reef only to present conditions (such as nutrient levels) when historical data do not exist because the current state could be due to a variety of causes such as past storm damage or long-term decline in grazers through overfishing. Experimental additions of nutrients resulted in algal blooms only when grazing fish were excluded (Hatcher and Larkum 1983) as grazing fish could ameliorate the increases in algal biomass following elevated nutrient levels. Thus while attention is often focused on a single stressor, the real world scenario involves a combination of complex, interacting stresses (e.g., overfishing plus altered nutrient loads, plus other natural stresses) (see also Section 6.9).

[122] A summary of the nutrients in the GBR lagoon etc. can be found in Cosser (1997).

Eutrophication can have significant impacts on other marine ecosystems, including benthic communities. Heip (1995) discussed the possible effects of eutrophication on benthic dynamics, describing three successive states based on the amount of organic matter reaching the sediments: a) slight increases in biomass and few or no changes in species composition over the "normal" situation; b) large increases in biomass and replacement of "normal" species by opportunistic species; c) disappearance of benthic animal species and azoic sediments. Increases in benthic biomass and changes in species composition over decades attributed to eutrophication have been documented in a few cases (see Heip 1995). Mesocosm studies tend to show much more rapid changes (weeks to months), but often responses to experimental nutrient additions are inconsistent. Benthic fauna are important for benthic-pelagic coupling, bioturbation, benthic mineralization, and release of nutrients and dissolved organic matter in shallow waters. Consequently the disappearance of this fauna due to increased organic loading and/or anoxia could exert a significant influence on shallow water energy and matter cycles (Heip 1995). Significant changes in benthic communities as a result of eutrophication from aquaculture facilities have been documented in several cases (e.g., Everett et al. 1995; Lu and Wu 1998; see Section 6.11).

***ENCORE*:** In one series of experiments, coral patch reefs at One Tree Island, in the southern GBR, were treated using controlled additions of nitrogen and/or phosphorus (Larkum and Steven 1994). During the initial, low loading, phase – in the first year of the experiment –biotic responses generally were not significantly affected, with the exception of coral reproduction, which declined in all nutrient enrichment treatments. An increased nutrient dosage – used in the second year – resulted in a variety of significant biotic responses (Koop et al. 2001). For instance, organisms containing endosymbiotic zooxanthellae (corals and giant clams) assimilated dissolved nutrients rapidly and were responsive to added nutrients. Coral mortality was evident. Nitrogen addition stunted coral growth, but phosphorus increased coral calcification and growth but reduced skeletal density, making corals more susceptible to breakage (see also Hoegh-Guldberg et al. 1997b). Settlement of coral larvae was reduced (see also Ward and Harrison 1997) in nitrogen treatments, but settlement of larvae from brooded species was enhanced in phosphorus treatments. Recruitment of stomatopods living in coral rubble was reduced in nitrogen and N+P treatments.

Artifical nutrient loads and farming the oceans

Given the evidence on deleterious effects of anthropogenic nutrient loading on marine ecosystems, it is of concern to note recent suggestions to "farm" the oceans by adding massive amounts of fertilisers in the hope of increasing phytoplankton growth and hence (in theory) fish production (e.g., Matsuda et al. 1999; Schueller 1999). According to Schueller (1999), a floating fertiliser pellet has been developed that releases iron and phosphorus over several days, and there are plans to commercialise this product. It is claimed that 25 000 tonnes of this fertiliser could increase fish production by 50 million tonnes and reduce global warming into the bargain (Schueller 1999). According to Schueller (1999), attempts by the developer to get property rights to part of the Gulf Stream off the US east coast to test the product failed, but he has since negotiated an agreement with the Republic of the Marshall Islands to use their waters. Critics of the idea (quoted in Schueller 1999) argue that there is a potential to trigger toxic blooms and deleterious changes to ecosystems, and that the species attracted to the plankton bloom may not be commercially valuable. Cavender-Bares et al. (1999) showed that a plankton bloom induced by iron fertilisation caused the dominant species to increase only slightly or to decrease while the number of initially rare pennate diatoms increased 15-fold. This differential response led to a dramatic change in the phytoplankton community structure with consequent impacts to food web structure. In addition, certain diatoms can suffocate fish directly by clogging their gills or indirectly by depleting oxygen. As dying planktonic blooms decay, they can produce ammonia and sulphides harmful to marine life (Schueller 1999). The claim that it will counter greenhouse emissions through uptake of CO_2 by algal blooms that then sink to the sea floor is unlikely as any significant CO_2 reduction might be negated by the production of other greenhouse gases. Fertilising can create local zones low in oxygen in which anaerobic bacteria produce methane and nitrous oxide, which have respectively 21 and 2000 times the global warming potential of CO_2. This is not just an isolated idea, as similar projects are underway at other locations. At the University of Sydney, the Ocean Technology Group is planning a similar pilot project off Indonesia, and in Norway, two companies also are investigating ways to maximise fish harvests (Schueller 1999). In a related approach, Matsuda et al. (1999) argued that open sea productivity could be enhanced through artificial upwelling of deep-ocean nutrients. If such a technology was developed then a 1000 MW ocean ranch, fed by artificial upwelling, could produce over $1 billion a year in seafood alone – or so it is claimed. While such ideas currently seem to be in the realm of science fiction, the critical state of global fisheries, the need to increase aquaculture production and the ever-improving nature of marine technology suggest they may one day become feasible. The current paucity of information on oceanic systems or their biological communities, and the fact that potential environmental / ecological impacts usually are not considered (Matsuda et al. 1999 refer to a "lack of nullifying obstacles") is a matter for considerable concern.

Sediments

Increased sediment loads result in less light penetration and siltation. The process originates from human activities in catchments (e.g., vegetation clearance, excavation, construction – Section 6.16) that lead to greater volumes of terrigenous material being carried into coastal areas by rivers, or from physical disturbance of marine sediments (e.g., by trawling, dredging, or drilling). For example, dredging increases water turbidity while the dredging is in progress and the effect is prolonged when the dredge spoils are resuspended by wave action (Fitzpatrick and Kirkman 1995).

Poor management practices in catchments can result in markedly increased sedimentation (Goulay and Hacker 1986; Arakel et al. 1989; Frouin 2000), and may result in the expansion of mudflat and mangrove habitat at the expense of other habitats (Brodie 1997). In addition, when suspended sediments settle, benthic organisms and aquatic vegetation can be smothered and bottom substrates altered (e.g., from sand to mud).

The turbidity caused by suspended soil particles, like that resulting from algal blooms (Section 6.8.3), can have direct and indirect effects upon organisms and habitats. Direct impacts include clogging of gills, reduction in larval settlement and reduction in feeding ability of predators that hunt by sight (Anonymous 1989; Fisk and Harriot 1989; Mapstone et al. 1989; Hopley et al. 1990; Stafford-Smith 1993; Stafford-Smith et al. 1994; Oliver 1995; Anonymous 1995a; Nowlis et al. 1997; Ayling and Ayling 1998). Indirect impacts include loss of aquatic vegetation due to poor light penetration, notably declining seagrass beds. Soft corals are also affected by increased rates of sedimentation (Fabricius and De'ath 2001).

Effects on seagrasses

Seagrasses are susceptible to sedimentation due to both light reduction and direct smothering (Robertson and J. 1991). Depth distributions of seagrasses are linked to light penetration and they can be impacted by events that reduce the amount of light reaching the seagrasses for long periods (Fitzpatrick and Kirkman 1995). Burial following large-scale sedimentation events can result in dramatic losses in some instances; for example, over 1000 km^2 of seagrass beds at Hervey Bay were destroyed in 1992 after large quantities of sediment were washed down from the mainland (Preen et al. 1995; State of the Environment Advisory Council 1996). While this was the result of a cyclone, land clearing probably resulted in a markedly increased sediment load.

Light deprivation occurs through increased turbidity or increased epiphyte growth on seagrass leaves (often related to increased nutrient levels). Longstaff et al. (1999), in an experimental study on the seagrass *Halophila ovalis* in Queensland, showed that it declined in biomass after 3-6 days of darkness and died after 30 days of continuous darkness. Thus the presence of this species may depend on how transient the light deprivation events are. Fitzpatrick and Kirkman (1995) found that experimental shading of a *Posidonia australis* meadow in Jervis Bay, New South Wales, caused significantly lower leaf growth rate, shoot density, shoot weight and epiphyte weight. The epiphyte community also changed from being mainly fleshy macroalgae to being dominated by encrusting invertebrates. Epiphytes are an important component of seagrass assemblages, both as primary producers and as habitat and food for invertebrates. Changes to their community structure may have impacts on the seagrass community (Fitzpatrick and Kirkman 1995; see also Section 5.3.2).

The effects of sedimentation and light deprivation on seagrass communities will undoubtedly be dependent on the duration and intensity of shading, whether it is continuous, periodic, irregular or infrequent. However, very little is known regarding these parameters and more experimental studies are needed to provide this information.

Effects on coral reefs

Coral reefs can be severely affected by sedimentation and turbidity but, conversely, some corals are tolerant of low light and have efficient sediment rejection mechanisms that allow them to thrive in muddy conditions on inshore reefs (Stafford-Smith and Ormond 1992), adjacent to rivers or areas of coastal development (Craik and Dutton 1987; Fisk and Harriot 1989; Ayling and Ayling 2000). There is considerable anecdotal evidence and some historical photographic evidence that inshore reefs are muddier and have less coral and more algal cover than previously (Brodie 1997).

Cortes and Risk (1985) studied a reef in Costa Rica under siltation stress from adjacent deforestation. They found low coverage and growth rates for live coral. Wesseling et al. (1999) examined damage to and recovery of corals in the Philippines subjected to experimental sediment burial for up to 68 hours. After twenty hours of burial corals showed tissue discolouration; after 68 hours about 50% of tissue disappeared leaving bare coral skeleton exposed or covered with algae whilst up to 90% of the remaining tissue was bleached. Recovery to the preburial state occurred after three to four weeks. *Acropora* sp. which is found at 5m depth died after 20 hours of burial. These authors concluded that complete burial caused considerable whole-colony mortality, at least in *Acropora*, and thus may result in a permanent loss of some coral taxa from reefs subject to intense or brief (less than one day) sedimentation events. Less sensitive taxa incur substantial damage but show significant recovery after several weeks (Wesseling et al. 1999). Periodicity of the sedimentation events is an important component about which there are few data. The available data show that impacts and the recovery periods between stress events for individual taxa will obviously vary but details must await further experimental studies.

Effects on the benthos
Benthic communities can also be affected by sedimentation events through smothering and burial. For instance, Stephenson et al. (1977) documented the effects of a major flood in 1974 on the benthos of Moreton Bay. The average number of individuals and species per site was significantly lower immediately following the flood, although the number of individuals subsequently increased to a level significantly above pre-flood numbers. The observed changes resulted from a combination of stresses associated with the flood and cyclone (e.g., reduced salinity, which fell to at least 24‰), but sedimentation certainly was a key factor. It was recorded, for instance, that a transient cover of soft silt occurred after the flood, with an overall increase in mud (Stephenson et al. 1977). Dredging in Botany Bay has changed the sediment distribution within the Bay, for example increasing the amount of fine sediments, which in turn has led to changes in the composition of benthic communities (Jones and Candy 1981).

Oil and petroleum

Approximately 3.8 billion litres of oil enters the oceans yearly nearly all due to human activities (Suchanek 1993). Only 8% of this input is derived from natural sources such as seeps. By comparison, at least 22% is intentionally released through normal tanker “operational discharges” and 12% from accidental tanker spills. Other sources include inputs from municipal and industrial sources (c. 31%), atmospheric deposition (c. 9%), urban and river runoff (c. 5%), offshore oil production (c. 2%) and ocean dumping (c. 1%) (Suchanek 1993). Thus, although accidental spills of oil and petroleum compounds focus attention on the impacts of transient oil pollution events, a far greater annual volume is discharged into the sea through routine shipping operations. Oil and petroleum products impact on marine invertebrates by smothering (especially when the oil washes ashore and affects intertidal organisms) and toxicity, with both acute (lethal) and chronic (sub-lethal) effects. Some fractions are soluble and disperse throughout the water column, while heavier fractions may sink and cover benthic communities. Dispersants sprayed onto oil spills may simply facilitate the oil's absorption into sediments and tissues and have been reported to be toxic to reef corals (e.g., Ballou et al. 1989). In addition, the immiscible surface fraction could affect light penetration and hence surface/upper pelagic communities of invertebrates and algae.

Studies of actual pollution events and manipulative experiments, on the effects of oil spills on intertidal communities were summarised by Suchanek (1993). Oil spills have been responsible for the destruction of entire coastal shallow-water communities that have taken years to recover (Peters et al. 1997). The effects of disasters such as the *Exxon Valdez* spill in Prince William Sound in Alaska (1988) have been well studied (e.g., Paine et al. 1996; Jewett et al. 1999). While the effects of such incidents (such as smothering of intertidal habitats) are obvious, chronic contamination by oil can also have an impact. For instance, sediments in areas of frequent shipping activity can be affected, as can interstitial species living in the littoral zone where large amounts of oil debris are common. Ponder (1990) described a mollusc community from upper littoral interstitial gravels at Ceuta in the Straits of Gibraltar, which survives only in a few small patches due to the large amount of oil and grease found in the interstitial spaces over most of the shore. Two species of interstitial slugs of the genus *Smeagol* (see Section 4.4.1) are known only from restricted areas in the upper littoral zone on the southern shore of Phillip Island (Tillier and Ponder 1992). This was the site of a recent serious oil spill which highlights the fact oil spills can result in the contraction of suitable habitat for certain species or potentially even cause extinction if the species has a restricted distribution within the affected area.

It is known that metamorphosis in some sea anenomes (Chieu and Berking 1997) and gamete production in some echinoderms (Nicol et al. 1977) are adversely affected by hydrocarbons.

A national plan has been formulated to combat pollution of the sea by oil and other noxious and hazardous substances[123] (AMSA 1997).

Effects on mangroves, saltmarshes and seagrasses

Oil spills have a major impact on saltmarshes (IPIECA 1994) and on intertidal seagrass beds (Lee Long and Coles 1997) and their associated fauna through loss of habitat, although subtidal beds may be at lesser risk. McGuinness (1990) examined the short and long-term effects of oil spills on molluscs and crustaceans in saltmarshes and mangroves in Botany Bay and found some short-term reductions in abundance but few long-term effects (the exception being *Salinator solida*, which still showed reduced abundance five months after the second oiling). Clarke and Ward (1994) studied the response of two common saltmarsh plants and gastropods to experimental contamination by weathered petroleum hydrocarbons designed to mimic an accidental spill. The plants were severely

[123] Guidelines for Acceptance of Oil Spill Dispersants. Available at http://www.amsa.gov.au/me/natplan/toolbox/dispersa/spill3.htm#14

affected for many months, and there was an initial high mortality of gastropods. Migration from the edges of the experimental area restored gastropod densities to pre-treatment levels within a few months. Studies that examined the effects of oil in mangroves and saltmarsh in the Gladstone area with and without application of an experimental bioremediation protocol showed that bioremediation was helpful in biodegrading the oil in both saltmarshes and mangroves (Duke et al. 1998; Duke et al. 2000; Ramsay et al. 2000).

Effects on coral reefs

The effects of oil on reef corals in their natural environment are poorly understood. Field studies suggest the effects on oiled coral populations are harmful and result in long-term damage, as measured by abundance, mortality, reproduction, and recruitment (e.g., Rinkevich and Loya 1977; Teal and Howarth 1984; Bak 1987; Guzmán et al. 1991; Harrison 1994; Kushmaro et al. 1997; Harrison 1999; Heyward et al. 1999). Observations supporting such findings generally are for intertidal corals and lack baseline measurements. One study for which baseline measurements were available was a refinery spill in Panama during 1986 that occurred near the Galeta Marine Laboratory, Smithsonian Tropical Research Laboratory. Oil slicks from the refinery landfill and mangroves were still there after 2½ years. Guzmán et al. (1991) studied the short-term effects on common shallow subtidal reef corals, at the individual, population, and community levels and found significant short-term effects. Numbers of corals, total coral cover and species diversity decreased with increased amounts of oiling. The large branching coral *Acropora palmata* decreased most in terms of cover. Frequency and size of recent injuries on massive corals increased with the level of oiling, and growth of three massive coral species was less at oiled reefs in the year of the spill than during the nine previous years.

Detergents and other chemicals used to disperse the oil are often considered to be as much, or more of, a problem than the oil itself (e.g., Singer et al. 1993; Singer et al. 1995a; Singer et al. 1995b; Siron et al. 1996; Singer et al. 1998; Wolfe et al. 1998). However, results of experiments on the effects of oil and dispersants in the laboratory or field give contradictory results. The effects of oil and dispersants on corals in such studies range from little or no mortality to persistent sub-lethal effects (e.g., Teal and Howarth 1984), to more serious and lasting damage (e.g., Johannes et al. 1972; Jackson et al. 1989). Experimental studies on the effects of petroleum hydrocarbons on fertilisation rates, larval settlement and metamorphosis of corals and coral larvae include Rinkevich and Loya (1979), Harrison (1994; Harrison 1999), Heyward et al. (1994) and Kushmaro et al. (1997). However, extrapolation of such experimental results to the field is hampered by the necessarily unnatural spatial scale and duration of the manipulations (e.g., Capuzzo 1987).

Effects on rocky shores

Edgar and Barrett (2000) studied the effects of the grounding of the bulk carrier *Iron Baron* on Hebe Reef in Northern Tasmania, which resulted in the release of approximately 350 tonnes of Bunker C fuel oil. However, the release of oil did not appear to substantially affect populations of subtidal reef-associated organisms in the near vicinity, with the major impact resulting from physical abrasion in the area of the grounding (see Section 6.12.1).

Effects on sandy shores and sublittoral sediments

The effects on sandy beach amphipods from the oil spill from the *Laura d'Amato* in Gore Cove, Sydney Harbour in August 1999 have been studied (Jones in press). This incident, which involved the spillage of about 250 tonnes of light crude oil, has apparently reduced populations of amphipods on affected beaches. Although no pre-spill data are available, all reference sites were found to support amphipods whereas polluted sites, following the event, did not. Likewise, lightly polluted sites appear to have made a good recovery whereas heavily polluted ones have not. In another study on amphipods, their recovery in sublittoral sediments was studied following the oil spill from *Amoco Cadiz* wreck in 1976 in Britain (Poggiale and Dauvin 2001).

Effects in the Southern Ocean

Smith and Simpson (1995) and Simpson et al. (1995) described the effects on invertebrates of a small light marine diesel spill, resulting from the grounding of a supply ship at Macquarie Island. Shortly after the time of the spill many marine invertebrates were washed up dead along two kilometres of shoreline. One year later, the effects on algal and invertebrate populations on the littoral and sublittoral rocky shore, and in holdfasts of the giant kelp *Durvillaea antarctica*, were investigated. On rocky substrate the effects of the spill were restricted to some biota of the lower littoral and sublittoral zones, particularly echinoderms and a patellid limpet. There were also differences between clean and contaminated sites in the invertebrate communities inhabiting *Durvillaea antarctica* holdfasts, with residual levels of hydrocarbons apparently restricting polychaete species to opportunistic taxa which were not found at control sites. Seven years after the spill, Smith and Simpson (1998) found no significant differences between oiled and control sites on the rocky shore, although holdfast macrofaunal communities still showed evidence of impact. These results appeared to indicate that even small oil spills can have long-lasting consequences for marine communities at Macquarie Island, and presumably also other areas at high latitudes.

Sewage

Sewage effluents contain a complex mixture of substances and their composition is highly variable over time and from place to place. Sewage is composed primarily of organic matter and water, but may also contain a variety of contaminants, some potentially hazardous, such as oils and grease, ammonia, pesticides, herbicides, heavy metals, bacteria and viruses.

The impacts of sewage effluent on marine benthic communities have been well studied in the Sydney region due to the decision to move the three main outfalls at North Head, Bondi and Malabar approximately three kilometres offshore (primarily for reasons of visual amenity and human health associated with the use of beaches). From 1973-1975, a team from the Australian Museum carried out a Shelf Benthic Survey to collect baseline data on ecological systems near the proposed deep-water outfalls (Jones 1977). Following construction of the outfalls, the NSW Environment Protection Authority established a monitoring program and a number of studies have been published (e.g., Koop and Hutchings 1997).

Few studies have demonstrated correlations between sewage discharge and community change, a fact which has either been attributed to the difficulty of distinguishing impacts from high natural spatial and temporal variability, or actual low impacts from sewage disposal through these outfalls. Underwood and Chapman (1996) found that, despite widespread concerns about the effects of sewage discharged into Sydney's coastal habitats, there was no indication of it being an ecological problem affecting the distribution and abundance of common sessile fauna or the composition of assemblages in immediately adjacent subtidal rocky habitats. They described changes to subtidal assemblages on rocky reefs around a cliff-face sewage outfall at Sydney, after the outfall was turned off, relative to two reference locations. Although most taxa showed significant changes over the study period of 20 months, most of these were unpredictable and varied interactively according to the taxon, plot, depth and location. For most measures, the effects due to sewage lay within the bounds of natural variability shown by the reference locations. However, the study was limited because of low replication at the site level and the first samples were taken four months after decommissioning so important changes might have already occurred. In addition, many organisms were identified from quadrat photos only to higher taxonomic groups (e.g., bryozoans or ascidians), so small mobile organisms (the majority of the taxa present in such communities) were not considered. Other studies have found significant changes. Roberts (1996a; 1996b) and Roberts et al. (1998) described rapid changes in encrusting marine assemblages in response to sewage discharge, with marked declines in algae and sponges within three months and a shift to a community dominated by silt and ascidians. Similarly, a change from an algal dominated community to one dominated by suspension feeding sponges and ascidians was reported by Fairweather (1990). A more recent study on the Sydney outfalls (Archambault et al. 2001) found that invertebrate diversity was lower at outfalls than at control sites but that recovery occurred following the closure of an outfall. Overall, these studies suggest that sewage disposal is not a significant problem for marine invertebrate communities (e.g., NSW EPA 1997). Other studies: Otway (1995), has shown that deep-water sewage outfalls off the coast of Sydney, caused an impact on soft bottom communities, some species increased in abundance whereas others decreased with the major environmental change an increase in suspended solid loads. He suggested that over time the sediments around the outfalls will become greatly enriched with nutrients and this would lead to changes in the abundances of the benthic fauna. Otway et al. (1996) discussed the experimental design of monitoring programs to detect such changes and ways in which estimates of spatial and temporal variation could be incorporated.

Elsewhere in NSW, Smith (1994) investigated the impact of domestic sewage effluent versus natural background variability in Jervis Bay, and Smith (1997b) studied marine communities in the vicinity of pipes discharging secondarily treated domestic sewage at Coffs Harbour. The latter study found that algal species richness and changes to the structure of invertebrate communities living in kelp holdfasts generally were restricted to within about 300m of the outfall. Sewage can even be a local problem for invertebrate communities at high latitudes such as at McMurdo Station, Antarctica, where effects such as bacterial concentrations in invertebrates and sediment accumulation were found up to several hundred metres from the outfall (Edwards et al. 1998).

Litter

There is increasing concern over the extent to which surface waters and shorelines everywhere have been progressively fouled by debris of human origin. Unsightly accumulations of such debris are now occur at the most remote and generally inaccessible shores, e.g., the uninhabited Oeno and Ducie Atolls and Henderson Is in the Pitcairn Group, as well as seldom visited subantarctic islands (Slip and Burton 1991; Gregory 1998). For instance, Ryan and Moloney (1993) surveyed litter washed up on the shores of Inaccessible Island (37º 15'S, 12º 30' W), a remote and uninhabited island in the south Atlantic Ocean. They found an exponential increase in stranded litter between 1984 and 1990, from less than 500 items per kilometre in 1984 to almost 2500 pieces km^{-1} in 1990.

Land-based sources of marine debris are probably a much more significant factor than are vessels (Liffmann 1994), which are often claimed to account for only 10-25% or less of all marine pollution (Wace 1995; Gregory 1998). Nevertheless, there can be little doubt that locally, a significant proportion of marine debris may have its origins in nearby offshore maritime activities, such as the fishing industry and shipping (Gregory 1998). For instance, beach surveys in Fog Bay in northern Australia suggested that commercial fishing, merchant fishing and recreational boaters were the likely source of over 85% of all debris items (Whiting 1998). The impact of marine fishing debris on the Australian marine environment was reviewed by Jones (1995), while Wace (1995) reviewed the extent of stranded litter in general on Australian coasts. Marine debris is dominated by persistent synthetic materials, particularly plastics. In 1975 the US National Academy of Science estimated that 6.4 million tonnes of litter were jettisoned from ships at sea each year, and in 1982 a daily input of more than 600 000 plastic containers into the oceans was attributed to shipping (Laist 1987). These figures have undoubtedly risen in the intervening decades.

While such material is clearly unsightly and is well known to cause problems for some vertebrates through tangling, choking etc., the impact of rubbish on invertebrates is less obvious. Plastics and fishing gear also have been reported as entangling some crustaceans (Laist 1987). Plastic bags and fishing nets may smother seagrass beds and infaunal species, and annual clean-up days provide a vivid reminder of how much rubbish actually ends up in the marine environment. Litter also provides novel attachment sites for sessile invertebrates (Ryan and Moloney 1993), and may serve as a vector for the transport of exotic species (Gregory 1998).

6.6.2 Information, research needs and management

Information and research needs

The information and research base to assess and monitor the impacts of pollution on marine invertebrate communities is woefully inadequate. While increasingly sophisticated studies have been undertaken on coral reef systems, most other marine ecosystems have been largely ignored with, at best, only a handful of well formulated studies from around Australia, or even internationally.

Peters et al. (1997), in a review of the effects of chemical pollutants on tropical marine systems worldwide, stressed that information is lacking, particularly with regard to:

- Long-term recovery;
- Indicator species; and
- Biomarkers for marine communities, and suggested that critical areas that must be addressed include:
    - Development of appropriate benchmarks for risk assessment;
    - Baseline monitoring criteria; and
    - Effective management strategies to protect marine ecosystems in the face of mounting anthropogenic disturbance.

In addition, there is a need for more experimental studies on the toxicity and sublethal effects on a range of invertebrates of the hundreds of different chemicals and other pollutants being introduced to marine ecosystems. Many studies focus on single taxa but these cannot be used as surrogates given the huge phylogenetic and physiological diversity in invertebrates.

Management

The management of water quality involves development of policy, cooperation between the levels of government and government departments, and effective enforcement. Management strategies need to focus on the links between terrestrial and marine systems, notably catchment, agricultural and soil management. One particular issue is the reduction of terrestrial run off to reduce the input of nutrients and pollutants into estuaries and the sea. Although state environmental agencies are ultimately responsible for regulating pollution from run off, effective catchment management schemes at the local level are probably the best solution to this problem. Coordination of guidelines resulting in better land management methods (improved agricultural methods, rehabilitation of riparian zones and wetlands, reafforestation, reduction in the use of agricultural chemicals) should lead to improvements in the current levels of pollution and erosion.

Despite the complexity of the diverse sources and kinds of pollutants and their differing impacts on different taxa, management programs tend to focus on the alleviation of a particular problem. This can cause difficulties. For example, degradation of coral reefs is often associated with declining water quality (e.g., Bennell 1979; Walker and G. 1982; Bell 1991; Kinsey 1991a, 1991b), but the complex nature of the inputs to coastal areas makes it difficult to identify the components (e.g., nutrients, sediment, heavy metals) responsible for the changes and, hence, the development of management strategies to address the problem (Koop et al. 2001).

Sewage disposal from coastal communities is one of the major causes of pollution. Tertiary treatment (nutrient removal) or land irrigation of sewage effluent should be encouraged, as has been the case in the GBR Marine Park where policies of maximum re-use and minimum discharge has led to reductions in nutrient fluxes in coastal systems (Brodie 1991, 1995, 1997).

Pollution from shipping requires Commonwealth and International regulation and can be considerable, particularly the disposal of bilge water, oil spills, and the dumping of rubbish. International legislation implemented in 1988 (MARPOL, the International Convention for the Prevention of Pollution from Ships[124]) and Australian legislation implemented in 1990 (Protection of the Sea Act 1983 – Prevention of Pollution from Ships) regulates the discharge of debris in Australian waters. Under these laws, discharge of plastics is prohibited and the discharge of other items is restricted to a minimum distance from shore (Whiting 1998). These laws, however, are difficult to enforce and illegal dumping of debris continues (Ryan and Moloney 1993), with smaller commercial and recreational vessels probably being significant contributors.

Economic instruments offer another approach to overcoming the problem of pollution. For instance, the Ecologically Sustainable Development Working Group on Tourism (1991) and the Australian Water Resources Council (1992) have proposed a number of measures, including:

- Charges for water use and sewage treatment should adequately reflect the costs of supplying these services. These charges should also incorporate the cost of preventing undesirable pollution;
- Non-compliance fees or fines should reflect the profits obtained by polluters by failing to comply with established standards;
- Subsidies or low-interest loans should be used to encourage the development or adoption of waste minimisation technologies;
- Differential taxation rates should promote the use of environmentally preferred products or technologies; and
- Development charges should reflect the social and environmental impacts of the development.

The Australian Maritime Safety Authority (AMSA) has a "National Plan to Combat Pollution of the Sea by Oil and Other Noxious and Hazardous Substances." Under this plan, all States and the NT have embarked upon programs to record and document their coastal and marine resources in the form of a Coastal Resource Atlas (CRA). These “provide a means of determining marine and coastal areas of sensitivity that could be impacted in the event of a pollution incident as well as providing valuable resource and

[124] This is the most far-reaching step to control marine pollution (including persistent plastic litter) from marine sources. It banned the disposal of persistent wastes at sea. However, there have been few attempts to monitor its effectiveness, at least with regard to litter (Ryan and Moloney 1993).

logistical information for combat authorities" (AMSA 1997). However, the CRA as currently formulated does not appear to take into account invertebrate (e.g., benthic) communities, these not having been mentioned in the list of data sets. The utilisation of invertebrate data is possible, as illustrated by the recent bioregionalisation for the GBR with the aim of establishing a system of representative marine areas (Fernandes et al. 1999; GBRMPA 1999; Day et al. in press). This process has used accessible data, including much of the available information on the distribution and abundance of invertebrates, to define the various habitats within the GBR region. We therefore recommend that invertebrate resources be added to the Coastal Resource Atlas wherever possible.

Whatever the means, reduction of pollution of the environment is an urgent goal if the long-term sustainability of marine communities is to be realised. While the evidence is often poor for demonstrating, the specific culprit (eg. toxins, vs sediment, vs nutrients) in observed impacts there is ample evidence that the sum of human-induced perturbations causes deleterious effects to marine communities. There is a widespread deterioration in marine ecosystems and to wait for irrefutable scientific evidence before meaningful action may result in a case of 'too little, too late'.

6.7 Long-term environmental change

The effects of global warming due to anthropogenic activities are becoming increasingly well documented. For the marine environment, potential effects are increases in sea level, temperatures and storms (Ray et al. 1992; Reid and Trexler 1992; GESAMP 1997; IPCC 1998; US EPA 2000). Despite the increasing evidence that sea and atmospheric temperatures are rising (e.g., Hughes 2000), global warming is still not universally accepted. There is a range of information available on global warming scenarios, potential impacts and management on several websites, including the US Environmental Protection Agency's global warming site[125] and the homepage of the Intergovernmental Panel on Climate Change[126].

6.7.1 Potential effects of global warming and climate change on marine invertebrates and their habitats

The potential effects of global warming were noted above. The potential impacts of these changes on marine organisms include:

- Direct effects of increased atmospheric CO_2 on some biological processes (e.g., coral calcification, photosynthesis by phytoplankton);
- Direct effects of increased sea temperature on survival of species living close to their upper thermal limit (e.g., coral bleaching);
- Changes in the distributions of species, resulting in changes to local invertebrate communities through the local disappearance or decline of taxa and the introduction of new taxa;

[125] http://www.epa.gov/globalwarming/
[126] http://www.ipcc.ch/.

- Loss (or alteration) of low-lying coastal habitats as a result of rising sea level and increased storm frequency.
- Consequent creation of 'new' habitats as sea level rises.

In each of these cases, the responses by human communities may comprise an additional and significant threat to habitat and invertebrate populations. These would result from increased coastal modification by humans (beach works, seawalls etc.) to minimise the damage to human habitations and infrastructures.

Increased atmospheric CO_2

Most research has addressed how increasing atmospheric CO_2 might affect terrestrial plants; indeed, some of that research illustrates the potential for positive effects through enhancement of plant growth (e.g., IPCC 1998; Rosenzweig and Hillel 1998). There is also the potential for increased CO_2 levels to enhance phytoplankton growth and perhaps increase the overall productivity of the oceans, although the implications of this for oceanic ecosystems are uncertain.

There is evidence that coral reefs may be directly affected by increasing atmospheric CO_2 entering the oceans (e.g., Kleypas et al. 1999; Wilkinson 1999) through effects on seawater chemistry and coral calcification rates. Evidence to support this includes:
- The fact that latitudinal limits to reef development correlate not only with temperature, but with saturation state as well;
- Geological evidence that aragonite sedimentation rates correlate latitudinally with changes in saturation states;
- Geological evidence that Cainozoic coral reef development did not flourish until after the Eocene (the last geological period thought to have had unusually high atmospheric CO_2 levels);
- The observation that reef cements and other inorganic precipitates of $CaCO_3$ are more prevalent in regions of high saturation states; and
- Direct experiments that demonstrate the relationship between $CaCO_3$ saturation state and coral / algal calcification (Joan Kleypas, quoted in Wilson 1999).

Rising sea temperatures

Various studies on the west coasts of the USA and Hawaii have considered the impact of rising seawater temperatures on the distribution of intertidal organisms. Sagarin et al. (1999) showed that the abundance of macroinvertebrates in a rocky intertidal community between surveys in 1931-1933 and 1993-1996 had changed, with southern species increasing in abundance, northern species declining, and cosmopolitan species showing no clear trend. They related these changes to rises in seawater temperatures, and suggested that ENSO (El Niño-Southern Oscillation) events are relatively unimportant compared to these temperature rises.

El Niño-Southern Oscillation

The El Niño-Southern Oscillation (ENSO) is one of the most important sources of inter-annual climate variability. The past 20 years have experienced the two strongest El Niño events in recorded history, leading many to speculate that ENSO may be changing in response to global warming. Nevertheless, there is evidence to show that strong ENSO events have occurred over a much longer time frame (Cole 2001). Some of the best evidence comes from work by Tudhope et al. (2001) who studied the extensive raised fossil coral reefs on the Huon Peninsula in PNG. Using annually banded corals from Papua New Guinea, Tudhope at al. (2001) have shown that ENSO has existed for the past 130,000 years operating even during “glacial" times. During the 20th Century, ENSO has been strong compared with previous cool (glacial) and warm (interglacial) times. However, these data from Papua New Guinea are sparse and need to be supplemented by adding records from other locations and time-periods before it will be possible to attribute the sensitivity of ENSO to specific aspects of global change. This is necessary if we are to successfully predict the consequences of future greenhouse gas-induced warming on ENSO.

Rising sea levels

Increases in sea level are an expected consequence of rising temperatures. The Intergovernmental Panel on Climate Change has forecast sea level rises ranging from 15 centimetres to almost one metre. The lower limit of these estimates corresponds to the rate of sea-level rise that has been occurring for the past century or two. The upper-limit estimate represents a substantial acceleration that may happen but for which, as yet, there is no strong evidence (IPCC 1995).

The main issue for marine invertebrates is the loss or change in coastal habitats. In particular, the biota restricted to or dependent on the narrow band of habitat close to sea level will be subjected to rising sea levels from below as well as increased pressure from development above (Reid and Trexler 1992). Rising seas will stress coastal habitats including wetlands, barrier islands, coral reefs, coastal lagoons, mangroves and saltmarshes. The areal extent of some of these habitats could well be contracted, if the rate of change is faster than the adoption of new available habitats.

There is an extensive literature on how mangroves might be affected by sea level change[127]. Nevertheless, stable sea levels are required for the formation of extensive mangrove swamp forests and have occurred only intermittently over the late Quaternary (Crowley 1996). Semeniuk (1994) discussed the effects of sea-level rise on mangroves in northwestern Australia and concluded that the mangroves’ response would depend on the environmental setting, including the homogeneity (geomorphologically and sedimentologically) of the coast, its tidal range, stability, and history, as well as the variety of species present and their reproductive strategies. In some places (e.g. King Sound in WA, where natural erosion, progressing at 1-3 cm/yr, simulates the effects of a rising sea), mangroves are able to migrate landwards, generally keeping pace with the

[127] see http://possum.murdoch.edu.au/~mangrove/sea_level.html

retreat, through colonisation (by seedling recruitment) of habitats made available by increased inundation. However, mangroves in other parts of NW Australia, where more heterogeneous assemblages have developed, are not likely to adjust so rapidly, and hence would be disrupted (Semeniuk 1994).

Changes in storm patterns etc

The effect that global warming might have on circulation patterns in the major oceans has been a topic of much speculation and research (Rahmstorf 1999), in part because of the link between the major current systems and global and regional climate. Recently, Wood et al. (1999) presented a new climate model and greenhouse scenarios that suggest possible dramatic changes in the large-scale circulation patterns and currents in the Atlantic in response to increases in greenhouse gas concentrations.

Property and other damage from cyclones, storms and associated wind, erosion and flood hazards appear to have increased markedly in recent years (e.g., David et al. 1999). In Australia, cyclones currently cause periodic damage to coral reefs (e.g., Connell et al. 1997) and – due to increased rainfall and river flows – sedimentation events, which impact on marine invertebrate communities (see Section 6.6.1). A greater frequency or intensity of such events could, in combination with other anthropogenic stresses, reduce the opportunities for normal recovery leading to long-term changes in habitat type and community compensation (Hughes and Connell 1999).

Loss of, and changes to, coastal habitats

Fluctuating sea levels and changing coastlines have occurred throughout geological history. Marine invertebrates are, overall, able to adapt to such changes through their capacity for dispersal and recolonisation, but much depends on the pace of environmental change. The potential to alter their distributions to suit new coastlines is complicated by the rapidity of the projected human-induced changes and the fact that large areas of coastal habitat have been alienated by human development, creating barriers to dispersal and recolonisation (e.g., saltmarshes, Hutchings 2001).

6.7.2 Ozone depletion and UV radiation

The decline in stratospheric ozone at high and mid latitudes has been correlated with increases in biologically damaging ultraviolet-B radiation (UVBR). The effects of increased UVBR on natural ecosystems have not been adequately assessed, although it has (rightly or wrongly) been implicated in the decline of invertebrates in freshwater ecosystems (e.g., Bothwell et al. 1994; Karentz et al. 1994) and frogs at high altitudes (Blaustein et al. 1994). The majority of marine organisms in deeper or more turbid waters are unlikely to be affected, due to the attenuation of UVR in the water column. UVB, however, does have the potential to damage DNA and chromatophores of marine plants and animals as deep as 20 m in clear oceanic waters and epibenthic invertebrates in shallow tropical waters (Hovel and Morgan 1999). The significance of this threat will depend not only upon the rate of ozone depletion (which is expected to peak soon and recover to pre- ozone depletion levels over the next 50 years; Brown 1997), but also any changes in climate patterns brought about through global warming. These later changes may affect the amount of UV reaching the surface, for instance through increased cloud cover (Brown 1997).

Marine organisms have evolved to live in a wide range of habitats and those on coral reefs are functionally adapted to extreme tropical conditions. Many groups of organisms living on reefs have developed efficient defences against potential damage from chronic solar exposure. This protection includes the elaboration of natural UV-absorbing compounds (sunscreens) and other mechanisms such as antioxidative enzymes and small molecule antioxidants. This has led to the synthesis of the UV absorbing compounds and their trailing as sunscreens for humans and in other applications (Dunlop et al. 1999; Shick and Dunlop 2002).

The relatively few studies on the effects of UV radiation on marine invertebrates have focused on corals, which are likely to be vulnerable given that they occur exclusively in clear, shallow tropical waters.

Studies on corals

Being sessile, and therefore unable to respond behaviourally to avoid UV damage, corals depend on physiological mechanisms such as screening by UV-absorbing compounds and recovery due to photoreactivation (Siebeck 1988; Gleason and Wellington 1993; Norris 1999). Siebeck (1988) investigated UV tolerance in hermatypic corals (Scleractinia) from the Great Barrier Reef, including the effect of depth. The most conspicuous symptom of serious UV radiation damage to polyps was extrusion of mesenterial filaments through the mouth opening and entire body surface. Corals taken from 1.5 m depth had, on average, 2-6 times the UV tolerance of those collected at 18-20 m. Gleason and Wellington (1993) transplanted colonies of a common reef-building species (*Montastrea annularis*) from 24 m to depths of 24, 18 and 12 m depth, at each depth leaving some colonies exposed to ambient UV radiation while protecting others using an acrylic cover. They found that colonies exposed to UV at 12 m depth began to show signs of bleaching within 7 days, whereas no changes were observed in exposed colonies at greater depths or any of the protected colonies. However, Brown (1997) has argued that the lack of controls prohibits the conclusion that this effect was solely due to UVR.

The effects of UV on survival of coral larvae were examined by Gleason and Wellington (1995), who exposed planktonic *Agaricia agaricites* larvae spawned from different depths to natural intensities of UV (A and B) radiation. They found that larvae originating from 3 m survived better than those from 24 m, which corresponded with tissue concentrations of UVB-absorbing compounds. These results suggested that sensitivity to high intensities of UVB radiation might affect the survival of *A. agaricites* larvae in shallow reef-waters. UV has also been suggested to play a role in inducing coral bleaching (see Section 6.7.3 below).

Studies on other marine invertebrates

In one of the few studies on the effects of UV on marine invertebrates other than corals, Hovel and Morgan (1999) examined the susceptibility of estuarine crab larvae to UV radiation. UV and photosynthetically active radiation were all rapidly attenuated in the turbid saltmarsh, but UVBR still decreased larval survival of all three species. However, because larvae in this marsh were released during nocturnal ebb tides, most newly hatched larvae were flushed from the marsh before daybreak and therefore avoided extensive UVB exposure. Just how widespread this larval susceptibility is to UVB will have to await further studies.

6.7.3 Examples – effects of global climate change on marine communities

We discuss in more detail below the impacts of global climate change on two important communities: coral reefs and saltmarshes.

Coral bleaching

The phenomenon of extensive coral bleaching was first described by Glynn (1984), after reefs along the Pacific coast of Panama bleached in response to the El Niño-Southern Oscillation event of 1982-83 (see also Glynn 1988), although more localised events had been reported earlier. Only isolated bleachings had been reported previously since monitoring began there in the 1950s (Sebens 1994). There have been a series of global bleaching events since the early 1980s whose frequency, scale and severity are unprecedented in modern times (for reviews see Berkelmans and Oliver 1999; Hoegh-Guldberg 1999). Bleaching events similar to that associated with the 1982-83 El Niño were observed throughout the Caribbean in 1987, 1989 and 1990 (Williams and Bunkley-Williams 1990).

Bleaching occurs when the corals lose the pigments or cells of their symbiotic zooxanthellae (photosynthetic algae), leaving the corals pale to white in appearance. This may occur, for example, when the thermal tolerance of corals or their zooxanthellae is exceeded, resulting in the mass movement of the zooxanthellae from the coral tissues. Although many corals can recover from this, by repopulation from other or remaining zooxanthellae, their loss makes the coral tissue far more sensitive to light damage, with extreme or prolonged stress resulting in death. Several studies now show that even with recovery, sublethal chronic effects on growth and reproduction may be apparent (Hoegh-Guldberg 1999). The vulnerability of corals to bleaching varies: some massive taxa such as *Porites* are more resistant than the branching *Acropora* species, the former having a thicker layer of coral tissue covering the skeleton. In addition, pigmented species, such as some of the pocilloporids, also appear more resistant to bleaching (Wilkinson 1998b; Hoegh-Guldberg 1999). Extensive bleaching occurred during early 2002 on the GBR, especially on inshore reefs[128].

[128] Extensively documented on the GBRMPA web site http://www.gbrmpa.gov.au/corp.site/bleaching.

A wide variety of factors provoke this stress response, including altered seawater temperatures, increased irradiance (including UV), decreased salinity, and bacterial and other infections (Table 6.3). However, it is important to distinguish between factors that cause extensive ("mass") bleaching as opposed to localised events (e.g., over a few hundred metres, in intertidal areas, river mouths etc.), and between primary causes versus compounding stresses. Most large-scale bleaching has mostly been attributed to a rise in sea temperature; in particular when sea-surface temperatures (SST) exceed their summer maximum, or through some combination of temperature and irradiance[129] (Brown 1997). The apparently increased episodes of coral bleaching over the last two decades are well correlated with increased SST (e.g., reviews by Williams and Bunkley-Williams 1990; Glynn 1991; Brown 1997; Hoegh-Guldberg 1999; Reaser et al. 2000; Fitt et al. 2001; Woesik 2001). Decreased SST and reduced salinity generally cause much more localised bleaching. Bacterial infections have been cited as a cause of bleaching, but it is difficult to attribute this unequivocally as a causal agent (Brown 1997).

Table 6.3: A summary of selected papers which have identified bleaching-related responses to various stressors (after Brown 1997).

STRESSOR	FIELD	LABORATORY
Elevated sea water temperature	Glynn (1993) for reviews see Hoegh-Guldberg (1999) and Berkelmans (1999).	Hoegh-Guldberg and Smith (1989); Glynn and D'Croz (1990); Lesser et al. (1990); Iglesias Prieto et al. (1992); Fitt and Warner (1995); Warner et al. (1996); Jones et al. (1998).
Decreased sea water temperature	Coles and Fadlallah (1991); Kobluk and Lysenko (1994)	Muscatine et al. (1991); Gates et al. (1992)
Increased irradiance (including UV)	Fisk and Done (1985); Gleason and Wellington (1993); Brown et al. (1994); Gleason and Wellington (1995)	Hoegh-Guldberg and Smith (1989); Lesser (1989); Lesser and Shick (1989); Lesser et al. (1990)
Combination of elevated temperature and irradiance	Harriott (1985); Brown and Suharsono (1990); Williams and Bunkley-Williams (1990) for review; Glynn (1993) for review; Brown et al. (1995)	Lesser et al. (1990); Glynn et al. (1992)
Reduced salinity	Goreau (1964); Van Woesik et al. (1995); De Vantier et al. (1996)	Fang et al. (1995); Kushmaro et al. (1996)
Bacterial and other infections	Upton and Peters (1986); Kushmaro et al. (1996)	Kushmaro et al. (1996)

[129] Interaction of high temperature and irradiance is likely to be a common feature of bleaching events but is poorly documented because of difficulties of accurate *in situ* monitoring, especially of irradiance (Brown 1997). As highlighted by Glynn (1993), many workers have reported coral bleaching during periods of low wind velocity, calm seas and low turbidity; conditions which favour heating of shallow waters and high irradiance penetration. Other evidence is the fact that colonies often do not bleach evenly, the upper sides tending to bleach first and with the greatest intensity (Goenaga et al. 1988). Given that water temperatures are unlikely to differ between the top and sides of a coral colony, other explanations are required, such as differences in the amount of PAR (photosynthetically active radiation) received by different parts of the coral colony (Hoegh-Guldberg and Jones 1999).

Coral reefs around the world are subject to a range of chronic stresses, including pollution, disease, predation by *Acanthaster*, etc., and it has been suggested that these may lower the corals' resistance to the bleaching process (e.g., Williams and Bunkley-Williams 1990), as well as hampering recovery. In general, most evidence points to increased global temperatures as the primary cause of global bleaching, with irradiance/UV as an important secondary variable (O. Hoegh-Guldberg pers. comm.).

While SST is the most common factor believed to be responsible for extensive bleaching, there has been some controversy in attributing the cause to global warming (Glynn 1993). In part, this has been due to an apparent lack of correlation between bleaching and surface temperature anomalies (Brown 1997). However, there are problems with using satellite data alone to determine SST, the most significant of which are lack of resolution and difficulties in ground-truthing. In addition, satellite data only relate to the top few millimetres of surface waters, which may correspond more closely to solar radiation than bulk sea temperature (Brown 1997). Nonetheless, during the recent 1997-98 El Niño-induced global bleaching event, the US National Oceanic and Atmospheric Administration were able to accurately predict where bleaching would occur using satellite SST data (Baird and Marshall 1998; Hogarth 1999). There is now an on-line service that accurately shows ocean hot spots (Coral Reef Bleaching Indices: - NOAA's National Environmental Satellite, Data, and Information Service (NESDIS), inaugurated 22 February 2000).

Another problem in explaining mass bleaching events has been the lack of continuous long-term data (Glynn 1993). In Phuket, Thailand, long-term continuous environmental monitoring has been in place since 1942 (Brown 1997). At this site, elevated seawater temperatures are thought to be primarily responsible for observed coral bleaching. The highest seawater temperatures on record occurred in 1991 and 1995 and were associated with bleaching events (Brown et al. 1996). The monthly mean SST for Phuket during the last 50 years showed a long-term and significant ($p<0.001$) decadal increase of 0.126ºC. This is consistent with positive trends in the Indian Ocean area and tropical seas as a whole (Bottomley et al. 1990; cited in Brown 1997).

The Great Barrier Reef appeared to be largely immune from mass coral bleaching prior to 1998, although more localised bleaching certainly has occurred at several locations. For instance, recurrent bleaching was reported on the fringing reefs of Magnetic Island during the summers of 1979/80, 1981/82, 1986/87, 1991/92 and 1993/94 (Jones et al. 1997) and again during 1998 and 2002 (Berkelmans and Oliver 1999; and GBRMPA website). Continuous *in situ* water temperature recordings suggest a close correlation between bleaching and periods of average daily seawater temperatures approaching 32ºC. Each of the bleaching events has occurred during periods of unusually high air temperatures and there has been a significant increase in annual summer and winter air temperatures in this area since the middle of last century. Differential coral survival may be explained, at least in part, by differences in local currents, not only providing different temperature regimes but also differences in diffusion boundaries between the coral and the water (Nakamura and Woesik 2001).

Intensive and extensive coral bleaching on the GBR occurred during 1997-98, (although extensive bleaching has also occurred in early 2002- see GBRMPA web site) when it was reported along the length of the GBR, particularly on inshore fringing reefs in the central GBR. Aerial surveys of 654 reefs (approximately 23% of all GBR reefs) showed that 87% of inshore reefs were bleached at least to some extent – 67% had >10% loss of coral cover and 25% had >60% loss of coral cover. This compared to 28% of offshore (mid- and outer-shelf) reefs with at least some bleaching, 14% with >10% loss and none with >60% loss (Berkelmans and Oliver 1999). Ground surveys showed that the aerial survey data underestimated the extent and intensity of bleaching (Berkelmans and Oliver 1999). Around Orpheus Is, where 103 scleractinian species were recorded from transects, colonies from 101 species were found to be bleached, as well as those of three common Alcyonacae genera and a hydrocoral. Acroporid corals were the worst affected; for example, all colonies of *Acropora hyacinthus* and *A. gemmifera* were bleached and 70-80% were dead within five weeks after the initial bleaching reports. Such high mortality of the dominant acroporids suggests this event was more severe than bleaching in 1982 (Baird and Marshall 1998). After the 1998 bleaching there was no coordinated reef-wide follow up to determine the rate of recovery and mortality of bleached coral colonies. With the onset of the 2002 bleaching, GBRMPA has established video transects on 30 affected reefs scattered along the reef and will continue to monitor these over the next few months to determine overall rates of mortality and recovery. On some of these reefs, it appears that between 50-90% of the bleached corals have died (P. Marshall, pers. comm.).

Given the finding that coral bleaching can occur in response to increased sea temperature, global warming has potentially serious consequences for coral reefs worldwide. Presently it is a matter of debate whether this would lead to catastrophe, including, potentially, "the complete loss of coral reefs on a global scale" (Hoegh-Guldberg 1999), or some form of adaptive response by corals. The report released by Hoegh-Guldberg (1999) predicted a grim future for coral reefs, in Australia and worldwide, if global warming continues unchecked, including severe annual bleaching events in most tropical oceans within 30-50 years (Hoegh-Guldberg 1999). Other researchers disagree, arguing that this fails to take into account natural selection or the potential for adaptive evolutionary change. For instance, it has been hypothesised (e.g., Buddemeier and Fautin 1993; Ware et al. 1996) that bleaching is an adaptive mechanism allowing the coral to be repopulated with a different, possibly more stress resistant, type of alga. The fact that some coral types do survive bleaching events suggests that some genotypes could take over if the seas do warm, while others have argued that the distribution of corals and coral reefs may shift to suit new conditions. Given the rapidity of the changes currently occurring it is unlikely, however, that such adaptive mechanisms can occur in time (Glynn 1993; Brown 1997; Berkelmans and Oliver 1999; Hoegh-Guldberg 1999) to prevent a decline in the distribution and extent of coral reefs. Corals clearly can adapt over evolutionary time scales, but such changes are expected to take many hundreds, if not thousands of years (Hoegh-Guldberg 1999) and any change in distribution would be limited to the availability of suitable environments.

Reefs worldwide are increasingly being subjected to a variety of anthropogenic impacts (e.g., eutrophication, increased sedimentation, mining, physical destruction, destructive fishing methods); with about 50-70% of all coral reefs being directly impacted by human activities (Wilkinson 1998; 2000 and references therein). Thus, climate-induced stresses cannot be considered in isolation and the interaction of anthropogenic and natural influences must be a major factor determining not only the mortality and recovery of reefs from bleaching, but also their ability to adapt to future change. While Australia's reefs are considered to be in relatively good condition (Wilkinson and Buddemeier 1994; Wilkinson 1998, 2000), with the exception of some inshore reefs, there is no room for complacency and active management of this increasingly used resource must be maintained.

Bleaching in other invertebrates

Soft corals, including those on the Great Barrier Reef, have also been affected by bleaching with tissue loss and widespread mortality (Fabricius 1999; Michalek-Wagner and Willis 2001b, a). Bleaching may also occur in sessile invertebrates with phototrophic symbionts including sponges (Fromont and Garson 1999) and Giant Clams (Addessi 2001).

Saltmarsh communities

The impacts of global warming on saltmarsh communities have been reviewed by Hutchings (2001). Increasing temperature is predicted to result in a change in community structure in Australia, through an increasing dominance of *Spartina*, and with it, presumably, a change in the structure of the associated animal communities. While the invertebrate fauna of saltmarshes in Australia is poorly documented, it appears to be heavily influenced by the type of saltmarsh plant community present (Hutchings 1994a, 1994b). There is some evidence that bird populations also are influenced by plant species composition, with dense beds of *Spartina* not being preferred habitats (Davis and Moss 1984); this has implications for predation of invertebrates by birds, i.e. both "top down" and "bottom up" impacts from population regulatory processes.

Increasing temperatures are also predicted, in most cases, to be associated with rising sea levels. Under natural conditions, the saltmarsh would usually just extend landwards. However, increasingly this landward drift is prevented by man-made structures such as seawalls or other retaining walls. It thus seems likely that the total area occupied by saltmarshes will decline, although measuring this accurately will be difficult as there is, as yet, no detailed inventory of the distribution and extent of saltmarshes in Australia. Such a decline could have consequences in terms of the net export of organic matter into nearby coastal waters, and perhaps a change in the quality of the detritus. Increasing storm activity may also lead to increased levels of erosion of saltmarshes, with the additional suspended matter being washed out onto nearby inshore seagrass beds.

Mangrove communities will also be affected, with changes in species composition as temperatures rise, and encroachment onto saltmarshes as sea levels rise.

6.7.4 Information, research needs and management

Climate change (and its causes) is potentially one of the most serious of the threatening processes and cannot be dealt with on an industry-by-industry basis but requires multifaceted national and global programs and solutions.

- To ensure the conservation of coastal biodiversity global warming must be slowed as much as possible.
- Steps must also be taken to establish coastal zone policies that allow adaptive responses to rising seas by making way for the shoreward movement of coastal ecosystems as sea level changes.

Research into the likely impacts of rising sea levels and both atmospheric and oceanic warming is needed so that effective forward planning can be undertaken to alleviate, where possible, the impacts.

International and national management programs

Increasing international concern about the implications of climate change and a recognition that it was an international issue resulted in the UN Framework Convention on Climate Change and the Kyoto Protocol. The Australian Greenhouse Office (AGO - http://www.greenhouse.gov.au/ago/) was established as a separate agency within the environment portfolio to provide a whole of government approach to greenhouse matters. The National Greenhouse Strategy (NGS), (http://www.greenhouse.gov.au/policy/) is a policy initiative of the Commonwealth, State and Territory governments and provides the strategic framework for Australia's greenhouse response. The Greenhouse Gas Abatement Program (http://www.greenhouse.gov.au/ggap/) announced in May 1999 will assist Australia in meeting its commitments. A National Greenhouse Gas Inventory (http://www.greenhouse.gov.au/inventory/index.html) is an annual inventory of national greenhouse gas emissions since 1988 as part of commitments under the United Nations Framework Convention on Climate Change. They provide a baseline for monitoring targets. Up to date information on greenhouse gas and global warming issues can be found at http://www.greenhouse.gov.au/ .

6.8 Problems with complex, unknown or debated origins

6.8.1 Diseases and parasites

In a review of this issue, Harvell et al. (1999) discussed climate links and anthropogenic factors and emerging marine diseases. They noted that in the past few decades there has been a worldwide increase in the reports of diseases affecting marine organisms.

Epidemiologists recognise an interrelationship between:

- Host (reduced health, increased susceptibilities);
- Disease (new virulent strains);
- Environment (modifies existing host-pathogen interactions).

Thus, while disease is a natural phenomenon, its incidence can be facilitated by human perturbation, for instance through:

- Environmental changes that reduce the health of individuals or populations making them more susceptible to disease and/or parasites;
- Introduction by human activity of diseases or parasites not previously found in an area (e.g., through deliberate or accidental introduction of hosts or vectors);
- Human activities that result in the emergence of new, more virulent strains of diseases (eg. the use of antibiotics in aquaculture).

Diseases in the marine environment

Harvell et al. (1999) cited as examples coralline algae in the Indo-Pacific, marine mammals in the North Atlantic, and ecologically and economically important species from temperate oceans, such as seagrasses, oysters and sea urchins, all of which have been affected by large-scale epidemics. Although there is an increased frequency of such accounts, whether these occurrences are indeed "new" or are simply artefacts of improved detection, requires further evaluation. As with most other marine issues, the paucity of baseline and epidemiological information on normal disease levels in the ocean makes it impossible to properly assess the novelty of recent disease outbreaks. Similarly, the relative importance of increased pathogen transmission versus decreased host resistance in facilitating the outbreaks cannot be determined.

Diseases affecting benthic marine species such as corals and seagrasses will have disproportionate impacts by altering habitat and ecosystem function. In spite of the impact, little progress has been made in identifying the causative agents for marine diseases or in applying standard epidemiological methods to assess impact or mode of transmission.

Corals

Over the past couple of decades, various coral diseases have been recorded in various parts of the world. Coral diseases were first recognised in the Caribbean, but have now been recorded throughout the Indo-Pacific (Antonius 1995), including, more recently, the Great Barrier Reef (Wachenfeld et al. 1998; Baird 2000). For example, a newly described coral disease on Indo-Pacific reefs is caused by *Halofolliculina corallasia*, a coral-killing ciliate (Antonius 1999a). The disease damages the skeleton and it is found on a wide variety of massive and branching corals and is somewhat similar to Black Band Disease. This disease was found on reefs of the Sinai (Red Sea), Mauritius (Indian Ocean) and Lizard Island, GBR (Baird 2000). Hutchins (1999) reported catastrophic mortality of *Pocillopora* coral at Rottnest Island, Western Australia that could have been caused by disease as it quickly spread through a large area. Previously there had been only one report from Western Australia (Simpson et al. 1993) of coral 'disturbance' of any kind, and no previous records of disease-related mortality.

Coral diseases are not necessarily caused by microbes. *Metapeyssonnelia corallepida*, is a recently described coral-killing red alga on Caribbean reefs (Antonius 1999b) and is one of the causes of the recently described syndromes of epizoism on reef corals particularly on reef crest areas (Antonius and Ballesteros 1998).

Fossil evidence also seems to suggest that some coral 'diseases' are novel. For example, the rapid replacement of the coral *Acropora cervicornis* with *Agaricia* in Belize and with *Porites* in the Bahamas, taken as a 'signature' of epidemics, was found to be absent from geologic cores representing several thousand years of reef development (Harvell et al. 1999).

Dust from African deserts may be responsible for spreading disease across the world's coral reefs. This is facilitated by prolonged drought in the Sahel region since the mid '70s has increased by fivefold the amount of atmospheric dust containing bacteria, viruses and fungi that can kill coral (Pearce 1999). Outbreaks of diseases such as white band and black band disease and the bacterial infection known as "coral plague" have coincided with years when the dust load was highest, the Caribbean being particularly badly affected. The strongest evidence for this hypothesis is the spread of a soil fungus, *Aspergillus*, in the Caribbean. It first appeared in 1983, an exceptionally dusty year, and since then has killed more than 90% of the Caribbean's sea fans (gorgonian soft corals). Iron in the dust can also trigger algal growth by stimulating nitrogen fixation. Fabricius (pers. comm.) claims that while some workers link this mass mortality to land run off, others suggest it is caused by the fungus *Aspergillus* (Smith et al. 1996a; Nagelkerken et al. 1997a; Nagelkerken et al. 1997b; Geiser et al. 1998; Harvell et al. 1999).

Other marine invertebrates

Other marine invertebrates are obviously affected by diseases but on the whole these are very poorly documented except for economically significant taxa such as oysters and prawns. The Crown-of-Thorns starfish is affected by a poorly understood disease (see Section 6.8.2) and a disease wiped out large populations of diademnid urchins in the Caribbean in the 1970s-1980s (Lessios 1988). Trematode infection may have played a role in the mass mortality of two common soft-bottom invertebrates (a gastropod mollusc and an amphipod) on an intertidal mudflat in Denmark (Jensen and Mouritsen 1992).

Diseases attributable at least partly to human activity or to stresses directly resulting from anthropogenic environmental degradation have been reported in several taxa. For example, Aguado and Bashirullah (1996) recorded the incidence of shell diseases in wild penaeid shrimps in eastern Venezuela. Shell diseases – resulting in degradation of the exoskeleton – are commonly caused by bacterial genera such as *Vibrio*, *Aeromonas*, and *Pseudomonas*, but may also be associated with certain environmental parameters and be an indicator of a stressful environment (Aguado and Bashirullah 1996). Chu and Hale (1994) examined the relationship between pollution and susceptibility to infectious disease in the eastern oyster, *Crassostrea virginica*. The disease studied is caused by a protozoan parasite and results in significant oyster mortalities in the mid-Atlantic region of the United States. Exposure to pollutants was found to enhance pre-existing infections and increase the oysters' susceptibility to experimentally induced infection in a dose-dependent manner (Chu and Hale 1994). In Norway, Bustnes and Galaktionov (1999) found that fishing industry complexes and fish farms appeared to be associated with a prevalence of trematode parasites in intertidal gastropods, due to the tendency of gulls (the final hosts) to concentrate in these areas to feed on offal.

6.8.2 Population outbreaks of invertebrate predators or grazers

Occasional outbreaks of species occur, some apparently "naturally", others probably as a result, directly or indirectly, of human disturbance. Many such population explosions may have little impact and largely go unnoticed. Others can have quite devastating impacts. Outbreak species typically involve invertebrates, but others that pose a significance threat in marine habitats include some dinoflagellates. The most publicised invertebrate involved in sporadic outbreaks has been the destruction of living coral by the Crown-of-Thorns starfish on the GBR and other parts of the Indo-west Pacific, although the whelk *Drupella* in WA, another coral eater, is also receiving some attention. Both of these are described in more detail below. Although there has been much debate over the causes of these outbreaks, with anthropogenic factors ranging from overfishing of predator species to increased pollution being blamed, some are probably, at least in part, natural boom-and-bust cycles. Problems with outbreak species arise when the prey has limited opportunities for recovery, recruitment, recolonisation, or dispersal compared to undisturbed ecosystems. That is, pressure from the effects of these species (e.g., grazing, predation etc.) is compounded by other, often anthropogenic, pressures on the species or ecosystem.

Acanthaster *– the Crown-of-Thorns starfish*

There have been outbreaks of the coral eating Crown-of-Thorns starfish *(Acanthaster planci*) on Indo-Pacific coral reefs since the mid-1960s (Birkeland and Lucas 1990). It has devastated reefs in many parts of the western Pacific and anthropogenic causes have been implicated including overfishing of predators (fish or triton shells) (Lassig and Engelhardt 1994) and enhanced survival of the larval stage (Birkeland and Lucas 1990; Brodie 1992) due to nutrient induced phytoplankton blooms.

Acanthaster planci was first collected in Australian waters during the 1913 American Expedition to Torres Strait, and was reported by H. L. Clark in 1921 (Rowe and Gates 1995). Local interest in this starfish was galvanised in the mid sixties when concerns were raised about its explosive increase in numbers and destructive effects on the corals of the GBR and since then the causes, effects and solutions have become a highly politicised environmental issue with continuing monitoring and fine-scale survey (Engelhardt et al. 2000). There is an extensive literature on this subject, with important reviews provided by Birkeland and Lucas (1990), a Special Issue of *Coral Reefs* (9(3) 1990), Johnson (1992) and Sapp (1999).

The Crown-of-Thorns can have a significant impact on coral reefs. A status report of long-term monitoring of the GBR show that changes in their abundance reflect effects of cyclones and activity of the Crown-of-Thorns (Sweatman et al. 1998; Ninio et al. 2000). There have been repeated outbreaks of Crown-of-Thorns observed on the GBR since the mid 1960s and several authors have conjectured that these may cause long-term degradation of reef community structure. Seymour and Bradbury (1999) analysed data from the AIMS annual synoptic surveys (1985 – 1996) and concluded that the average reef recovery time is lengthening over the period for which data are available. They interpret this as evidence that it is harder for reefs to recover from outbreaks in later years than in earlier years, other things being equal, indicating that key features of reef community structure have been damaged over time. A study by Lourey et al. (2000) showed highly variable recovery rates with estimated recovery times ranging from 5 to >1000 years. A study on the outer GBR (Fabricius 1996) on *Acanthaster*-impacted and non-impacted reefs showed that neither abundances nor cover of soft corals, sponges, tunicates, zoanthids, and macro-algae were increased in areas where stony coral cover had been low for 5 to 20 years. Unlike turbid, near-shore reefs, soft corals did not invade space rapidly on these outer reefs following stony coral mortality and substratum may remain vacant until fast-recruiting stony coral taxa colonise the patches. Fabricius (1996) concludes that with no additional stress these shallow outer reefs are resilient enough to return to their pre-outbreak state.

A number of authors have questioned whether the devastating outbreaks of *Acanthaster* are novel contemporary events or part of an ecological pattern that has persisted for millennia (Moran et al. 1986; Moran and Bradbury 1989; Walbran et al. 1989; Cameron et al. 1991; Keesing et al. 1992). Cameron et al. (1991) compared size and damage frequency of massive scleractinian corals on affected and unaffected reefs in the central GBR. There were fewer and mostly smaller massive coral colonies on the outbreak reefs. These authors argue that because most massive corals are slow-growing, long-lived and have lower rates of recruitment than other corals, continuing starfish re-infestation coincident with re-establishment of a coral cover by the faster growing, more opportunistic corals will not allow sufficient time for recovery of the massive coral assemblages in the intervals between outbreaks. They conclude that the recent devastating outbreaks appear to be abnormal perturbations coincident with large-scale human activities on the GBR, rather than integral features of reef ecology. Repeated outbreaks of this intensity could not have occurred in the century prior to the 1960s because of the high number and large size of the massive corals observed on the unaffected reefs.

The evidence for outbreaks of the Crown-of-Thorns starfish having occurred in the geological past was reviewed by Moran et al. (1986). He reassessed the data presented by Frankel (1977; 1978) as evidence of past outbreaks, and used data from extensive starfish surveys conducted prior to his research. He showed that the remains of *A. planci* in recent sediments occurred independently of whether or not the reef from which the sample was collected had experienced a recent outbreak. Thus, it was not possible to infer from Frankel's data the occurrence of past outbreaks from similar material in much older sediments, although *A. planci* has clearly existed within the Great Barrier Reef for at least several millennia. Walbran et al. (1989) examined sediment cores to establish a link between the occurrence of outbreaks and the number of skeletal elements recovered. They argued that substantial populations of *A. planci* have had a long history on GBR reefs (at least 8000 years), probably influencing the morphology and species richness of the GBR during this period, and that "past patterns are likely to have been similar to those presently observed". They also suggested that *A. planci* predation may have been a factor in suppressing coral growth 9000 – 7000 years ago when sea level was rising about 9 mm per year and the capacity of framework growth to keep pace was under stress. These conclusions were disputed by Keesing et al. (1992), who disagreed with the ecological basis for some of Walbran et al's. assumptions. For instance, they disputed the idea that reefs that had suffered recent *A. planci* outbreaks could be discriminated from those that have not by using the abundance of starfish skeletal remains in recent sediments. They argued that Walbran et al. (1989) had insufficient data to infer the outbreak history of *A. planci*, or to discount the possibility that recent outbreaks are unprecedented.

Evidence that outbreaks may not necessarily be related to anthropogenic factors comes from occurrences in isolated areas well away from direct human influence. For example, in 1987 considerable numbers of adult Crown-of-Thorns were observed on Elizabeth and Middleton Reefs, far from human activities (Hutchings 1992b). Genetic studies showed that these individuals were more closely related to the population on the southern GBR than on nearby southern reefs of the GBR (Benzie 1999).

Ayukai et al. (1997) argue that food availability usually limits the growth and development of Crown-of-Thorns starfish larvae in the GBR but that heavy rainfall on land transports a large amount of nutrients into the sea and often triggers phytoplankton blooms. Such an event may increase the available food for larval Crown-of-Thorns starfish, leading to the establishment of an outbreak population (Birkeland 1982). Birkeland's hypothesis has been classified as a 'natural causes' (as opposed to a 'human causes') hypothesis (Ayukai et al. 1997), but is pertinent to the concern over the effect of eutrophication on Crown-of-Thorns starfish population dynamics. A critical assumption in the terrestrial runoff hypothesis is that larval food usually is limited. Ayukai et al. (1997) constructed a carbon budget model for the larvae which predicted that the ambient concentration of phytoplankton and dissolved free amino acids is usually too low for them to meet even half of their basic energy requirements. Semi-natural rearing experiments suggest that food limitation in larvae can commonly occur, but its extent is not as severe as the model predicts with larvae growing in similar food levels to those observed in the study area (Ayukai et al. 1997).

Predator removal hypotheses also received some recognition on the GBR since the Giant Triton (*Charonia tritonis*) is known to prey on adult Crown-of-Thorns. Populations of *C. tritonis* are thought to have declined from heavy collecting (Endean 1977). However, numbers of this species were probably never very high and we consider it unlikely that it was ever a significant predator. Fishes that prey on juvenile Crown-of-Thorns starfish may reduce their densities and fishing these predators of juvenile starfish may lead to periods of increased starfish recruitment, possibly leading to outbreaks of adults in subsequent years (Ormond et al. 1991). It has also been observed that trapezid crabs, that form an obligate association with corals, will defend their coral hosts (Pratchett et al. 2000; Pratchett 2001). It is likely that no single factor is responsible, with outbreaks probably due to several interacting factors (e.g., better larval food supply; lower predation), that leads to a synergistic effect on populations.

Outbreaks of *Acanthaster planci* often end with rapid population declines throughout the Indo-Pacific. In Fiji, *A. planci* declines have been attributed to disease (a sporozoan pathogen - Zann et al. 1990) and disease has been implicated in the mass-mortalities of numerous other echinoderms. In January 1999, at Lizard Island, GBR, a single diseased Crown-of-Thorns starfish was found to infect healthy nearby starfish. The symptoms are similar to those reported by Sutton et al. (1988), who implicated bacteria in infections of captive Crown-of-Thorn starfish. Research is continuing to determine the pathogen of this disease. There have been no previous reports of disease among *A. planci* populations on other reefs throughout the Indo-Pacific (Pratchett 1999).

Coral eating whelks* - Drupella *spp.

Drupella species are known to be agents of large-scale disturbance to coral reefs, particularly in Western Australia and Japan, where population outbreaks have drastically reduced coral cover (Turner 1994; Cumming 1999). These snails also occur in other reefs, including the GBR. *Drupella* are perceived as a serious destructive agent to live corals, like the Crown-of-Thorns starfish, and ecological research on *Drupella* has focused on these high-density populations (Cumming 1999).

Saueracker (1997) described how the population of *Drupella* at Ningaloo reef, NW Australia, increased from 100-200 / km^2 in the 1970s to 1-2 million / km^2, and stated that it had destroyed 90% of corals in parts of the northern reef. *Drupella* has a thick shell that only large fishes could crack, so overfishing (predator removal hypothesis) may have allowed their population to increase. The fish population of Ningaloo has declined under fishing pressure. Since Ningaloo is isolated from human development, it is unlikely that extraneous human impact is responsible for the abnormal numbers.

Drupella spp. prefer *Acropora*, and some other corals seem to be unaffected or even benefit from reduced competition (Saueracker 1997). The pelagic larvae of *Drupella* settle on digitate corals such as *Acropora* and, when they are larger with a thick shell, graze openly on staghorn and plate corals. *Drupella* may actually assist in maintaining the diversity on some coral reefs because *Acropora* grows faster than other corals and can dominate the reef through shading (Saueracker 1997).

Cumming (1999) argues that even at lower densities, such corallivores still potentially affect the dynamics of whole coral reef communities because their prey, reef-building corals, provide the main structural framework of the reef. While this is in a strict sense true, many other organisms feed on living coral and such activity is part of the natural reef ecology. Cumming (1999) rightly argues that research is needed to identify the variation in *Drupella* density to distinguish normal from outbreak populations, and to quantify the impact of *Drupella* on coral reefs.

In Caribbean reefs another corallivorous snail, *Coralliophila*, has some impacts on *Acropora* (Miller 2001). Several members of this genus occur in Australian waters but do not appear to be a problem.

Other invertebrate outbreaks

Outbreaks of several species of sea urchins have been recorded. The formation of grazing fronts is well documented among temperate sea urchins of the genus *Strongylocentrotus* (e.g., Watanabe and Harrold 1991; Hagen 1995). Such fronts have also been observed in various other benthic marine invertebrates including Queen Conch snails, *Stombus gigas*, in the Central Bahamas (Stoner 1989; Stoner and Lally 1994), the Caribbean starfish *Oreaster reticulatus* (Scheibling 1985), and another starfish, *Asterias ruber* (Dare 1982). The destruction of Florida Bay seagrasses by a grazing front of the sea urchin *Lytechinus variegatus*, was described by Maciá and Lirman (1999). This aggregation, several kilometres in length and with densities of up to 600 / m^2, was first detected in August 1997. Normal densities are usually <10 / m^2 Seagrass cover was reduced from 100% to <5% in sample plots and eventually completely eliminated from the affected areas.

Jellyfish and ctenophores may cause economic problems by forming dense schools (Shushkina et al 1990; Mills 1995). A species of the ctenophore *Bolinopsis* occurs in huge numbers in the upper Spencer Gulf, SA, a nursery area for many commercially important species. Large blooms of jellyfish and ctenophores can cause potentially devastating effects on larvae or food items of other species (Mills 1995), or on human activities. They have voracious appetites and many are generalist feeders, able to eat any live or dead organic material in the water column (Mills 1995; L. Gershwin, pers. comm. 1999). Blooms caused significant mortality in Tasmanian salmon farms in January 1999 and blocked power plants in the Northern Kimberley during the latter part of 1999 (L. Gershwin, pers. comm., 1999). Blooms of stinging jellyfish can adversely impact on human activities, notably tourism.

6.8.3 Red tides (dinoflagellate blooms)

Worldwide there has been an increase in the frequency and extent of blooms of harmful marine microalgae and heterotrophic dinoflagellates (Burkholder 1998). The reasons for these blooms are poorly understood. Many taxa of “red tide” dinoflagellates appear to increase under suitable environmental conditions and independently of any human influences (Burkholder 1998). These environmental conditions may include strong salinity and temperature stratification in the upper layer and a bloom in the phytoplankton or diatom food source (e.g., Cloern et al. 1994; Crawford et al. 1997). On the other hand, some newly discovered toxic or otherwise harmful taxa have been correlated with anthropogenic factors such as eutrophication in poorly flushed areas such as estuaries and coastal waters (Burkholder 1998). Outbreaks of certain warm-optimal species have coincided with El Niño events, suggesting that warming trends in global climate change may stimulate their growth and extend or shift their range (Burkholder 1998). Another important human influence is transport. In the late 1980s, various exotic dinoflagellate species, both toxic and non-toxic, were discovered in ballast water, and subsequently in Australian harbour waters and sediments (Hallegraeff and Sumner 1986; Hallegraeff et al. 1990; Hallegraeff and Bolch 1991). Of particular concern was the discovery of a group of dinoflagellates that can produce paralytic shellfish poisoning (PSP), as these species have the ability to produce resistant resting spores that can be easily transported in ballast water sediment (Hallegraeff et al. 1988). PSP affects people who consume contaminated seafood. For instance, Hallegraeff and Bolch (1991) reported that among 80 cargo vessels that were sampled by Quarantine officers as they entered Australian ports (1987-1989), 40% contained viable dinoflagellate cysts and 6% carried the cysts of the toxic dinoflagellates *Alexandrium catenella* and *A. tamarense* (up to an estimated 300 million cysts per ship).

Some other dinoflagellates secrete toxins that can result in massive fish and invertebrate kills. In addition to direct toxic effects, these blooms – like those of other microplanktonic species – can also harm other marine species through local depletion of oxygen resulting in zones of hypoxia or anoxia. In addition to acute or lethal effects, accumulating evidence indicates that there may be substantial sublethal and chronic impacts to both marine species and human health from these organisms, such as long-term behavioural alteration, increased susceptibility to cancers and other diseases, depressed appetite, and impaired reproduction (Burkholder 1998). For some harmful species, there may also be significant indirect impacts resulting from habitat loss or disruption of the microbial food web balance (Burkholder 1998).

6.8.4 Information, research needs and management

The causes of population outbreaks of marine species are poorly understood and urgently require study.

- Studies on the basic biology, ecology and distribution of at least keystone taxa would greatly improve the knowledge base so that at least basic information was available when required. For example, although much research has been carried out on *Acanthaster,* the reasons for population outbreaks in this taxon are complex and/or variable and remain poorly understood.

Dinoflagellate blooms pose a major threat to the aquaculture industry due to their risk to human health. They can be very effectively transported in ballast water as they encyst and can survive long periods in dark, hypoxic waters.

- We know relatively little about the species of dinoflagellates that occur naturally in Australian waters, let alone the potential dangers of these taxa if they bloom. The early recognition of problematic or exotic taxa is also rendered difficult because there are very few people undertaking research on these organisms.

There is very little information available regarding diseases of marine invertebrates, other than for a small number of commercial species. Monitoring is almost solely limited to a few commercially important taxa. There is scant information on what diseases are present, let alone how to control them.

- This issue raises important questions regarding the translocation of aquaculture stock.
- There is a considerable risk that the introduction of exotic species (either as pests or for aquaculture) will bring with them undetected diseases that could affect native species.

Managing disease and other outbreaks is a very difficult, if not impossible task, given our very limited to non-existent knowledge of what is involved in all but a very small number of cases.

- Guidelines on reporting and responses should be established for significant events.

6.9 Synergistic / cascading effects of multiple threatening processes

The effects of individual stresses or processes may be studied in isolation in the laboratory or staged manipulative field experiments, but it is often difficult to generalise the results of these experiments to the natural environment, where several stresses may be acting simultaneously.

It is also possible for two or more processes to have opposing (when two processes largely negate each other) and neutral (when two processes do not interact) effects or do not interact. In contrast, synergism is when the effect of two or more processes combined is greater than the sum of the individual effect.

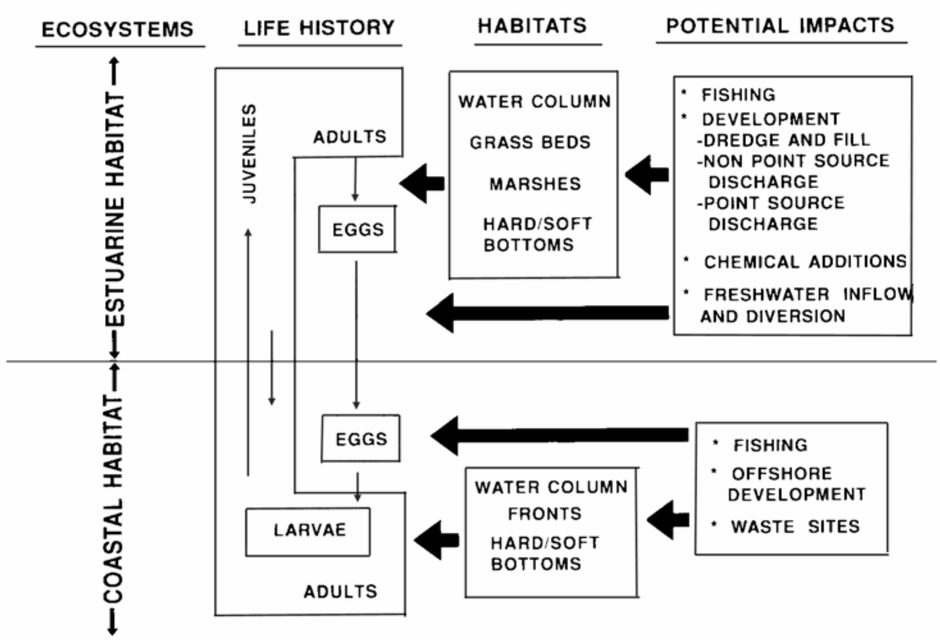
The potential for synergistic effects, involving two or more different threatening processes, cannot be over emphasised and is probably a common situation in natural and disturbed systems. Synergism occurs when two or more processes work together simultaneously to great effect, unlike indirect or secondary impacts in which processes occur sequentially. Two examples are given below.

Macro-algae and corals
On coral reefs, the balance between macro-algae and coral is a delicate one. Increased algal biomass can indicate problems with the reef because coral reef degradation often involves replacement of corals by macro-algae, this resulting in declines in economic and aesthetic values of the reef (e.g.,Szmant 2001). Such changes may result from impacts such as eutrophication, sedimentation from terrestrial run off (McCook 2001) or over-fishing, all of which are more prevalent on inshore reefs (McCook et al. 1997). McCook et al. (1997) surveyed the macroalgal composition of 77 GBR reefs and discussed distribution patterns and their correlation with biotic and environmental variables. They demonstrated that the floras of both inshore and offshore reefs normally include some fleshy macro-algae and that experimental work showed that the abundance of *Sargassum* on inshore reef flats is not due to direct enhancement by nutrients or sediments, but to isolation from herbivorous fish and to availability of substrate. They concluded that macro-algae appear to often be consequences, rather than the direct causes, of coral mortality.

Blue Crab
Engel and Thayer (1998) describe the effects of habitat destruction on the Blue Crab, *Callinectes sapidus*, an estuarine and coastal species in the USA that lives in a wide range of environmental conditions. Like other crustaceans, this species has a complex life cycle and each stage is vulnerable in different ways to chemical and physical changes to their habitat and is dependent on different parts of the food chain. Effects on one or more of these parts of the life cycle can include human induced impacts such as toxic chemicals

and pesticides, nutrient loading, alterations of freshwater inflow, and physical destruction of estuarine and coastal habitat (see Figure 6.2).

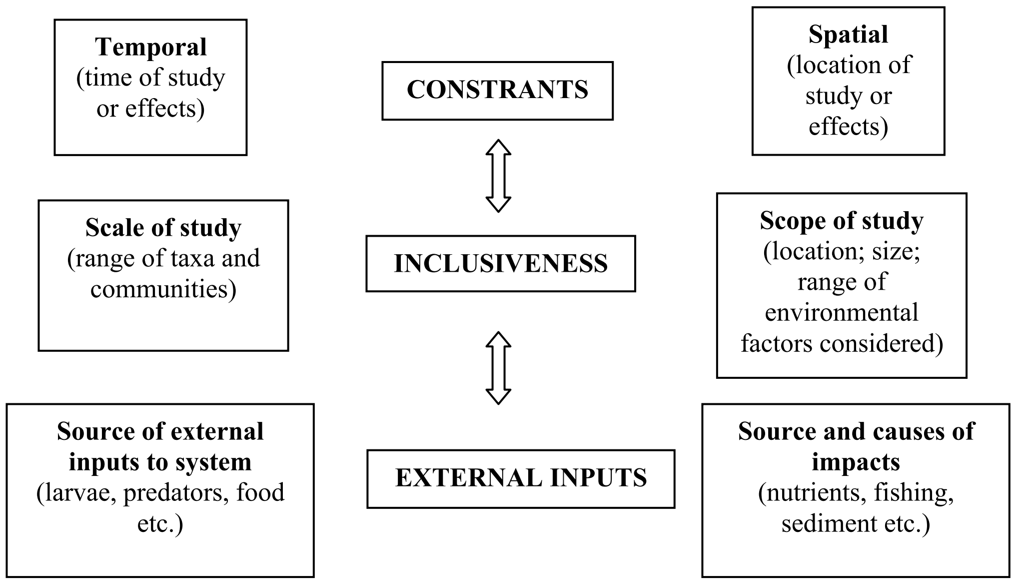
Figure 6.2: A schematic diagram depicting the interactions between the life history of the Blue Crab, estuarine and coastal habitats, and potential impacts on both the habitats and the crabs themselves (from Engel and Thayer 1998)



6.9.1 Information, research needs and management

Recognition of synergistic effects is critical to the identification of adequate research questions and the adoption of effective management options. While the state of knowledge regarding a few marine systems is reaching a sufficient state to identify some of the processes involved in particular geographic areas, it is unlikely that the specifics of these findings can be transferred to other areas even if the same or very similar ecosystems are involved. Management options are further frustrated by the wide range of agencies responsible for the observed impacts on a particular environment.

Figure 6.3: The main components of a study examining synergistic effects to illustrate some of the complexity of such studies. While an increase in all of these components (time, area, scale and scope) will theoretically improve the understanding of a particular situation the amount of effort required to achieve a result increases exponentially.



ACTIVITIES (AGENTS) RESPONSIBLE FOR THREATENING PROCESSES

We have identified eight activities (agents), five mainly marine-based industries and three other land-based activities, largely responsible for the main threatening processes of significance in the marine environment, and thus having direct consequences for marine invertebrate conservation. These are:

- Fisheries and other forms of biotic exploitation;
- Aquaculture;
- Shipping/transport;
- Petroleum, gas or mineral exploration and production;
- Recreational use and tourism;
- Waste disposal;
- Coastal development and modification;
- Land use in catchments.

These industries and activities (summarised in Table 6.1) differ in the extent to which they affect marine habitats and invertebrates. It is also often difficult to differentiate and distinguish between the effects of particular threatening processes (Yen and Butcher 1997) because several may be involved.

6.10 Fisheries and other forms of biotic exploitation

"The uncertainty inherent in the scientific method has made scientific data extremely vulnerable in the face of the economic harm that has been predicted by the region's commercial fishing industry in response to proposed govt regulation". The image of the fisherman "as one of contemporary society's last rugged individualists….. has resulted in the industry's opinion…. being viewed as more credible than scientific data" (Brailovskaya 1998).

World fishery landings increased through the 1980s, but have now levelled at about 90 million tonnes per year, apparently the maximum sustainable yield. Thus, catches have not increased with population (and therefore demand), and prices have risen dramatically relative to other meat. Also, about 2/3 of the important commercially fished stocks are fully or over-exploited and many have been depleted, some to the point of economic extinction. Aquaculture now contributes more than 20% to global aquatic food production. World fishing fleets are still far too large, and many governments are (often covertly) subsidising them. The FOA calculates that US $124 billion is being spent per year to catch US $70 billion worth of fish (Tickell 1997).

Australia's ocean waters are renowned for their low nutrient status. Although we have the third largest EEZ in the world, our total fisheries catch ranks about 50th. Australia lacks upwellings and discharges from large rivers resulting in a lack of nutrients to drive productivity. This, combined with poor results from exploratory fishing and research surveys, reinforces the pessimism about additional significant sustainable fisheries from the deep-sea (Newton 1999).

Earlier (Section 6.3), we dealt with over-exploitation of marine invertebrates as a threatening process. This section deals with all fisheries (finfish as well as invertebrates) and the various effects they can have, directly or indirectly, on populations, communities or ecosystems.

Issues with declining fisheries and impacts from fishing

Pauly and Christensen (1995) calculated the mean annual world fisheries catches for 1988-1991 to be 94.3 million tonnes and split this into 39 species groups, which were assigned to fractional trophic levels ranging from 1.0 (edible algae) to 4.2 (tunas). The primary production required to sustain each group was computed based on mean energy transfer efficiency between trophic levels of 10%. The primary production required to sustain the reported catches, plus 27 million t of discarded bycatch, amounted to 8% of global aquatic primary production. Ecosystem-specific estimates ranged from 1.8% for the open ocean to 8.3% for coastal and coral reef systems to 24-35% for shelf and upwelling systems. Any relatively low values were due to high productivity, large catches of low trophic level species (seaweeds; bivalves; other invertebrates) and overfishing, leaving insufficient biomass to use the available production.

Jackson et al. (Jackson et al. 2001) have suggested that ecological extinction caused by overfishing precedes all other pervasive human disturbance to coastal ecosystems, including pollution, degradation of water quality and anthropogenic climate change. Historical abundances of large consumer species have changed dramatically over time. Using palaeoecological, archaeological and historical data, they show that time lags of decades to centuries occurred between the onset of overfishing and consequent changes in ecological communities. This was because unfished species of similar trophic levels assumed the ecological roles of overfished species until they too were overfished or died of epidemic diseases. They suggest that retrospective data not only help to clarify underlying causes of ecological change, but they also demonstrate achievable goals for restoration and management of coastal ecosystems. They further suggest, based on their data, that current gloomy estimates of overfished fish stocks are almost certainly far too low. The shifting baseline syndrome is thus more insidious and ecologically widespread than is commonly realized.

Several writers (e.g., Roberts 1997; Pauly et al. 1998) have criticised marine fisheries management and made the observation that, on a global scale, human exploitation proceeds more or less involuntarily down the trophic pyramid, and the more sought-after species are being replaced by less valuable, smaller ones that grow more quickly. In most fisheries in the world, the trends are in the wrong direction: decreasing catch per unit effort despite improved technology, reduced fish abundance, average size and reproductive output, loss of genetic variation, replacement of high-value species by ‘trash’ fish, increased bycatch mortality, recruitment failures and habitat degradation (Roberts 1997). From the invertebrate point of view, species targeted by the fishing industry are not the only ones affected, as many non-commercial small fishes and invertebrates are being captured as bycatch (see Section 6.10.4).

Blaber (2000) provides an extensive overview of fisheries in the tropics, including Australia, and discusses the impact that fishing has on estuarine habitats, and associated invertebrate communities, and how the impacts relate to various anthropogenic inputs. Christensen (2000) discusses indicators for assessing the impact of fisheries on ecosystems.

6.10.1 Effects of fishing on non-target species, communities and ecosystems

Predators at the top of the food chain are particularly at risk of overexploitation (Section 6.3.1) but unexploited species also suffer from fishing activities - either directly (as bycatch) or indirectly through habitat damage (Section 6.10.2) or the removal of key species.

Bycatch of non-target species

Bycatch is the term used for the non-target species captured in fishing operations and typically discarded. Pauly and Christensen (1995) calculated that some 25-35% of the primary productivity of continental shelves (which yield 95% of global catches) is required to sustain reported catches plus the discarded bycatch. The amount of bycatch, and how it is treated, varies between the kinds of operation and the gear used. Prawn trawl fisheries generate 6-10 kg of bycatch for every kg of prawns caught (Poiner et al. 1998); thus, each prawn trawler in northern Australia catches and discards 300-500 kg of bycatch per night (see review by Dayton et al. 1995). Andrew and Pepperell (1992) describe the effects of shrimp bycatch on populations of the non-target organisms caught and the effects that the discards have in increased levels of scavenging and changes in community structure. Hill and Wassenberg (1990) discuss the fate of discards from prawn trawlers in Torres Strait and note that the fishing operations make large quantities of benthic material available at the surface for scavengers. Liggins et al. (1996) report on bycatch from prawn trawling in Botany Bay and Port Jackson, although they mainly focus on finfish. Probert et al. (1997) describe the benthic invertebrate bycatch from a deep-water (662-1524 m) trawl fishery for Orange Roughy (*Hoplostethus atlantina*) on the Chatham Rise, New Zealand. Despite the large mesh size used, 82% of the tows included large benthic invertebrates comprising 96 species (Ophiuroidea, 12 spp., Natantia, 11 spp., Asteroidea, 11 spp., Gorgonacea, 11 spp., Holothuroidea, 7 spp., and Porifera, 6 spp.). The composition of the bycatch from different topographies differed significantly.

While the bycatch of trawling operations have generated most attention, pelagic fisheries can also have impacts on non-target taxa. While the impacts on species such as dolphins, seabirds and sharks have generated the most publicity, some larger pelagic invertebrates such as cephalopods could also be affected. Owing to the generally great reproductive potential and dispersive ability of this latter group of species, this is unlikely to pose a serious threat, but the issue requires further research to clarify impacts.

Effects on habitats

Fishing can alter habitats, usually because of direct damage from the equipment used (especially by trawling and dredging - see Section 6.10.2). Some types of hand collecting for bait (e.g., turning over rocks in intertidal zone; use of yabby pumps on soft shores or in seagrasses) or damaging coral habitat to collect fish for live fish trade, can also have serious localised impacts. Indirect effects on habitats result from fishing activities such as dumping of rubbish, oil spills etc.

Effects on communities and ecosystems

Exploitation impacts the communities to which targeted species belong, through alteration of competitive interactions and community and trophic structure. However, the broader impacts of fishing activities that may cause profound changes, even at the ecosystem level, are less well understood and difficult to study (Frid et al. 1999; Linnane et al. 2000). Effects at this level may operate through the removal of larger individuals in both fish and invertebrate communities, altering the population structure and reducing the abundances of fish and invertebrate predators and herbivores. The ecosystem shifts that result from exploitation may be reversible or not within reasonable time scales (Jennings and Polunin 1996). These changes to ecosytems and communities occur because:

- Key predator or herbivore species are targeted;
- Key prey species removed;
- Fishing methods cause disturbance, alteration or destruction of communities;
- Fishing damages or destroys metapopulations that act as sources for repopulating fished areas, even if they are not fished directly (Thrush et al. 1995).

All of these changes in communities are probably reflected in knock-on effects within trophic cascades (Frid et al. 1999).

The removal of predators by commercial fisheries could also have a significant impact on invertebrate communities. Similarly, the removal of large quantities of invertebrate prey from marine systems can potentially have an impact on the other predators that utilise these resources. This is illustrated by the concern generated over the developing Antarctic krill fishery, which represents a potential threat to the food source of vertebrate predators such as marine mammals and seabirds. An ecosystem-based approach to fisheries management may be required, where food webs are taken into account, including the requirements of taxa that share the resource being targeted, as has already been developed for Antarctic fisheries (see Sections 2.2.22 and 2.3.5).

The large body of work on ecological interactions on rocky shores, and the effects of human collecting, have clearly demonstrated that the removal of key species such as grazing limpets or predatory whelks can lead to phase shifts in community composition (see Section 6.10.3).

Jennings and Polunin (1996) discussed the complex impacts of fishing on tropical reef ecosystems and note that existing fisheries management strategies, which focus primarily on target fish populations, may not be appropriate when fishing initiates shifts in the reef ecosystem. Such shifts may not be reversible, and can impair the processes that guarantee future fish production. For example, sea urchins and fishes are the dominant herbivores in reef ecosystems and their interrelationships appear to be readily affected by fishing. Urchin eating fishes are closely reef-associated and are frequently targets or bycatch of fisheries. The persistence of herbivorous fishes on reefs appears to depend on the presence of sea-urchin predators that maintain sea urchin populations at a level where their low gross production makes them inefficient competitors with herbivorous fishes (in French Polynesia and Levitan 1992 in the Caribbean; e.g., Pari et al. 1998). This in turn can led to other cascading impacts - in the Caribbean the increased sea urchin abundance resulted in echinoderm disease and die off which in turn led to an increase in algal growth and coral smothering (Hughes 1994b). Starfish fish predators may reduce densities of juvenile Crown-of-Thorns starfish and correlative evidence suggests that fishing these predators may lead to periods of increased starfish recruitment. Higher survival of the juvenile starfish could lead to outbreaks of adults in subsequent years (Ormond et al. 1991). The effects of *Acanthaster* selectively grazing on stony corals on the GBR can lead to ecosytem changes such as an increase in soft coral abundance (Fabricius 1997) (see also Section 6.9).

The relative dominance of herbivorous fishes and invertebrates will have profound influences on the rates of reef accretion and bioerosion (Jennings and Polunin 1996). The grazing activities of herbivorous fishes may clear space for coral settlement and enhance the survival and growth of young coral colonies. Conversely, urchin-grazing leads to bioerosion that may exceed the rate of carbonate accretion (Pari et al. in press). The ecological release of invertebrate populations following the capture of their predators can also affect diversity.

There have been relatively few studies to determine directly the effect of removal of fishes from an ecosystem on invertebrate communities, although several have examined the importance of fish predation. For instance, Connell and Anderson (1999) found that predation by fish was intense on oysters and directly or indirectly reduced the density of a gastropod (*Bembicium auratum*) that uses the oysters as substrate. Algal cover was positively affected as a result. Other studies have found considerable variability in the effect of fish predation on epibiota (Sutherland and Karlson 1977; Russ 1980; Choat and Kingett 1982; Menge et al. 1985). In one of the few studies specifically designed to test the effects of fishing on invertebrates, Edgar and Barrett (1999) examined fish, invertebrate and algal species within the Maria Is marine reserve in Tasmania relative to external reference sites. They demonstrated that the densities of large fishes and rock lobsters, and their mean sizes, all increased significantly over a six year period following declaration of the reserve. Their results provided the first clear evidence that shallow Tasmanian reef ecosystems are overfished, and that unfished coastal ecosystems differ substantially from those where fishing occurs. They suggested that ecosystem change associated with fishing of shallow coastal reefs might be a widespread phenomenon.

Before-After Control-Impacted (BACI) type experimental programs (Underwood 1991) or short-term manipulations of the benthos in experimental areas have been used to observe the direct impacts of fishing, but the results do not lend themselves to the much larger spatial and temporal scale of the fishing ground (Thrush et al. 1995). As ecological changes due to fishing impacts occur over large areas and time scales, they may only be detected by comparing long time series that have been collected in fished and unfished areas, to take account of natural factors which may be changing over the same period.

6.10.2 Commercial fisheries

While a wide range of methods are employed including trawls, dredges, pots and traps, purse seine nets, jigging, long lines etc. only some of these are used to directly harvest invertebrates. However, all these methods have some direct or indirect affects on invertebrate communities. In particular, trawling and scallop dredging have been identified as being of serious concern in recent literature (e.g., Dayton et al. 1995; GESAMP 1997; Hall 1999; see Section 6.10.2). Some illegal fishing methods used by some countries in the Indo-west Pacific region are seriously destructive, and include the use of explosives and cyanide on coral reefs (Barber and Pratt 1998; Jones and Hoegh-

Guldberg 1999; Pet-Soede et al. 1999). Fortunately, as far as we are aware, these are not used in Australian waters. Other impacts from commercial fishing include rubbish dumping and other pollution such as oil spills. An overview of the ecological aspects of fisheries in temperate Australia is provided by Bell (1995).

Table 6.4: Summary of the impact of the main types of commercial fishing on invertebrates, their communities and their habitats.

Method	Examples of Invertebrate target species	Examples of invertebrate bycatch	Impact on community	Impact on habitat
Bottom trawl	Prawns	Octopus Squid and cuttlefish Crabs Specimen shells + other species large enough to be retained by the mesh (sponges, corals, bryozoans, crustaceans, molluscs, echinoderms, polychaetes etc).	Major disruption; removal of larger species; destruction, removal or damage to many benthic organisms Removal of 3D structure and replacing it with effectively a 2D structure—removal of small isolates of diversity	Impact gear dependent but can be severe removing most epibenthic organisms
Dredge	Scallops Various edible "clams"	Octopus Specimen shells + other invertebrates large enough to be retained by the mesh (sponges, corals, bryozoans, crustaceans, molluscs, echinoderms., polychaetes etc).	Removes larger organisms benthic and shallow burrowing infaunal organisms. Probably damages or kills many others by destroying their burrows, displacing shallow infaunal organisms etc.	Destruction of habitat by scooping through upper sediment layers.
Pots/traps	Crayfish Crabs ??Octopus	Scavenger species such as certain gastropods, hermit crabs, and some other crustaceans.	Removal of scavengers/predator s	Damage to coral skeletons if pots are thrown onto reefal areas-such as the Abrohlos
Purse seine nets	None	Squid	Removal of predators from food chain	?
Jigging	Squid	Squid	Removal of predators from food chain	?
Long line	Various finfish, e.g., tuna	?	Removal of predators from food chain	?

Some methods of fishing have much greater impacts than others. A study in the UK showed that beam trawls and scallop dredges did much more damage than the lighter otter trawls and that areas closed to towed fishing methods were significantly different from those where these methods were used (Kaiser et al. 2000). Some indication of the impacts of each of the main fishing methods relevant to invertebrates, or their communities and habitats, is given in Table 6.4.

Impacts of trawling and dredging for biota

The impacts on benthic invertebrates from fisheries practices such as trawling can be devastating and affect a very large range of taxa. While these impacts are not widely appreciated, they are increasingly becoming recognised. Without adequate control areas it is very hard to gauge the true impact of fishing on extant benthic populations. In large part, these communities have been heavily modified historically and so, unsurprisingly, further experimental trawling produces little change (Roberts 1997). Some areas, such as the North Sea, have been fished for so long that nobody knows what they were originally like, but even in these areas during recent memory the benthic communities have changed (Frid et al. 1999; Linnane et al. 2000). There are considerable problems in differentiating the effects of trawling and dredging from natural variability, and to find areas which have not been previously been trawled to act as controls because most areas suitable for trawling have been trawled at some stage (Dayton et al. 1995). A CSIRO report (Poiner et al. 1998), on the effects of trawling discusses the difficulties of designing a sampling program that has the power to statistically prove that trawling has been impacting benthic communities. Nevertheless, it clearly is having an impact (see Underwood 1997). Damage to substrate from fishing, and the fishing activity itself may have also assisted in the spread of exotics in Australia (see Section 6.5).

Trawling

Benthic trawling occurs throughout Australian waters for a wide variety of fish and invertebrates. The method involves towing a net just above the bottom while a tickler chain in front drags along the bottom to disturb the target organisms. These move up into the water column and are trapped in the net that is held open by the use of otter boards. The net and the otter boards also often drag on the bottom. However as well as disturbing the target organisms the tickler chain dislodges or damages other organisms living on the sea floor, and many of these are also caught in the net as bycatch (see Section 6.10.1). Such fishing techniques have the ability to rapidly and completely change benthic communities, which consist largely of invertebrates (Thrush et al. 1998) and, in particular, significantly reduce the three dimensional structure of these communities (Hutchings 1990; Thrush et al. 2001). Trawling may also resuspend sediments and increase turbidity (Churchill 1989).

Benthic trawling is one of the most important methods of fishing worldwide, and annually impacts an area equivalent to perhaps half of the world's continental shelf. However, its continuation and sustainability are being questioned because of its possible environmental impacts (Dayton et al. 1995; Poiner et al. 1998). In the most heavily trawled areas, such as parts of the North and Baltic Seas, the tracks of otter trawl boards criss-cross the entire seabed so that the seafloor is continually being trawled preventing little recovery of benthic communities (Krost et al. 1990). Despite this intensity of fishing, apparently sustainable trawling has continued for decades (and even centuries) on some Northern Hemisphere grounds, although there are no data regarding changes from the original state of these areas. In other areas declines have been linked to the impact of trawling (e.g., Bradstock and Gordon 1983). Recent advances in technology, especially the widespread use of GPS, allows boats to trawl almost up to the margins of reefs and have virtually eliminated what were de-facto refuges from trawling.

***Impacts from trawling*:** Trawling directly affects colonial and sessile invertebrates such as sponges, tunicates, bryozoans, hydroids and corals through smothering by sediment, dislodging them, breakage or prevention of further growth. It also affects mobile epifauna that are either captured in the trawl or damaged, or their habitat or food source is removed.

There is a substantial amount of literature documenting the effects of trawling on the benthos. Watling and Norse (1998b) have likened heavy trawling to forest clear cutting. They identify the impacts, both immediate (physical damage and destruction of organisms and communities; reduction of habitat complexity and 3-D structure; sediment resuspension and burial by settling plumes; water quality; bycatch) and long-term (changes in community structure and composition; reduction in numbers of sensitive long-lived, slow-growing, fragile and "tall" (epifaunal) species; increases in numbers of opportunistic species). Watling and Norse (1998a) argued that the "use of mobile fishing gear is on a par with agriculture as humankind's most important physical disturbance of the biosphere" (p. 1178). A comprehensive review of the environmental effects of trawling is given by Jones (1992). It includes a history of trawling, effects of scraping off the epibenthos and ploughing soft sediments; sediment resuspension; destruction of non-target benthos; dumping of bycatch and how this may modify feeding behaviours seabirds and fish predators, as well as theoretical considerations including the rate of recolonisation, recovery, and the types of communities that are most sensitive. Harris and Ward (1999) have recently reviewed the information on bycatch collected during Australia's commercial fisheries activities, and provides breakdown of the bycatch in all the major fisheries.

Several studies have been undertaken to attempt to quantify the impact of trawling. However, as pointed out by Dayton et al. (1995), there are difficulties distinguishing effects from natural variability, because the long fishing history makes it difficult to find undisturbed control areas. A study on the effects on seafloor habitat and associated invertebrate taxa in the Gulf of Alaska, USA, found that boulders were displaced, and large epifaunal invertebrates removed or damaged just by a single trawl pass. These structural components of habitat were the dominant features on the seafloor. There was a significant decrease in density, and an increase in damage, to sponges and anthozoans in trawled versus reference transects. Changes in density, or damage to most motile invertebrates were not detected (Freese et al. 1999).

CSIRO undertook a major review of the fisheries on the Northwest Shelf during the 1960-1970's (Sainsbury 1987). Comparisons of sponge catch rate in 1967-73 (Shu et al. 1973a, b) with that recorded by a CSIRO survey in 1979 showed that during the 16 year interval there was a significant reduction in sponge frequency. Along with the loss of these sponges and associated benthos there was a reduction in certain fish species (Sainsbury 1988). The bycatch in terms of diversity and biomass was significantly greater in "lightly" fished areas than in "heavily" fished areas (Russell et al., in Sainsbury and Poiner 1988). This suggests that habitats with three-dimensional structure tend to be more sensitive to fishing disturbance than communities with mobile sandy sediments and little three-dimensional structure (Collie et al. 1997; Hansson et al. 2000; Jennings et al. 2001). Loss of three-dimensional structure changes the habitat, leading to reductions in populations of animals dependent on it for a range of biotic reasons including shelter, food or spawning. Another study in Torres Strait also recorded a similar phenomenon (Poiner unpublished data quoted in Hutchings 1990) and anecdotal evidence suggests that large sponges were also abundant on the shelf areas around much of NSW, Victoria and South Australia prior to the advent of frequent trawling.

Demersal fishing activities provide food for scavengers in the form of dead or damaged animals left in the tracks of the trawl or dredge or discarded as bycatch. Responses by motile benthic invertebrate scavengers to trawling can vary, as studies on benthic scavengers to experimental trawling in the UK have shown. For example, the numbers of hermit and swimming crabs, as well as starfish decreased in some sites but the density of hermit crabs increased in others (Ramsay et al. 1998).

Dredging and bottom trawling affect water quality and turbidity. Pilskaln et al. (1998) suggest that the resuspension of sediment from trawling may have important implications for regional nutrient budgets (input of nitrogen and silica into the water column). While trawling had no detectable effect on sediment grain size, tracks made by trawl doors were readily visible on the sea floor 10 weeks later; in some cases they were still faintly visible after one year. Trawling had also increased roughness of surficial sediments, reduced surficial biogenic sediment structure and the abundance of flocculated organic matter (Schwinghamer et al. 1998). Studies of trawling and mussel dredging in Danish waters have shown that these activities increased particulate matter and ammonia levels and decreased levels of oxygen, which may affect phytoplankton primary production (Riemann and Hoffmann 1991).

In New South Wales there are plans by State Fisheries to investigate the effects of trawling (most of the NSW shelf and upper slope are heavily impacted by trawling), but the proposal, which proposes to investigate the infauna as well as the epibenthos, has yet to be funded.

***Trawling on the GBR*:** Hutchings (1990) reviewed the effects of trawling on epibenthic communities in the GBR. She highlighted the limited information available on their rate of recovery and suggested areas where information was urgently needed. To date, this information has still to be collected. Most of the trawling in the GBRMP and in the Gulf of Carpentaria is targeting prawns, with the type of trawling varying according to the target species (Hutchings 1990). About 800 trawlers are licensed to work on the Queensland east coast and a large number of these work within the Great Barrier Reef Marine Park. Trawl grounds in the GBRMP cover 153,000km^2, but the intensity of trawling varies throughout the region. About 27% of the lagoon and inter reef seabed is not trawled at all, and for that part of the seabed which is trawled, more than 50 % of the effort is concentrated in less than 20% of the area. Thus, about 70% of the trawled grounds have been trawled by less than one pass per year since the fishery has been established (Poiner et al. 1998). However, as pointed out by Hutchings (1990), many of the sponges, corals and gorgonians that form the three dimensional structure (or "mini-islands") in which other animals aggregate, are long lived, slow growing and have low levels of recruitment. Thus, even relatively low levels of trawling can be devastating to these communities.

During the early 1990s, the impact of trawling on the Great Barrier Reef became a major issue for the Authority. There was increasing pressure to reduce the area where trawling was allowed, or to even entirely ban this activity within the Marine Park. In response to this issue, a major study by CSIRO and the Queensland Department of Primary Industry compared an area in the Far Northern Zone that had been closed to trawling since 1985 (the "Green zone"; 10,000 km^2) with nearby areas that were regularly trawled over five years (Poiner et al. 1998). The major findings of the study are complicated by the fact that much illegal fishing was occurring within the "Green Zone'. One positive flow on effect of this investigation has been the implementation of the use of the VMS (Vessel Monitoring System). This system will allow the Authority to document illegal fishing incursions into protected areas and these recordings will be acceptable as evidence in court.

In the above study, there were considerable problems in the sampling design and obtaining statistical confirmation to prove that the observed differences were solely attributed to trawling, as the habitats varied considerably within the open versus closed areas both across the shelf and latitudinally. Much criticism has been levelled at the sampling strategy employed and the suitability of the Before-After-Control-Impact (BACI design) (Underwood 1991; Fabricius and Kelley 1999). Nevertheless, as a result of these studies there is now much better information as to where trawling actually occurs. In addition, generalised descriptions of the epibenthic communities and bycatch revealed the high diversity of the benthic fauna. There is also now reasonable sediment distribution data for the northern part of the GBR, which were previously lacking.

No studies on the infaunal communities have been undertaken in these heavily trawled areas of the GBR or even on the impact of the removal of the original three dimensional structure of the communities. Proposals to undertake this work have been developed, but to date have not been funded or given permits.

The coral reef communities on Low Isles are now covered in fine sediment. Historical data on these reefs (from the Great Barrier Reef Expedition) show that they were pristine reefs in the late 1920s. While some development has occurred in the catchment at Port

Douglas resulting in increased land run off during the wet season, trawlers also continually resuspend the fine sediments nearby.

Lowe (1999) commented on this GBR study by Poiner et al. (1998) and noted that trawling generated 6-10 kg of bycatch for every kg of prawns caught. One trawl can remove 10% of all seabed fauna (though since some species are more vulnerable than others, proportion of each species taken varies from 5-20%) while 13 trawls over the same area removed up to 90% of the initial biomass of epifaunal communities. Currently about 3% of trawling effort is expended in the most lightly used 20% of the GBRMP, and therefore these areas would be relatively easy to close to trawling from a political point of view. Such alarming figures of bycatch removal reinforce the need for information on recovery rates from trawling of epibenthic communities. It is likely that recovery will be patchy and vary according to the group of organisms. A large proportion of this bycatch represents undescribed species and, unfortunately, as in many of these studies, relatively little of the material was deposited in research collections in museums. In addition, the species recorded in such studies are only the larger taxa, which represent only a small fraction of invertebrate diversity. Most of the smaller epibenthic taxa would also be disturbed, damaged or killed by the trawling operation. Thus, the tentative species list given in Poiner et al (1998) undoubtedly greatly underestimated the number of species affected.

Negotiations were carried out by GBRMPA and Queensland Fisheries to reduce the area over while trawling is allowed on the reef and to gradually reduce the effort annually (Tanzer 1998). A new management plan for the East Coast Trawl Fisheries came into effect on the 1st January 2001. 96,00 km^2 of the GBR is now closed to trawling for the first time to prevent expansion of the trawl area. Half of the GBRMP is now closed, as a result of zoning through Queensland Fisheries legislation. This plan caps the fishing effort at 1996 levels involving a 15% reduction. The plan allows the introduction of tradeable effort units, with penalties on trading of effort units, license transfer and vessel replacements to combat effort creep due to improvements in technology. In 2001 this resulted in a 5% reduction in effort units. The number of trawlers has been reduced and overall there are 258 fewer trawlers that there were in 2000 over the entire region. Within the World Heritage Area, there are now only 500 trawlers. In August 2001 there was an agreement between the Authority and Queensland Fisheries Service on effort levels within the WHA. In September 2001, Queensland Fisheries Service legislated for an automatic closure of the trawl industry in the WHA once the approved yearly allocation is reached, and this will be decreased by 3% per year. The implementation of the trawl plan has been incorporated into the recently approved Far Northern Zoning (FNS), so that now only 21% of the FNS is open to trawling, the remainder being closed. In addition, the Authority is undertaking auditing, including stock assessments, to assess the performance of the trawl plan.

***Trawling in deep-water habitats*:** With increasing over exploitation of shallow water fisheries, deep-water areas are now being trawled (e.g., for Orange Roughy, *Hoplostethus atlantina*). Kaiser (1998) suggested that deep-water habitats are likely to be severely impacted as these areas are subjected to low levels of natural disturbance and organisms tend to be slower-growing than in shallower waters. Probert et al. (1997) reviewed the environmental effects of deep-water trawling and described the impacts of Orange Roughy fishing in 662-1524 m depths of the Chatham Rise, New Zealand where the bycatch consisted of large sessile epifauna (see below). They pointed out that these organisms may significantly increase the complexity of benthic habitat and trawling damage may thereby depress local biodiversity, with coral patches perhaps requiring about 100 yr to recover. There conclude that there is an urgent need to assess more fully the impact of trawling on seamount biotas and necessary conservation measures.

Off Southern Tasmania, a large number of submerged seamounts have recently been discovered and already trawl operations appear to have significantly impacted some of them (Koslow and Gowlett-Holmes 1998; Koslow et al. 2001). In these cases the reef aggregate has been mostly removed from the slopes or turned to rubble. The benthic biomass from heavily fished seamounts was 83% less than from lightly fished or unfished seamounts and the number of species per sample was 59% less. The fauna is highly vulnerable to trawling and is likely to have limited resilience, as its slow growth and low natural mortality are adapted to an environment with little natural disturbance (Koslow and Gowlett-Holmes 1998; Koslow et al. 2001). This was taken into account when the area was declared as the Tasmanian Seamount Marine Reserve in May 1999. The reserve fully protects a zone from 100 m below the sea floor to 500 m under the ocean surface from fishing and from petroleum and mineral exploration. About 20% of the reserved seamounts are included in the reserve and all occur well below depths of 500 m and thus will be protected from the damaging effects of trawling.

Dredging

The effects of experimental scallop dredging on the infaunal benthic community structure of Port Phillip Bay, using a BACI design, has been investigated by Currie and Parry (1996). The abundances of seven of the 10 most common species changed significantly, with six decreasing and only one increasing. Dredging impacts varied among species, but in general most species initially decreased in abundance by 20 to 30% then recruited within six months of the dredging impact (after which impacts became undetectable). A small number of species, however, still had not recruited after 14 months and appeared to be responsible for a persistent change in community structure. The general conclusion was that scallop dredging had profound impacts on the benthos and probably cannot be sustained (see also Section 6.3.2). In the US, Thrush et al. (1995) conducted field experiments to test the short-term impacts of commercial scallop dredging on macrobenthic communities. Dredging decreased the density of common macrofaunal populations, and resulted in significant compositional changes, with some differences still apparent three months later. These findings were considered to provide a conservative assessment of the impact from dredging.

6.10.3 Recreational and minor commercial or semi-commercial fishing

Recreational fishing is a very popular pastime in Australia, involving over 4.5 million Australians each year. Of these, 800 000 fish on 20 or more days per year and can be regarded as regular fishers (McNair 1992). While most of this activity is focused on finfish, molluscs, crustaceans and echinoderms also are directly targeted for food, and a range of invertebrates are used for bait. Activities such as bait collection have a minor commercial involvement.

While most forms of recreational fishing are unlikely to directly drive any targeted invertebrate species to extinction, they can remove considerable biomass, particularly from the accessible intertidal zone. This can lead to reductions in the density of harvested species, as well as effects on community structure (e.g., through removal of predators) and damage to habitats (e.g., through destructive collection techniques or access routes).

In the Cocos (Keeling) Islands, for instance, there is little commercial fishing but fish, clams, coconut crabs and mud crabs are heavily collected by local artisanal and recreational fishers and there are concerns that sites may be overfished (Caton et al. 1998; Fishery Status Reports for Commonwealth fisheries). Hand-collected giant clam (*Tridacna gigas*), coconut crab (*Birgus latro*) and mud crab (*Scylla serrata*) are very depleted locally. Other popular species, including rock lobsters (*Panulirus* spp.), spider shells (*Lambis* spp.) and clams are potentially vulnerable to overfishing; for instance, an estimated 50 000 to 70 000 spider shells are taken each year for food (Caton et al. 1998).

Methods employed by recreational fishers are outlined in Table 6.5 with an overview of the possible impacts that each method may have on invertebrate communities and habitats.

Intertidal collecting

Intertidal habitats are unquestionably the most vulnerable to activities by the general community because of their accessibility. A range of seashore communities is threatened, especially in easily visited areas. Keough et al. (1993) and Keough and Quinn (1998) monitored human activity (collecting for food and bait) on rocky shores near Melbourne and found high proportions of exploitative activity – 25% of visitors were actively collecting, despite protective regulations.

The types of habitats in which harvesting occurs differ from place to place and depend on the animals being targeted. In recent years, the abundance of invertebrates in the intertidal zone around metropolitan centres has declined largely due to collecting for food or bait by recreational fishers (e.g., NSW Fisheries 1998a). There are particular concerns about octopus, cunjevoi and molluscs from rocky shores, and pipis and cockles from sandy and muddy shores respectively. Some forms of bait collecting are also of concern, with a large proportion of recreational fishers obtaining their own bait from pumping ghost shrimps or worms from sand flats and collecting algae, crustaceans, gastropods, bivalves and ascidians from rocky shores (Fairweather and Quinn 1995). Of particular concern is suction pumping for "yabbies", a method that destroys sand flats and seagrass beds. Declines through over-collection have been exacerbated by destructive collection techniques such as the use of crowbars. Other impacts include damage to individuals and habitats from trampling (Keough and Quinn 1998) or overturning of rocks.

Table 6.5: Summary of the impact of various types of recreational fishing on invertebrate communities and their habitats.

Method	Examples of invertebrate target species	Impact on community	Impact on habitat
Intertidal line fishing	N/A	Removal of predators; removal of bait species	Minimal
Bait collection	Burrowing crustaceans; beach worms; ascidians; limpets; other gastropods; various bivalve species (e.g., pipis).	Removal of target species from the ecosystem.	Usually local damage only, except for digging and pumping methods used to obtain burrowing taxa on intertidal flats and seagrass beds- if extensive these can cause considerable damage.
Spear fishing and SCUBA diving	Crayfish; crabs; sea urchins; abalone; octopus; cuttlefish; other edible molluscs. Specimen shells; coral.	Removal of target species	Usually minimal. Sometimes some damage either directly or indirectly (e.g., from boat anchors).
Intertidal food collection	Sea urchins; abalone; other edible molluscs; crabs.	Removal of target species.	Usually minimal - may result in overturning of rocks etc.
Miscellaneous intertidal collecting	Specimen shells Coral etc	Removal of target species	Usually minimal - may result in overturning of rocks, removal of coral etc.

Rocky shores

Activities of people on rocky intertidal areas are diverse and have different levels of impact on the biota. Tide pooling; collecting for food, aquaria or research; educational field trips; seaside strolling; photographing and fishing are probably the most common activities on rocky shores. Heavy human use of the intertidal zone can be very destructive through trampling, overturning rocks (and failing to return them), and the intensive collection of certain species.

The effects of collecting by humans on Southern Hemisphere rocky intertidal shores have been examined in a number of studies, particularly in Chile, South Africa and to some extent Australia (e.g., Castilla and Durán 1985; Kingsford et al. 1991; Underwood 1993b, a; Addessi 1994; Siegfried 1994; Davis 1995; Griffiths and Branch 1997; Castilla 1999).

In Chile, comparison of human-excluded 'no-take' areas with human-impacted sites showed an increase in the abundance of keyhole limpets (*Fissurella* spp.) coupled with a dramatic decline in the abundance of mid-intertidal macroalgae that resulted in extensive food-web modifications (Moreno et al. 1984; Moreno 1986; Oliva and Castilla 1986; cited in Castilla 1999). This led Castilla (1999) to argue that humans, as top predators in these systems, should be considered a keystone species and incorporated in ecological studies and models just like any other species (See Section 5.4.4). Davis (1995) discussed human-exclusion experiments in Chile and their effects on community structure and diversity of marine invertebrates, and how these findings were applied as management strategies. For example, densities of the "Piure" (*Pyura chilensis*), a commercially exploited tunicate, were more than three orders of magnitude higher within protected reserves than outside them, and only around 6% of individuals in the harvested populations grew large enough to reach sexual maturity (Davis 1995). This was indicative of over-exploitation. In South Africa, comparisons between paired human-exploited and non-exploited rocky shores demonstrated that selective predation by humans of the mussel *Perna perna* and the limpets *Cellana capensis* and *Patella* spp. increased species richness. Overall it led to a significantly greater cover of unexploited sessile species such as macroalgae because the remaining limpet population could no longer control algal growth at the sporeling stage (Hockey 1994). A review of the effects of exploitation of coastal invertebrates and seaweeds in South Africa (Griffiths and Branch 1997) found that direct impacts, such as radical changes in population densities and size distributions of many target species, have generally been well documented, although effects on community dynamics are far less well appreciated. In San Diego, California, the long-term effects of human disturbance on a rocky intertidal community included reduced density of all species in heavily visited sites, increased numbers of some small gastropods, disappearance of five species of echinoderms and decline in the density of predators such as octopuses (Addessi 1994).

Targeted organisms on Australian temperate rocky shores include molluscs (e.g., mussels, limpets, abalone, octopuses), echinoderms (sea urchins), and ascidians (cunjevoi). Human harvesting of intertidal and subtidal species of invertebrates and algae on the rocky coast of NSW is widespread and can be destructive. The patterns and consequences of harvesting in NSW were summarised by Underwood (1993b). Direct effects included the loss of the individuals actually taken and the potential loss of breeding populations. Indirect effects included the loss of food for other species, which depend on the harvested species, and the loss of habitat for non-exploited species caused by removal of harvested species with which they interact (Underwood 1993b).

The activities of humans on rocky reefs along the coast of NSW were also surveyed by Kingsford et al. (1991), who found that intertidal invertebrates (ascidians, crabs and gastropods) were primarily taken by fishermen for bait, whilst some echinoids and gastropods were used as food. In particular, large numbers of the ascidian *Pyura stolonifera* (cunjevoi) were taken for use as bait (see also Fairweather 1991). The collection of this species, in addition to the effect on the local population from the removal of large reproductively active individuals, may have an ecological impact, although there is currently poor understanding of such effects. For instance, the removal of cunjevoi may result in loss of habitat for other species as *P. stolonifera* are an important substratum for the growth of algae in areas grazed by chitons. They also serve as habitat for a wide range of other organisms and as prey for certain species, including the sooty oystercatcher, a bird listed as vulnerable (A. Davis pers. comm.) and several species of large triton whelks (*Cabestana spengleri* and *Charonia lampas* (Ranellidae)) (WFP pers. observ.). The impacts of collection for bait on the population dynamics of cunjevoi were examined by Fairweather (1991), whose results suggested that declines in *Pyura* beds at some of the sites examined (in NSW) were linked to severe and chronic harvesting by fishermen.

Keough et al. (1993) examined the effect of human collecting for food and bait on mollusc populations in northern Port Phillip Bay, Victoria. Three of four collected species (*Cellana tramoserica*, *Austrocochlea constricta*, and *Nerita atramentosa*) were significantly larger and one (*N. atrementosa*) was markedly more abundant at protected than heavily visited sites. The only species that showed no significant difference (*Turbo undulatus*) has a distribution extending into the subtidal zone and may have its intertidal populations replenished from deeper water (Keough et al. 1993).

Soft shores

While most studies have focused on rocky shores, there are also concerns about the harvesting of pipis and cockles from sandy and muddy shores respectively (NSW Fisheries 1998a). Bag limits are imposed in many areas. For invertebrates in NSW, these range from a bag limit of two (e.g., painted, slipper or eastern rock lobster) to 50 (e.g., oysters and pipis). For species with an unspecified limit, the default is 20. The activities listed below are mainly all carried out by amateurs so few figures are available as to the intensity of each activity.

Suction pumps**:** Fishermen collect *Callianassa* spp. (burrowing prawns) for bait by using a "yabby" or suction pump at low tide on muddy flats which often have sparse seagrasses. In NSW, NSW Fisheries has closed many beaches to pumping because of the destruction of the seagrass beds. This activity is carried out by fishermen for their own use, and, as far as we can determine, this bait is not sold commercially.

Hand-collection of beach worms**:** Beach worms are collected by the millions by recreational fishermen and professional collectors, but this bait fishery appears to be sustainable (H. Paxton pers. comm.). Paxton (1979) found that the single species of beach worm (Polychaeta, Onuphidae) previously recognised by authorities was actually several species, most of which had been recognised by fishermen as separate varieties. The Giant Australian Beachworms *Australonuphis teres* and *A. parateres* are widely used as bait and occur in large numbers intertidally on sandy surf beaches along the southeastern Australian coast. Due to their great length (up to 200 cm) and muscular body, the beachworms are highly valued as bait. The current method of collecting involves attracting the worms to the surface using bait, and then extracting them by hand, one worm at a time. Although some beaches have been heavily collected for a considerable period, the worms are still plentiful. This is probably due to the non-destructive collection method - it only occurs in the intertidal zone (the population extends subtidally) and mainly only large individuals are taken. The harvesting technique employed causes virtually no disturbance to the substrate in contrast with digging for other species of bait worms (H. Paxton pers. comm.). However, any trials of mass collection (e.g., using mechanical diggers) should be discouraged and some management action taken to restrict the collection of beachworms to the present method (H. Paxton pers. comm.). Currently there are studies being carried out at the University of Queensland on the ecology of these species and their potential value as a sustainable fishery[130].

***Digging for worms*:** Digging for polychaetes occurs widely throughout Australia, with the species collected varying between regions. In Moreton Bay in Queensland there are licensed bait diggers who dig the seagrass beds to collect *Marphysa* sp. (commonly known as blood worms, but such activity has largely been banned in NSW because of the damage it causes to seagrass beds. In South Australia, the species collected are *Diopatra* spp. (Hutchings and Karageorgopoulos in press). While primarily collected by amateurs, there are some professional collectors. Pilot studies are being carried out to investigate the feasibility of breeding these worms commercially. In Europe another polychaete, *Nereis virens*, is being successfully bred and exported as bait throughout northern Europe.

***Raking for bivalves*:** Collection of intertidal and shallow sublittoral bivalves for food, while important in many parts of the world, is not a major industry in Australia largely because few edible species have commercially viable populations. The few species collected in temperate Australia and sold in relatively small volumes for both food and bait are bivalves including the Pipi (*Donax deltoides*), "cockle" *Kateleysia* spp and the Mud Cockle (*Anadara trapenzia*).

Hall and Harding (1997) described the effects of mechanical intertidal harvesting of cockles (*Cerastoderma edule*) on non-target benthic infauna in the UK. Manipulative field experiments are used to compare hydraulic suction dredging and tractor dredging with undisturbed controls. They found that the faunal structure in disturbed plots recovered (i.e. approached that of the undisturbed controls) after 56 days. They concluded that, although mechanical harvesting methods imposed high levels of mortality on non-target benthic fauna, recovery of disturbed sites was rapid and the overall effects on populations were low. No similar experiments have been carried out in Australia to our knowledge.

Shell collecting and the commercial shell trade

The collection of shells, particularly for the commercial trade, was identified as a significant threat in early conservation literature (e.g, Starmühlner 1985). Reports on the specimen shell trade in Australia have been produced by Willan (Willan 1986), Davey (1993) and Ponder and Grayson (1998)[131]. Willan (1986) estimated that the annual value

[130] www.geocities.com/CollegePark/Residence/5422/project.html.
[131] http://www.ea.gov.au/biodiversity/trade-use/wild-harvest/pubs/pondrpt1.pdf.

of the shell trade was $2.5 million in the mid 1980s, but since then the number of specimens exported has doubled (Davey 1993).

The specimen shell trade and collectors in general are only concerned with a very small proportion of the molluscan fauna. The great majority of molluscs are small (< 1cm) and of no interest to collectors or the general public. In their analysis of official export records, Ponder and Grayson (1998) reported that in the years 1996-7 only 1682 species were exported, compared to the more than 10,000 species in the Australian fauna. 1036 of these species, belonging to 21 families, made up 83.9% of the total number of specimens exported. Only 27 species had 300 specimens or more exported in 1996-7 and these comprised 28.8% of the total specimens exported. Twenty-two families with more than 500 specimens exported over two years (1996-7) represented 84% of the total specimens exported and 61.8% of the species. By far the most specimens exported were members of the Cypraeidae (20.7%), with the Volutidae (11.2%) and Haliotidae (7.9%), which together with Muricidae, Conidae, Turbinidae and Trochidae, had in excess of 2000 specimens exported over the two years. All of these trends would probably also be reflected in the local usage of specimen shells by Australian collectors.

Ponder and Grayson (1998) argued that there is little evidence that collecting for the purposes of obtaining specimen shells has any appreciable impact on the long-term survival of species, except in a few special circumstances. They suggested that the impact of habitat disturbance due to fishing trawling and dredging, and other impacts such as development in coastal areas, pollution etc., are much more significant and may ultimately affect the long-term survival of even some relatively common taxa. They also identified the factors that might lead to over collecting and used these as the basis for criteria to assess the vulnerability status of targeted species. These included restricted distribution, direct development, accessible habitat and high market value. Other relevant criteria that could not be used, due to lack of data, were fecundity and abundance. Overall, the species exported that were identified as being in the top vulnerability categories comprised 7.8% of the total, but the number of specimens exported for these species was 23.7% of the total. Of considerable concern is that of the top 27 species exported (>300 specimens in 2 years), close to half (45.3%) were in the top three categories of vulnerability.

Aquarium trade

While the aquarium trade in marine invertebrates is significant in some parts of the world, it appears to be a relatively minor activity in Australia and much of it is restricted to amateur activity.

The global trade in coral, which involves more than 2000 species, is monitored by the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES). Analysis of trade in black corals and stony corals from the early-mid 1980s through to 1997 (Green and Shirley 1999) found that during this period, 70 nations imported a total of 19,262 tonnes from 120 exporting nations. Most imports originated from Southeast Asia (especially Indonesia), followed by the Pacific, Caribbean, and

Indian Oceans; Australia is an insignificant source in global terms and was not mentioned. While dead coral (mainly branching forms) accounted for more than 90% of the trade up to the early 1990s, there has since been a large increase in the amount of coral shipped live for the aquarium trade, the quantity having increased tenfold since 1985 to make up half of the global trade in 1997 (i.e. 600-700 tonnes). The dimensions of coral pieces suggest the typical live coral in the aquarium trade is at least three years old. Green and Shirley (1999) argued that in comparison to other impacts (such as mining and dynamiting), the effects of collecting live coral for the aquarium trade are very small. For example**,** it has been argued that the loss of coral export potential because of the CITES ban has led to a loss of income to coastal communities in countries such as Indonesia and the Philippines. The resultant lack of commodity value for coral may have encouraged the wide adoption of blast fishing throughout the Indo-west Pacific, a practice that has greater adverse non-target impacts.

Collection of coral for the tourist trade is illegal in Australia, but a small, highly controlled live coral trade industry occurs on the Great Barrier Reef, with permits issued by GBRMPA for selected reefs. Currently, about 40 operators are licensed to collect in a certain area and take about 50 tonnes of live coral (including living rock and soft corals, anemones and their associated fish) annually (GBRMPA 2000; S. Breen pers. comm.). Harriott (2000) concludes that the current live coral trade, which currently involves 50 small fixed leases, is sustainable and suggests that roving licenses should be investigated, to spread the collection effort over wider areas. However, the Authority is opposed to this because of difficulties in assessing compliance and in monitoring impacts. The industry currently has some latency and the Total Allowable catch (TAC) should be reduced to reflect more closely the current, much lower level, of catch within the fishery, thus removing this latency. Separate catch quotas should be set for the living rock/rubble and live coral component. Harriott stresses that the management of real or perceived conflicts with other reef users is a significant issue for the fishery. Designation of collection areas must minimise areas of conflict. Re-designation of most collection sites to deeper, turbid locations would benefit the fishing industry and reduce conflicts with the tourism activities. Finally, the farming of coral is probably not feasible within Australia but should be encouraged overseas.

6.10.4 Management issues and recommendations

Management strategies applicable to coastal marine resources need to be at the level of ecosystems rather than species, should include regulations that accommodate biological and social realities, and adopt novel approaches such as the encouragement of co-management of resources (e.g., Griffiths and Branch 1997). In Australia, integrated management is rendered more difficult because management of the coastal zones is state-based and beyond three nautical miles is the responsibility of the Commonwealth. Thus, management strategies may differ considerably on either side of a state or territory border. In addition, such issues as indigenous access to marine resources are mostly still in the process of being resolved.

Many management regimes take into account recreational as well as commercial harvests, for example by incorporating size and bag limits for casual collectors. Hence, while the discussion below of management options is divided (as above) into sections for commercial versus recreational or semi-commercial fisheries, there is some overlap.

Commercial fisheries

Conventional approaches to fisheries management have had limited success in many cases, as shown by the declines in stocks of many exploited species (both finfish and invertebrates), even those supposedly under management regimes (Roberts 1997). Some of the problems with traditional approaches, as discussed in some detail by Roberts (1997), include the species by species assessment of stocks. Many fisheries are still managed based on *Maximum Sustainable Yield* (MSY), an approach discredited more than two decades ago (Larkin 1977). Calculation of MSY assumes that equilibrium can be achieved between fishing effort and stock size. In practice, fish (and invertebrate) stocks are highly variable as a result of variations in recruitment success and subsequent survival of juveniles and because stock size is often driven more by environmental variation than directly by exploitation (Roberts 1997). For example, although predictive models have been developed which accurately estimate the year's catch of overfished (Caton et al. 1998) Tiger Prawns (Rothlisberg et al. 1985) a major problem with the management of this fishery is the difficulty in evaluating its sustainability through catch size due to the huge natural variation.

There are several conflicting and inter-related goals in any successful fisheries management program, including:

- Sustainability;
- Minimising environmental impact;
- Maximising return for effort (profitability); and
- Meeting demand.

While balancing commercial or recreational interests and environmental concerns can be difficult, sustainability is in the long-term interest of the fishing industry and the consumer. Minimising environmental damage can also enhance productivity (e.g., by protecting the communities directly or indirectly supplying the food and/or shelter on which the target species depend). Similarly, productivity can be enhanced by the environmental management of habitats in which the target species (or their food species) live at different stages in their life cycle (e.g., mangrove habitats act as nurseries for many species of fish and invertebrates including commercially important species).

For a fishery to be sustainable it must at least address the following three issues:

- Manage breeding stocks of the target species;
- Minimise effects on non-target species (minimisation of bycatch); and
- Minimise impacts on habitats.

There are several options available for meeting these issues on the sustainable management of fisheries:

- Control of catch (e.g., size limits, bag limits, total allowable catch);
- Control of fishing methods (e.g., limitations on size and type of equipment or boat used);
- Control of fishing period (e.g., seasonal closures);
- Control of number of fishers (e.g., licensing) and control of effort- i.e. fishers can only work so many days a year;
- Harvest refugia (e.g., fisheries "no-take" zones; marine protected areas see Section 6.10.4)
- Regulation of exports and imports;
- Artificial seeding (e.g., hatcheries); and
- Vessel Monitoring Devices (VMD)—which allow the managers to know exactly where the boats are, and what activities they are undertaking.

We do not discuss all of these as these strategies are well covered elsewhere. Instead, we focus on those management strategies that may be important for the conservation of the target and non-target marine invertebrates. Because many of the non-target species provide the food for target species, references in this discussion to commercial and recreational fishing relate to all such activities, not just those targeting invertebrates.

Maximum sustainable yield

MSY and other recent approaches (e.g., attempts to maintain reproductive output at a certain proportion of unexploited stock capacity) have many drawbacks (Roberts 1997), including:

- Considerable biological information on stocks is required to make predictions and such data often are absent or expensive to obtain;
- The location of MSY at the apex of the catch-effort curve means that even a small amount of uncertainty/error risks overfishing the stock;
- The precision of predictions is compromised by the complexity of species interactions;
- Ecological uncertainty is interpreted as an invitation to do nothing, or insufficient efforts to safeguard stocks; and
- Even where the political will is present, the resources necessary to police and enforce complex quotas, size, gear or bag limits, sometimes are not applied.

Management of effects on harvested species

A variety of methods for limiting effort or catch (known as input and output controls respectively) are already in place for different fisheries within Australia. *Input controls* limit effort and include restrictions on the number of licences, type of gear, or times when harvesting may take place (e.g., seasonal closures). *Output controls* limit the quantity or characteristics of the stock that may be collected and may include individual bag limits, size limits, and /or a Total Allowable Catch (TAC) or Individual Transferable Quota. While these options are fairly well developed, they have had limited success in many instances due to problems with enforcement and dealing with single species management.

Since the fecundity of many marine invertebrates increases with age and size, the choice of legal size limits is critical. While minimum legal sizes are widespread, and aim to preserve individuals until they have at least obtained reproductive age, it can also be important to restrict collection of very large individuals, as their removal can have a disproportionate effect on the reproductive output of the population. For example, overfishing of scallops in Queensland might be addressed by selective protection of brood stock as an alternative to restricting effort or actively enhancing the stock (Dredge 1988). Consequently, some management plans have introduced maximum as well as minimum size limits (e.g., Western Rock Lobster fishery in WA). Such maximum size limits may be increasingly necessary as improvements in technology (and economic imperatives) enable (or force) fishers to exploit populations in increasingly distant and difficult areas, which might formerly have served as refuges for some large individuals. For instance, the imposition of a maximum size limit in the WA rock lobster fishery was in response to the advent of GPS, which enabled fishers to accurately locate deep offshore reefs.

In NSW, commercial fishing for the blacklip abalone, *Haliotis rubra* is regulated by a variety of controls such as minimum size limits and a Total Allowable Commercial Catch (TACC). While the catch has remained stable for about the last five years (as a result of the TACC being fixed at 333 tonnes), the size-structure and growth of individuals are suggestive of high exploitation rates (Andrew et al. 1997). The fishery is dominated by individuals that have recently recruited to the fishery (i.e. reached the minimum legal size of 115 mm). These recent recruits make up a large proportion (50-95%) of the commercial catch in all zones (Andrew et al. 1997), the accumulated stock of old, large abalone apparently having been seriously depleted over the life of the fishery. Catch rates in NSW (about 20 kg hr^{-1}) are also much lower than in Tasmania and Victoria (above 50 kg hr^{-1}), due mainly to the additional time required finding abalone above the length limit (Andrew et al. 1997). The NSW abalone fishery appears to be very different to those of Victoria and Tasmania, which have annual catches of over 1500 tonnes and over 3000 tonnes (respectively). This probably results from a combination of biological, ecological and other factors, as well as patterns of management. However, it is interesting to note that the legal minimum sizes in both Victoria and Tasmania are considerably higher (120 mm and 132 mm respectively) than in NSW, resulting in the presence of greater numbers of larger (more fecund) individuals in the population (Andrew et al. 1997).

In cases where stocks appear to be over exploited, one management option is to reduce fishing levels, for instance through a reduction in the number of licenses. This has occurred in several fisheries, including the NSW abalone fishery. A reduction in fishing levels is currently being pursued on the Great Barrier Reef, with a planned gradual reduction through buy-back of some licenses, restriction in the area available for trawling, and limitation on the number of days of trawling. While current levels of exploitation may be sustainable, any new incentive for new or latent license-holders to enter the fishery (e.g., increases in price, depletion of alternative species) could result in significant depletion beyond the immediate control of managers. In Queensland, for instance, it is well known that there are a number of latent (“sleeping”) licenses to operate on the GBR, a fact which has had to be taken into account in discussions and negotiations about the reduction of fishing pressures on the Reef, and calculations of the effect of proposed buy-backs etc. In the future, perhaps, fishers may need to use their licenses to at least a minimal level each year in order to be keep them (see above).

Case Study: Western Rock Lobster fishery

The Western Rock Lobster (*Panulirus cygnus*) fishery is the most valuable single-species fishery in Australia and usually represents about 20% of the total value of Australia's fisheries. Virtually the entire catch of the Western Rock Lobster is caught up to 60km off the coast between Augusta and Shark Bay. The open season is between 15 November (Abrolhos Islands area closed until 15 March) and 30 June and fishing is done using baited pots. Commercial diving for lobsters is banned. The catch is mainly exported. Fishing has been under a management plan since March 1963 when licence and pot numbers were frozen, with a 10% reduction in pots between 1987 and 1992 a decline in boat numbers from 836 in 1963 to 596 in February 1999. The sustainable catch is estimated at between 10,000 and 11,000 tonnes per year, although the size of the actual catch has varied between 8,000 and 13,000 tonnes.

In the early 1990s it was estimated that egg production had fallen to 15-20% of what it had been before serious commercial fishing began in the 1950s, whereas 25% was considered sustainable. Measures to boost numbers of breeding females were introduced in 1993/4 and are still in force. Pot numbers were reduced temporarily by 18% and a slight increase in minimum size was set for the first 2.5 months of the season. It is also illegal to take mature females in breeding condition. There is also a maximum size limit applicable for females to boost egg production by protecting the remaining very large females. The configuration of pots and the size and number of escape gaps are also regulated to allow undersize lobsters to escape and pots may only be pulled during specified daylight hours.

Any new technology that may increase fishing success (e.g., underwater video cameras and refined pot design) must be assessed and approved before use. Recreational fishers, who take an estimated 3-8% of the commercial catch, must also be licensed and are subject to a strict bag limit.

As the price of lobster produces a high seasonal income, licences and gear have become valuable assets. Depending on the zone fished, a 100-pot licence, together with boat and gear, is valued at about $3m (1997/98 season) so new entrants face very high costs. Existing rock lobster fishermen are thus keen to conserve their livelihoods and have been supportive of conservation measures designed to maintain stocks. The 1997/98 catch of 10,500t represented an increase of 6% from the 1996/97 catch of 9,900t, despite the same fishing effort (10.7 million pot lifts)[132]. This is one of the few fisheries that has not exhibited any decline in landings, probably because this is a very well regulated industry with few players operating over a relatively small area. For instance, during the fishing season most of the fishers live on one small island (Rat Island in the Abrohlos) and in this close-knit community self-regulation plays a very important role.

Limitation of bycatch –equipment modification

[132] WA Fisheries website: http://www.wa.gov.au/westfish/comm/broc/lobster/lobcm.html

Minimising catches of non-target animals (bycatch) in fisheries reduces the impact on a marine community and may help to sustain the fishery in the long term. Various authors such as Brewer et al. (1996) and Kenneally (1997) have discussed ways in which the bycatch problem could be addressed. Various devices to reduce bycatch have been developed. These include exclusion devices designed to exclude larger animals (e.g., dugongs), avoidance mechanisms (e.g., raising a trawl slightly off the bottom), and filtering using a targeted mesh size. The last two approaches are particularly pertinent to invertebrates. Roosenburg and Green (2000) discuss the use of a bycatch reduction device on crab pots.

There have been relatively few studies on the invertebrate component of bycatch (see Section 6.10.1). Assessment of the bycatch (as well as target species) would be a convenient way of determining ecosystem changes over time and could be used as a management tool.

No matter what avoidance techniques are used, bycatch of non-targeted invertebrates will still be an issue. Typically, once caught the mortality of the animals in the bycatch is very high to total, even when returned to the sea quickly.

Area closures / harvest refugia

Because conventional management involving restrictions on catch and fishing effort require biological information, some argue that the use of marine reserves is an alternative, as discussed in Section 5.5.3. Harvest refugia – also known as fisheries reserves or no-take zones – are a form of marine protected area designed primarily to conserve breeding stocks of exploited species, with the intention that these serve to replenish harvested stocks in surrounding waters.

The concept of marine refugia as a management strategy for sustainable fisheries has gained considerable support in recent years (e.g., Dredge 1988; Underwood 1993b; Roberts 1997) with many examples showing their effectiveness in enhancing local fisheries (e.g., Quinn et al. 1993; Castilla 1999). For instance, Dredge (1988) argued that selective protection of brood stock could be a more effective means of managing overfished scallop stocks, rather than restricting effort or actively enhancing the stock. Roberts (1997) argued that marine reserves are one of the most effective ways of providing safety factors in fisheries management, thereby integrating the precautionary principle, providing the reserves are of adequate size, in the right places, and adequately policed. Refugia may be important for maintaining breeding stock and ensuring the survival of juveniles (e.g., as suggested by Dredge 1988; Shepherd and Brown 1993) or as an area with a dependable food supply for commercial species. In 1990, the US South Atlantic Fishery Management Council, in what was then considered a radical departure from conventional fisheries management, recommended that 20% of the region be set aside as marine fishery reserves: areas in which 'no take' of reef fishes would be permitted (Roberts 1997). Under the current management plans for the GBR, about 4.5% of the Marine Park area is designated as "no-take" zones, in which no form of fishing or harvesting is permitted; however, most of this covers reef (rather than inter-reefal) areas (GBRMPA 1999; Day et al. in press). Hopefully, the current Marine Representatives

Area program, which is currently in the first phase of public participation, will see an increase in the area of no take and better represent the diversity of habitats represented within the Park. This management option (i.e. prohibition of fishing) has, however, been used in relatively few marine areas around the rest of the Australian coast.

Recently NSW State Fisheries declared six new aquatic reserves in the Sydney region, which complement eight other aquatic reserves in the State. Each aquatic reserve is unique, and the type of protection varies. In some areas, diving and observing are the only activities permitted. In others, activities such as fishing are also allowed. Details of the activities allowed in each of these reserves are on the NSW Fisheries web site[133].

Depth refuges can also be an important strategy in some fisheries, depending of course on the fishing methods used and the potential of developments in technology to allow access to formerly inaccessible stocks. Karpov et al. (1998) emphasised the importance of depth refuge in the management of the Red Abalone in California. In the southern and central part of California, the use of SCUBA has been permitted for both commercial and sport fisheries, allowing abalone to be taken throughout its depth range. In contrast, in northern California, commercial fishing has not been allowed, nor has the use of SCUBA by sport fishers, resulting in a refuge from harvesting at depths below 8.4 m. The former fisheries were recently closed as a result of stock declines, while the latter continue to provide high yields.

Ideally marine reserves should be established on the principle of ‘no-take’, and these areas should be, where possible, large areas due to the increased conservation potential, enhanced recruitment and ease of management of larger areas - see Section 5.5.3. Reserves that allow some kinds of exploitation may be politically more acceptable but ‘no-take’ reserves generally are a more practical option because:

- They are much easier to implement and enforce;
- It is difficult to predict what kinds of “take” will have minimal impacts; and
- ‘No-take’ reserves can also provide reference areas for large-scale experiments in which human impacts on marine ecosystems can be estimated.

Marine reserves must not, however, replace conventional management options but rather complement them. They can provide insurance against stock collapse given the uncertainty in estimating stock status, and a safety margin against overshoots in fishing effort (Roberts 1997).

Illegal fishing is a problem in some protected areas. For example, Gribble et al. (1998) reported considerable levels of illegal prawn trawling in closed areas of the tropical Great Barrier Reef Marine Park and calculated that approximately 47 boats fished illegally on a consistent basis with an estimated yearly total of 3260 days illegal trawling occurring (about 69 days illegal trawling per regular offender). However, the management of even remote no-go areas are now feasible with GPS technology and a "black box" system installed on fishing trawlers that record or monitor positions during fishing operations. The gradual introduction of Vessel Monitoring Devices (VMD) as proposed on the GBR

[133] http://www.fisheries.nsw.gov.au/conservation/mpas/ar_about.htm.

will make it possible to accurately track boats and record their activities, making management of the industry and protected areas much more effective and efficient.

Management of trawling and dredging

Fishing methods that use trawling and dredging are undoubtedly the single most serious impact on sublittoral benthic communities. Approaches that can be used to reduce these impacts include:

- Equipment modification. Recent advances in more environmentally "friendly" trawl gear in Australia are discussed by Eayers et al. (1997);
- Restricting areas available for trawling - the provision of "no go" areas;
- Developing alternative harvesting methods - e.g., aquaculture, traps; and
- Better policing to reduce illegal trawling.

The impacts of trawling and dredging on the benthos are considerable and have affected most of the continental shelves around the world. While we strongly support the idea of refugia as an important long-term option, the reality is that most of the accessible sea floor is already seriously damaged and most of it will continue to be heavily fished. To minimise future damage, and allow some recovery of benthic invertebrate communities, there must be regulations that enforce the use of gear that does not drag across the bottom, or at least minimises this impact. Although the benthic invertebrate communities will never recover fully to pre-trawling status whilst fishing continues, a change in fishing practices could allow the partial recovery of these populations and hence the food supplies for the fish being targeted.

Given the enormous damage that scallop dredging does (see Section 6.3.2), we recommend that this method be banned from all but a few strictly controlled areas, especially as scallop aquaculture is now a viable source of these molluscs.

Another management tool would be to require Environmental Impact Statements (EIS) for commercial fishing operations (see Section 8.2.3 for a recent case where this was required).

Recreational and minor commercial or semi-commercial fisheries

Intertidal collecting

Intertidal collecting is mainly for bait, food, specimens or ornaments. Underwood (1993b) briefly reviewed the possible policies for management and conservation of intertidal assemblages on rocky shores, including:

- Use of general or selective bag limits, or size limits;
- Bans on harvesting for food and bait on the whole coast; or
- Bans on harvesting in selected patches of habitat (this being considered to be the only realistic option)

He also identified a series of measures necessary for ensuring the success of such policies, including:

- Policing – without proper, patrolled policing of closed areas, bans on harvesting will be ineffective;
- Identification of sites in need of protection – through use of criteria such as the perceived threat, proximity to urban centres, current use, possibility of effective policing, and representativeness of habitats;
- Public education – including signs at closed areas, advertising, agreed penalties etc.
- Evaluation of the effectiveness of the policy, in the form of scientific experiments.

Bag limits and size limits on some species have been in place for a number of years in different jurisdictions, but many of these have been inappropriate (e.g., in South Australia, 1 for the Black Cowry, *Zoila friendii thersites*, 200 for scallops (Mattei and Pellizzato 1996; J. Brook pers. comm.) and in other cases, bag limits are non-existent. Many older bag-limits were actually volume (e.g., bucket) based and proved ineffective. The response in some jurisdictions (e.g., NSW) has been to develop a variety of strategies, to protect habitats (by banning certain collection techniques that damage them), protection of whole communities (declaration of intertidal protected areas) and protection of heavily harvested species (by imposition of species size and bag limits) (NSW Fisheries 1998a). However in many places these restrictions have been ineffective due to language problems, lack of enforcement and lack of community awareness, especially amongst newly arrived migrants. The most critical issue in ensuring the effectiveness of any management strategy is effective enforcement, and its lack is the most common reason for failure.

The current method of collecting beach worms individually (for bait) appears to be sustainable (see under 6.10.3 above) and any trials of mass collection should be discouraged. We recommend that action be taken to restrict the collection of beachworms to the present method.

Shell collecting / shell trade

The management of the specimen shell industry is the responsibility of the various State fisheries agencies under the *Wildlife Protection (Regulation of Exports and Imports) Act 1982*, which requires that the fishery be conducted under Management Plans enacted in each State and Territory. To date only Western Australia and Queensland have management plans in place. A draft management plan has been available in NSW for more than two years but has not yet been released for comment.

Because a very low priority is given to this "fishery" by most of the state/territory agencies (some do not have draft management plans), there is unlikely to be a significant allocation of resources to this issue. Thus these management plans need to be workable and practically enforceable. There is general acknowledgment that shell collecting is a popular and generally harmless occupation and should be allowed to occur without undue restriction, while at the same time protecting species and habitats recognised as being vulnerable. An option favoured by the Malacological Society of Australasia is to licence collectors and impose a code of ethics. Bag limits should be placed on the species targeted by collectors as specimen shells or for food.

Separation of live and dead collected specimens has been identified as an issue in some management plans. Willan (1986) detailed the ways in which live collected and beach collected specimens can be differentiated, providing a potential means of settling any disputes. In Western Australia, however, the definition of specimen shells includes empty beach shells. Ponder and Grayson (1998) argue that there is little, if any, justification in preventing or limiting the collection of empty shells from beaches. The implementation of such a definition could potentially make any child or tourist picking up a few shells on the beach liable to be fined for breaking the law. This is the case, for example, on the GBR where it is illegal to collect shells or corals dead or alive.

Another management option (for example in the Queensland Management Plan) involves banning the sale of bycatch obtained from fishing trawlers. Ponder and Grayson (1998) argued that by excluding using bycatch, the legal source of most of the shelf taxa will disappear, driving the prices up and the market underground.

Some reports on the shell trade have recommended banning some species from trade (e.g., Willan 1986). While Ponder and Grayson (1998)[134] argued that there is no immediate need to ban any species from export, there was a clear need to have management plans in place for targeted, vulnerable species. Of these, some members of the families Cypraeidae (cowries) and Volutidae (volutes, including the baler shells) are most at risk. The Australian volutes are all direct developers, as are several endemic cypraeid taxa but there is no evidence to suggest that all volute species are at risk and many species of cowry are widespread and very common.

Licensing of commercial specimen shell operators is a management option that is used in some states - for example, there are five licenses for people to take specimen shells (about 600 shells) from the GBR.

Information and data needs

The data available to Ponder and Grayson (1998) were less than adequate with 17.1% of names, comprising more than half the total specimens exported, unable to be allocated to a valid Australian species. The listing by states (the only locality information recorded in the export forms) has little or no relationship to the actual source of the material but is related to the location of the exporter. In addition, there is a general lack of basic information about the biology and ecology of the key species of specimen shells. These same problems also relate to other exported invertebrate fauna.

Davey (1993) also noted the problem of misidentifications, either deliberate (to avoid management implementation) or accidental. She noted that Customs do random checks on shipments, but only for species listed on CITES. Ponder and Grayson (1998) recommended that effective reporting should be encouraged by a few random checks on parcels containing shells for export. These spot checks should include a check for clearly misreported material involving:

- Identification check to prevent threatened species being passed off as some other taxon;

[134] http://www.ea.gov.au/biodiversity/trade-use/wild-harvest/pubs/pondrpt1.pdf

- Source of the material (grossly misreported sources can be easily identified); and
- Number of specimens.

Such checks could be minimal (a few each year), but given the volume of the trade this should be sufficient incentive to encourage a better standard of reporting.

Ponder and Grayson (1998) also recommended that better information regarding potentially vulnerable species be made available to the public and fisheries inspectors and managers. They suggested that this could be done using a variety of strategies; at the state level, pamphlets illustrating the taxa involved could be a viable option; at the national level, a CD ROM, web facility or booklet would be desirable and readily achievable. These facilities should include colour illustrations, key identification characters and a distribution map of each taxon listed.

Other exported invertebrates
There are moderate to small exports in marine invertebrates other than the main commercial species. These include various holothurians (some are not listed as commercial species and require an export licence), corals, seastars and various molluscs. The data on these exports have not been analysed in any useful way other than for the shell trade and presumably will present similar problems.

6.11 Aquaculture

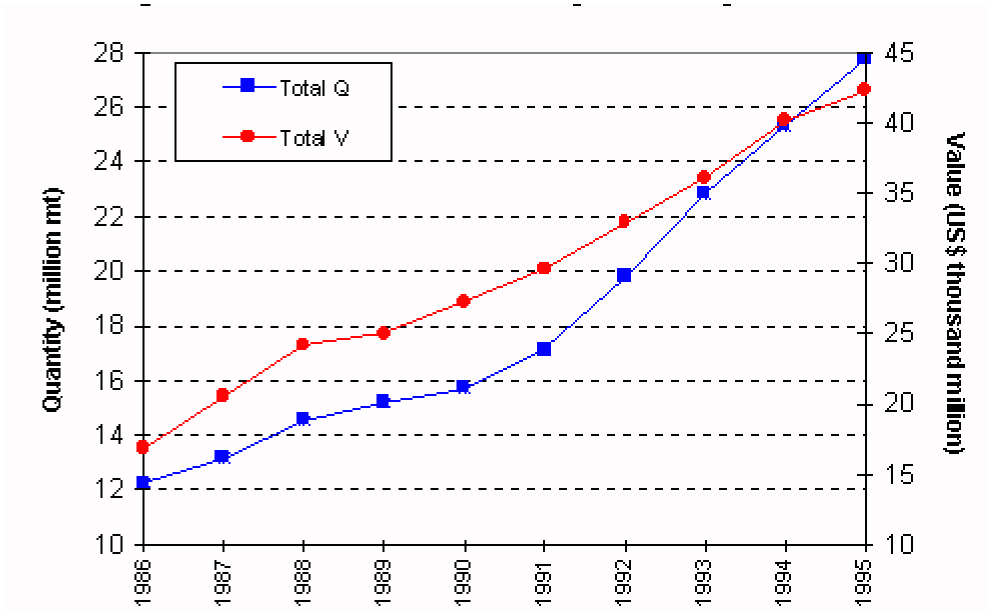
Marine aquaculture[135] (mariculture) involves the farming of seaweeds, shellfish, or fish. There are two major culture techniques; intensive and extensive. The term `intensive aquaculture' is generally used to describe projects where stock is confined (usually at high density) and artificially fed; this is the case for most fish farming. 'Extensive aquaculture' is the term used where stock is not closely confined and naturally available food is used; this describes most shellfish farming (NSW Fisheries 1998a). Aquaculture has been growing rapidly worldwide and in Australia (see Figure 6.4).

6.11.1 Effects of aquaculture

Aquaculture has the potential to impact on the marine environment in a number of ways, depending largely on the species, culture method, stocking density, feed type, hydrography of the site and husbandry practices (Wu 1995). These impacts can include habitat damage during construction, and pollution, introduction of non-native species and diseases, consumption of natural resources and ecosystem changes during operation. The potential impacts of extensive aquaculture are quite limited given that food supplements are not used during the production process, though it is important to minimise any potential impacts which may occur during the construction or operation phase (NSW Fisheries 1998a; see below). The implications of aquaculture for biodiversity, particularly in relation to the threat posed by wetland destruction, use of chemotherapeutants and translocation of exotic plants and animals, are discussed by Beveridge et al. (1997).

[135] See Bell (1995) for an overview of aquaculture in temperate Australia.

Figure 6.4: Global trends in aquaculture production, 1986 – 1995 (source: FAO Fisheries Department 1997)



Habitat destruction during construction

Clearing of wetlands – particularly mangroves – for the construction of aquaculture facilities is an important issue in some parts of the world, particularly in Southeast Asia where aquaculture production is increasing rapidly and severely impacting coastal areas such as mangrove habitats (e.g., Ong 1995). While generally less significant in Australia due to the lesser prevalence of aquculture here, it can be locally very significant in areas where new facilities are being developed. For instance, prawn culture is an expanding industry along the coasts of Queensland. The total farm ponded area increased by 9 % from 600 hectares in 1999/00 to 657 hectares in 2000/01 and the number of farms increased from 32 to 33. This increase in area coincided with a drop in the value of marine prawn production in Queensland from $45.2 million to $44.6 million in 2000/01[136].

Pollution

One of the most frequently cited problems associated with aquaculture is pollution of the local environment, through the release of wastes comprising uneaten food, faecal and urinary products, chemicals and chemotherapeutants. For instance, Wu (1995) calculated that in general some 85% of phosphorus, 80-88% of carbon and 52-95% of nitrogen input into a marine fish culture system may be lost into the environment through feed wastage, fish excretion, faeces production and respiration. Cleaning of fouled cages may also add

[136] Information from www.dpi.qld.gov.au/fishweb/9306.html.

an organic loading to the water, albeit periodically, and the use of chemicals (therapeutants, vitamins and antifoulants) has also raised environmental concerns. The major impact is on the sea bottom, where high organic loading can result in high sediment oxygen demand, anoxic sediments, production of toxic gases elimination of benthic organisms (Wu 1995; Lu and Wu 1998). TBT contamination and the development of antibiotic-resistant bacteria have also been reported near farms. Nonetheless, Wu (1995) argued that the significant impact is normally confined to within 1 km of the farm. In addition, a study (Lu and Wu 1998) on the recolonisation and succession of marine macrobenthos in organic-enriched sediment deposited from fish farms suggested it was unlikely there would be a long-term impact once farming activities ceased. Recolonisation occurred rapidly and temporal changes over subsequent months resembled the spatial changes seen in benthic communities along a decreasing pollution gradient (e.g., Pearson and Rosenberg 1978).

Introduction of exotic organisms and pathogens

The introduction of pathogens and exotic species through transport of aquaculture stock, and mixing of genetic strains as a result of movement of spat or stock from place to place, has also raised concerns. For instance, the Sydney Rock Oyster is moved between estuaries, altering the distribution of strains of wild populations of the species. Another species of oyster was introduced to Tasmania from New Zealand between 1885 and 1930 and carried with them several exotic species that have since become established. These include mud worms (*Polydora* spp.) and several species of mollusc (Dartnall 1969; Carlton 1992; Furlani 1996; R. Willan pers. comm.). Similarly, the slipper limpet *Crepidula fornicata* was introduced into Southern England by the shipment of oysters from the USA to the UK (Fretter and Graham 1962). In French Polynesia there are regulations restricting movement of the Black Lipped Pearl Oyster between islands, but these are not enforced (Hutchings and Salvat 2000).

Alteration of local habitats and communities

The clearing required and the building of cages, stakes, racks etc. used in aquaculture facilities can lead to significant alterations to local habitats and ecosystems, including major alteration of habitat or more subtle effects such as shading and changes to hydrography. For example, an experimental investigation of the effects of oyster aquaculture in a northwest Pacific estuary in the US (Everett et al. 1995) found that both stake and rack methods resulted in significant (>75%) decreases in submerged aquatic vegetation (SAV), *Zostera marina*, after 1 year of culture, and complete loss from rack treatments after 17 months. The stake plots caused significantly greater sediment deposition, and this, combined with direct physical disturbance during placement and harvest, was probably responsible for the effect on SAV. The effects of rack plots appeared to be related to greater erosion and perhaps shading. Similar impacts are likely from oyster cultivation in eastern Australia.

The Australian pearl industry produces about a quarter of the world production and generates exports valued at more than $200 million annually and employs approximately

1000 people. The industry is a quota-based fishery, cooperatively managed by Government and Industry to maintain the resource. Much of the activity is located around Broome where licenced companies fish for wild pearl oyster stocks for their shell supply. The total combined quota is 572,000 shells per annum. Individual company quotas range from 15,000 shells up to 330,000 shells[137].

In Queensland, the value of the pearl oyster *(Pinctada spp)* industry decreased by over 37 % from $781 000 in the 1999/00 season to $489 000 in 2000/01. The main species cultured is the gold lip oyster *(Pinctada maxim*a). Other species produced in small quantities included black lip oyster *(P. margaritifera,)* gem pearl oyster *(P. radiata)* and penguin oyster *(Pteria penguin*)[138].

Dense pearl culture in closed lagoons can result in changes to ecosystems, through removal of planktonic food, which is then not available to other organisms (Souchu et al. 2001).

Ecosystem impacts

While most discussion of the impacts of aquaculture has focused on relatively localised and direct impacts such as pollution and degradation of local habitat, intensive aquaculture may also have more indirect but widespread effects through the consumption of natural resources for feed or base stock. Many farmed fish, for example, are fed with products derived from wild-caught fish or shrimp, and many forms of aquaculture rely on collection of wild spat, juveniles or adults for culture or brood stock (e.g., oysters for pearl culture, blue fin tuna caught as juveniles and grown to adult size). Imported food stock (as with the southern Australian Tuna industry) can potentially be a means of introducing new diseases[139]. Folke et al. (1998) calculated that intensive fish farming is dependent, for its daily survival, on marine ecosystem areas as large as 10 000-50 000 times the area of the cages for producing its food and 100-200 times for processing parts of its waste.

Cultures of algae or filter feeding shellfish are not constrained by the same requirement for food supplementation, but its expansion may be limited by deterioration in environmental quality. For example, the Sydney Rock Oyster industry (the most valuable form of aquaculture in NSW) has declined steadily since 1979, mostly in relation to a decline in estuarine water quality (NSW Fisheries 1998a) as well as competition from the introduced Pacific Oysters.

Oceanic enrichment

Aquaculture is carried out in coastal areas with relatively localised impacts. Of considerable concern are several proposals being aired at present to initiate large-scale

[137] http://www.costellos.com.au/pearls/industry.html
[138] Information from www.dpi.qld.gov.au/fishweb/9306.html.
[139] Such as the pilchard disease in southern Australia in 1995 and 1998. See http://www.frdc.com.au/pub/reports/files/99-227.htm.

oceanic enrichment experiments involving stimulation of zooplankton production (and hence, in theory, biomass of higher trophic organisms) by artificially augmenting the nutrient status of ocean waters. These proposals range from ocean "ranches" using artificially upwelled sub-surface waters (see Matsuda et al. 1999) to methods of "fertilising the oceans" (Schueller 1999 - see Section 6.6.1). The lack of consideration given to the broader ecological and environmental consequences of such proposals is astounding and highlights how little we know and care about these "out of sight, out of mind" ecosystems.

6.11.2 Management issues and recommendations

Issues

Aquaculture, while reducing pressure on wild stocks in some cases, has environmental problems of its own. These include:

- Destruction or alienation of habitat;
- Pollution;
- Introduction of pests - quarantine procedures - possible problems even within states let alone between states; and
- Mixing of genetic stocks - genetics of existing populations of the species involved should be surveyed so that management options can be assessed based on actual data.

Management options

Buschmann et al. (1996) pointed out that regulations to protect the environment from the impacts of aquaculture, such as have been adopted in Chile in the last few years, can only be effective if other human activities, such as urban discharge, intensive agriculture fertilisation and pesticide utilisation, are taken into consideration, in an integrated perspective. Active research is currently being undertaken into new cultivation strategies, such as the use of integrated cultivation and the recycling of nutrient-rich waters, which should permit the diversification of this economic activity in Chile, while minimising the environmental impact. Wu (1995) discussed management options to ensure that aquaculture is sustainable, particularly in relation to pollution impacts. These included:

- ***Keeping stocking density and pollution loadings under environmental capacity.*** Carrying capacity depends on tidal flushing, current and assimilative capacity of the water body to pollutants. e.g., assuming that dissolved oxygen in seawater is 7 mg O2 l-1, at least 17-57 m3 of fresh seawater would be required to compensate for the oxygen consumption alone of 1 t of culture fish, not to mention the additional oxygen demand exerted by wastes from the farming activities. Water quality modelling has been used to estimate the maximum permissible stock.
- ***Improved artificial feed formulation.*** Wastage can be reduced by increasing the stability and reducing the sinking rate of feed, and providing feed of an optimal size and digestibility during each stage of development.
- ***Harvesting pollutants by integrated culture.*** e.g., by growing marine macroalgae to remove nitrogen, and filter feeders (e.g., bivalves) to remove particulate matter / phytoplankton (although care needs to be taken to avoid bacterial contamination if shellfish are to be used for human consumption).

- ***Environmental impact assessment, monitoring and control.*** EIA needs to be applied to all aquaculture development proposals. Effluent standards for land-based fish farms have been well established but it is more difficult to control effluent quality in open-water cage farming systems; the most practical option for managing water quality is to control stocking density. Regular monitoring of sediment and water quality is also necessary.

To these should be added:
- Location of aquaculture activities should be sympathetic and not destroy or impact otherwise important habitat.
- Minimising translocation to reduce mixing of genetic stocks.
- Transport of aquaculture stock needs to be carefully regulated, as well as any seawater used to transport the animals, in accordance with the National Policy for the Translocation of Live Aquatic Organisms[140] to minimise the risk of spreading exotic species.

6.12 Shipping and transport

As discussed in Section 6.5, shipping is the main vector for transport of exotic organisms from one region of the world to another. This industry is also responsible for a significant proportion of marine oil pollution, as well as some other forms of pollution including rubbish and waste disposal. Other impacts specific to the shipping and boating industry include physical damage (especially on coral reefs) due to anchoring and shipwrecks or groundings, and the toxic effects of anti-fouling paints, particularly TBT.

6.12.1 Shipwrecks and groundings

Shipwrecks and grounded vessels can cause local habitat damage but the main threat is usually from fuel or cargo spillage. The resulting contamination from such accidents can be broad scale, especially if a tanker (e.g., Torrey Canyon in the Scilly Isles, UK) or other type of bulk transporter is involved (see Section 6.6.1 for a specific discussion of the impact of oil and related products on marine invertebrates and communities).

On the ‘positive’ side, shipwrecks can form artificial reefs that add to the structural diversity of an area and provide good habitats for fish and invertebrates. In some areas, ships have been deliberately scuttled to provide such habitats.

Examples of groundings

[140] The “National Policy for the Translocation of Live Aquatic Organisms: Issues, Principles and Guidelines for Implementation” (Ministerial Council on Forestry, Fisheries and Aquaculture) was produced in 1999. Translocation occurs for example in the development of aquaculture ventures, the stocking of waterways for sport and recreation, and the release or escape of aquarium organisms. Until recently, assessment of proposals for translocation have been treated in an *ad hoc* way and assessed individually without any guidelines. Requirements vary among States and Territories. These guidelines provide a policy framework and risk assessment process. All State and Territory fisheries agencies have agreed to use it as a basis from which to develop their own policies and guidelines.

In 1995, however, when the bulk carrier *Iron Baron* ran aground on Hebe Reef in Northern Tasmania releasing approximately 350 tonnes of fuel oil, the main impact appeared to be from the grounding itself rather than the oil. Edgar and Barrett (2000) found that physical abrasion from the ship's hull during grounding caused the complete destruction of the subtidal reef community over an area of about 3400 m^2, whereas other areas affected only by the oil showed no significant changes. Post impact monitoring showed that plant and invertebrate communities in the grounding area had still not reached pre-disturbance levels after two years, apparently because wave disturbance was hindering reestablishment of large macroalgae over part of the abrasion zone where the reef substrata had been converted to unstable rubble (Edgar and Barrett 2000).

On November 2000, the 184 m Malaysian cargo ship *Bunga Teratai Satu* ran aground on Sudbury Reef within the Great Barrier Reef World Heritage area. The ship was re-floated three days prior to the annual mass spawning of corals. Fortunately none of the ships cargo of fungicides, pesticides, glycerol and polystyrene beads was lost, nor was any fuel. However, the reef sustained extensive structural damage. The impact of the collision caused the reef matrix to shatter and large amounts of coral rubble were generated. During both the initial grounding and subsequent refloating anti-fouling paint was scraped from the ship's hull. Subsequent analysis revealed that tributyltin (TBT), and copper and zinc, in rubble/sediment samples was dispersed up to 250 m from the grounding (GBRMPA 2001, in Negri et al. 2002).

To assess the potential impact of these chemicals on coral larvae, experiments were set up to mimic the concentrations found at the impacted site. At these levels, significant inhibition of larval settlement and metamorphosis occurred. These results indicate that the contamination of sediments by anti-fouling paint at Sudbury Reef has the potential to significantly reduce coral recruitment in the immediate vicinity of the site and that this contamination may threaten the recovery of the resident coral community unless the paint is removed (Negri and Heyward 2000).

In the case of the *Bunga Teratai Satu*, the ship owners were fined $400 000, after pleading guilty to causing serious environmental harm under the Queensland Environmental Protection Act. This was the heaviest fine ever imposed under Queensland environment protection laws. Similar charges against the captain and first mate were dropped. The company had previously agreed to pay clean-up costs of $1 million.

On 6 November, 2000 the Commonwealth Transport Minister ordered a review of measures to promote ship safety and pollution prevention in the waters of the Great Barrier Reef. A steering committee presented an interim report on 15 December. A final report, incorporating a public consultation phase, is due by the end of June 2001. On 15 November, 2000, the Government announced that it would fast track the introduction of a new shore to ship tracking system by mid 2001. The system would mean that licensed coastal pilots would carry portable transponders enabling real-time monitoring of the ship's position while it was under the control of a pilot. However, while certain areas of the GBR require a pilot others do not, and in fact the pilot had just left the *Bunga Teratai Satu* prior to its grounding.

6.12.2 Cruise ships

Cruise ships, as with other shipping, pose a serious threat to marine communities such as coral reefs and associated organisms. The effects of increasing numbers of cruise ships on corals, fisheries and tourism are discussed by Smith (1988). Qualitative and quantitative observations made under cruise ships anchored over disturbed and undisturbed sections of reef off Grand Cayman Island, West Indies showed 3150 m^2 of previously intact reef destroyed by one cruise ship anchoring on one day (Smith 1988). From follow-up observations, and additional data from other cruise ship anchorings, recovery periods of more than fifty years were postulated (Smith 1988).

The number of large cruise ships and dive boats is increasing in Australian waters. In addition to the problems identified by Smith (1988), they may also introduce non-indigenous organisms via either hull fouling or ballast water. Currently only the GBRMPA controls the locations within the Park, where cruise ships may visit and anchor; regulations elsewhere are far less stringent.

6.12.3 Transport of exotics

See Section 6.5.

6.12.4 Antifouling paints

Biocidal antifouling paints are used on vessels to discourage or prevent the settlement and growth of fouling organisms (barnacles, algae, sponges, tubeworms, etc.). Fouling poses a significant problem for commercial and recreational shipping and boating, increasing drag, reducing the efficiency and manoeuvrability of affected ships, and increasing operational costs through higher fuel use and cleaning.

Over the years, a variety of active ingredients have been used in antifouling paints, including organomercury, arsenic and lead. Since the late 1960s, however, the mostly widely used antifouling additive has been *tributyl tin* (TBT), either in conjunction with other agents in copper-based paints or as the sole biocidal agent.

Tributyl tin (TBT)

Unfortunately, the effectiveness of TBT – like other compounds that prevent the settlement and growth of fouling organisms on man-made structures – relies on its toxicity to a variety of marine taxa. TBT enters the marine environment primarily through leaching from hulls during passage and at docks and moorings, although wastes from slipway operations (as a result of paint application, disposal and hull cleaning) can be a localised source of contamination. Like other trace metals and organometals, TBT has the potential to accumulate in sediments and biota. The highest concentrations are found near ports and marinas where shipping densities are greatest, but it has also been found many kilometres from the most likely site of contamination. TBT is especially persistent in estuaries, marinas and similar waters where circulation is poor and flushing irregular, and may take up to 10 years to degrade to safe levels in some sediments (SA EPA 1999).

TBT is almost universally toxic and has been used variously as a fungicide, bactericide, insecticide and wood preservative (Evans et al. 1995). Concentrations as low as 1 ng TBT per litre of seawater have been found to affect organisms from a wide variety of marine phyla (Rexrode 1987; Langston et al. 1990; Wester et al. 1990; Kelly et al. 1990a; Kelly et al. 1990b; Kohn et al. 1999; Carnevali et al. 2001; Meador and Rice 2001), including microalgae (Beaumont and Newman 1986). Major documented biological and environmental impacts of TBT contamination include (SA EPA 1999):

- Bioaccumulation in some species, particularly deposit feeders and filter feeders. For instance, some molluscs have been found to contain levels up to 250,000 times higher than surrounding sediment or seawater, causing shell deformities, reduced growth rates and impaired reproduction.
- Biomagnification up the food chain, from primary producers (bacteria and algae) through to top consumers. TBT can become concentrated in large predators (e.g., sharks, seals and dolphins), and has been linked with the deaths of some of these.
- Health effects in humans – TBT contamination may inhibit liver and spleen function and suppress immune function. This is of concern given that TBT has been found in marketed fish and crustaceans.

TBT has also been shown to inhibit fertilisation and larval metamorphosis in corals (Negri and Heyward 2000) and to impair the ability of heterotrophic microbes in sediments to degrade organic matter (even minute amounts, below current detection limits, affected this process; Dahllöf et al. 1999).

By far the mostly widely documented effect of TBT, however, is “imposex”, or *pseudohermaphroditism*, in which female gastropod molluscs develop male features. In advanced stages this can block the genital pore leading effectively to female sterility (Kohn et al. 1999).

***Pseudohermaphroditism*:** First reported by Blaber (1970) for the marine gastropod *Nucella lapillus* (L.), imposex is now a worldwide problem affecting many species. Fioroni et al. (1991) and Oehlmann et al. (1991) listed reports of imposex in 72 gastropod species from 49 genera, and more recent reports have added additional species. The resultant reproductive failure has caused substantial population declines in some gastropods (Kohn et al. 1999), notably in *N. lapillus* in the United Kingdom and along the European Atlantic coast (Bryan et al. 1987) and in the oyster drill *Urosalpinx cinerea* in England (Gibbs et al. 1991). The community consequences of the TBT-induced decline in the dogwhelk, resulting in part from a reduction in predation, were discussed by Spence et al. (1990). One of the first published reports of imposex in gastropods in Australia was by Kohn and Almasi (1993), and subsequent studies on sediments and biota have found extremely high levels of TBT contamination and frequency of imposex in some locations (e.g., up to 1350 ng TBT l^{-1} in Cockburn Sound, Fremantle Harbour, and Thomson Bay, Rottnest Island (see Kohn et al. 1999 for references). Wilson et al.

(1993) examined imposex in neogastropods along the NSW coast as an indicator of TBT contamination and found high levels in many locations.

6.12.5 Management issues and recommendations

Shipwrecks and groundings

Regulation that increases safety in shipping (e.g., introduction of double hulls for tankers so that in the event of grounding they are less likely to be holed; increasing the use of pilots - currently in force for parts of the GBR- but perhaps needed elsewhere) may help to reduce some of the impacts from shipping accidents. However, these measures may be offset by the increasing volume of shipping and the widespread use of "flags of convenience".

The minimisation of oil spills during groundings of tankers can be achieved by the use of double hulled vessels, as now being advocated by some jurisdictions.

Transport of exotics

Ballast water
While Australia was one of the first countries to bring in voluntary mid ocean exchange of ballast water, the effectiveness of this treatment is unknown. AQIS, the Federal agency that polices this practice, relies on ship logbooks, as there is no easy way of checking that the water in the ballast water tanks is really of oceanic origin.

A number of possibilities for the treatment of ballast water are being explored (O'Hara 1999b); these include heat treatment (to 37-41°C - sufficient to kill most larvae, spores and cysts if prolonged – although these temperatures also tend to promote the growth of bacteria (e.g., cholera). Other options include a combination of UV light and filtration (expensive) or port-based filtration plants to purify ballast water. Chemical solutions are expensive and environmentally damaging. The management option in use today uses the environmental differences between the ballast water and destination port to minimise risk of introductions, e.g., by discharging freshwater into salt or vice-versa, tropical into colder water, or in mid ocean, etc. but this may not be effective for species with broad physiological tolerances. Flow-through exchange, where tanks are filled and emptied at the same time, is safer for most ships, although three tank volumes need to be exchanged to remove 95% of the original organisms (Rigby and Hallegraaef 1993).

A toxic dinoflagellate bloom in New Zealand, which closed their shellfish industry, resulted in a voluntary agreement between Australian and New Zealand ships that no ballast water would be taken on board in New Zealand waters or discharged into Australian waters. This agreement may have prevented the bloom from being introduced into Australian waters (O'Hara 1999b).

Hull fouling, sea chests

Despite the emphasis on ballast water management, this is only one method by which species are introduced into Australian waters by shipping. Historically up to 60% of our introduced marine species have arrived here on the hulls or in the sea chests of ships, or as part of the live seafood trade (e.g., Striped-Black False Mussel, sabellid Fan Worm, Pacific Oyster, even Northern Pacific Seastar). We have made little progress on these issues and research on hull fouling or aquaculture vectors is *ad hoc*. With increasing oil prices ships tend to be cleaned regularly so that maximum speeds may be obtained. In Australia, any such cleaning should occur in dry dock with scrapings prevented from being washed into the marine environment.

While a cruising yacht may have introduced the Striped-Black False Mussel into a marina in Darwin Harbour, Willan (pers. comm.) suggests that boats from Java are a more likely agent. A process of public education and public awareness is required as vessels are often cleaned outside marinas and enforcement of protocols for private vessels is difficult. With slipways being closed in the Sydney Harbour region as they fail to satisfy EPA regulations for retention of scrapings, illegal cleaning is set to increase. There are also reports of large tankers in Botany Bay being cleaned on the moorings by divers while waiting to unload, with the scrapings dropping onto the seafloor. There are similar issues with fishing boats, or for floating rigs etc. associated with the gas and oil exploration industries.

Following the Black-Striped False Mussel incident (see above), marinas in Darwin are planning to instigate a system whereby boats entering the marina are checked for hull fouling. With the increase of yachts for cruising, including the large super yachts, the potential for new routes of introduction is increasing.

To date the emphasis on managing the introduction of pests via shipping has been largely on commercial vessels, but needs to be expanded to include yachts (Floerl 2001), fishing boats, offshore platforms and naval shipping.

Antifouling paints

TBT (see 6.12.4) continues to be used on larger vessels and alternatives are urgently needed. Continued research into low impact antifoulants is should be a priority for the shipping industry.

Tributyl Tin (TBT)

Worldwide recognition of the significance of the environmental problems caused by TBT has led to the introduction of legislative controls in a number of countries, including France (1982), UK (1987), USA (1988), New Zealand (1988), Australia (1989) and Norway (1989) (Evans et al. 1995). In Australia, most states have implemented a recommended ban on the use of TBT-based paints on small craft (less than 25m in length) (State of the Environment Advisory Council 1996). Across Australia, the principal control measure on larger boats is to restrict the release rate of TBT from marine coatings to below 5 micrograms per square centimetre per day (by standard test rate). Many states have other controls on vessel size and hull material. States also licence shipyards and slipways to cover the application and disposal of hull coatings (SA EPA

1999). These restrictions appear to have had an effect in reducing background contamination in many coastal rivers and harbours (e.g., Evans et al. 1995; State of the Environment Advisory Council 1996; Evans 1999; Kohn et al. 1999). Nonetheless, both the Australian Government and the International Maritime Organisation have proposed total bans on the use of TBT, the latter agreeing in November 1998 to ban the use of TBT in antifoulants by 2003, with a 5-year phasing-out period. Both Japan and New Zealand have already implemented a complete ban on the use of TBT in their waters.

Alternatives. Despite the clear need to reduce the impact from TBT contamination, the imminent global ban has raised some serious concerns, relating mainly to the fact that there are currently – despite considerable research effort – no generally acceptable alternatives. This lack of alternatives has led some authors to question the sagacity of the decision to enforce a global ban on the use of TBT (e.g., Lewis 1999), and to argue that it is premature and should at least be delayed until a suitable alternative product becomes available (Evans 1999). While concerns expressed by the shipping industry are based primarily on the consequences for the economic and operational performance of vessels, other issues – principally the problem of minimising the introduction of exotic marine pests via hull fouling – are also of concern.

A serious problem is that existing toxic alternatives to TBT, such as copper oxide paints and triazine herbicides, have similarly unacceptable environmental impacts including accumulation in marine systems and a range of effects on marine biota. Medical antibiotics, such as tetracyclines, are also being used in some 'home brew' marine protective coatings, but are not suitable alternatives because they promote antibiotic resistance in bacteria (SA EPA 1999). Since the restrictions on TBT, there has been an increase in the number of formulations containing 'booster' herbicides (e.g., Irgarol 1051, not registered for use in Australia but found in high concentrations in seagrasses off the east coast of Queensland; Scarlett et al. 1999). However, herbicide formulas are particularly detrimental to mangroves and seagrasses (Haynes et al. 2000), adversely affect the endosymbiotic algae in corals, and may have adverse consequences for long-lived herbivores such as the dugong.

Current research is focusing on less toxic alternatives, such as foulant-release ('non-stick') coatings, ceramic-epoxy coatings, epoxy-copper flake paints, electrical current systems and natural antifouling compounds (produced by marine algae and invertebrates to avoid becoming fouled themselves). However, development problems include the cost to secure registration for natural biocides and the amount of time needed to properly test products for efficacy and non-toxicity; in addition, many potential products are costly and / or unsuited to particular hull types or conditions.
Some alternatives include (from SA EPA 1999):

- Foulant-Release ('non-stick') Coatings. Silicon elastomers are effective non-stick antifoulants, with similar service life to TBT copolymer, but are costly and best suited to fast craft.
- Ceramic-Epoxy Coatings. They can provide an effective barrier, and can be used on fibreglass and metal hulls.

- Epoxy-Copper Flake Paints. A bonded-copper system which may protect boats for up to 10 years but cannot be used on aluminium hulls.
- Electrical Current Systems. These are based upon the generation of hyperchlorite from seawater on the surface of the structure and can sterilise the surface for up to four years. Another similar system uses alternating current and a conductive hull coating.
- Biological compounds. Marine algae and invertebrates do not become fouled themselves because they secrete chemicals, many with anti-bacterial, anti-fouling, and anti-algal properties. Some of these are currently being tested around the world. At least 50 natural substances have been identified as potentially useful antifoulants. Australia is prominent in this research.

Sea tests are under way in Europe to determine which method offers the most effective antifouling action with least environmental damage. The Australian and New Zealand Environment and Conservation Council (ANZECC) prepared a 'code of practice' that sets out best practice in the application, use, removal and disposal of antifouling paints to minimise adverse environmental impacts.

6.13 Petroleum, gas or mineral exploration and production

There are now many Acts, regulations, conventions and protocols associated with mining, including the Australian Environmental Protection Agency's series on "best practice environmental management in mining". Exploitation of the presumably vast mineral and oil resources on the ocean floor is in its infancy at present. Even so offshore petroleum is worth approximately $8 billion per annum and supplies 85% of Australia's petroleum needs and contributes $2.4 billion in tax revenue each year (Commonwealth of Australia 1998b). There is great potential for expansion of this industry in the future given dwindling land-based reserves and improving technology. Given our rudimentary state of knowledge about even the most fundamental issues regarding offshore ecosystems the potential for environmental damage is virtually impossible to estimate (e.g., Moore and Zeidner 1999). The ocean also has valuable mineral deposits, many of which occur in the deep ocean or on underwater volcanoes and ocean ridges, or on the sides of seamounts. These latter habitats are likely to harbour endemic taxa (see Section 5.3.1). Similarly, recent press reports regarding proposed mining operations on deep-sea "smokers" near New Ireland would be endangering a unique community of organisms restricted to this habitat.

The issues relating to oil (and other hydrocarbon) pollution are discussed in Section 6.6.1.

6.13.1 Oil and gas

Impacts associated with construction of drilling platforms and other infrastructure (damage to habitats and communities) and toxic and smothering effects of drilling fluids, wastes and leaked oil products have been long recognised (e.g., Davies et al. 1984). Lissner et al. (1991) point out the potential impacts on deep-water hard-substrate communities, primarily from anchor damage and sedimentation. Several studies have shown that drilling fluids have a detrimental impact on survival and recruitment of marine invertebrates, the effects mostly impacting benthic organisms as these wastes tend to settle rapidly (e.g., Anonymous 1995b; Raimondi et al. 1997; Cranford et al. 1999).

R. F. May (1992b) examined the issues resulting from petroleum exploration and development and oil spills on marine conservation reserves in Western Australia while Pendoley (1992) detected hydrocarbons in oysters and sediments in the remote Rowley Shelf.

Northwest Shelf Gas Project

The Northwest Shelf Gas Project is the largest single resource development project in Australia and extensive efforts have been made to limit environmental impacts (Wright and Nunn 1996; Burgman and Lindenmayer 1998; Heyward et al. 1999; Sherwood et al. 1999). Natural gas and liquid condensate is extracted using drilling platforms located 130 km off the coast and the gas is piped to treatment plants near the towns of Karratha and Dampier (Wright and Nunn 1996). The region supports coral reefs and mangrove forests, and the islands of the Dampier Archipelago have important conservation value. Thus, the minimisation of environmental effects was considered an important part of the Project and a number of measures were instigated, at a cost (to 1996) of about $20 million. Most of these measures focus on mitigating the impacts at the onshore processing plant, but a few are relevant to the marine environment (Wright and Nunn 1996):

- Treatment of effluents and runoff from the gas processing plant before they are discharged into the ocean;
- Long-term monitoring of heavy metals in the marine environment; and
- Seabed condition and sediment drift surveys of the areas around the offshore drilling platforms.

Effects on corals from dredging and spoil-dumping activities were found to be restricted to a distance of 1.5 km from the nearest point of operations, while assessment of environmental impacts (e.g., sediment hydrocarbon and metal levels) from the offshore operations showed that affected sediments occurred up to several kilometres from the site in the direction of prevailing currents (Wright and Nunn 1996).

Although extensive environmental assessments have been conducted (e.g., Hubbert et al. 1994; Kinhill Engineers Pty Ltd 1997; Phatak and Palmer 1998), relatively little detailed information on the marine invertebrates communities have been provided. A recent marine biological workshop in July/August 2000 organised by the WA Museum, is part of an overall survey of the area by the WAM (Wells et al. in press).

Bass Strait

Bass Strait has been Australia's premier oil and gas production area for more than 30 years, supplying more than three billion barrels of oil and over three trillion cubic feet of gas. These fields are operated via off shore platforms. To date, there have been remarkably few incidents. A serious oil spill in 1999 was from a ship and had nothing to do with the oil and gas operations.

6.13.2 Mineral exploration and mining

Marine mining for minerals is still in its infancy but is likely to increase in significance as terrestrial sources dwindle and technology improves.

Deep-sea mining

The deep-sea contains extensive deposits of commercially important minerals, including ferro-manganese nodules (rich in Co, Ni and Mn) found on the abyssal plains of all oceans, metalliferous sediments (rich in Zn, Cu, Au, Ag, Co) and massive sulphide deposits (rich in Zn, Cu and Fe) found along the mid-ocean ridges (Gage and Tyler 1991; Newton 1999).

It is of considerable concern that the disturbance created by mining activities may profoundly modify the ecology of these delicate environments. The impacts of these disturbances have been considered to be serious (e.g., Thiel and Schriever 1990; Gage and Tyler 1991; Newton 1999) and include removal of sediment and fauna and discharge from tailings into the water column. The suspended material is likely to smother animals, interfere with physiological activities and change microbial and chemical activity in the lower water column (Gage and Tyler 1991). According to Gage and Tyler (1991) removal of 4000 tonnes of sediment is required to gather 1000 tonnes of nodules. Given the low rate of recolonisation and slow growth rates in the deep-sea (see Section 5.3.1) recovery of these habitats would occur over a very long time period indeed. Massive sulphide deposits develop as a result of hydrothermal activity at deep-sea hot vents and seeps. These habitats are also biodiversity hotspots with large and unique faunas associated with them. While we deplore the idea of mining these habitats, none are known within the Australian EEZ.

Sand dredging

Poiner and Kennedy (1984) studied the impact of dredging for fine sand on marine benthos of a subtropical, sublittoral sandbank (Middle Banks, Moreton Bay, Qld). They found that there were significant decreases in richness, abundance and diversity of benthos in the dredged area, and significant increases in richness and abundance in adjacent areas. The distribution and the predicted deposition rates of the sediment plume correlated precisely with the area of enhancement, suggesting a response of the benthic biota to an increase in resources from the deposited plume.

A recent study of the impact of dredging in Botany Bay NSW for the construction of the Third Runway showed that while some of the dredged sites did show some recovery over the length of the study (2 years), others did not recover (Wilson 1998). The sites exhibiting no short-term recovery were deep holes where anaerobic conditions were created.

Newell et al. (1998) undertook a review of the impact of dredging in coastal waters, concentrating on the nature of benthic communities, their sensitivity to disturbance by dredging and land reclamation works, and on the recovery times likely to be required for the re-establishment of community structure following cessation of dredging or spoils disposal. Impacts mainly relate to the physical removal of substratum and associated organisms from the seabed along the path of the dredge, and partly on the impact of subsequent deposition of material rejected by screening and overspill. The impact of dredging to maintain shipping channels will depend upon the frequency of dredging necessary.

Morton (1996) compared subtidal benthic molluscan community in Hong Kong before and after extensive suction dredging for major construction projects and associated commercial trawling. There was approx. a 2/3 decline in numbers of species and individuals of gastropods and bivalves within two km of dredged sites. Most losses in the gastropod fauna were of specialist neogastropod predators, with fauna becoming dominated by opportunistic scavengers. Bivalve fauna post-dredging was dominated by a few species resistant to disturbance, which were of no commercial value and some of which have solid shells (adaptations to avoid predation) resistant to trawl damage. It was postulated that settling silt plumes associated with dredging have exacerbated the problems of a sea-bed already disturbed as a result of trawling and pollution.

6.13.3 Management recommendations

There are already tight controls regulating mining activities in Australia. However, environmental impact statements and environmental considerations prior to approval should:

- Give greater attention to impacts on the benthic and pelagic invertebrate communities likely to be affected.
- Mining exploration and operations should be done sympathetically, with impacts confined to as small an area as possible.
- Dumping of spoil (from dredging for channel deepening etc. or from mining operations), will destroy any marine invertebrate communities in the area of dumping. Such activities should not be undertaken without an environmental impact assessment that takes account of the extent of the habitats being used for dumping, and the impacts on the fauna in those habitats.
- Procedures should be put in place to minimise the impacts of drilling muds (e.g., Burke 1994).

6.14 Recreation and tourism

Over a quarter of Australians live within three kilometres of the sea and three quarters live within 50 kilometres of the coast (Zann 1995). Recreational activities that centre on the marine environment are an important pastime and part of our culture. Popular recreational activities include swimming, snorkelling and diving, sailing and related water sports (surfing, water skiing, windsurfing etc.), fishing and spear fishing, and a variety of intertidal activities such as tide pooling and shell collecting. Areas such as the GBR, Ningaloo Reef and Shark Bay in WA, Jervis Bay in NSW and the South Australian Bight, to name just a few areas, attract considerable numbers of tourists, both domestic and international, and are an important source of revenue.

Recreation and tourism are potentially among the most benign uses of the marine environment, and can in many respects be beneficial, serving to encourage an awareness of and enthusiasm for the marine environment, providing a forum for public environmental education, and supplying an economic motive (or even a means, in cases where user fees are levied) for its protection. Excessive or poorly managed use, however, can have significant impacts (e.g., Sala et al. 1996).

The economic and financial benefits of tourism are well known, and the impacts of visitors on protected areas has been widely recognised as an issue that requires careful management (e.g., Driml and Common 1995). Coral reefs are an especially popular attraction and considerable concern has been voiced over the potential impacts of excessive visitation (particularly from SCUBA diving impacts (Schleyer and Tomalin 2000) or general degradation resulting from tourism-associated development. For instance, Hawkins and Roberts (1997.) pointed out that the declaration of marine protected areas is often partly in response to their amenity value and increasing recreational use; yet their establishment often serves to attract further visitors, exacerbating the problem of over-use. For example, the Ras Mohammed Marine Park in Egypt received wide publicity in diving magazines and the National Geographic, so acting as a focus for the development of a thriving tourist industry based around scuba diving. At high intensities, however, this can cause damage, with recreational diving leading to wear and tear on reefs, pleasure boats increasing water pollution and anchor damage, and the hotels and other infrastructure harming the very resources tourists come to see (Hawkins and Roberts 1997). While coral reefs have received the most attention, many other types of habitats – particularly in the intertidal zone – also have the potential to be impacted by overuse or misuse (see State of the Environment Advisory Council 1996 for a series of case studies on the impact of tourism on the coastal environment).

The sections below deal with the impacts of various recreational and tourism activities on marine invertebrates and their habitats. The effect of recreational collecting of marine organisms is discussed in Section 6.10.3. Indirect effects of tourism, such as those resulting from increased development, damage during construction of tourist facilities, increased pollution loadings, etc., are dealt with in the sections on "Coastal development" (Section 6.16) and "Land use in catchments" (Section 6.17), below.

Intertidal activities

Activities of people on the rocky intertidal zones are diverse and consequently there are different levels of impact on the biota. Tide pooling; collecting for food, aquaria or research; educational field trips; sea side strolling; photographing and fishing are probably the most common activities on rocky shores. Human use of the intertidal zone can be destructive due to trampling, overturning of rocks, and the intensive collection of some species. All activities tend to remove (intentionally or unintentionally) fauna and flora from the seashore. A recent paper by Schiel and Taylor (1999) has studied the effects of trampling on a rocky intertidal algal assemblage in southern New Zealand and while it focuses on the algae, this must have implications for the associated invertebrate communities. Walkers trampling coral reefs and fossickers disturbing habitats have been shown to have serious impacts (Kay and Liddle 1984a, 1984b; Kay and Liddle 1987; Liddle and Kay 1987).

Supralittoral habitats are narrow, often fragile and very readily accessible but are typically neglected in the management of littoral habitats.

Beach cleaning and vehicles

Many Councils regularly use tractors with rakes to clean popular tourist beaches to remove natural debris and human rubbish deposited on the beach. The impact on the infauna is unknown but presumably exposure of the infauna, albeit briefly, must make them more susceptible to predation by birds. In some cases, the distribution of the sediment will be changed which may have implications for the infauna. The specialised fauna associated with drift algae and seagrass is impacted by beach cleaning through the removal of most of the habitat and food of these animals.

The impact of vehicles on the infauna of sandy beaches is unknown. There would certainly be an impact on strand-line taxa and the delicate and narrow band of habitats comprising the supralittoral fringe.

Recreational boating activities

Activites such as recreational sailing and boating, tourist boats, boats for diving and snorkelling etc. cause anchor damage and groundings, as well as pollution from hydrocarbons, sewage and rubbish. The activities also require infrastructures such as the building of moorings; pontoons and marinas. See also "Shipping and transport" - cruise ships (Section 6.12.2).

Dive impacts

The potential impacts of recreational divers have been assessed in underwater surveys at four major dive sites in marine protected areas in Eastern Australia (Heron Is, Lady Elliot Is, Gneering Shoals, Solitary Islands Marine Reserve) by Harriott et al. (1997). Recreational divers were followed for standard periods and all contacts with and damage to the substratum or biota were recorded. The mean number of contacts with the substratum per 30 minute dive at each site ranged from 35 to 121, with a maximum of 304 in a single dive. The majority of contacts were made with fins. Breakage of coral ranged from an average of 0.6 per dive to 1.9 per dive. Most divers damaged no coral, but a small minority of divers broke between 10 and 15 corals each per 30 minute dive. The level of damage to the sites studied appeared to be sustainable at present levels of use by divers. However, at intensively dived, coral-dominated sites, the potential exists for considerable environmental impact.

Sala et al. (1996), working in the Mediterranean, found that coral communities at heavily dived sites were significantly affected by diving, with these sites having fewer and smaller colonies of an erect, foliaceous bryozoan. Similar effects were recorded in Spain by Garrabou et al. (1998). These studies indicate that sublittoral benthic communities in the northwestern Mediterranean constituted by sessile organisms with fragile calcareous or corneous skeletons are not adapted to the disturbances caused by continuous and intense diving and recovery is slow. Hawkins and Roberts (1997) have attempted to estimate the carrying capacity of coral reefs for scuba diving and to use this as a tool for managing tourist development on coral reefs. It implies there is some level of use below which an ecosystem can cope with the amount of disturbance or stress but above which degradation ensues. It can be applied to both direct and indirect consequences of tourism.

Research in the Red Sea and Caribbean suggests that reefs can sustainably support around 5000-6000 dives per dive site per year, but that greater levels of use cause a rapid rise in diver damage. However, carrying capacity can be influenced both positively and negatively by a number of factors. For example educating divers on how to minimise the damage they cause underwater considerably reduces impacts and thus raises the carrying capacity of an area. Carrying capacity will vary according to site characteristics (e.g., sloping bottom vs vertical drop-off) and wise management will incorporate monitoring of damage levels to keep site use within sustainable intensities. In Bonaire, Dixon et al. (1993) developed a threshold model for diver use of reefs based on differences in coral cover and diversity among sites subject to different diving intensities. Although based on different measures to the Hawkins and Roberts (1997) study, they also suggested that there was a threshold of 4000 – 6000 dives per site per year above which diving caused a detrimental shift in the structure of the coral community.

Divers are not equal in the damage they do. Some studies have found that novice divers are more damaging than experienced ones. The former, with their limited underwater experience, tend to have poor buoyancy control and so a tendency to crash into the reef inadvertently breaking coral. Divers with cameras (trying to take photos) also tend to be highly destructive. Medio (1997) has demonstrated conclusively that a short educational talk on the reef environment at the beginning of a diving holiday can dramatically reduce the frequency of damaging interactions with the reef by divers.

Hawkins et al. (1999), in Bonaire (Caribbean), compared coral and fish communities in undived reserves and environmentally similar dive sites where maximum use reached 6000 dives per site per year. They found only minor direct physical damage to reefs; although there were more loose fragments of living coral in dive sites than reserves and more abraded coral in high- than low-use areas. Between 1991-1994 diving intensity increased 70% and coral cover declined at two-thirds of dive sites and in all three reserves, suggesting a background stress unrelated to tourism. However, there was a significant decline in the proportion of old colonies of massive coral species within dive sites (19.2% loss) compared to a smaller loss in reserves (6.7%) and branching corals increased by 8.2% in dive sites compared with 2.2% in reserves. Thus, despite management of these reefs, diving is changing their character. Higher rates of abrasion recorded in dived sites could have rendered corals more susceptible to disease and influenced the decline of the massive corals. The Hawkins et al. (1999) study shows that even relatively low levels of diving can have pronounced effects that may be manifested in shifts in dominance, rather than loss of overall coral cover.

While the above studies were restricted to scuba divers, snorkellers may have a greater impact, especially novices and often snorkellers stand on corals.

6.14.1 Management issues and recommendations

Increasing use of marine protected areas for pursuits such as recreational scuba diving may lead to biological damage and reduced amenity values in popular locations (e.g., Davis and Tisdell 1996). The relationships between biological and amenity values are discussed by Dixon et al. (1993) while Driml (1995) discussed the economic and financial values of the GBR. These authors argue that a blend of regulation and the use of economic instruments will provide better overall management of popular marine recreational sites than is presently the case. Education will also have a significant role to play by increasing environmental awareness and reducing the damaging impacts caused by users.

Limitations on the amount of anchoring, limits on access, requirements for visiting defined sensitive locations, limits on motorised water sports etc. tourism operator permits etc., are some of the ways in which recreational activities likely to damage sensitive sites can be controlled. However, there must also be an awareness of the need to allow users to obtain maximum benefits, as well as limiting their impact on other user groups. While there is a considerable current awareness of the need to limit damage in coral reef habitats, in determining areas at risk more attention needs to be paid to other habitats of significance to marine invertebrates.

Trampling in fragile habitats (particularly coral reefs, but also mangroves and saltmarshes) can be controlled by the designation of pathways or the construction of platforms for viewing, board walks etc., although even boardwalks have some effects (Kelaher et al. 1998a; Kelaher et al. 1998b). The construction of facilities designed to improve access and at the same time minimise impact is an effective management strategy - in that they can also achieve educational, "involvement" and "experience" goals. Devices such as boardwalks in mangroves and artificial pontoons on the reef achieve these goals and, if properly and sympathetically constructed and managed, have few significant impacts. There are, however, some impacts on benthic invertebrate communities in close proximity to the board walk, within distances of 3 m (Kelaher et al. 1998a; Kelaher et al. 1998b). Platforms on GBR reefs are permitted with stringent conditions, and monitoring must occur during the construction and operational phases. The GBR Authority has restricted the number to limit the impact on the “wilderness effect” for tourists, and regulation includes all rubbish and waste products being transported back to the mainland.

Specifically we recommend that:

- More attention be paid to education and public awareness regarding the damage that can be done by diving and snorkelling on coral reefs and other habitats with fragile colonial organisms;
- Trampling be minimised in sensitive habitats in popular areas by the construction of walkways, viewing platforms etc.;
- That damage to habitats important to marine invertebrates be considered when restrictions to various boating activities (including water sports) are being formulated; and
- Research be carried out on the impacts of various recreational activities on marine ecosystems so that this can be used as a basis for management.

6.15 Waste disposal

The sea is increasingly becoming a repository of waste. Its volume and physical properties result in enormous dilution effects and its chemistry and biology give it the capacity to recycle much of the deposited waste. In addition, the remoteness of the deep-sea makes this huge part of the ocean attractive for the disposal of waste that is unsuitable for disposal on land or coastal waters (Gage and Tyler 1991). But increasingly it is being shown that the oceans can disperse wastes and poisons, with, for example, DDT and other pollutants being recorded in Antarctic animals, plankton and seawater (e.g. Lukowski and Ligowski 1987; Joiris and Overloop 1991; Cripps 1992; Gupta et al. 1996; Inomata et al. 1996; Zimmermann 1997).

The oceans are treated as a "sink" for many of the wastes generated by society. These not only include sewage, but many industrial effluents and by-products, stormwater (carrying urban runoff and rubbish), and solid wastes, sewage sludge and, overseas, even include household garbage and radioactive wastes. In some cases these wastes are deliberately disposed of through ocean outfalls or by authorised dumping at sea; in other cases they are discharged into streams and rivers from which they are eventually carried to the sea. Even if sewage is treated, the sludge residues may contain elevated levels of toxic substances such as Cd, Cu, Pb, Hg, Zn or polychlorinated biphenols (Gage and Tyler 1991).

In the absence of any global policy to reduce the production of waste, the pressures on the terrestrial and inshore marine habitats are considerable and the use of the deep-sea as a repository for often dangerous waste is growing. Gage and Tyler (1991) discuss the problem of waste disposal in the deep-sea. They point out that on a larger scale (i.e. excluding shipwrecks etc) the proximal deep ocean is being used to dispose of dredge spoil, sewage sludge, pharmaceuticals and industrial wastes, and low level radioactive waste. The disposal of waste into the marine environment is controlled by the 'Convention on the Prevention of Marine Pollution from Dumping of Wastes and other Matter', better known as the 'London Dumping Convention', which has been in force since 1975 and provides a legal framework (involving 57 countries) to regulate dumping at sea. Although the deep-sea is considered remote from human activity, and therefore safe as a dumping ground, toxic effects such as decrease in oceanic productivity and biomagnification of toxic elements in the food chain could occur.

6.16 Coastal development

6.16.1 Impacts from coastal development

The impacts of coastal development are obvious and well documented (e.g., RAC 1993), these being responsible for the destruction or modification of coastal marine invertebrate habitats and communities, and its combined impacts are probably the most significant cause of the deterioration in coastal marine ecosystems. Almost 90% of Australia's population lives in the coastal zone, about 90% of all building activity in Australia (1983-1991) took place there and it experienced very rapid rates of population growth (Burgman and Lindenmayer 1998), with the continuing development of a largely continuous, heavily urbanised ribbon along the coast from southern NSW to Central Queensland[141].

Coastal development can result in several threatening processes (e.g., habitat destruction (Section 6.4), pollution (Section 6.6), etc.). It includes urban and industrial development, and can involve activities that directly impact on marine environments such as land clearing, reclamation, beach constructions (e.g., seawalls, bridges and wharves), dredging estuarine waterways and sand dredging for construction or fill.

Examples of some of the impacts include:

- The building of seawalls, groynes or other structures - these not only directly impact on littoral and supralittoral habitats but can result in the loss of beach, seagrass and mangrove habitats when current and wave patterns are altered.
- Land clearing and/or loss of mangroves in estuaries leads to increased erosion resulting in increased sedimentation and smothering of benthic organisms, and turbidity (see also Section 6.6.1).
- Increased nutrient run off and sewage disposal results in eutrophication (see Section 6.6.1).
- Spoil from construction of roads etc. smothers habitats and results in increased turbidity (see Section 6.6.1).
- Loss of coastal wetlands (e.g., for agriculture, particularly sugar cane) results in the loss of the beneficial filtering of the wetlands and leaching of acid sulphate soils (see Section 6.6.1).
- The dumping of dredge spoil for land reclamation from busy harbours or offshore from industrial sites can be a problem when it is heavily contaminated with heavy metals and oil.

Studies by Glasby (1997), Glasby and Connell (1999; Glasby and Connell 2001) and Connell and Glasby (1999) have shown that artificial structures such as pontoons and

[141] See also Resource Assessment Commission Coastal Zone Inquiry - Final Report (RAC 1993)

marinas may increase the abundance and diversity of subtidal epibiota in the shallow areas of an estuary, they are not surrogate surfaces for epibiotic assemblages that occur on nearby natural rock. They suggest that urbanisation of estuarine habitats has consequences for the identity, diversity and abundance of subtidal epibiota. Glasby and Connell (1999) suggest that a great deal more research is needed to understand fully the consequences of adding new habitats to the marine environment.

6.16.2 Management issues and recommendations

It is vital that coastal planning instrumentalities consider the impacts of development proposals on all aspects of the marine environment (including marine invertebrate habitats and communities) and introduce regulations that minimise these. While developments in enclosed bays and estuaries are likely to cause the most damage, all developments leading to the loss of supralittoral, intertidal and shallow subtidal habitats must undergo assessments that include:

- Their direct impact on marine invertebrates and their habitats;
- An assessment of amount of similar habitat in the area; and
- Whether any species in the estuary or bay affected are likely to be threatened within that area as a result of the development.

Given that there are relatively few marine ecologists or other marine biologists capable of undertaking such detailed assessments, this expertise should be developed at regional levels as a matter of urgency.

In order that such assessments can be better expedited, baseline inventories should be conducted on major bays and estuaries around the coast.

6.17 Land use in catchments

6.17.1 Impacts

The declining quality of terrestrial run off as a result of human modification of terrestrial systems (particularly through loss of native vegetation and overgrazing) can be one of the most significant anthropogenic threats in many coastal areas, including the GBR[142] region (Bennell 1979; Baldwin 1990; Bell 1991; Yellowlees 1991; Brodie 1995a; 1997), although there is debate about the severity of the problem (Brodie 1997; Section 6.6.1).

Moss et al. (1992) estimated that 15 million tonnes of sediment, 77 000 tonnes of nitrogen and 11 000 tonnes of phosphorus are discharged annually to the coastal waters of the GBR by mainland rivers. Overall, grazing lands contribute about 80 % of nutrients,

[142] The recently enacted EPBC Act provides a potential mechanism whereby land run and water quality which impacts on a World Heritage Area such as the GBR could be controlled. The Commonwealth could act to improve water quality in the GBR region, which has been identified by the GBRMPA as a key threatening process. Although this a widespread problem for all marine coastal areas in Australia impacted to varying levels from coastal run off, the EPBC Act can only be invoked in Commonwealth waters or World Heritage Areas.

sugarcane areas 15 %, and sewage approximately 1%. Most of the nutrients originate from erosion rather than from added fertilisers. Grazing lands can lose larger amounts of sediment than woodlands or forests (1979). Much of the transportation of sediments occurs during flooding. On the GBR, Brodie (1997) points to a fourfold increase in sediment and nutrient transport in the last 40 years combined with a dramatic increase in fertiliser use. These trends probably are typical of Australia generally.

The estuarine parts of most river systems have been subjected to considerable pressures and changes. Recher et al. (1993) attempted to collect long-term data on the marine fauna of the Hawkesbury estuary, a river system that has been subjected to major changes including considerable development in some of the catchment. These authors determined that reduced freshwater runoff due to the construction of dams, enabled increased penetration of seawater upstream and, consequently, the upstream migration of marine taxa.

6.17.2 Management recommendations

We make the following recommendations concerning catchment management and how it relates to marine ecosystems and invertebrate communities. These largely follow standard recommendations for catchment management so are only briefly outlined.

- More attention to be given to preventing habitat destruction and changes to catchments which then impact downstream in terms of changed hydrography and increased rates of run off
- Reduction of terrestrial runoff by improved land management practices, reduction in clearing vegetation, restoration of vegetation, especially riparian strips;
- Changes to agricultural practices including reduction in fertiliser and pesticide use and practices that reduce erosion;
- Removal of nutrients and heavy metals from sewage discharge;
- Educational programs leading to an increased awareness of the connections between land and the sea via rivers and coastal waters; and
- Increased research into the effects of terrestrial inputs on marine ecosystems.

6.18 Summary of recommendations

Ensure that planning and management mechanisms are in place to reduce habitat alteration and destruction. All developments leading to the loss of supralittoral, intertidal and shallow subtidal habitats should be required to undergo assessments that include:

- Their direct impact on marine invertebrates and their habitats;
- An assessment of amount of similar habitat in the area; and
- Whether any species in vicinity of the area affected are likely to be threatened within that area as a result of the development.

The detailed extent and condition of critical coastal habitats should be monitored on at least a statewide or, preferably, national level.

Encourage and facilitate experimental studies to assess likely impacts from habitat fragmentation, particularly on maintenance of taxon diversity and recruitment success.

Encourage and facilitate genetic studies on taxonomically and biologically different invertebrates to assess patterns and differences in the spatial extent of significant genetic structuring.

Continue with surveys for exotic invertebrates but broaden their scope and reassess the methodologies being used. Programs relating to the assessment of introductions should aim to:

- Undertake baseline studies on native fauna;
- Determine the effects of introduced taxa on native fauna and communities; and
- Develop effective control strategies.

Improve the information and research base to assess and monitor the impacts of pollution on marine invertebrate communities. Information is required, in particular, on:

- Long-term recovery;
- Indicator species;
- Biomarkers for marine communities (development of appropriate benchmarks for risk assessment; baseline monitoring criteria);
- Development of effective management strategies to protect marine ecosystems; and
- Experimental studies on the toxicity and sublethal effects on a range of invertebrates of a wide range of chemicals and other pollutants in marine ecosystems.

Gather and utilise invertebrate data in the Coastal Resource Atlas (CRA).

Development of policy to encourage and enable cooperation between different levels of government and government departments, including those responsible for land-based activities - particularly with regard to pollutants, sediments, development, resource utilisation, recreational activities etc.

Utilisation of international regulations, protocols and law where necessary to reduce the levels of relevant impacts (e.g., pollution and exotics from shipping, global warming, etc.).

Recognise that global warming and the resultant rise in sea levels and temperatures is one of the most serious threats facing coastal marine biodiversity.

- Ensure the conservation of coastal biodiversity global warming must be slowed as much as possible.
- Facilitate and conduct research into the likely impacts of rising sea levels and atmospheric and oceanic warming to enable forward planning to alleviate impacts.
- Establish coastal zone policies that allow adaptive responses to rising seas by making way for the shoreward movement of coastal ecosystems as sea level changes.

Continue and enhance research into the causes of population outbreaks of destructive or harmful marine species.

- Studies on the basic biology, ecology and distribution of at least keystone taxa would greatly improve the knowledge base so that basic information was available when required.

Improve the knowledge base regarding diseases of marine invertebrates.
- Address issues regarding the spread of diseases by translocation of aquaculture stock and the introduction of exotic species (either as pests or for aquaculture).
- Guidelines on reporting diseases and suitable responses should be established for significant events.

Recognise that synergistic effects are important and that their recognition is critical to the identification of adequate research questions and the adoption of effective management options.
Management strategies applicable to coastal marine resources need to be at the level of ecosystems

Address unresolved issues relating to indigenous access to marine resources.

There is a need for better data so that the degree of threat posed from harvesting for most marine invertebrates can be more accurately established. This should include:
- Accurate catch data (both professional and amateur);
- Knowledge of species biology, population genetics, structure and size;
- Historical data;
- Suitable reference areas for comparison of exploited with unexploited populations; and
- Certainty of taxonomic status.

Fisheries (whether commercial or recreational), if they are to be sustainable, should at least address the following:
- Manage breeding stocks of the target species;
- Minimise effects on non-target species (minimisation of bycatch); and
- Minimise impacts on habitats.

Develop marine refugia as a management strategy for sustainable fisheries, including coastal, shelf and slope trawl fisheries.
- Marine reserves should be established on the principle of 'no-take', and these areas should be, where possible, large areas.
- Marine reserves must not, however, replace conventional management options but rather complement them.
- Ban scallop dredging from all but a few strictly controlled areas (especially as scallop aquaculture is now viable).

Require Environmental Impact Statements (EIS) for any new commercial fishing operations.

Manage intertidal collecting and the conservation of intertidal assemblages on rocky shores by:

- Use of general or selective bag limits, or size limits;
- Bans on harvesting in selected areas;
- Improve public education – including signs at closed areas, advertising, agreed penalties etc.; and
- Evaluate effectiveness.

Ensure that aquaculture developments have management that minimises:

- Destruction or alienation of habitat;
- Pollution;
- The introduction or translocation of pests; and
- Mixing of genetic stocks.

Encourage and enforce relevant regulations relating to shipping (both private and commercial) such as:

- Those that encourage safety and will reduce impacts from shipping accidents (such as oil spills);
- Discharge of ballast water in oceanic water;
- Hull cleaning should be carried out in such a way that will prevent any exotic organisms being able to survive; and
- Use of the least toxic and damaging antifouling measures.

Encourage and facilitate research on environmentally friendly anti-fouling measures.

Exploration and development of new mining and oil and gas resources activities should give greater attention to impacts on the benthic and pelagic invertebrate communities likely to be affected.

Management of coastal resources should ensure that impacts from visitors and recreational activities (including boating) are properly managed to cause as little harm as possible to marine and supralittoral environments.

- More attention be paid to education and public awareness regarding the damage that can be done by various activities; and
- Research should be carried out on the impacts of various recreational activities on marine ecosystems so that this can be used as a basis for management.

Develop expertise in marine biological assessment at regional levels as a matter of urgency.

- To expedite assessments baseline surveys should be conducted in key locations around the coast.

With regard to catchment management and how it relates to marine ecosystems and invertebrate communities, we recommend that:

- Attention to be given to downstream impacts by reduction of terrestrial runoff, improved land management practices and reduction in pollutants such as sewage and fertilisers;
- Educational programs be encouraged and developed to increase awareness of the connections between land systems and the sea; and
- Research into the effects of terrestrial inputs on marine ecosystems be encouraged and facilitated.

PART 3 - INFORMATION AND IMPLEMENTATION

CHAPTER 7 – THE INFORMATION BASE

7.1 Introduction

The information base on marine invertebrates comprises taxonomic, biological, ecological and distributional data, including their identity and the phylogenetic relationships between taxa, as well as their conservation status, economic values etc. In this chapter we outline the shortfalls, strengths and weaknesses in the information available for marine invertebrates and briefly describe some of the resources and programs available.

Under the United Nations Convention on the Law of the Sea (UNCLOS), Australia is entitled to claim an extra 5.1 million square kilometres within the Australian Marine Jurisdiction (AMJ) (see Section 1.1.2), but only if it can demonstrate (by November 2004) that it will look after it responsibly (i.e., all activities within the area must be ecologically sustainable) and has the necessary scientific understanding. Despite the diversity and dominance of Australia's marine invertebrate fauna and its ecological and economic importance, it is on the whole poorly known. One writer states that "we are still discovering species at the rate of 80 per year, and squid common enough to be frequently served on marinara pizzas have still not been described scientifically" (Luntz 1999). While the number of new taxa "described" each year is in fact much higher than estimated by Luntz, the general point, regarding lack of knowledge of such a significant portion of the Australian fauna, is well made. While the last century of research has seen major progress in the understanding of this fauna, several significant problems remain:

- At least half of all marine invertebrate species are not yet formally described, and some groups are almost completely unknown;
- Little or nothing is known about the ecology and biology of the great majority of Australian marine invertebrates;
- Many parts of the Australian marine environment are unsampled or poorly sampled for invertebrates, including virtually all areas below 3000m;
- There is a serious lack of accessibility to existing information.

Good quality information is required if effective decisions are to be made. Such information can be obtained from data already in hand (published information, databases, collections) or from new studies (e.g., field inventories (surveys), ecological or biological studies). The utility of existing data is heavily dependent on the accessibility of the information – e.g., whether the collections and literature are databased or not, and whether these databases are accessible and accurate.

Part of the problem relating to access to information is knowing (1) what questions to ask and (2) where the information can be obtained. The following issues need to be considered:

- **Information needs** will depend on the questions being asked, and these in turn will be dependent on the goal(s) of the inquirer. Needs will vary from those with broad-based, long-term goals such as documenting global or national biodiversity, to national or international issues (e.g., coral reef or seagrass conservation), to those with a more localised or immediate focus (e.g., documenting fauna of a particular location, assessing changes in response to an impact, etc.).
- **Information availability**, including accessibility, must be determined. Good information will enable:
    - Refinement of the questions being asked;
    - Determination of what new data are needed within the constraints of the timeframe, resources etc.
    - Better planning of a research/sampling program.

7.1.1 Information needs

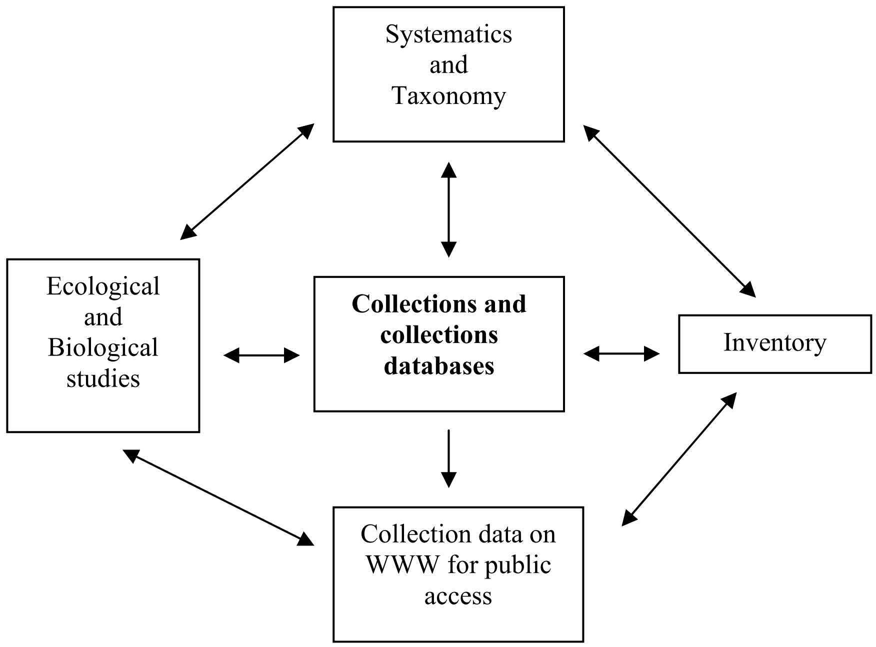
Some of the major kinds of information relevant to the conservation of marine invertebrates include issues relating to their diversity (numbers and kinds of species), ecology (factors that determine their distribution and abundance), evolutionary relationships (which give predictive power), conservation status, and biological attributes (feeding, breeding etc.).

In order to reach a position where better information regarding the marine invertebrate fauna can be provided to end users (e.g., conservation managers), it is essential that research activity be increased. The following research areas are important:

- **Systematics** encompasses **taxonomy** (the naming, classifying etc. of organisms) and **phylogeny** (the determination of the relationships of organisms); both are fundamental to the information base. These sciences make it possible for information on species to be standardised and compared, and because classifications are constructed based on evolutionary relationships, they have predictive power (i.e. closely related species are more likely to be biologically similar than distantly related species). Many ecological, fisheries and biological studies are rendered useless by misidentifications or failing to recognise that what was treated as a single taxon is in fact two or more taxa.

- **Collections** (here defined as significant and accessible collections of marine invertebrates held in a public institution) contain most of the information that currently exists on marine invertebrates. Collections have fundamental scientific roles including the housing of vouchers (specimens referred to in the literature that enable future workers to check identifications of species used in systematic, ecological and biological studies). In addition, valuable information relating to distributions and variation (in time and space, seasonally, by habitat etc.) can be determined from data associated with specimens in collections. Collections should be considered as an integral part of the research component (Figure 7.1).

Figure 7.1: The relationship between collections and the major kinds of research on non-commercially important marine invertebrates.



- Field surveys or **inventories** are important sources of new information. While ideally their planning should be based on existing information (e.g., from museum collections), often it is not. In addition, the material obtained from such surveys should be (though again, often it is not) lodged in a museum collection so that it can be available for future confirmation and other studies.

- **Ecological and biological studies** provide essential information regarding life history, food and feeding, habitat, and behaviour essential for successful management and accurate determination of conservation status. This kind of research is necessary if we are to understand some of the biological processes operating in the marine environment, the ecology of marine communities and the factors determining the distribution and abundance of taxa. This information will allow predictions to be made about the effects of particular changes or impacts. For instance, "a prediction of functional change within an ecosystem after depauperisation depends on knowledge of how that system's diversity was generated and is maintained. Merely counting species or phylum numbers within each system, though interesting from a descriptive point of view, must be interpreted with careful consideration of an array of historical, life-history, and contemporary processes" (Ray 1991). A better understanding of the ways in which combinations of environmental gradients influence patterns of biodiversity at different scales could enable better predictions, more successful conservation of ecosystem function, and associated benefits (Ray 1991; Ray and Grassle 1991).

- **Knowledge of processes and mechanisms**, rather than the traditional approach based on monitoring patterns of abundance, should ideally form the basis for management of natural resources (Hughes et al. 1999). For instance, it could be important to understand how hierarchically scaled spatial and temporal processes may give rise to or maintain the diversity of natural, semi-natural and disturbed systems (Grassle 1991). Unfortunately, however, practical considerations (such as financial constraints, lack of data etc.) necessitate that most research related to management issues involves only basic inventory, monitoring etc., and most of this takes little or no account of invertebrates.

- **Applied research**, such as that related to economic issues and application of basic knowledge, for example:
    - Assessment of economic worth (fisheries potential, useful chemicals, antibiotics etc.);
    - Pest control, including control of non-indigenous species introductions; and
    - Fisheries management.

7.1.2 Information availability

Access to information, knowledge and data is a fundamental issue. While there is considerable information already available (in collections, literature, or via specialists), much of this information is not readily accessible for various reasons, including:

- Collections are often not databased or only partially databased and there is currently no national database facility with linked invertebrate databases using common protocols.
- Much of the published information base is not readily accessible with only a handful of libraries having most of the key journals; many specialist journals are held in only one or two libraries nationally, or in some cases, none at all.
- Much of the information is difficult to obtain because it is not published (University theses, unpublished government reports etc.), or is part of the "grey literature" and held only in a small number of locations. The biggest problem with this source of information is finding out that it exists at all.
- Identifications of marine invertebrates (e.g., for ecological surveys) have typically been done by specialists. Due to increasing demand and decreasing numbers of specialists, current demand cannot be met. There are few facilities (such as identification guides or interactive keys) available for use by non-specialists in fauna surveys or other applications.
- Web-based information, while increasing, is in general not coordinated or focused and is highly variable in content and reliability.

7.2 Systematics, taxonomic information and collections

7.2.1 Systematics and taxonomy

Systematics[143], which incorporates taxonomy, is an essential science that underpins all biological, ecological, environmental, biogeographical and other related forms of research, because they all require discrimination and identification of taxa and an assessment of their relationships. The knowledge generated by systematics is critical for the conservation and sustainable use of the components of biodiversity (DIVERSITAS 2000).

Systematics is a dynamic process involving naming, classifying and discovering the natural relationships among species, using both extant and fossil material (Vane-Wright and Cranston 1992). Dedicated to the discovery, organisation, and interpretation of biodiversity, systematics is a key discipline in the documentation of biological diversity and, hence, in underpinning conservation (Morris 1986; Systematics Agenda 2000 1994; Bisby 1995; Patrick 1997; New 1999). Taxonomy supplies the formal names that facilitate information retrieval (Patrick 1997) and phylogeny discovers the relationships and evolutionary history of organisms. Together, these two disciplines help us to understand our biodiversity and facilitate its maintenance and rational use (Yen and Butcher 1997). Not only must we be able to manage the information about species, but through the estimation of phylogenetic relationships, supply the basic data for testing hypotheses on evolution and biogeography (Richardson 1984).

The process involved in systematics usually involves three steps (Richardson 1984). First, taxa (usually species) are identified and described and named if necessary. Second, the phylogenetic relationships of the taxa are estimated and a classification adopted. The third step involves the preparation of keys or other aids to the identification of the taxa involved (preferably for use by non-taxonomists).

Recognising and naming species underpins assessment of species richness, patterns of diversity, distribution and endemism, and the definition of assemblages and communities, and helps set priorities for conservation (New 1999). A phylogenetic classification gives us the capability to make predictions about the characteristics of species based on what we know about the biology of close relatives.

The need for more taxonomic research

As discussed in Section 2.1.1, understanding of the marine invertebrate fauna in general is hampered by the so-called "taxonomic impediment". This term refers both to the existence of a large proportion of undescribed taxa, and the lack of trained taxonomists

[143] The terms taxonomy and systematics have been variously applied. Systematics, as used here, encompasses both taxonomy and phylogeny. Taxonomy relates to the identification, naming and classification of organisms. Phylogenetic studies estimate the relationships of organisms and these hypotheses of relationships can be applied to classification. Others (e.g., Richardson 1984) have used taxonomy as equivalent to systematics.

and funds which make it virtually impossible to complete the task of describing the fauna within any reasonable timeframe. As New (1999) has noted, "the magnitude of describing Earth's biota has become increasingly apparent over a period marked by rapid decline in the taxonomic workforce and the lessening of interest in the science of systematics in university teaching". There is also a decline in support for classical taxonomic investigation, which is often seen as dull and repetitious, with an overemphasis on more expensive molecular techniques, despite the need (in applied ecology, conservation, and environmental management) for obtaining identifications based on visible phenotypic characters (Green 1992).

Some initiatives

On a worldwide scale, recognition of the taxonomic impediment has led to the implementation of a program known as the **Global Taxonomy Initiative (GTI)**, which was endorsed by the parties to the Convention on Biological Diversity as being fundamental to the aims of this Convention (ABRS 1998). A workshop was held in Darwin in January 1998, under the auspices of the Smithsonian Institution and Environment Australia, to develop an action plan for implementing the GTI, and resulted in the Darwin Declaration, which begins by stating that:

> "*The governments of the world that recognise the Convention on Biological Diversity have affirmed the existence of a taxonomic impediment to sound management and conservation of biodiversity. Removal of this impediment is a crucial, rate-determining step in the proper implementation of the Convention's objectives. There is an urgent need to train and support more taxonomic experts, and to strengthen the infrastructure required to discover and understand the relationships among the world's biological diversity.*"

Other international (and Australian) taxonomic initiatives are described in Section 7.4, including the Australian Biological Resources Study (ABRS). The ABRS is a Commonwealth program with the objectives of coordinating work on the collection, description and classification of Australian plants and animals, determining their distributions and publishing this information (see Section 7.3).

Overall, however, in Australia, taxonomy remains a low priority and the resources allocated to taxonomic research are inadequate, especially in light of the exceedingly diverse Australian fauna and in comparison with other developed countries in Europe and North America. This, coupled with the relatively short time that marine science has been undertaken in Australia, explains in part the lack of knowledge of the fauna. In contrast in Europe and North America there has been a much longer tradition of undertaking marine science.

Prioritisation of taxonomic studies

Given the limited resources available, it is essential that there be some prioritisation of taxonomic studies to provide the greatest possible benefits (Yen and Butcher 1997) for the available resources. ABRS has a set of criteria for funding priorities, which includes groups of conservation significance and of value as environmental indicators. Yen and

Butcher (1997) argue that "from an invertebrate conservation aspect, it is recommended that priority be given to groups of use in (1) assessing sites for conservation, and (2) biomonitoring". While important, to prioritise taxonomic funding on the perceived value of these two activities seems to us to be too restrictive, because (1) taxonomic information has a much wider application; and (2) many groups of potential value (for these activities) are currently being overlooked either because they are poorly known or because of the biased interests of researchers.

Criteria to consider when selecting taxonomic or functional groups on which to focus should include as many as possible of the following:
- Ecological importance;
- Abundance and/or biomass;
- Conservation significance;
- Potential for monitoring;
- Economic importance;
- Existing taxonomic knowledge;
- Availability of collections/material; and
- Availability of expertise.

Selection of individual proposals should take into account additional criteria including the obvious ones such as ability, experience and so on. They should also look at:
- Opportunity (e.g., making use of an available post doctoral student or a person at the end of their career);
- The scope (e.g., major monograph versus regional revision);
- Value-adding of the product - with emphasis on widest possible availability and use (e.g., scientific publication only, or supplemented by keys, handbook, CD ROM or web-based guide or key).

Yen and Butcher (1997) recommended that funding be made available for taxonomic research into invertebrates found in "sites under extreme threat (which) are of prime scientific and conservation significance". While this is an excellent aim in general, we doubt the practicality of this piecemeal approach as a basis for sound taxonomic research, particularly in the marine environment where such sites have rarely been identified. We would prefer to see this problem tackled as an inventory of the area concerned with the material being made available to relevant experts for informed comment and broader-based research.

Training of systematists

It is now widely recognised (e.g., Tilling 1987; Yen and Butcher 1997; New 1999) that, to its detriment, taxonomy has been widely seen as "less prestigious" than many other sciences, and is often considered a 'service' discipline. Tilling (1987) suggested that the problem is even deeper rooted, with few teachers having the "confidence or inclination to tackle the subject with school students, thereby affecting the number of students interested in taxonomy as an option in higher education." This situation is now slowly changing, especially in some European and North American universities, where phylogeny and other evolutionary research are combined with broader systematics and biodiversity studies.

In Australia, the new Bachelor of Science in Biosystematics[144] (B.Sc. (Biosystematics)) at University of New England (ENE), developed through partnerships between UNE, the Australian Museum and the Royal Botanic Gardens Sydney, is specifically designed to address the lack of training opportunities in systematics/taxonomy. This course is currently the only one of its kind in Australia.

While increasing the number of specialist taxonomists is a laudable goal, it is also important to increase the number of graduates with invertebrate expertise, and find relevant employment for these graduates (Yen and Butcher 1997). This can be done in part by establishing a work force of well-trained, not necessarily graduate level, technicians to undertake the primary sorting and identification but it is also important that ecologists and environmentalists have some training in the basics of systematics.

The National Strategy for the Conservation of Australia's Biological Diversity has training as one of its Actions (No. 4.1.6). The aim is to:
Facilitate and support the development of collaborative taxonomic training programs by existing institutions such as museums, herbariums and universities for:

- Specialist taxonomists, particularly to work on inadequately studied groups;
- Biological diversity technicians;
- Ecologists;
- Members of the public and community organisations assisting in biological diversity projects.

Improving taxonomic efficiency
Yen and Butcher (1997) made a number of recommendations relating to increasing the efficiency of the taxonomic work that is being done. These include:

- Providing more technical assistance and encouraging the use of newer technologies such as computer image enhancement techniques, scanning electron microscopy and new software for taxonomic and systematic research;
- Providing assistance to persons who are engaged in taxonomic work but are not employed as taxonomists;
- Encouraging overseas taxonomists to work on Australian groups;
- Ensuring that ecological and applied projects include a budget for taxonomic assistance.

To expand on the first of these points, the development and use of computer-aided technology has enormous potential. For instance, the DELTA computer program (Dallwitz et al. 1993) has been developed as a taxonomic tool by a team in CSIRO Entomology. It includes sophisticated data handling, natural language and key generation and the interactive key program Intkey. Unfortunately, the funding for this program was

[144] Website: http://www.austmus.gov.au/une/

terminated in 2000 after more than 20 years of development. Currently, many of the features of this package are being bundled into CSIRO's Biolink[145].

While new technology, such as scanning electron microscopes (SEMs) have been available for several decades and their use can markedly improve efficiency, they are still not being used by some taxonomists whose work would benefit by them doing so. In addition, molecular techniques – though not necessarily a replacement for traditional approaches – should now be considered a standard tool of taxonomists.

7.2.2 Identifications

Identifications of taxa in samples obtained from surveys are frequently required for ecological studies, environmental impact assessments, monitoring programs etc. The key processes required for successful identification of invertebrates are (Yen and Butcher 1997):

- The ability to identify a particular specimen to the species (or morphospecies) level with authority;
- To be up to date with species names by co-ordinating and communicating name changes; and
- To be certain of consistent universal name use.

To fulfil these criteria currently requires a background in the field of expertise required and reliance upon an expert on the group of invertebrates in question (Yen and Butcher 1997). *Voucher material* should be lodged in a museum collection following certain protocols.

Voucher material

Identification of all material by a specialist is not always practical. Increasing demand and decreasing numbers of specialists mean that it is becoming impossible for these experts, who work mostly in museums, to act as consultants to all members of the wider community who seek information from them. Efficiency can be increased if only voucher material needs to be examined. To do this, users need to undertake the sorting and the preliminary identifications. While this can be assisted by the use of identification guides and reference collections (Yen and Butcher 1997), serious errors can be introduced, for example by the unwitting lumping of two or more similar taxa, with perhaps only one included in the voucher material.

Aside from the time and difficulty involved in making identifications, a serious deficiency in using data compiled by different research groups and organisations is the lack of taxonomic consistency. Formal scientific names for animals provide standards with which to measure diversity or monitor environments. When available, all environmental scientists can use them and others know what is meant. But, for at least half of the marine invertebrate species in Australia, Linnean names are not available and each institution, consulting company and survey scientist devises his or her own coding system to deal with the problem.

[145] http://www.biolink.csiro.au/

***Protocols*:** The following guidelines should be applied if a project requires the identification of material:

- Discuss the project with the expert(s) prior to sampling - they will be able to offer valuable advice that will result in a better outcome.
- Maintain all samples separately - do not combine separate samples until the identity of the taxa in the samples has been confirmed.
- When selecting vouchers, do so from as wide a range of the variation of the putative taxon as possible.
- Always lodge voucher material in an appropriate museum collection that has long-term curatorial capability. The material should be properly fixed and labelled using protocols determined at the beginning of the project from discussion with the appropriate collection manager(s).
- Budget for additional funds that may be needed to cover the expert's time and to assist with collection management.

7.2.3 Collections

Collections have a number of actual and potential uses aside from their primary scientific function as taxonomic reference collections. One of those that has been increasingly emphasised in recent years is their value in biodiversity research (e.g., Hoagland 1989; Hawkesworth and Mound 1991; Nielsen and West 1994; Ponder 1999). The Darwin Declaration (see above, 7.2.1) states that:

> "Information derived from biological collections held in the world's taxonomic institutions underpins the global, regional and national efforts to conserve biological diversity. The collections, staff and associated information serve as an essential resource for countries in fulfilling their obligations under the Convention on Biological Diversity."

As recognised by the 'Diversitas' program (DIVERSITAS 2000), although the world's natural history collections (museums, herbaria, living culture facilities, and seed banks) currently house nearly two billion specimens, including voucher specimens (types etc.) which can be re-examined at a later date, very little of the information associated with these specimens is available electronically to all countries of the world. At the national level, increased capacity to undertake systematics research will promote the documentation of the components of biodiversity, identification of patterns of diversity and endemism, recognition of regions of critical conservation concern, and support for efforts to manage habitats, ecosystems, and landscapes as well as agroecosystems and fisheries or bioprospecting for new natural products (DIVERSITAS 2000).

Although museums typically have a legislative responsibility to maintain their collections, in general the level of funding for core responsibilities such as collection management is on the decline. This funding problem may relate (at least in part) to the fact that the values of museum collections are overall little appreciated in the broader community and government. In an era when the roles of museums are being redefined to reflect the current priority passion for serving "customers" or "stakeholders", the priority assigned to the once core responsibilities of development and maintenance of collections is, in many institutions, in a state of decline.

Voucher specimens: The responsibilities of non-taxonomists to lodge voucher material in museums must not be overlooked (Robinson 1975; Yen and Butcher 1997). However, for deposited voucher material to be of value to other workers (ecologists or taxonomists) it should:

- Include the name used in the study;
- Identify the study and/or publication/report;
- Be deposited in an accessible and well-maintained public collection;
- Have correct and informative locality and collection data; and
- Be preserved using standard methods.

Access to museum data

Museum collections are effectively huge databases, and most have been accumulated over a long period, so they not only reveal valuable biodiversity information such as distribution and habitat, they also provide a historical perspective impossible to obtain with contemporary field surveys (Ponder 1999; Ponder et al. 2001a).

Biodiversity information**:** For many taxa – particularly invertebrates – collections contain by far the largest repository of data on biodiversity and distributions (Ponder 1999)[146]. These data can be used, for example, to determine areas rich in species or endemics for conservation purposes (e.g., Prendergast et al. 1993; Väisänen and Heliövaara 1994; Kress et al. 1998). Another application is in the comparison of existing fauna with that historically found in an area, in order to determine the presence of introduced species (e.g., Coles et al. 1999); without such historical reference collections, it would be virtually impossible to differentiate between native and non-indigenous species. Existing collections are often spatially and temporally inadequate to enable as much analysis or comparison as is needed for sophisticated ecological studies or detailed local distributions, but as the only such mechanism for historical comparison, they are a valuable resource. In addition, museum data used in conjunction with GIS technology can be a valuable research tool for determining distributions (Ponder et al. 2001a).

Whereas other work such as reports or published papers can also be useful, museums provide physical reference collections where the specimens themselves can be directly examined at a later date, which can be essential in situations where species have previously been unidentified or misidentified or the group has undergone significant revision.

However, one difficulty is that data held in museum collections are often not easily obtainable, the great majority being neither published nor available in any electronic form. Collections are traditionally stored in taxonomic groups, making retrieval of distributional and habitat information very time-consuming due to the need to read labels

[146] See Appendix 1 for a summary of data on collections of marine invertebrates in Australian museums.

or refer to catalogues (Ponder 1999). The advent of electronic databasing has opened new possibilities for the retrieval of valuable information on biodiversity and distributions, although to date most natural history museums have been slow in databasing their collections – particularly those of invertebrates, in part because of lack of funds.

There is therefore a need to provide funding to database collections so that vast amount of information stored in museums is made available/accessible for use in management decisions etc. (see Ponder 1999). This transfer of information from specimens in collections is one of the priorities of DIVERSITAS Sub-Programme 3 (Management of Systematic Knowledge Bases) (DIVERSITAS 2000).

Sorting centres

Yen and Butcher (1997) argued that the effective use of specimen-based information for invertebrate conservation could be enhanced markedly by establishing a network of regional sorting centres (or biodiversity laboratories, as they are termed in the National Strategy for the Conservation of Australia's Biological Diversity) across Australia. There are arguments for and against such centres, but in general we believe that a more useful and practical approach would be to enhance already established facilities.

Arguments for and against

Such laboratories would (Biological Diversity Advisory Committee 1992):

- Identify gaps in knowledge;
- Strengthen the network of biological diversity laboratories;
- Rapidly assess poorly known elements;
- Increase training and monitoring;
- Assist quick dissemination of information; and
- Conduct research into the development of new methods to sort, identify and store large numbers of unidentified specimens.

Yen and Butcher (1997) argued that these steps would help to overcome the current lack of infrastructure to assist rapid and efficient handling of bulk samples - one of the great impediments in invertebrate biodiversity research. Samples would be sorted to a particular taxonomic level and the sorted material distributed to the relevant institution/expert for lower level sorting, identification and storage. They argued that it is "important that the sorting centres be regional so as to be directly accessible to the main users. They need to be networked so that there is consistency in identifications and standards, but there is scope for each centre to become a centre of specialisation in one or more groups of invertebrates" (Yen and Butcher 1997, p. 165). They recommended that the establishment of regional biodiversity assessment laboratories for invertebrates should be a matter of priority under the Biodiversity Strategy, and that "regional laboratories should be associated with institutions that have existing expertise, and laboratories should be networked to prevent duplication of effort" (Yen and Butcher 1997, p. 165).

While generally agreeing that sorting centres are a useful concept, there are some complicating factors that need to be considered including:

- Lack of available expertise in many groups;
- Duplication of interests in others resulting in problems with eventual deposition of material;
- Much increased work loads for the likely major recipients of most of the material - the state natural history museums (Note that we consider it would be a retrograde step to separate the museums from this exercise, if it were to proceed, as this is where most of the existing collections and expertise reside); and
- Lack of facilities/space and other resources in the existing institutions.

Nevertheless, from the marine perspective, there are also many advantages of a national sorting centre system. Much fieldwork is expensive, and this particularly applies to sampling involving ship time. Such samples can be of value and relevance to a number of scientists / projects. There are many instances where samples are obtained for a particular purpose that would be of enormous value to other workers - for example sediment samples obtained by geologists contain some living marine organisms, as well as the hard parts of molluscs, brachiopods, bryozoans, corals etc. Processing in a sorting centre would markedly “value add” to such samples. Other samples collected for a particular biological program may contain many organisms of no interest to those involved in the program, but of great value to other scientists. Similarly, bycatch from deep-water exploratory fishing has produced many new taxa and rare species. Unfortunately much of this valuable material has simply been thrown over the side of the research vessel once the data on the target species have been extracted.

7.3 Other basic research needs

The great gaps in our knowledge of the diversity of the marine invertebrate fauna are exacerbated by an even greater lack of information on their ecology and biology (Section 2.1). Basic information on distribution, food and feeding, reproduction, habits, preferred habitat etc. are lacking[147] for the vast majority of taxa, as is fundamental information on the communities that they make up and the habitats that they occupy, let alone hard science backed up with experimental and other data.

7.4 Global and national programs designed to increase knowledge and improve access to information

7.4.1 Global programs

The last few years has seen the development of several global initiatives (the relationships between which can be quite confusing) designed to increase knowledge and improve the accessibility of information on biodiversity and/or systematics. Such

[147] The decline in interest in "whole animal" studies in universities is one reason for the lack of basic biological information.

programs include DIVERSITAS[148], the Global Taxonomic Initiative (GTI)[149], Species 2000[150] ("Indexing the world's known species"), the Global Biodiversity Information Facility (GBIF)[151], the Census of Marine Life or Ocean Biogeographical Information System (OBIS)[152], and the Intigrated Taxonomic Information System (ITIS)[153]. Generally these are focused around information systems, with the aim of developing web-based access to checklists of names, distributional data and other taxon-based information. To date however, despite the rhetoric, there has been little or no funding provided to enhance research in systematics and taxonomy. More information on these programs is readily available on the web.

7.4.2 Australian programs

National Strategy for the Conservation of Australia's Biological Diversity (NSCABD)

This strategy recognises that the "Full and effective implementation of many of the actions identified in this Strategy requires considerable improvement in our knowledge and understanding of Australia's biological diversity in terrestrial, marine and other aquatic environments. Accordingly, there is a need for a significant increase in research into biological diversity at the genetic, species and ecosystems levels." It identifies the need for major research initiatives in the areas of compilation and assessment of existing knowledge, conservation biology, rapid assessment and inventory, long-term monitoring, and ethnobiology. It also recognises that it is "essential that there be adequate mechanisms to ensure that the results of research are rapidly disseminated and rapidly incorporated into current and future actions." (Commonwealth of Australia 1996).

Aims and actions.
These aims are formalised as Objective 4.1: "Provide the knowledge and understanding of Australia's biological diversity essential for its effective conservation and management." This objective has nine Actions:
"4.1.1 *Compilation of current knowledge.* Undertake to coordinate, collate and synthesise available data and information from collections, survey results and geographic information systems to provide a basis for assessing research needs and priorities. This will include knowledge held by industry (for example, from environmental impact assessments and rehabilitation activities), community groups, local government and experts. Particular attention should be given to:

- Identifying inadequately understood components of biological diversity;
- Identifying those components of biological diversity important for its conservation and sustainable use (see Objective 1.1);

[148] http://www.icsu.org/DIVERSITAS/
[149] http://www.biodiv.org/programmes/cross-cutting/taxonomy/links.asp
[150] http://www.sp2000.org/aboutsp2000.html#Organisation
[151] http://www.gbif.org/
[152] http://marine.rutgers.edu/OBIS/
[153] http://www.itis.usda.gov/

- Accelerating the activities of agencies and institutions involved in the development and networking of complementary environmental geographic information systems and databases and ensuring that there are adequate resources for the storage and maintenance of collections;
- Developing and implementing, by the Australian and New Zealand Environment and Conservation Council, mechanisms for the improvement of research and coordination and dissemination of information about biological diversity. "

These actions are "consistent with the governments' undertakings in the National Strategy for Ecologically Sustainable Development to cooperate in developing analysis and information technologies and systems for optimising the use of natural resource databases and to use these in pursuit of ecologically sustainable development.

Australian Biological Resources Study (ABRS)

The stated aims are to:
- Facilitate at a national level the identification of Australia's biodiversity.
- Support studies of the origins, evolution and relationships of Australia's biota.
- Foster knowledge on Australian biodiversity, including poorly known groups.
- Enhance Australia's capabilities in taxonomy including training and mentoring of new scientists.
- Document and disseminate information on the taxonomy and biogeography of Australia's biota, providing increasing accessibility to our knowledge base.
- Foster partnerships with other organisations to build capacity in taxonomy.
- Meet Australia's national and international taxonomic and biogeographic responsibilities as custodian of a mega-diverse biota.
- Increase public awareness of taxonomy and ABRS's role as a fundamental provider of information on Australia's biodiversity

A major role of this very important program is cataloguing and documenting the biota in a program related to GBIF, the Australian Biodiversity Information Facility (ABIF). Data in ABIF are available on the ABRS website[154]. See also 7.5.2.

7.5 Making use of existing information - improving the accessibility and availability of data

Action 4.1.9 (Information: access, dissemination and use) of the NSCABD states that by "Taking intellectual property rights into account, ensure that as information about Australia's biological diversity accumulates it is published and otherwise disseminated in ways readily accessible for national and regional planning, development, management and decision making, in both the private and public sectors, including through computer networks. " It states further: "Ensure that the accumulated information is used to evaluate

[154] http://www.gbif.org/linkfram.htm

and improve the effectiveness of current management, to meet the objectives of ecologically sustainable use of biological diversity, including its protection."

While there is much that we do not know about marine taxa and systems, the considerable information that we do have is widely scattered and generally inaccessible. While specialists have reasonable access to their areas of interest, the transfer of information between specialists is often difficult. Three steps are necessary if information is to be accessible nationally:

- Collation of available information;
- Electronic databasing of the information; and
- Linking of the databases containing the information.

Clearly, to compile all information in such a way is a very long term or even impossible task. However, priorities can be set to achieve this facility in some areas that would provide considerable general benefit. Some of these areas are detailed below.

7.5.1 Identification tools

Marine invertebrates need to be able to be identified by a range of groups including environmental groups, schools, universities, and naturalists; as well as ecologists, biologists, physiologists, organic chemists, and many other scientists and technicians. The following economic and environmental applications all require that marine invertebrate taxa be accurately and readily identified:

- Biomonitoring and use of taxa or communities as indicators, for instance to monitor sewage outfalls, oil or gas wells, or the effects of pollution incidents (involves detection of variation or changes in the distribution or abundance of vulnerable animals or the composition of communities);
- Understanding the impacts of fishing (e.g., incidental harvesting or bycatch);
- Recognition, early detection and control of introduced marine species, including pests such as fouling organisms, pests etc.[155]

However, non-specialists wishing to make even the most elementary identification of a common species or family are currently forced to use a variety of sources which, in addition to being frequently difficult to obtain, may be inappropriate or out-of-date. Although there are an increasing number of “picture books” illustrating some of the more common and conspicuous marine invertebrates, these only identify a minuscule proportion of our diverse benthic fauna and they are not particularly useful for community groups or management agencies. Comprehensive guidebooks, such as those readily available for birds, reptiles and mammals, are not generally available for marine

[155] . For instance, CRIMP has asked, through its community liaison officer, for community groups to watch out for new arrivals. The success of this apparently laudable venture is unlikely except in cases involving large, unusual and conspicuous animals. Not only will it be difficult to recognise most invasive species because of the lack of accessible faunal guides, many potential problem species are easily confused with native taxa (e.g., the Pacific Oyster *Crassostrea gigas* with the commercial rock oyster on the east coast, *Saccostrea glomerata*; and the black-striped mussel *Mytilopsis* sp., recently introduced to Darwin, with some native species of Mytilidae such as *Xenostrobus securus*).

invertebrates[156]. Moreover, many species are endemic to particular parts of Australia so that identification guides written for other parts of Australia or the Indo-Pacific region are not necessarily relevant.

The absence of accessible identification guides to the fauna is a serious issue for the environmental industry, for community groups, students, naturalists and others with a need to monitor the marine environment. There is an urgent need for up-to-date identification tools for, at the very least, the shallow-water Australian marine macroinvertebrates. Moreover, production of these guides is expensive and relies largely on some level of commercial success.

7.5.2 Other sources of information

Bibliographies

Bibliographies of particular taxonomic groups, regions or subjects can be very useful for those not working on a daily basis with a particular subject area. Up-to-date bibliographies relevant to Australia are rare. Ideally these should be available electronically in a searchable database with key words.

Checklists and checklist-based information systems

State-based printed checklists (mostly out of date) exist for some taxa. One of the objectives of the Australian Biological Resources Study (ABRS), established in 1973, was to provide an inventory of the fauna. Of the 20 catalogues published to date, only three cover marine invertebrate taxa (Vol. 12 Porifera (Hooper and Wiedenmayer 1994); Vol. 33 Echinodermata (Rowe and Gates 1995); Vol. 34 Hemichordata, Tunicata, Cephalochordata (Wells and Houston 1998)), Crustacea: Malacostraca (Davie 2002a, 2002b) although others, such as the polychaetes (Hutchings and Johnson 2001), are available on the ABIF website or are in preparation. The cataloguing of a number of other groups are currently in progress or have just been completed. More recently, ABRS has provided on-line checklists (Australian Biological Information System - ABIF). ABIF-Fauna is a Web based source of taxonomic and biological information being compiled for all animal species known to occur in Australia. It comprises the Fauna of Australia (a family-level synthesis that has now been suspended)[157], the Census of Australian Vertebrate Species and the Australian Faunal Directory[158] (formerly the Zoological Catalogue of Australia database, a species-level directory of data).

[156] Some examples illustrate the problem. Some attempt to identify polychaete worms from written for the South African fauna in 1967 (Day 1967). The only comprehensive text covering crustaceans was written in 1927–29 and books on the Japanese fauna are essential reading for decapod crustaceans. There is, currently, no easy way to identify isopods to family or species and most non-experts find amphipods impossible even at the family level. Molluscs must be identified from picture books covering only a fraction of the fauna, or from outdated regional guides.

[157] Although the following invertebrate groups were completed - the molluscs (Beesley et al. 1998) and polychaetes, sipunculans, echiuroids, mogonophorans and myzostomes (Beesley et al. 2000).

[158] http://www.environment.gov.au/abrs/abif-fauna/intro.htm.

Several global systems have an aim to produce web-based online system for all biota. These include the Integrated Taxonomic Information System (ITIS)[159], The Global Biodiversity Information System (GBIF)[160] and Species 2000[161]. The Ocean Biogeographic Information System (OBIS)[162] is the only one with a similar aim that is exclusively marine. It has funded work on sea anemones and Indo-Pacific molluscs as well as fishes.

Distributional information

While much distributional information exists, most of it is not readily accessible because it is not published, or it is in collections not databased or the database(s) are not accessible (Yen and Butcher 1997; see above). Until we have more data that are comprehensive on marine taxa so that rarity and/or decline can be determined, conservation should initially focus on the identification and preservation of restricted species and special habitats that support suites of invertebrates currently considered uncommon[163].

The idea that linking collections databases around Australia can create a National Marine Invertebrates Database is suggested in Section 7.2.3. Such a database could provide distributional and other data in real time in a similar way to that done by the Virtual Australian Herbarium[164]. This facility can provide information on the distribution of plants found in coastal habitats (mangroves, saltmarshes etc.).

Other computer-based sources of information

Other biotic and physical data can be put on line. The Australian Coastal Atlas (ACA)[165] is a web mapping tool that directly accesses databases of marine and coastal data and documentation to enable mapping of information about Australia's coastal and marine environments. Currently, geographical and physical information about all open coast beaches (from Surf Lifesaving Australia), Maritime boundaries information, World Heritage areas and National Estate places, important wetlands (including RAMSAR sites) and IMCRA, bathymetry and topographic data. The capability is being regularly upgraded.

The Marine and Coastal Data Directory of Australia[166] was coordinated by ERIN as part of the *National Marine Information System* (NatMIS), an initiative of the Ocean Rescue 2000 Program. The intention was that the NatMIS would consist of a series of distributed

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[159] http://www.itis.usda.gov/
[160] http://www.gbif.org/
[161] http://www.sp2000.org/
[162] http://marine.rutgers.edu/OBIS/
[163] For example, mapping of mollusc distributions has been used as a conservation tool in the UK (Light 1998).
[164] http://www.anbg.gov.au/avh.html
[165] http://www.environment.gov.au/marine/coastal_atlas
[166] http://www.marine.csiro.au/marine/mcdd/index_text.html

databases. However, while the Marine and Coastal Data Directory exists, NatMIS seems to have foundered.

National Marine Information System (NatMIS). The aim was that web-based information would be made available to meet the requirements of a wide range of users. The project was to include meso-scale biophysical regionalisations of Australia's Exclusive Economic Zone (EEZ) that were being developed by a Commonwealth Technical Consortium involving the:

- Environmental Resources Information Network
- Australian Oceanographic Data Centre
- Australian Geological Survey Organisation
- CSIRO Division of Fisheries (the CSIRO Division of Marine Resources - combined Fisheries and Oceanography Divisions)
- CSIRO Division of Oceanography
- CSIRO Division of Wildlife and Ecology.

According to their website, the aim was that NatMIS would contribute to answering key questions such as:

- What is the current status of Australia's marine and coastal environment?
- Where can information be found on coastal and marine issues?
- What are the main regions and habitats and where do they occur?
- Where do endangered species occur, and are they conserved?
- How can the effects of pollution be minimised?
- What are the 'indicator species' for marine and coastal degradation?

One of their roles was "facilitating the coordination of marine data collection and management throughout Australia" although the intention" is definitely not to usurp the roles of other marine agencies and coordinating bodies, but to show an active interest and assist and add value to these processes wherever possible." However, despite a stated aim of having this laudable system operational by 1996, it has not eventuated to date (mid 2002).

Indigenous knowledge

The NSCABD[167] has as its Action 4.1.8, Ethnobiological knowledge of Aboriginal and Torres Strait Islander peoples. This recognises "the value of the knowledge and practices of Aboriginal and Torres Strait Islander peoples and incorporates this knowledge and those practices in biological diversity research and conservation programs" by:

- encouraging the recording (with the approval and involvement of the indigenous people concerned) of indigenous peoples' knowledge and practices;
- assessing the potential of this knowledge and these practices for nutritional and medicinal uses, wildlife and protected area management and other purposes; and
- applying the knowledge and practices in ways that ensure equitable sharing of the benefits arising from their use.

[167] National Strategy for the Conservation of Australia's Biological Diversity.

There are, apparently, very few reviews of indigenous knowledge of marine invertebrates and this is clearly an area that needs attention. Gray and Zann (1988) provide an overview of the traditional knowledge of marine environments in northern Australia.

Yen and Butcher (1997) emphasise that "invertebrate inventories and research in Australia should, where applicable, consider Aboriginal perspectives on invertebrates." They state that indigenous knowledge is important in the following areas:

- Ethnozoology. The names and classification of invertebrates from an indigenous perspective.
- Biological and ecological information. Observations on the biology, behaviour, seasonality and distribution of invertebrates by indigenous peoples will provide valuable information.
- Utilisation of invertebrates is important both from a biological and ecological perspective, and for many marine taxa is a conservation issue because of the increased pressures on coastal edible invertebrates.
- The cultural side of indigenous knowledge on invertebrates cannot be overlooked.

Yen and Butcher (1997) also note that "the documentation of indigenous knowledge requires the approval and involvement of the communities concerned. This includes resolution of issues such as intellectual property rights. The preparation of guidelines in this area would assist the process and would reduce the chances of misunderstandings."

7.6 Obtaining further information – Inventories

The inventorying of biodiversity is the fundamental starting point for the conservation, sustainable use and management of biodiversity (Stork and Samways 1995). It provides information on the presence and abundance of animals in a given area (local, state, national or even global). An inventory involves the surveying, sorting, cataloguing, quantifying and mapping of entities that range from genes to landscapes or their components, and the synthesis of the resulting information for the analysis of processes (Stork and Samways 1995; Yen and Butcher 1997). We, like Yen and Butcher (1997), use the term herein to apply to inventorying invertebrate biodiversity. Inventory is a necessary prerequisite for monitoring that assesses how, and to what extent, biodiversity changes over time (Stork and Samways 1995).

Inventory is recognised as an Action (4.1.5) in the NSCAB[168] which states that research should be accelerated "into the taxonomy, geographic distribution and evolutionary relationships of Australian terrestrial, marine and other aquatic plants, animals and micro-organisms, priority being given to the least known groups, including non-vascular plants, invertebrates and micro-organisms. Where appropriate, methodologies should be standardised."

[168] National Strategy for the Conservation of Australia's Biological Diversity

It goes on to state that this "can best be achieved by strengthening the role of the Australian Biological Resources Study[169], including an extension of the Study program to cover micro-organisms[170]. Resources available to Commonwealth and State and Territory institutions involved in taxonomic work and in the study of ecosystem and genetic diversity should also be maintained or enhanced." Further, that "The completion of a comprehensive inventory of Australia's ecosystems should be treated as a matter of urgency in research funding." To date there has been little or indication that this research funding is a priority.

Inventorying and monitoring can be undertaken for single species, multiple species in one or more groups, or, much more rarely, for large components of the biodiversity. However, field surveys, whether inventories or ecological, are generally time-consuming (and thus expensive) and are therefore usually impractical if information is required rapidly. Practical and/or economic constraints result in surveys often focusing on only the largest organisms (representing only a tiny fraction of the biodiversity) and often employ techniques involving identification to higher-level taxa making their results useless for species-level information.

Some groups of invertebrates (e.g., nematodes and some of the other very poorly collected and researched groups) remain largely beyond the practical bounds of surveys because there is very little available information and this lack of knowledge is compounded by a lack of expertise. However, while these enormous gaps regrettably exist in our knowledge, some groups (larger molluscs, echinoderms, decapod crustaceans) are sufficiently well known to identify them to species-level. Other groups (e.g., isopods, amphipods, small molluscs, polychaetes) are becoming reasonably well known although still contain many undescribed taxa.

There is no state or region of Australia where an even reasonably complete list of the marine invertebrate fauna is available. Not only is there no such list for our most populous state NSW, but the only one produced for an area within the state is a list of the invertebrates from Sydney Harbour, compiled by Whitelegge in 1889 more than 100 years ago and effectively useless today.

Types of inventory

Nielsen and West (1994) listed four phases of sampling (1) opportunistic (early explorers, settlers and collectors); (2) general surveys (expeditions); (3) specialised collecting (experienced field collectors concentrating on their group(s) of interest); and (4) baseline sampling (either intensive non-quantitative collecting techniques and/or standardised quantitative sampling methods used for longer periods). Inventory in its broadest sense is the outcome from any one of these phases of sampling or survey.

The simplest form of inventory is simply a list of taxa (species list) for one or more groups. Such lists are useful, and often all that there is available in the literature to give

[169] The budget of ABRS was substantially cut this year.
[170] ABRS have, in the last few years, funded programs involving protists.

an historical perspective and may perhaps give clues to significant changes in species composition. Remarkably, there are few such lists available in the public arena. Worse, there are very few well-constituted inventories for marine invertebrates in the literature for any part of Australia, although the Port Phillip Bay survey in Victoria being a notable exception (e.g., Wilson et al. 1998). Other reasonably comprehensive documentations of fauna include some of the estuaries of New South Wales (Stephenson and Cook 1979; Stephenson 1980; Day and Hutchings 1984 – Merimbula and Pambula Lakes; Hutchings et al. 1989 – Twofold Bay; Hutchings 1992c - Middleton and Elizabeth Reefs; Hutchings and Jacoby 1994 – Jervis Bay; Davie and Hooper 1997 - Moreton Bay; Wilson 1998 – soft benthos of Botany Bay).

Broad survey inventories describe what species are present and where they are found. Such information is vital for conservation planning and management but can also provide information that will assist with scientific experimentation or modelling; environmental impact statements; baseline biological surveillance or monitoring; information for contingency plans and damage assessment (Baker et al. 1987; Yen and Butcher 1997).

There have been attempts to undertake biological inventories of marine invertebrate faunas in North America and Europe over the last few years. For example, Theroux and Wigley (1998) have provided an inventory of the macrobenthic invertebrate fauna off NE USA detailing their distribution, relation to bottom sediments, etc. for all taxonomic groups. Programs such as OBIS (see Section 7.4.1) are designed to incorporate this information.

In Australia, there has been little funding for field-based marine survey work on invertebrates. With the current emphasis on the conservation of biological diversity, the necessity for broadly based marine invertebrate surveys is obvious. In particular, there is a great and urgent need for rigorous *baseline studies* to establish what fauna is currently present in key areas such as major ports, marine parks, etc. This will enable future comparisons to be made in either follow-up surveys or full-blown monitoring programs. Areas that require biodiversity inventories are not necessarily those furthest from the centres of population; areas near large cities may more urgently require surveying due to imminent threats associated with human impacts.

Yen and Butcher (1997) give the following arguments in favour of broad-based invertebrate surveys:

- They are a cost-effective method of obtaining information for a wide spectrum of invertebrate groups, even if that information is not realised immediately.
- Effective use of scarce fieldwork resources, especially in remote or difficult to reach areas.
- Environmental data relevant to a wide group of invertebrates can be repeatedly sampled at a number of selected sites.
- They obtain information that may assist in enabling community or habitat-based conservation programs rather than single species conservation.

In addition to protocols relating to sampling and sorting, other issues of importance when planning a survey inventory include a clear definition of the aims; whether or not species or morphospecies identifications will be used; target groups and the level of taxonomic focus. Material collected for a particular survey may have only been identified to higher taxon levels but if this material is deposited in an accessible museum collection, it is then available for future studies involving species-level determinations (provided adequate fixation and preservation protocols are followed appropriate to individual taxa).

New (1996) emphasises that defining the aims of the inventory are crucial in conducting surveys. There are two kinds of survey: taxonomic and environmental, but these are not mutually exclusive (Kim 1993). An aim to find out what is there is a perfectly adequate reason to conduct a survey, as long as the specimens and data are collected and stored in a manner that can be used for conservation management (Yen and Butcher 1997). The attainment of this objective is enhanced by collaboration between ecologists and taxonomists (New 1996).

Base-line studies

The primary aim of a baseline study is to document the biodiversity of an area at a particular point in time (i.e. to provide an inventory) and has certain minimal requirements that enables it to be used as a benchmark in the future and also to make comparisons with similar studies in other areas. These include:

- The use of adequate and repeatable quantitative sampling design involving adequate temporal and spatial replication;
- Reference material to be placed in the appropriate state museum so that it is accessible to future research workers for the verification of identification;
- Identification to species level where possible; and
- Information obtained to be publicly accessible.

In addition, the following should also be objectives if an assessment of the biodiversity of an area is to be taken seriously:

- Inclusion of at least the major groups of marine invertebrates;
- Inclusion of species down to at least 1 mm in size;
- Coverage of the entire range of environments in an area, including the sublittoral; and
- Inclusion all components - pelagic, epibenthic, infaunal etc.

A completely comprehensive survey for invertebrates is probably unattainable, partly because of the variables associated with sampling methods and effort (Yen and Butcher 1997), and partly because we lack the expertise to adequately separate biologically meaningful *morphospecies* of morphologically uniform, but never-the-less megadiverse, groups such as nematodes. While collecting success (number of species caught for a given effort) can be easily measured, sampling efficiency (percentage of total number of species) is not (Disney 1986). Invertebrate surveys can also require considerable laboratory time for sorting and identification.

Morphospecies
Assessment of biodiversity does not necessarily entail formal species identification, and instead may involve the identification of "morphospecies" that obtain informal codes or numbers instead of species names. Such studies are difficult if not impossible to compare with other studies carried out by different investigators, as the informal codes being used are typically not comparable. Comparison of vouchers, or the common use of keys or images, can, however, reduce these difficulties. However, it is important that vouchers of all taxa be deposited in an accessible museum collection for future workers to access.

Sorting and identification
Broad-based invertebrate surveys often amass a large amount of unworked material, reflecting inadequate methodologies and funding (Yen and Butcher 1997). While most of the terrestrial survey work in Australia has focused on higher plants and vertebrates, many being identifiable in the field, marine surveys are usually restricted to fishes and/or a few large invertebrates, such as molluscs, decapods, corals and echinoderms. Such surveys involve long periods in the field, and less time involved in laboratory-based identification, specimen handling, data analysis, and report preparation (Yen and Butcher 1997). In comparison, broadly based invertebrate surveys gather more specimens and taxa in relatively short periods in the field, whereas the sorting and identification for most groups is typically done in the laboratory and the need for the input of specialist taxonomists is much greater. Funding for invertebrate surveys must allow for the laboratory-based work that follows fieldwork, which can be many times the cost of actually obtaining the material.

It has been argued, that due to the diversity of invertebrates and the lack of taxonomists, future collecting should be focused on groups of "importance" and for which taxonomists are available. This argument is flawed (Yen and Butcher 1997), although some setting of priorities is necessary for practical reasons. Yen and Butcher (1997) argue (and we agree) that it is important that non-target taxa also be collected and vouchered because "our knowledge base will only be improved if we have material on hand for future study. Nobody can predict the future importance of collections of any particular group of invertebrates, even if they lie unstudied for many years". We whole-heartedly endorse this recommendation, if these "generalist" collections are adequately fixed and preserved to allow subsequent species determinations. This is unfortunately not the case for many existing "general" marine invertebrate collections.

7.6.1 Assessment of biodiversity for conservation

Conservation cannot depend on prior total inventory because this goal is practically impossible (see Section 7.6.3). Assessment of biodiversity for conservation purposes must address the following questions - the distribution and abundance of (representative) species present; what taxa must we conserve; how do we conserve them; and how do we monitor change?

There is an important distinction between surveys to assess biodiversity for conservation and conservation of threatened species. The former is aimed at conservation of a site or habitat while the latter is aimed at conserving site/habitat for a particular species (Yen and Butcher 1997).

7.6.2 Assessing the numbers of species of marine invertebrates

Estimates of the number of protist, plant and animal 'species' described vary from about 1.4-1.8 million (Stork 1988; Minelli 1993; May 1994; Stork 1999) while estimates of total diversity reach 30 million or more (see Stork 1999 for review). Such estimates are difficult to make and most have focused on terrestrial organisms, particularly because of the belief that insects are by far the most diverse group. Most species described to date are either insects (56%) or plants (14%) (May 1994), although this may simply indicate our ignorance of the marine biota (Clarke and Crame 1997).

Grassle and Maciolek (1992) estimated that the number of species of marine macrofauna on ocean floors may be 10 million, and, although they considered their estimate conservative, it was disputed by May (1992a). Their estimate was based on an extrapolation from the number of new species found from just 21 m^2 off the east coast of the USA to the whole area of the deep-sea. Poore and Wilson (1993) estimated marine species richness as possibly even higher than Grassle and Maciolek's (1992) figure, a claim also disputed by May (1993) who reassessed these data and suggested that there were probably only around 500 000 undescribed species. However, these estimates of marine species diversity are, of necessity, based on information obtained from sampling only a tiny percentage of marine habitats, especially for the small-sized taxa and the deep-sea. Thus, there are major problems extrapolating from the inadequate information that we have to the enormous unstudied areas and taxa resulting in possible orders of magnitude errors in available estimates.

Estimates of marine species diversity

Estimates for global marine species are discussed by Grassle and Maciolek (1992); May (1992a); Poore and Wilson (1993) and May (1993). These estimates rely on extrapolations based on relatively little detailed information. For example, Grassle and Maciolek (1992) based their figures on sampling of soft-sediment communities along a 176 km transect at 1500-2500 m depth off the east coast of the United States. They found a typical species accumulation (or 'rarefaction') curve which suggested, in the latter stages, about 100 species were being added per 100 km along their transect. They generalised this to one new species per square kilometre, giving a total of several hundred million macrobenthic species for the 3×10^8 km^2 of ocean floor deeper than 1000 metres. This was scaled back to a tentative estimate of about 10^7 benthic species because of the order-of-magnitude lower densities of living things known to occur in the deepest, most oligotrophic parts of the ocean. These estimates were criticised by May (1992a).

Poore and Wilson (1993) showed that Grassle and Maciolek's data were atypically low on a global scale and showed that the shelf faunas of southeast Australia are rich in species. This latter study collected samples in three areas along 50 km of the Victorian coast, from 11-51 m (in medium to coarse sand) and found that the fauna was extremely species rich – the 104 samples (= a total sample area of 10.4 m^2) yielded 60 258 individuals and 803 species. Few species were highly abundant and 51% of species collected are apparently undescribed. Such an area represents one of the richest marine habitats known worldwide.

The above estimates are based mainly on sampling in the deep-sea but shallow-water marine ecosystems are also much more diverse than indicated by the numbers of described species. Reaka-Kudla (1997) estimated that there were 93,000 described species living in coral reefs but that this might represent only 1-15% of the actual number of species. Gosliner (1996) estimated that 60% of all marine invertebrates in the Indo-Pacific were molluscs and Bouchet (2000) found 2781 species of molluscs at a site in New Caledonia and estimates that there are in excess of 3000 at another site. A single dredge site in the Swain Reefs contained over 1100 species of shelled molluscs (WFP, pers. observ.).

High species richness of benthic invertebrates in the shallow marine waters of southeast Australia has been recorded (Poore and Wilson 1993; Coleman et al. 1997) with 800 species found in a few square metres in Bass Strait and 700 in Port Phillip Bay sediments. Hooper et al. (in press) considered the distribution of Australian tropical sponges and found no evidence of latitudinal variation but rather that the diversity was related to environmental factors (such as substrate, water clarity and water currents). They also found that a large percentage of the fauna is apparently endemic to a particular area.

While benthic and coastal diversity levels are obviously high, the diversity of pelagic organisms is relatively low (Angel 1993; Pierrot-Bults 1997). However, genetic diversity in widespread pelagic species might be considerable and cryptic species may occur. Only 12 phyla contain pelagic taxa, and none are entirely so, with crustaceans being the most diverse group in marine macrozooplankton (Pierrot-Bults 1997). Price et al. (Price et al. 1999) showed that while echinoderm species diversities are similar in coastal zones and in the deep-sea, there are considerably greater differences in diversity between different shelf regions than between lower bathyal/abyssal regions. Their results do not support the idea that the deep-sea has exceptionally high diversity, as least for echinoderms.

7.6.3 What do we need to know? - the possible and the impossible

The Systematics Agenda 2000 report (Systematics Agenda 2000 1994) stated that total global inventory could be achievable if a 25 year program were instigated with an annual investment of approximately US$3 billion. This view is supported by some such as Wheeler (1995), who argued that with modern taxonomic methods, high capacity computers and supportive technologies, it should be possible to document the Earth's biodiversity. Others (e.g., New 1993; Yen and Butcher 1997; New 1999) consider that it is an impossible task to know all species, even with massive funding. New (1993) argued that the only realistically obtainable goal was to assess biodiversity in a focused, objective manner using selected priority groups.

The idea of assessing Australian biodiversity has been included in the NSCABD (see Section 7.4.2). Yen and Butcher (1997) argued that Australia was in a unique opportunity to achieve this.
However, there are two quite separate tasks -

- Obtaining basic information about the taxa; and
- Biodiversity assessment, including rapid biodiversity assessment (RBA).

Each of these is discussed briefly below.

Assessing Australian biodiversity

Yen and Butcher (1997) stated that the main issues, besides funding, staffing and appropriate infrastructures to facilitate and coordinate the work, are:

- Determining priority taxa; and
- Developing rapid biodiversity assessment methods[171].

Obtaining basic information about the taxa
This aspect of biodiversity assessment includes information mainly concerning species-group taxa that includes taxonomy (description, name), systematics (classification, phylogeny), distribution, biology and ecology. Due to the magnitude of the task, New (1994) and Yen and Butcher (1997) advocated the determination of ecologically important priority taxa (criteria being: taxonomically well known, high diversity, geographically widespread, abundant/dominant, etc.) and establishing an infrastructure to coordinate the work (possibly ABRS). They suggested that the determination of priority taxa will require some research and that this will require collaboration between taxonomists and ecologists. Further, they suggested that the establishment of coordinated sorting centres would assist in meeting this objective. These are all laudable objectives but we caution against the emphasis on "ecologically important" as a major criterion as this can lead to oversimplification and may not take into account regional and ecosystem diversity - for example, corals are not ecologically important in the southern half of Australia but are of utmost importance in tropical seas.

Rapid biodiversity assessment (RBA)
Biodiversity assessment protocols include rapid biodiversity assessment (RBA) and work on methods for freshwater (Cranston and Hillman 1992; Trueman and Cranston 1994) and terrestrial invertebrates (Andersen 1990; Beattie et al. 1993; Trueman and Cranston 1994; Andersen 1995; Oliver and Beattie 1996) are relatively well documented. In addition, the Marine BioRap Guidelines (Rapid assessment of marine biological diversity) have been developed (Ward et al. 1998b) and its application is discussed by Faith (2001). Such developments were identified as an Action (4.1.2) in the NSCABD. The stated aims are to:

- Establish a joint Commonwealth and State and Territory program to carry out rapid assessment of Australia's biological diversity. This will include:

[171] Although recognising that some marine taxa cannot be differentiated accurately to species-level without employing specific techniques, such as histology or SEM examination.

- Strengthening the network of biological diversity laboratories and agencies to assist in the rapid processing of collected material[172];
- Research into and development of new methods of processing large numbers of unidentified organisms;
- Research into environmental and biological models and groups of organisms that may be used to assess biological diversity.

Rapid biological diversity assessment is described in the NSCABD as using "a range of methods that facilitate rapid field survey work and classification. The fieldwork normally involves a multidisciplinary team, including experienced field scientists and people with local knowledge, in surveying component groups representative of biological diversity. Techniques and procedures are employed to quantify the variety of organisms collected by classifying them into recognisable taxonomic units (RTU's). These techniques overcome the large time requirements of formal classification. By establishing the relationship between these recognisable taxonomic units and the formal species they represent, rapid estimates of biological diversity will become available." However, many RBA protocols do not subscribe to species level RTU's but use higher categories (genus, family, order etc.) in assessing diversity. This issue of *taxonomic resolution* is contentious but ultimately depends on the question being asked.

RBA methods have been developed for marine organisms including invertebrates (e.g., Ward et al. 1998b) and are useful for certain types of assessment (monitoring, impact assessment, recovery, comparison of diversity in different areas etc.). However, there are many shortcomings. In particular they are limited in their ability to provide suitable information regarding particular species-group taxa and comparisons between many studies can be difficult or impossible because of the lack of standard protocols, uniform taxon identifications and lack of, or lack of access to, vouchered material.

The aims of some RBA's include monitoring to detect change. Probably the best known of these that involves invertebrates is the *National River Health Program*, instigated by the Commonwealth Government in 1992. Fairweather and Lincoln Smith (1993) discussed the difficulty of assessing environmental impacts prior to them occurring and discuss some solutions to this problem. Ward (2000) described indicators for the assessment of sustainability in Australian marine ecosystems.

Other RBA methods use non-taxonomic approaches and surrogates.

***Shortcomings*:** The methods employed in RBA will clearly depend on the aim of the assessment so data are not necessarily compatible. In addition, surveys involving a variety of groups of invertebrates will need to employ a number of different collecting, fixation and taxonomic analytical techniques. These vary according to the group being targeted and the habitats involved. Emphasis on determination of higher taxonomic categories (e.g., order, family etc) has no value for the assessment of threatened species. In addition, even if morphospecies-level identifications are made, rare species may be

[172] The major flaw in this approach is that many conservative groups may have major genetic differences not necessarily manifested at the morphological level.

missed or excluded as noise. Although they may arguably be of secondary importance in community function, rare species can constitute a large proportion of species diversity.

***National River Health Program*:** This involves monitoring standard sites using standard methods over several years, the aim being to detect changes in diversity that might indicate changes in river health. Taxa are identified to family level so the data are effectively useless for other types of biodiversity assessment such as determining changes in the distribution of particular species.

Non-taxonomic approaches: Saiz-Salinas and Ramos (1999) used "biomass size spectra" as a non-taxonomic RBA method to studying macrobenthos in Antarctica (South Shetland Islands). They argued that this was a useful alternative for understanding the functioning of such ecosystems because the taxonomic approach is time-consuming, requires expertise and is difficult in remote areas like the Antarctic where invertebrate taxonomy is still in early stages. The size spectrum was bimodal in shallow waters (<100 m) due to the presence of large filter feeders and unimodal in deeper waters. They argued that this suggested a predictable and regular supply of food in shallow water (allowing accumulation of biomass in larger organisms) and more unpredictable / fluctuating food supplies in deeper waters.

Surrogates: RBA methods often advocate the use of particular groups of invertebrates as surrogates for other invertebrates (e.g., ants are used in some terrestrial surveys as a surrogate for all terrestrial invertebrates) a protocol that, in our opinion, greatly reduces the effectiveness of the method given the huge diversity in habits, biology and habitat of invertebrate taxa. Whether vegetation, substrate or certain species can be used as reliable surrogates for the whole community is very debatable. For example, the faunal composition of saltmarshes in Tasmania could not be predicted by the plant communities, nor were there any strong relationships between individual animal and plant species (Richardson et al. 1998).

***Taxonomic resolution*:** The level of taxonomic resolution that should be used for bioassessment has been debated in recent literature (e.g., Warwick 1988a, 1988b, 1988c; Smith and Simpson 1993; James et al. 1995; Vanderklift et al. 1996; Chapman 1998; Olsgard et al. 1998; Hutchings 1999a). This concerns the level samples should be identified at and is increasingly being done to major groups (phylum, class, order or families) in order to minimise costs and sorting time because much of the fauna is unidentified or difficult to identify by "non-specialists". There is no single answer to this problem as it depends on whether or not the question being asked can be resolved by the level of resolution adopted. Thus, for example, faunal changes resulting from significant impacts (e.g., oil spills and other serious pollution)[173] can probably be detected at any level of resolution, as can faunal diversity along strong environmental gradients (e.g., salinity and substrate changes along an estuary). Finer resolution examining more subtle

[173] A study of the impacts of sand mining (for diamonds) in depths of 110-130 m of the coast of Africa (Savage et al. 2001) showed that major disturbance could not be readily detected using phylum-level meta-analysis. The authors suggested that this approach may be better suited to detecting organic or chemical impacts rather than physical disturbance.

impacts, looking for early warning of impacts, or trying to detect differences between similar habitats in different locations will usually require resolution at species level. In addition, Hutchings (1999a) points out that we need to take into consideration that:

- Most of our knowledge of how organisms function is at species level and may not be relevant to other members of that family or genus; and
- There are important functional levels below species (i.e. populations), and there may exist significant differences between populations of the same species.

Thus, while short-cut approaches involving the use of higher taxonomic categories in RBA may be appropriate for some questions - such as detecting significant impacts, they are not appropriate, for example, in assessing changes in dominant species or more subtle changes.

7.7 Monitoring and the use of invertebrates as indicators

7.7.1 Monitoring

Monitoring assesses how, and to what extent, biodiversity changes over time (Stork and Samways 1995) and requires baseline inventory information (Yen and Butcher 1997) (see Section 7.6), replication and the use of control sites (e.g., Underwood 1991, 1995, 1997; Underwood and Chapman 1999; Chapman 2000; Underwood 2000). The aims of any monitoring program must be clearly defined and the groups used for monitoring must be carefully selected so that they are sensitive to the environmental changes being targeted (Yen and Butcher 1997). In addition, the background natural variability must be assessed and natural disturbances taken into account (Yoccoz et al. 2001). The conduct of such studies requires specialised training and careful attention to methodologies. Fortunately, there is no shortage of excellent, well-trained marine ecologists in Australia.

Measuring changes in community composition and assessing their significance (e.g., in relation to anthropogenic environmental impacts) is complicated by spatial and temporal heterogeneity and a general inability to predict outcomes for communities (e.g., after a stress or a removal). Problems identified by ecologists include the lack of baseline data, the identification of natural variation including difficulties separating “natural” versus “anthropogenic” effects, and they caution against the extrapolation of results from short-term studies. The level of taxonomic resolution necessary in monitoring programs has also been frequently discussed. Each of these issues is discussed briefly below.

Long-term studies

There is a need for long term studies to assess population variability in space and time (Lundberg et al. 2000) but, unfortunately, such studies are usually constrained for logistical and other reasons. Since so much variation is reported over inter-annual, seasonal or even shorter periods it is important to analyse data about changes over long time scales but this has been done very rarely, one of the few studies being that of Wilson et al. (1998) who assessed changes in benthic communities of Port Phillip Bay between 1969 and 1995.

Long-term monitoring of even simple systems or populations of single species of marine invertebrates is rare. Addessi (1994) notes that there are many observations on marine communities over time where significant declines in the populations of some species have been reported, but argues that usually there is not enough data to confirm or refute these impressions (e.g., Thorne-Miller and Catena 1991). Long-term, properly replicated field data, using controls (e.g., Underwood 1991, 1997; Underwood and Chapman 1999; Chapman 2000; Underwood 2000; Underwood et al. 2000), are needed to understand the distribution and abundance of species if we are to document changes through time in disturbed and undisturbed habitats. Sampling regularly over time at the same season at the same site, is highly recommended for determining the actual status of species or communities under the influence of increasing human disturbance. Even simple systems can be spatially patchy and temporally dynamic over short time scales (few years to decades)[174]. In addition, monitoring sites within habitats such as coral reefs can be effected by more than one disturbance regime, this having implications for the interpretation of results (Done 1997).

Another issue is whether the monitoring program itself is causing disturbance. Stephenson and Cook (1979), for example, in a survey of changes in the macrobenthos of Moreton Bay, questioned whether the grab sampling itself was depleting the macrobenthos.

Spatial and temporal variability versus anthropogenic and versus natural changes

There is considerable difficulty in distinguishing between “natural / unaffected” versus “impacted” communities, or between natural versus anthropogenic changes, given the high levels of spatial and temporal variability. Coral reefs, for example, are both highly diverse and dynamic, with their community structure varying greatly spatially and temporally. The extent to which this variation can be measured determines our ability to detect change, identify stresses, distinguish between anthropogenic and natural disturbances and make well-informed management decisions (Cumming 1999). Knowledge of the scales of natural variation in organisms is thus fundamental to understanding and managing the many pressures on marine ecosystems.

Hughes and Connell (1999) posed the question of how to recognise a “healthy” versus a “sick” or stressed coral reef, and argue that with few exceptions, the time-scale of most studies has been too brief to determine long-term trends. Instead, numerous researchers have employed a variety of indicators of the status of a reef, the most common being abundances of key organisms (e.g., corals, fish), population structures or species composition (e.g., species diversity, evenness), or physical or chemical variables (e.g., turbidity, nutrients). The problem with these approaches is in defining normal values, given the considerable spatial and temporal variability of many of these parameters. For

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[174] Butler & Connolly (1999) reported on changes in fouling organisms on the piles of a pier in South Australia over several years. Although species composition differed along the pier, probably reflecting environmental differences, the communities appeared to change in a similar way through time. Abundances of key taxa fluctuated markedly from site to site at any one time, and through time. The overall composition of the assemblage had not stabilised after 6.5 yrs; and it was unclear whether it had done so by 13.5 yrs.

example, coral cover on fringing reefs on the inner Great Barrier Reef can approach 100%, despite the turbidity and rate of sedimentation being higher there than that which kills most Caribbean corals (Stafford-Smith and Ormond 1992).

Patterns and mechanisms observed over 25 years in a large kelp forest suggest that definition of a meaningful benchmark is impossible (Dayton et al. 1998). This is because many of the larger animals (e.g., sea otters, various fishes, abalones) have been removed and others (crabs, snails, sea urchins) severely depleted by fishing. In addition, kelps are sensitive to large-scale, low frequency climatic events (such as El Niño). The plants continue to exist without any obvious effects from animal loss. There is very little documentation regarding what the natural community was like. Thus, in a case such as this, the ability to separate anthropogenic impacts from the "natural" dynamics of the system is severely compromised.

There is a need to identify expected values for indicators of ecosystem health and the expected response to increasing stress at global, regional and national levels. These are difficult to assess because of the temporal and spatial variability of communities but are priorities identified by The International Coral Reef Initiative (Done 1997). Done (1997) explores the issues of the expected status of coral reefs as benchmarks against which human influence must be measured, and identifies maintenance of reef framework and reef-building capacity as key criteria for reef health. He points out that there is a need to understand whether ‘normal’ changes have been prematurely initiated by human influence, or whether human influence has caused an abnormal change. Management or mitigation of impacts would be necessary in both these cases. Done (1997) also argues that, for large-scale, long-term monitoring programs, that a reasonable proportion of sites in various states and stages be included. He cites as examples of difference between changes in ‘stage’ and 'state’. Periodically storm-disturbed sites in Hawaii recover towards hard-coral dominance, i.e. at all times they were in some stage of a potentially reef-building succession, whereas on Jamaican reefs, Hughes (1994a) demonstrated a change in ‘state’ from hard coral dominance (a reef-building state) to algal dominance (a non-reef-building state) resulting from a combination of human impacts and natural disturbance.

In a study on the patterns of recruitment and abundance of corals along the Great Barrier Reef, Hughes et al. (1999) found a high predicability in the number and cover of adult spawners at the scale of sectors and reefs on the GBR. However, they point out that this does not mean that the processes (for example recruitment and mortality) controlling abundance are uniform. They further suggest that management of these resources must be based on a sound knowledge of process and mechanisms rather than one centred on monitoring patterns of abundance.

Need for baseline data

We have little idea of the full extent to which anthropogenic impacts such as fisheries have reduced populations of marine species, or effected ecosystems, because we have no proper baseline against which to judge declines (e.g., Roberts 1997). The detection of trends in ecosystems depends upon (1) a good description of the foundation or benchmark against which changes are measured and (2) a distinction between natural and anthropogenic changes (Dayton et al. 1998). This is particularly true in marine systems because they are sensitive to the availability of nutrients and often show considerable natural variation. However, measures of benchmarks can be "represented by a moving target of reduced expectations of what really is the natural standard. In addition, rare, episodic events have important consequences that result in a highly variable baseline amidst much background noise" (Dayton et al. 1998). This is illustrated in the kelp forest example above, where base line studies are not necessarily recording a "natural" state and are not necessarily meaningful unless they encompass a considerable time span. They only assist in measuring changes that have occurred in an ecosystem between the time of the initial and subsequent studies. Without unimpacted control areas, it is very hard to gauge the true impact of activities such as fishing. This explains, for example, the difficulty of demonstrating that trawling damages benthic communities - these communities have been heavily modified historically and so, unsurprisingly, further experimental trawling produces little change (Roberts 1997).

Taxonomic resolution

There is debate as to the appropriate level of taxonomic resolution that should be used in monitoring studies (see 7.6.3). Some authors have suggested that data at family, ordinal or even phylum level may be sufficient to detect impacts (such as pollution). Vanderklift et al. (1996) examined the effect of reducing taxonomic resolution on ordinations to detect pollution-induced gradients in macrobenthic infaunal assemblages. While the patterns associated with both habitat and pollution gradients were visible in ordinations based on data pooled to higher taxonomic levels, the similarity among ordinations based on different sets of pooled data was not fully supported by more detailed analyses. Smith and Simpson (1993) examined the discrimination at five taxonomic levels of the effects of pollution on kelp holdfast macrofauna. The samples, from a gradient of domestic effluent at Coffs Harbour, NSW, showed separation of close outfall sites from controls at all taxonomic levels. In contrast, the effect due to sites within locations, although significant up to order, was not significant for analyses at higher levels. Smith and Simpson (1993) argue that their results support the hypothesis that anthropogenic effects modify community structure at higher taxonomic levels more than natural environmental variables and conclude that studies *of impacts* that focus on higher taxa may be more readily interpretable than those conducted at the species and family levels.

Monitoring and the NSCABD

Monitoring is identified as an Action (4.1.7) in the NSCABD. This strategy aims to "Establish a national coordinated program of long-term ecological monitoring to document patterns of change or lack of change in order to establish a baseline for understanding the impact of such change or lack of it on natural communities, ecosystems and ecological processes, and to detect changes in biological diversity and their causes." It states that the program will:

- Combine remote sensing[175] with a national network of secure field-based monitoring sites in representative habitats;
- Develop and encourage the application of national monitoring protocols involving standardised sampling designs and techniques for testing management regimes and strategies, including rehabilitation and reintroductions;
- Use biological diversity indicator groups to reveal the impacts of environmental disturbance;
- Establish properly constituted and supported assessment panels or monitoring committees, or both, comprising representatives of industry, non-government conservation organisations, other appropriate community groups and governments;
- Accelerate research into new, cost-effective methods of monitoring; and
- Integrate with an ecological research program aimed at improving our understanding of long-term and event-driven ecological processes.

7.7.2 Indicators and surrogates.

The concept of using indicators of ecosystem health and biodiversity assessment is discussed by Yen and Butcher (1997: 185-187). This approach is commonly used in a de facto way in marine ecosystems where one "keystone" group (e.g., fishes, corals) is considered an indicator of ecosystem health.

While a range of benthic marine invertebrates (such as corals) can and have been used as indicators of water quality and ecosystem health, vegetation such as seagrasses have also been used (e.g., Dennison et al. 1993; Lee Long and Coles 1997).

Fairweather (1999), in a discussion on the possible use of indicators of ecosystem health in estuaries, pointed out their potential use as legal evidence (e.g., checking compliance), for determining the utility of policy decisions or regulation, and for documenting degradation. They suggest that criteria for indicators of the assessment of ecological condition might include accordance with some hypothetical idea, legally fulfil mandated standards for environmental quality, be in harmony with observed background levels and variability of measured variables, be within theoretical limits of acceptable change, be below socially mandated thresholds of unacceptability, and meet some desired target set in advance. While in theory this seems like a reasonable approach, the use of indicator taxa will always be contentious, mainly because no surrogate(s) will adequately reflect the full range of responses exhibited by all members of the assessed community.

Attayde and Bozelli (1998) assessed the indicator properties of zooplankton assemblages to disturbance gradients by canonical correspondence analysis in a coastal lagoon. They found that changes in zooplankton species composition were significant indicators of environmental heterogeneity patterns. They found that one rotifer genus was a better indicator of these environmental gradients than the entire zooplankton assemblage and suggested that taxon could be used in monitoring and conservation planning. These

[175] Remote sensing is used successfully for assessing many marine habitats - for example, sea level rise (Cabanes et al. 2001), coral reef communities (Dustan et al. 2001), mass mortality of sea urchins (Phinney et al. 2001) and monitoring tidal flats, seagrasses and mangroves (Robinson et al. 2001).

authors suggest that their methods to assess the indicator properties of species assemblages and to select target taxa can be widely applied in any aquatic ecosystems to any group of organisms, spatial and temporal scales, and environmental gradients.

Environmental Indicators for National State of the Environment Reporting[176]

A stated aim is to "develop a set of environmental indicators that, properly monitored, will help us track the condition of Australia's environment and the human activities that affect it." To help develop these indicators, Environment Australia has commissioned reports recommending indicators for each of the seven major themes around which Commonwealth state of the environment reporting is based. The relevant ones are Biodiversity (Saunders et al. 1998) and Estuaries and the Sea (Ward et al. 1998a). The latter contains papers dealing with nutrients, seagrasses and macroalgae. The inclusion of some data on invertebrates in the future is recommended.

7.8 Main issues and recommended actions

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Issues

There are many reasons why it is imperative that we increase our knowledge of marine invertebrates. These include:

- They form the great bulk of marine biodiversity;
- They are already of considerable economic importance and in some areas, especially as a source of useful biochemical compounds, their potential has hardly been realised;
- Under UNCLOS, Australia is entitled to claim a large amount of extra territory within the AMJ if it can demonstrate that it has the necessary scientific understanding;

Good quality information is required if effective decisions are to be made. However, the state of knowledge of marine invertebrates is generally very poor with:

- At least half of all marine invertebrate species not yet formally described, and some groups remain almost completely unknown;
- Little or nothing known about the ecology and biology of the great majority of Australian marine invertebrates;
- Many parts of the Australian marine environment unsampled or poorly sampled for invertebrates, including virtually all areas below 3000m; and
- A lack of accessibility to or co-ordination of existing information.

Most marine invertebrates remain very poorly known, and existing information difficult to access, for a number of reasons:

- Surveys of marine organisms tend to focus on fishes and, in some cases, the more conspicuous members of the invertebrate fauna (e.g., corals, macro-molluscs).
- There are very few experts on marine invertebrates in Australia, despite the diversity of the fauna;

[176] http://www.ea.gov.au/soe/publications/envindicators.html

- The few available guide books deal with only a few of the common species and most of these are regional in focus; and commercial in outcome;
- The relevant literature is mostly in relatively obscure scientific publications;
- For most groups there are no up to date authoritative checklists of names;
- Public institutions such as museums and universities have totally inadequate resources devoted to the study of marine invertebrates while CSIRO and AIMS are largely focused on fisheries and coral reefs respectively.
- Ship time is so expensive and difficult to obtain that it is beyond the reach of most marine invertebrate workers.
- Indigenous knowledge regarding marine invertebrates is poorly understood.
- Whole animal and taxonomic studies have declined in universities in recent decades.

The information that is known about Australia's marine fauna is only accessible to a few experts. We see knowledge sharing and access to information as key issues that need to be addressed as quickly as possible.

Recommended actions

Access to information urgently needs to be improved. This can be done by:

- Web-based databases be constructed that provide basic information at a national level, preferably linking to other similar international databases.
- The production of identification guides (including interactive keys, handbooks etc.) for at least the major groups of shallow-water marine invertebrates be facilitated.
- National checklists be fast-tracked (as a basis for a more general information system) for marine groups using ABRS's ABIF facility. These should be regularly updated and should serve as a means of ensuring that taxonomic consistency is achieved.

Facilitate access to the collections held in state museums (the greatest source of information on the diversity and distribution of marine invertebrates) by:

- Databasing and validation of records; and
- Linking museum databases into a national virtual facility.

The critical importance of taxonomy and systematics in underpinning the rest of the research and knowledge base must be recognised.

- More invertebrate taxonomists must be trained and tenured positions established for them in museums, universities, marine research institutes and fisheries departments;
- An increase in the competitive funding available for taxonomic research (e.g., by increasing the allocation for grants for ABRS); and
- Encourage and facilitate phylogenetic studies which are an integral part of the knowledge base required for making informed management decisions regarding conservation.

Use invertebrates in assessment and monitoring:

- Implement surveys and inventories to determine biodiversity baselines that will serve as benchmarks for assessing change in faunal composition, assisting in the detection of introduced taxa and community change due to anthropogenic impacts. These must:
    - Use rigorous sampling protocols;
    - Cover a wide range of invertebrates; and
    - Be conducted in representative habitats and in major biogeographic areas around Australia.
- Identify relatively pristine (unimpacted) areas to use as control sites (de facto "baselines") because:
    - Successfully using invertebrates as indicators of, or to monitor, the health of ecosystems will require rigorous approaches including sampling that allows the assessment of natural variation, the use of external controls etc.

Test the validity of using surrogates such as sediment, marine vegetation, a particular invertebrate taxon, etc. as the basis for predicting benthic invertebrate communities before wide application.

Recognise that a knowledge of processes and mechanisms is critical for the management of natural resources.

Increase the number of long-term studies.

CHAPTER 8 – LEGISLATION, POLICY AND ADMINISTRATION

Different jurisdictions, in various countries and at all scales from locally to internationally, have taken a variety of different approaches to the conservation of natural resources and the protection of threatened fauna and flora. However, one of the most common approaches has involved the listing of threatened species (see Chapter 4). Depending on the definition used for "animals" and "wildlife" under these instruments, there may or may not be the possibility of listing marine invertebrates as "threatened species" or "protected wildlife".

The role of species protection legislation, as it relates to invertebrates, was discussed by Yen and Butcher (1997) and Hutchings and Ponder (1999). Yen and Butcher outlined various arguments for and against protective legislation.

Arguments in favour of protective legislation include:

- It provides a means of controlling (stopping or reducing) the illegal killing or collecting of listed species;
- Invertebrate animals should be given the same legislative protection that vertebrates have benefited from;
- It provides an impetus for obtaining more accurate data, enabling the development and implementation of relevant recovery plans;
- Protected invertebrates could be used as "flagship" taxa to raise awareness of invertebrate conservation; and
- Lists of threatened invertebrates can raise public awareness of the diversity and conservation needs of invertebrates.

Arguments against protective legislation include:

- Protection will not be effective unless relevant habitat is protected or unless threatening processes causing decline are stopped and the decline reversed;
- A management focus regarding listed species is to protect them from collectors, but there is little evidence to prove that properly controlled collecting will cause serious decline in most invertebrate species;
- It can create a black market trade in genuinely rare species in those groups of interest to collectors (notably some mollusc species);
- Trade in invertebrates is more difficult to control because they are easily transportable and identifications are easily falsified because they are difficult to identify; and
- Because of the large number of invertebrate species, there is the danger of long lists of protected invertebrate species eventuating. This would (a) be politically difficult to accept, (b) be practically difficult to enforce, and (c) discourage useful gathering of information by scientific collectors.

Shortcomings of legislation

Yen and Butcher (1997) argued that the value of protecting invertebrate species without protection of their habitat is at the very least dubious. Writers as early as Rawlinson

(1981) highlighted the deficiencies in protective legislation in Australia with its emphasis on protection of individuals rather than habitat and control of threatening processes, and emphasis on the conservation of rare and endangered species. Legislative protection alone will not ensure conservation of a species. For invertebrates in particular, protection *per se* is not generally synonymous with conservation, and in most cases is not effective (Yen and Butcher 1997). Yen and Butcher (1997) identified the priorities for species known to be at risk as: legal protection, identifying and alleviating the causes of decline, public education and involvement, and appropriate recovery plans. However, for the vast majority of invertebrates, conservation efforts need to focus more on a habitat protection approach rather than the individual species approach for the reasons outlined in Chapters 4, 5 and 6. Even passive habitat protection (such as reserve establishment), while a major first step, may be inadequate unless there is active management to reduce the causes of decline. Commonwealth, and some State, legislation now have provisions for the nomination of endangered populations, critical habitats and (in some cases) vulnerable communities, as well as key threatening processes (see Table 8.2).

The emphasis on protective legislation that promotes protection of individual organisms, as opposed to species protection, is based on the often false assumption that targeted collecting (or "harvesting") (with the exception of some commercially fished taxa) is the major threatening process to the species (Yen and Butcher 1997). Authorities responsible for the management of marine parks restrict the collecting of marine invertebrates and in permit applications, the exact number of individuals to be collected has to be given *a priori*, even though in the great majority of cases neither the research worker or the authority have any real knowledge of population levels or sustainable take. In the case of commercial species, or species targeted for food or bait, bag limits may be in force or restricted collecting seasons imposed (see Section 6.10).

8.1 International agreements – Conventions, Treaties and Protocols

Australia is a party to a wide variety of international agreements[177], some of which (such as CITES and the World Heritage Convention) have direct relevance for the conservation of marine invertebrates and their habitats. Others, while not directly relevant, nevertheless have some bearing, for instance through habitat protection or the control of threatening processes. Examples of the latter include:

- The Climate Change Convention 1992,
- The Convention on Wetlands of International Importance especially as Waterfowl Habitat (the RAMSAR Convention) 1971,
- The International Convention on Prevention of Marine Pollution by Dumping of Wastes and Other Matter (the London Dumping Convention) 1972,
- The International Convention for Prevention of Marine Pollution from Ships (MARPOL) 1973, and
- The United Nations Convention on the Law of the Sea 1982 (although Article 61 is of direct relevance).

[177] The Australian Treaties Library, contains a list of bilateral and multilateral treaties to which Australia is a party, can be found on the web at http://www.austlii.edu.au/au/other/dfat/.

The relevance of a range of international agreements to Australia's management of the marine environment was reviewed in a Background Paper to the Oceans Policy (Herriman et al. 1997), and (in more abbreviated form) in Parker (1995).

A summary of those International Agreements relevant to marine invertebrate conservation is provided in Table 8.1. Several of the more important international instruments, from the point of view of marine invertebrate conservation, are listed and discussed in more detail below.

Convention on International Trade in Endangered Species of Wild Fauna and Flora 1973 (CITES)

The aim of the Convention on International Trade in Endangered Species of Wild Fauna and Flora 1973 (CITES) is to establish worldwide controls over trade in endangered wildlife and wildlife products by listing species in three categories:

1. CITES Appendix I – all species threatened with extinction which are or may be affected by trade. Trade in these taxa is strictly regulated and commercial trade virtually prohibited.
2. CITES Appendix II - taxa likely to become endangered if trade is not strictly managed. Appendix II species can be commercially traded, but trade requires an export permit.
3. CITES Appendix III – species which any Party to the Convention identifies as being subject to regulation within its jurisdiction.

Australia became a signatory to CITES in 1974 and, as required, the first official list of endangered vertebrate animals was prepared in 1980.

One shortcoming of CITES is that there are a number of practical problems associated with gaining protection. The requirement that a species be nominated by a State Party and accepted by the Conference of the Parties has meant that the species listed are mainly those that do not have major commercial value, since there is considerable lobbying by industry for states not to nominate commercially exploited species (Herriman et al. 1997).

Another problem, discussed by Green and Hendry (1999), is that there is a widespread failure to record coral species on CITES trade permits, despite a legal obligation to do so, due to the complexity of coral taxonomy and the volume of coral specimens in trade. Green and Hendry (1999) experimentally tested the accuracy of non-specialists in identifying coral genera, using existing guides, and found that only 3 of 10 genera were identified to more than 67% accuracy. They recommended that guides designed specifically for this purpose, and a change in emphasis under CITES to genera rather than species, would improve the effectiveness of CITES in monitoring trade.

Invertebrates listed under CITES

Invertebrates listed under CITES include:

- All members of the family **Tridacnidae** (giant clams)
- ***Strombus gigas*** (Caribbean Queen conch)
- All corals in the orders **Coenothecalia** (1 species – *Heliopora coerulea* – Indo-Pacific blue coral), **Scleractinia** (reef-building corals), **Antipatharia** (black corals) and in the family **Tubiporidae** of the order Stolonifera (organ-pipe corals)
- Hydrocorals of the families **Milleporidae** (stony or fire corals) and **Stylasteridae** (stony or lace corals).

All of these are listed under Appendix II of the Convention.

World Heritage

The Convention for the Protection of the World Cultural and Natural Heritage 1972 ("World Heritage Convention") enables sites to be listed on the World Heritage List as natural heritage if they meet one or more of the following criteria:

- Outstanding examples representing the major stages of the earth's evolutionary history; or
- Outstanding examples representing significant ongoing geological processes, biological evolution and man's interaction with his natural environment; or
- Contain superlative natural phenomena, formations or features; or
- Contain the most important and significant natural habitats where threatened species of animals or plants of outstanding universal value from the viewpoint of science or conservation survive.

Nominations of Australian sites for the World Heritage List can only be made by the Australian Government. The Commonwealth is obliged to protect world heritage values, but States and Territories have no legislative obligations, although in most cases, management of these areas is (at least partially) in the hands of State authorities. Australian marine areas currently listed as World Heritage Sites include the Great Barrier Reef (Queensland), Shark Bay (Western Australia), Lord Howe Island, Heard and Macdonald Islands and Macquarie Island. All of these areas are important in terms of marine invertebrate diversity and conservation.

Biological diversity

The Convention on Biological Diversity 1992 has as its objectives the conservation of biological diversity, the sustainable use of its components, and the fair and equitable sharing of the benefits arising out of the utilisation of genetic resources. However, it has been criticised as having achieved little real progress. For instance, Guruswamy (1999) has argued that by rejecting legally binding environmental obligations and embracing highly qualified 'soft' commitments, the CBD as conceived is inherently flawed. For instance, it "(1) rejects the concept of sustainable development by prioritising economic growth over environmental protection; (2) denies state responsibility for damage to the global commons; and (3) repudiates the idea that the plant, animal, insect and genetic resources of the world (our biodiversity) are the common heritage of humankind and that it is the responsibility of the community of nations to protect this heritage" (Guruswamy 1999).

RAMSAR Convention

Some coastal habitats, especially mud flats and saltmarshes, are protected under agreements made under the RAMSAR Convention (the Convention on Wetlands of International Importance Especially as Waterfowl Habitat 1971). Although it is used primarily to protect wetlands of importance (for feeding or breeding) to birds, there is no reason why invertebrates cannot be used to identify wetland sites of conservation importance (Collins 1987). After all, these waterbirds feed, in large part, on the invertebrates of these habitats.

Other agreements (relating to habitat protection)

A number of other, bilateral agreements have been made which reinforce the RAMSAR Convention as well as extending Australia's commitment to protect migratory birds other than waterfowl, birds in danger of extinction, and their environments. These, which include an Agreement for the Protection of Migratory Birds and Birds in Danger of Extinction and their Environment between the Government of Australia and the Government of Japan 1974 (JAMBA), as well as one between the Government of Australia and the People's Republic of China 1986 (CAMBA), could be of indirect relevance to marine invertebrates through the protection of coastal habitat.

Table 8.1: International Conventions, Treaties and Protocols relating to protection of the marine environment and / or marine biodiversity

Instrument	Date of entry into force	Description / Purpose / Relevance to marine invertebrates or their conservation	Given effect in Australia by: (legislation)	Provisions (relevant to marine invertebrate conservation)
General				
Convention on the Continental Shelf 1958	Entry into force generally 10 June 1964	Sedentary marine organisms are protected from unjustified interference arising from the exploitation of gas, petroleum and other minerals on the Continental Shelf.	*Continental Shelf (Living Natural Resources) Act 1968*	The Minister may prohibit the taking of species, restrict the harvesting of certain species or sizes, restrict fishing gear as well as restricting the quantity of catch. Licences may be issued permitting the holder to take species in such quantities as specified. Does not apply to marine organisms subject to joint fishery agreements between the Commonwealth and the States
United Nations Convention on the Law of the Sea (UNCLOS) 1982	Entry into force for Australia 16 November 1994	This is a framework convention governing all aspects of ocean space and its uses, such as delimitation, environmental control, marine scientific research, economic and commercial activities, transfer of technology and the settlement of disputes relating to ocean matters. It allows coastal states to claim six maritime zones: Territorial Seas, Contiguous Zones, Exclusive Economic Zones (EEZ), Continental Margins, Archipelagic Waters and Internal Waters. Chapter XII of the Convention covers protection and preservation of the marine environment. It imposes international obligations to conserve and sustainably manage living and non-living resources in the Australian EEZ, the continental shelf and the territorial sea. The Convention also imposes obligations with respect to preventing, reducing and controlling pollution and also in preventing transfer of pollution through technologies or introduction of harmful species.	*Maritime Legislation Amendment Bill 1993* (Parts II, V and VI)	General relevance

Instrument	Date of entry into force	Description / Purpose / Relevance to marine invertebrates or their conservation	Given effect in Australia by: (legislation)	Provisions (relevant to marine invertebrate conservation)
Species and biodiversity conservation				
Convention on Biological Diversity 1992	Ratified by Australia 18 June 1993	The objectives of this convention are: the conservation of biological diversity, the sustainable use of its components, and the fair and equitable sharing of the benefits arising out of the utilisation of genetic resources. This includes appropriate access to genetic resources and, by appropriate transfer of relevant technologies, taking into account all rights over those resources and technologies.		General relevance
Convention on International Trade in Endangered Species of Wild Fauna and Flora 1973 (CITES)	Entry into force for Australia 27 October 1976	This Convention regulates trade in species threatened with extinction. There are three Appendices, each of which contain a list of species afforded certain protection from trade and are subject to varying degrees of regulation. Appendix I includes all species threatened with extinction which are or may be affected by trade; the export, import or re-export of these species requires a permit or certificate issued by the export, import and re-export state. These can be issued only if certain conditions are met, including that the trade will not be detrimental to the survival of the species. Appendix II includes species which although not now threatened with extinction may become so if trade is not strictly regulated, and other species that must be regulated in order to bring trade in the first category of species in the Appendix under effective control; the export of these species requires only an export or re-export permit or certificate. Appendix III includes all species that any State Party identifies as being subject to regulation within its jurisdiction.	The Environment Protection and Biodiversity Conservation Amendment (Wildlife Protection) Act 2001[178]	The object of this legislation is to comply with the obligations of Australia under the Convention and otherwise to further the protection and conservation of the wild fauna and flora of Australia and of other countries.
Convention on Conservation of Migratory Species of Wild Animals (Bonn Convention) 1979	Entry into force for Australia 1 September 1991	This Convention aims to provide a framework mechanism for international cooperation for the conservation and management of migratory species, and to identify endangered migratory species in need of urgent conservation measures at national level. Appendix II contains species to be subject of cooperative international conservation and management agreements. The Convention has mostly been used to list marine mammals such as whales and dolphins; some terrestrial invertebrates have been listed (e.g., the Monarch or Milkweed butterfly, Danaus plexippus, to protect over wintering sites in the USA and Mexico; Yen and Butcher 1997).		Possible but unlikely application

[178] Repealed the *Wildlife Protection (Regulation of Exports and Imports) Act 1982* (on 29 June 200) and incorporates its provisions, with amendments, into the *Environment Protection and Biodiversity Conservation Act 1999*.

Instrument	Date of entry into force	Description / Purpose / Relevance to marine invertebrates or their conservation	Given effect in Australia by: (legislation)	Provisions (relevant to marine invertebrate conservation)
Habitat / site protection				
Convention for the Protection of the World Cultural and Natural Heritage 1972	Entry into force generally 17 December 1975	This Convention establishes a scientific system for permanent protection of cultural and natural heritage of outstanding universal value. The most significant aspect of this Convention is the development of the **World Heritage List**. Natural heritage encompasses physical and biological formations of outstanding universal value for aesthetic or scientific reasons; geological and physiographical formations and the habitat of threatened species of animals and plants of outstanding value for scientific or conservation reasons; and natural sites of outstanding universal value from the point of view of science, conservation or natural beauty (Article 2). Member countries must ensure that they identify, protect, conserve and present their listed World Heritage properties.	*Environmental Protection and Biodiversity Conservation Act 1999* (replaced the *World Heritage Properties Conservation Act 1983*)	General relevance for marine invertebrates in World Heritage sites.
Convention on Wetlands of International Importance especially as Waterfowl Habitat (RAMSAR Convention) 1971	Entry into force generally 21 December 1975	The RAMSAR Convention aims to stem the loss of wetlands and to ensure conservation of wetlands for their importance in ecological processes as well as their rich fauna and flora. Other goals include establishing wetlands within nature reserves and providing adequate protection and wardenship of wetland areas. It also promotes the training of personnel in wetland research and management. It includes a List of Wetlands of International Importance, including 53 sites in Australia[179] , although most of these are freshwater. Although it is used primarily to protect wetlands of importance to birds, there is no reason why invertebrates cannot be used to identify wetland sites of conservation importance (Collins 1987).		Wetland conservation (including saltmarshes and mangroves)

[179] http://www.biodiversity.environment.gov.au/environm/wetlands/ramsar/siteindx.htm.

Instrument	Date of entry into force	Description / Purpose / Relevance to marine invertebrates or their conservation	Given effect in Australia by: (legislation)	Provisions (relevant to marine invertebrate conservation)
Particular geographical areas				
Antarctic Treaty 1959	Entry into force for Australia 23 June 1961	The objective of this treaty is to ensure that Antarctica (defined as the area south of 60°S latitude) is used for peaceful purposes and scientific investigation. This treaty allows freedom of movement and scientific research on and around the Antarctic continent. It does not directly provide for environmental protection although the Parties do have responsibilities with respect to the preservation and conservation of living resources in Antarctica under Article IX (Herriman et al. 1997).	*Antarctic Treaty Act 1960*	General relevance to Antarctic taxa and habitats
Protocol to the Antarctic Treaty on Environmental Protection (Madrid Protocol) 1991	Entry into force generally 14 January 1998	The Madrid Protocol is designed to ensure the 'comprehensive protection of the Antarctic environment and dependent and associated ecosystems' and to designate Antarctica as 'a natural reserve, devoted to peace and science' (Article 2), with priority given to scientific research. It adopts a set of environmental principles for the planning and conduct of all activities in the Antarctic area, including avoidance or limitation of adverse environmental or ecological impacts; regular monitoring; and environmental impact assessment prior to any notifiable activity in the area.	*Antarctic Treaty (Environment Protection) Act 1980*	Covers examination of proposals or actions taken by, or on behalf of, the Commonwealth government, protection of Antarctic wildlife, and protection of areas of outstanding ecological and scientific importance.
Agreed Measures for the Conservation of Antarctic Fauna and Flora		This establishes a system to protect wildlife and its habitats in Antarctica.		General relevance to Antarctic taxa and habitats
Convention on the Conservation of Antarctic Marine Living Resources 1980 (CCAMLR)	Entry into force generally 7 April 1982	This Convention was introduced in response to the fishing of krill in Antarctic waters. The core objectives of CCAMLR are conservation of marine living resources and protection of the ecosystem, as opposed to exploitation, although the term `conservation' includes rational use. CCAMLR introduces an `ecosystem' approach to the management of marine living resources, for instance by taking into account effects of fisheries on other Antarctic species (although the ability of the Convention to control overfishing has been challenged). The Convention covers not only the area south of 60°S latitude (as in the Antarctic Treaty 1959) but also `the Antarctic marine living resources of the area between that latitude and the Antarctic Convergence which form part of the Antarctic marine ecosystem' (Article I).	*Antarctic Marine Living Resources Conservation Act 1981*	Applies to all Australians and Australian vessels and to foreigners and foreign vessels within the 200 nm Australian EEZ. Makes it an offence to harvest or undertake research on any species of living marine organism without a permit, unless permitted under another Commonwealth law.
Convention for the Protection of the Natural Resources and Environment of the South Pacific Region (SPREP) 1986	Entry in force generally 18 August 1990	To protect and manage the natural resources and environment of the South Pacific Region.		General relevance (although not to Australia directly).

Instrument	**Date of entry into force**	**Description / Purpose / Relevance to marine invertebrates or their conservation**	**Given effect in Australia by: (legislation)**	**Provisions (relevant to marine invertebrate conservation)**
Particular geographical areas - continued				
Convention on the Conservation of Nature in the South Pacific Region 1976 (Apia Convention)	Entry into force generally 26 June 1990	This international convention establishes a broad framework for nature conservation in the South Pacific region. Of particular importance is the protection of migratory and endangered species, and the preservation and management of wildlife habitats and terrestrial ecosystems.		General relevance (although not to Australia directly).
Torres Strait Treaty 1978	Entry into force 15 February 1985	Australia and Papua New Guinea concluded this Treaty in 1978 and it came into force in February 1985. Both parties must initiate legislative or alternate measures to protect and preserve the traditional way of life of the native inhabitants and the marine environment in and around the protected zone established by the Treaty.	*Torres Strait Fisheries Act 1984*	In part administered by Fisheries Management Notices (e.g., Torres Strait Prawn Fisheries restriction on gear (2002)).

Instrument	Date of entry into force	Description / Purpose / Relevance to marine invertebrates or their conservation	Given effect in Australia by: (legislation)	Provisions (relevant to marine invertebrate conservation)
Threatening processes – pollution				
Convention on the Prevention of Marine Pollution by Dumping of Wastes and Other Matters (London Dumping Convention) 1972	Entry into force for Australia 30 August 1975	This Convention controls dumping from ships, aircraft or platforms.	*Environmental Protection (Sea Dumping) Act 1981*	Controls dumping from ships, aircraft, platforms or other man-made structures in Australian waters and from Australian ships and aircraft in any sea
Amendments to Annexes 1978, 1980, 1989, 1993	1979, 1981, 1990, 1994	The 1993 Amendment concerned the phasing out of sea disposal of industrial waste. It was accepted for Australia in 1994, except in relation to jarosite waste, where the option of dumping at sea was retained for a short while after the January 1996 deadline, but not beyond December 1997.		
International Convention for the Prevention of Pollution from Ships (MARPOL 1973), and Protocols I and II	Did not enter into force, but given effect, with modifications and additions, through subsequent signing of the 1978 Protocol	This Convention is designed to "prevent pollution of the marine environment by the discharge of harmful substances or effluents containing such substances" from ships. The Convention includes five technical Annexes. Annexes I, II, III and V, which deal with oil, noxious liquids, packaged harmful substances and garbage, entered into force between 1983 and 1992. Under the Convention, most sea areas are considered to have an adequate level of protection, but where additional protection is required, MARPOL can designate an area as a special area and impose correspondingly more stringent restrictions on the disposal of harmful substances. A special area is defined as "a sea area where for recognised technical reasons in relation to its oceanographical and ecological condition and to the particular character of its traffic the adoption of special mandatory methods for the prevention of sea pollution by oil, noxious liquid substances, or garbage…. is required". Important controls on oil and ballast water discharge are defined under Regulations 9 and 15 of the Schedule 3 Annex (Amendments to the 1978 Protocol). These deal with requirements for control and management of oil pollution and dirty ballast water discharge, including conditions under which ballast can be carried in cargo tanks. The International Maritime Organisation (IMO) is currently undertaking a major revision of Annex II MARPOL (see Crayford 1999), in response to developments (including improvements in ship technology, better knowledge of the impacts of chemicals on the marine environment, etc.) which have made it necessary for the IMO to reconsider the criteria used to assign "pollution category" and "ship type".	*Protection of the Sea (Prevention of Pollution from Ships) Act 1983*	This Act ratifies Annexes I, II, III and V, which deal with oil, noxious liquids, packaged harmful substances and garbage (respectively)

Instrument	Date of entry into force	Description / Purpose / Relevance to marine invertebrates or their conservation	Given effect in Australia by: (legislation)	Provisions (relevant to marine invertebrate conservation)
Threatening processes – pollution - continued				
Basel Convention on the Control of Transboundary Movements of Hazardous Wastes and their Disposal	Acceded to by Australia in May 1992	This convention puts an onus on exporting countries party to the Convention to ensure environmentally sound management of hazardous wastes in importing countries. Its purpose is to minimise such movements and control essential ones. It seeks to minimise generation of hazardous waste; ensure adequate disposal facilities are available; control and reduce international movements of hazardous waste; ensure environmentally sound management of wastes; and prevent and punish illegal traffic.	*Hazardous Waste (Regulation of Exports and Imports) Act 1989*	Relevant to control of marine pollution.
Protocol for the Prevention of Pollution of the South Pacific Region by Dumping	Australia has not yet ratified this Treaty			Relevant to control of marine pollution.
Threatening processes – environmental change				
Vienna Convention for the Protection of the Ozone Layer 1985	Entry into force 22 Sept. 1988	Formulated to protect the ozone layer from modifications due to human activities.		Relevant to mitigate impacts of ozone depletion (e.g., effects on corals).
Montreal Protocol on Substances that Deplete the Ozone Layer 1987 (plus Amendments in 1992 and 1994)	Entry into force January 1989	Multilateral international treaty to provide agreements for reducing and depleting ozone producing substances. Follows on from, and is related to, the earlier Vienna Convention.	*Ozone Protection Act 1989*	Controls the manufacture, import and export of ozone depleting substances.
United Nations Framework Convention on Climate Change (UNFCCC) 1992	Entry into force for Australia March 1994	The UNFCCC provides the focus for international action to address the threat of climate change. The objective of this treaty is to achieve ‘stabilisation’ of greenhouse gas concentrations in the atmosphere at a level that would prevent dangerous anthropogenic interference with the climate system. Such a level should be achieved within a time frame sufficient to allow ecosystems to adapt naturally to climate change, to ensure that food production is not threatened and to enable economic development to proceed in a sustainable manner.		Recognition of climate change issues which are seen as a major threatening process for marine invertebrates

Instrument	Date of entry into force	Description / Purpose / Relevance to marine invertebrates or their conservation	Given effect in Australia by: (legislation)	Provisions (relevant to marine invertebrate conservation)
Threatening processes – pollution - continued				
Kyoto Protocol to the United Nations Framework Convention on Climate Change 1997	Signed April 1998 but not yet ratified	The Kyoto Protocol on Climate Change committed Parties to achieving quantified emissions limitations and reduction targets, through various measures. Article 3 included requirements in relation to Parties individually, or jointly, to ensure that their aggregate anthropogenic carbon dioxide equivalent emissions of greenhouse gases listed under Annex A of the Convention did not exceed assigned amounts (calculated to commitments under Annex B) and with a view to reducing overall emissions of such gases in developed countries by at least 5% below 1999 levels in the period 2008 to 2012 (called the 2010 budget period). The conference adopted differential targets of which Australia's was 8% growth above 1990 levels.		Attempts to address climate change issues which are seen as a major threatening process for marine invertebrates

8.2 Australian legislation and its administration

Under the Australian Constitution, environmental matters – which include biodiversity conservation and natural resource management – have been interpreted as being primarily a State and Territory issue, Federal jurisdiction being limited to areas such as trade, Commonwealth waters, and international treaties (although Commonwealth governments have nevertheless enacted their own environmental legislation). Local government areas of environmental responsibility include catchment issues, development approvals and planning.

Australian legislation and how it relates to invertebrate conservation in Australia has been reviewed by Yen and Butcher (1997) and Hutchings and Ponder (1999). The following discussion is primarily concerned with those pieces of legislation directly relevant to marine invertebrate conservation (summarised in Table 8.2). This review is based upon, but expanded and updated from Appendix 1 of Hutchings and Ponder (1999). It is clear that there is:

- A considerable disparity in the approaches taken by different Australian jurisdictions;
- A lack of a consistent or coordinated approach to marine biodiversity conservation in Australia;
- A conflict of interest in some circumstances.

In addition, with the separation of agencies or legislation based on habitat (aquatic vs terrestrial, marine vs non-marine), the management of transitional habitats can be problematic - in particular estuarine or supralittoral habitats.

Overall, there is a need to develop greater integration and more consistent standards Australia-wide, particularly in relation to those taxa, including marine invertebrates, that at present receive inadequate legislative protection.

The fact that States and Territories are responsible for the management of conservation and related issues within their own borders has led to the development of a diverse body of legislation, with considerable disparity in the approaches taken by different jurisdictions. This has led to parochial anomalies in relation to threatened species, where (for instance) a species can be protected as rare in one jurisdiction, even if it is extremely abundant in another, or a genuinely threatened species can be afforded legislative protection in one jurisdiction but not in another where it occurs.

Many States have different government authorities responsible for different areas of conservation; for example, the terrestrial and aquatic/marine realms are frequently the responsibility of different departments (Hutchings and Ponder 1999). At the same time, one agency may have responsibility for both exploitation and conservation (as is the case, for example, with the Department of Conservation and Land Management in Western Australia, or Fisheries in NSW). Such an arrangement has the potential for serious conflict of interest.

The primary legislation under which marine invertebrates can be protected and conserved varies widely among different jurisdictions. For instance, it may fall into one (or more) of the following categories:

- **Threatened species** – legislation whose express purpose is to allow for the listing of threatened species[180] (e.g., as vulnerable, endangered, presumed extinct, etc); some also allows for the listing of threatened populations, ecological communities and / or threatening processes. Marine

[180] Under some legislation a species does not require a formal name to be listed, providing a formal description is pending and voucher material has been formally lodged in a recognised Institution. This is a potentially useful avenue for those marine invertebrates from groups under investigation to be listed, especially in situations where immediate attention is required.

invertebrates are eligible for listing under most Commonwealth and State threatened species legislation (see below), though in few cases have any marine invertebrates actually been listed.

- **Parks and reserves** – legislation that allows for the declaration of protected areas such as parks and reserves. In some cases, this includes marine areas, in other cases (e.g., NSW) there is separate marine legislation. For instance, in South Australia, there is no dedicated threatened species legislation, and although some species of "wildlife" can be protected under the *National Parks and Wildlife Act 1972,* invertebrates are not classified as "wildlife" under this Act.
- **Fisheries** – fisheries legislation, which has been enacted in all jurisdictions, generally contains provisions relating to the management of harvested species, which may be relevant to harvested marine invertebrates. It may also be used to prohibit the collection of protected species, although this rarely occurs for marine invertebrates, and to declare various types of aquatic reserves. In some cases (e.g., NSW), threatened species provisions for aquatic organisms are also contained in the fisheries legislation.
- **Definitions** employed in legislation need to be carefully framed to best achieve the desired conservation outcome. Very broad definitions of animals relating to particular conservation measures (total ban on collecting, bag limits etc.) can be impossible to enforce and may have little no biological rationale[181].
- **Recreational activities** - There is often an inconsistent approach to marine conservation. For example recreational fishing may be allowed in marine reserves but there are blanket bans on harvesting or collecting even super-abundant species, or the dead remains of various marine invertebrates (shells, echinoderm tests, etc.).
  - More flexible approaches to collecting and harvesting are required. For example, the imposition of bag limits for specific taxa as an alternative strategy to blanket bans;
  - Collecting (of shells[182] etc.) by responsible collectors[183], interested children etc. should be encouraged, rather than discouraged, as a means of gaining interest in invertebrates and as a potential source of valuable information[184].

Most of the other environmental legislation, which can comprise a vast array of Acts and Regulations[185] dealing with environmental planning and assessment, environmental protection and pollution control, resource allocation and special development approvals, are mostly not of direct relevance to invertebrates. However, some have the potential to be very important, particularly in relation to mitigation of threatening processes and instigation of planning controls. Even legislation that deals, primarily or wholly, with activities on land can affect organisms in the marine environment, for instance through its influence on land use, land clearing, run-off, pollution etc. (see Chapter 6). Yet, there is little integration or cooperation among the various agencies with responsibility for administration of the various Acts[186].

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[181] For example legislation aimed at reducing the impact from gathering intertidal and subtidal invertebrates for food should not specify "all marine invertebrates" as this would include microscopic interstitial animals etc.

[182] Dead (empty) shells are treated identically to living specimens in some jurisdictions. Similarly, legislation or management plans relating to shell collection should not specify "all molluscs" as this would include all tiny species, a great number of which are less than 2 mm in size and of no interest to shell collectors.

[183] Responsible collecting by amateur naturalists should be encouraged and could be managed by way of licensing societies that have adopted an appropriate code of ethics.

[185] Australian Acts and Regulations can be accessed on the web via the Australian Legal Information Institute (AustLII - http://www.austlii.edu.au/) or Commonwealth and State web pages. A summary is provided in Table 8.2.

[186] For example, the Great Barrier Reef Marine Park Authority has recognised that one of its major areas of concern is terrestrial run off, yet the Authority can only act as an observer on various catchment management authorities along the Queensland coast, having no direct jurisdiction over activities on land adjacent to the Park (Wachenfeld et al. 1998).

8.2.1 National coordination

Yen and Butcher (1997) suggested that the problem of species being listed as protected in one jurisdiction but not in another could be partially addressed if the Australian and New Zealand Environment Council (ANZECC[187]) provided a framework for invertebrates as they had with vertebrates because ANZECC maintained national lists of threatened Australian fauna, flora and ecological communities. However, the Commonwealth Govt. is currently creating a new council, the *Environment Protection and Heritage Council,* that will be an amalgamation of the *National Environment Protection Council* (NEPC), the non-Natural Resources Management (NRM) component of ANZECC and Heritage Ministers' Meeting. A second council, the *Natural Resource Management Council* will combine the NRM issues from the Agriculture and Resource Management Council of Australia and New Zealand (ARMCANZ), the Australian and New Zealand Environment and Conservation Council (ANZECC) and the Ministerial Council on Forestry, Fisheries and Aquaculture (MCFFA).

8.2.2 Commonwealth

The Commonwealth holds a range of powers, including the implementation of international treaties, which could theoretically be used to achieve considerable control over environmental matters (Dixon 1994). During the 1970s and 1980s, these were used to override State decisions on issues such as the damming of the Franklin River in Tasmania, mining of limestone in Moreton Bay and sand mining on Fraser Island. However, successive Commonwealth governments have been reluctant to use these powers, focusing instead on the prime responsibility of the States for environmental protection and the need for a cooperative approach among all jurisdictions (Woinarski and Fisher 1999). This position was formalised in the Intergovernmental Agreement on the Environment (1992)[188], which recognised that the Commonwealth's role was largely one of coordination.

The *Environment Protection and Biodiversity Conservation Act 1999* (EPBC Act)[189] gives the Commonwealth primary responsibility on matters of national environmental significance and is now the key piece of Commonwealth legislation with relevance for the conservation of marine invertebrates. The EPBC Act represents an attempt to explicitly define the environmental responsibilities of the Commonwealth, rather than relying on indirect triggers such as foreign investment approval and Commonwealth funding decisions. Under this Act, Commonwealth environmental responsibilities are limited, in accordance with an agreement given in-principle endorsement by the Council of Australian Governments in 1997, to "matters of national environmental significance". These are explicitly defined in the Act as:

- The Commonwealth marine area;
- World Heritage properties;
- RAMSAR wetlands of international importance;
- Nationally threatened species and ecological communities;
- Internationally protected migratory species; and
- Environmentally significant nuclear actions (Hill 1998).

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[187] ANZECC (formerly the Australian Council of Nature Conservation Ministers or CONCOM) was, until recently, a forum for consultation between the Commonwealth, State and Territory Governments on matters related to nature conservation and wildlife protection.

[188] http://www.environment.gov.au/psg/igu/pubs/igae.html

[189] This Act replaces five existing pieces of legislation; the *Environment Protection (Impact of Proposals) Act 1974*, *Endangered Species Protection* ("ESP") *Act 1992*, *National Parks and Wildlife Conservation Act 1992*, *Whale Protection Act 1980*, and *World Heritage Properties Conservation Act 1983*.

The Act also applies to actions on Commonwealth land and actions by the Commonwealth and Commonwealth agencies. However, it excludes from Commonwealth jurisdiction other major environmental issues of concern, such as land clearance and degradation, the allocation of water rights, climate change and greenhouse gas emissions and forest management and protection (EDO 1999). An activity which does not have a significant impact on one of the matters of national significance will no longer trigger Commonwealth involvement in the assessment and approval process, even if it requires a Commonwealth decision or approval such as foreign investment approval (Hill 1998).

Another key change in this Act involves the capacity of the Commonwealth to 'accredit' State and Territory procedures and decisions, if the appropriate outputs or standards of outcome are guaranteed, by means of bilateral agreements between the Commonwealth and each State and Territory. It has been argued by some that this allows the Commonwealth to further reduce its role in environmental protection (Münchenberg 1998), by 'delegating' to the States the responsibility for conducting environmental impact assessments, even with respect to matters of national significance. To date, as far as we can determine, no bilateral agreements have been entered into, despite an original intention to have draft agreements prepared for consultation during early 2000. The situation is currently uncertain with some States having stated that they will not enter into such an agreement.

Listing of threatened taxa and communities

Many of the key features of the *Endangered Species Protection Act 1992* are retained, although there are some key changes designed to address some of the identified shortcomings of the latter, including extension of the Commonwealth's authority in relation to listed species across their full Australian range (previously limited to Commonwealth land and waters).

Species (the definition of which includes a subspecies, or distinct population) may be listed as 'extinct', 'extinct in the wild', 'critically endangered', 'endangered', 'vulnerable', or 'conservation dependent'. Ecological communities may also be listed, in the categories 'critically endangered', 'endangered', or 'vulnerable' (under the ESP Act, only 'endangered' communities could be listed). The criteria for proclaiming a species or community as threatened, which were established by the Endangered Species Scientific Subcommittee, are listed in the *Environment Protection and Biodiversity Conservation Regulations 2000*. A Key Threatening Process can be listed if it could cause any native species or community to become eligible for listing (in any category other than conservation dependent), could result in a listed species or community being moved to a higher category of threat, or adversely affects two or more listed species or communities[190].

The public can nominate species, communities and threatening processes. Nominations are assessed by the Threatened Species Scientific Committee, who forward their recommendations to the Environment Minister for decision. Benefits of listing include a responsibility by the Commonwealth to develop recovery plans for all listed species or ecological communities. Recovery plans developed under the EPBC Act must identify critical habitat for listed species. Key Threatening Processes may also be listed through this process, and Threat Abatement Plans may be prepared if the Minister considers it feasible and effective to develop such a plan to address the problem (Dovey 1999).

To date only a few invertebrates, and no marine invertebrates, have been listed at the Commonwealth level.

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[190] The webpage of the Threatened Species section of Environment Australia has electronic versions of the Schedules containing Commonwealth listed species, communities and threatening processes, as well as links to the threatened species websites of each of the States and Territories.

The ability to list threatened ecological communities and threatening processes are potentially one of great value for the conservation of marine invertebrates and other taxa not well covered by the threatened species approach. However, these provisions have yet to live up to their promise. To date 27 ecological communities have been listed, none of which is marine[191]. There are currently 11 listed Key Threatening Processes (Schedule 3)[192], but again, none is of relevance to marine invertebrates. All but three (incidental catch of seabirds and turtles and land clearing) concern terrestrial feral pests and pathogens. However, land clearing has significant indirect impacts on coastal ecosystems. The EPBC Act differs from the ESP Act in enabling the listing of Threatening Processes even when preparation of a Threat Abatement Plan is not feasible – it is a separate decision whether a Plan should be prepared.

Aside from the threatened species provisions discussed above, the EPBC Act has a number of other features of relevance to marine invertebrate conservation. These include the ability to declare marine protected areas in Commonwealth areas, the provisions relating to protection and management of World Heritage areas and RAMSAR wetlands of international significance, and the requirement to prepare impact assessments for Commonwealth-managed fisheries. There are also new, specialised criteria to "assess the conservation status of marine biota" (Hill 1998), and special provisions (Part 13 Division 4) which make it an offence to take, kill or injure any "listed marine species" in a Commonwealth area except under limited conditions[193]. As currently constituted this list does not include any marine invertebrates, but it is clearly a potential vehicle that could be used to benefit marine invertebrate conservation.

Commonwealth Acts, which give effect to Australia's obligations under relevant international treaties, are listed in Table 8.1 (a table of relevant international treaties and their implementation in Australia). Additional Acts with some relevance to marine invertebrate conservation are listed in below.

[191] Source: http://www.ea.gov.au/cgi-bin/sprat/public/publiclookupcommunities.pl
[192] Source: http://www.ea.gov.au/cgi-bin/sprat/public/publicgetkeythreats.pl
[193] Section 248 of the Act contains a list of marine species (all vertebrates) for which it is an offence to kill, injure, take, trade, keep, or move any member of a listed marine species on Commonwealth land or in Commonwealth waters without a permit.

Other Commonwealth Acts with some relevance for marine invertebrate conservation

Legislation	Relevance	Provisions
States Grants (Nature Conservation) Act 1974	Financial assistance to States for conservation measures	Provides financial assistance to State and Territory initiatives in nature conservation. These special purpose grants apply to land acquisition, management, and the provision of facilities by the States (Parker 1995)
Environment (Financial Assistance) Act 1977	Financial assistance to States for conservation measures	Facilitates Commonwealth financial assistance to the States for environmental planning projects
Fisheries Management Act 1991 and Fisheries Administration Act 1991	Commonwealth fisheries, excluding organisms covered under the Continental Shelf Act	Replace the *Fisheries Act 1952* (repealed). Objectives are to: (a) implement efficient and cost effective fisheries management, (b) ensure that fisheries activities are conducted according to ESD principles, and (c) maximise economic efficiency in the exploitation of fisheries. Allow the Minister to gazette proclaimed areas, to prohibit and regulate the taking of fish or classified fish, to prohibit the possession of unlicensed equipment, etc.
Torres Strait Fisheries Act 1984	Control of fisheries in the Torres Strait Protected Area established under the Torres Strait Treaty	
Pollution of the Sea by Oil (Shipping Levy) Act 1976; Pollution of the Sea by Oil (Shipping Levy Collection) Act 1972; Protection of the Sea (Powers of Intervention) Act 1981; Protection of the Sea (Prevention of Pollution from Ships) Act 1983-86	Control of pollution from shipping	
Australian Biological Control Act 1984	Control of non-indigenous species	
Quarantine Act 1908	Control of import of non-indigenous species into Australia	Some voluntary agreements to encourage ships to undertake mid-ocean ballast exchange and also mechanisms exist whereby ships may be banned from discharging ballast water in port
Australian Heritage Commission Act 1975	Potential avenue for protection of sites	The Australian Heritage Commission advises the Commonwealth on the protection of Australia's National Estate, places of natural, historical or Aboriginal heritage value. The Commission keeps a list of such places on the Register of the National Estate. Listing in the Register gives some protection to a place under Section 30 of the Act, which aims to ensure that the Commonwealth Government does not unnecessarily damage Australia's heritage unless there is no prudent alternative. It does not cover the actions of State or Local Government, or of private owners (Yen and Butcher 1997).
Australian Institute of Marine Science Act 1972	Establishes AIMS to carry out research in marine science	
Great Barrier Reef Marine Park Act 1975 (Amendment Act 1991)	Establishes the Great Barrier Reef Marine Park and the Great Barrier Reef Marine Park Authority (GBRMPA)	Region extends to low water mark and is not affected by the Offshore Constitutional Settlement. Makes provision for Commonwealth financial assistance to Queensland for Park management. Requires compulsory pilotage for all vessels carrying oil or toxic chemicals as cargo and for large vessels navigating within the Park

8.2.3 New South Wales

In NSW, marine invertebrates are classed as 'fish' under the *Fisheries Management Act 1994, Marine Parks Act 1997* and *Fisheries Management Amendment Act 1997*, which define 'fish' as:

> "marine, estuarine or freshwater fish or other aquatic animal life at any stage of their life history (whether alive or dead) ... (including) oysters and other aquatic molluscs; and crustaceans; and echinoderms; and beachworms and other aquatic polychaetes, …. (but not) any species of whales, marine mammals, reptiles, birds, (or) amphibians".

Thus all marine (and other aquatic) invertebrates come under the fisheries legislation and their conservation is the responsibility of NSW Fisheries. Part of the purpose of the *Fisheries Management Amendment Act 1997* was to extend legislative protection (given to terrestrial animals and plants under the *Threatened Species Conservation Act 1995*) to threatened aquatic animals and vegetation. In essence, this amendment replicates the principles included in the *Threatened Species Conservation Act 1995*, so that similar criteria apply to fish and marine vegetation as apply to other biota in New South Wales (O'Connor 1999). The amendment also resulted in changes to the *Environmental Planning and Assessment Act 1979*, integrating the consideration of threatened "fish" (the definition of which includes both finfish and aquatic invertebrates) and marine plant (including mangrove, seagrass and algae) conservation into the environmental planning and assessment process. Thus, the effect of a development or activity on threatened species must be considered by a consent and/or a determining authority, and where there is likely to be a significant impact, the preparation of a species impact statement (SIS) is required.

The *Fisheries Management Amendment Act 1997* incorporated provisions for the listing of endangered species, endangered populations, endangered ecological communities, species presumed extinct, vulnerable species, and key threatening processes. Listing is the responsibility of a Fisheries Scientific Committee, though any person may nominate a taxon for listing. A recovery plan must be prepared for each listed species, population or community, with the aim of returning it to a position of viability in nature. Key threatening processes are processes that adversely affect two or more threatened species or which could cause a species to become threatened. A threat abatement plan must be prepared for each key threatening process listed, with the aim of managing the process in order to reduce or eliminate the threat. Draft recovery and threat abatement plans must be placed on public exhibition for community input before finalisation. Relevant recovery plans must be considered by consent and determining authorities when assessing applications for development under the *Environmental Planning and Assessment Act 1979*.

Schedule 4 contains lists of endangered species, populations and ecological communities and species presumed extinct, while Schedule 5 lists vulnerable species. To date no marine invertebrates have been listed under any category. Schedule 6 contains a list of key threatening processes; however, to date none have been listed relevant to marine environments[194].

The amending legislation also allows the Minister for Fisheries to declare 'critical habitat', defined as the whole or any part or parts of the area or areas of land comprising the habitat of an endangered species, population or ecological community that is critical to its survival. Once critical habitat is declared, relevant planning instruments must take note of it and any proposal under Parts 4 or 5 of the Environmental Planning and Assessment Act that is likely to affect critical habitat requires the preparation of a species impact statement (SIS).

[194] The relevant Schedules can be found on the web at http://www.fisheries.nsw.gov.au/pages/conservation/threasched.htm

Under the *Fisheries Management Act 1994*, NSW Fisheries has a range of other powers and responsibilities with relevance for marine invertebrate conservation, particularly with relation to habitat protection; these include the preparation of Habitat Protection Plans, declaration of aquatic reserves, control of dredging and reclamation, protection of mangroves and certain other marine vegetation, control of noxious 'fish' and the release or importation of 'fish', etc.

Other relevant Acts in NSW

Habitats such as saltmarsh can be protected under the *Environmental Planning and Assessment Act 1979* invoking SEPP (State Environmental Planning Policy) 14 or SREP (Regional Environmental Plan) 20 (NSW Fisheries 1998b). This and other relevant legislation is listed in the table below.

Legislation	Relevance	Provisions
Marine Park Act 1997	Declaration of Marine Protected Areas	Objects are to conserve marine biological diversity and marine habitats by declaring and providing for the management of a comprehensive system of marine parks, to maintain ecological processes in marine parks, and (where consistent with these), provide for ecologically sustainable use of fish and marine vegetation, and provide opportunities for public appreciation, understanding and enjoyment, in marine parks.
Crown Lands Act 1989	Activities on Crown lands (including the seabed)	The Department of Land and Water Conservation is responsible for all Crown Land including the seabed. The objectives of this Act include conserving natural resources, environmental protection, and the public use and enjoyment of Crown Lands
Environmental Planning and Assessment Act 1979	Preparation of environmental planning instruments such as SEPPs, REPs and LEPs	SEPP 14 (1985) relates to wetlands
Protection of the Environment Operations Act 1997	Control of threatening processes - pollution	Replaces the *Clean Air Act 1961*, the *Clean Waters Act 1970*, the *Environmental Offences and Penalties Act 1989*, the *Noise Control Act 1975*, the *Pollution Control Act 1970*, and incorporates the major regulatory provisions of the *Waste Minimisation and Management Act 1995*

Lord Howe Island, which is part of NSW, is administered by the LHI Board. Marine conservation matters are handled by the NSW Marine Parks Authority.

Bans on commercial fishing in estuarine waters in NSW

A recent landmark decision in the NSW Land and Environment Court (21 Jan. 2000) ruled in favour of a group of recreational fishermen (as represented by the 25,000-strong Sustainable Fishing and Tourism Inc.) seeking government legislation to prevent trawler operators from getting a licence without producing an environmental impact statement. Their concern related to the trawlers dragging their nets over seagrass beds because these areas were seen as important fish nursery areas. Court action was brought against a private trawler operator but it was also claimed NSW Fisheries Minister unlawfully granted a licence to him by not complying with the Environmental Planning and Assessment Act. The judgement was suspended for six months to allow Fisheries time to review its licence-issuing procedures (McIlveen 2000). More recently, Botany Bay and Lake Macquarie have been closed by NSW State Fisheries to commercial fishing, a decision upheld in the Land and Environment Court. This is part of the protection of 30 locations from commercial fishing, resulting in 27% of estuarine waters being substantially free of this activity. Commercial fishing was banned from these areas from 1 May 2002 , except for the

Clarence River haven which will come into effect on 1 September 2002. The stated purpose of these areas is to improve recreational fishing by banning commercial fishing in key areas of significance to recreational fishers[195].

8.2.4 Victoria

In Victoria, the key piece of legislation for the listing of threatened species is the *Flora and Fauna Guarantee Act 1988*[196]. This Act has been the subject of considerable discussion, particularly with regards to invertebrates (e.g., Butcher et al. 1994; Clunie and Reed 1995; O'Hara 1995; Reed and Clunie 1997; Yen and Butcher 1997), as it was the first of its kind in Australia to specifically recognise invertebrates as 'fauna' and to allow for the listing of threatened communities and threatening processes. It thus served as a model for the development of the Commonwealth *Endangered Species Protection Act* 1992 and is also currently one of only two pieces of legislation in Australia under which any marine invertebrate species have been listed as threatened. Previously, wildlife protection in Victoria was controlled by the *Wildlife Act* 1975 and while the potential to list invertebrates did exist, all attempts to do so were unsuccessful (Butcher et al. 1994).

Under the Act, a taxon, community, or Potentially Threatening Process can be nominated for listing by any person or organisation, and is assessed by a Scientific Advisory Committee (SAC). This assessment is based on whether or not a taxon is declining and likely to become extinct, or whether or not it is significantly prone to future threats likely to result in extinction. There are no categories such as extinct, endangered or vulnerable. Public comment is invited on the SAC's recommendation, but the final decision is made by the Minister for Environment and Conservation. An Action Statement must be prepared following a listing, these identifying those actions necessary to conserve the species or community, or manage Potentially Threatening Processes.

Threatened taxa and communities of flora and fauna are listed in Schedule 2.

There are more than 170 listed species of fauna, these including two marine opisthobranchs (*Platydoris galbana* and the genus *Rhodope*), both members of a listed community (San Remo). Neither is yet the subject of an Action Statement. There are 36 communities listed[197], including one marine community, the San Remo Marine Community[198]. San Remo is the subject of Action Statement No. 18.

Potentially Threatening Processes are listed in Schedule 3. There are currently 28 listed, five of direct relevance to marine invertebrates:

- Input of organotins to Victorian marine and estuarine waters;
- Input of petroleum and related products into Victorian marine and estuarine environments;
- Introduction and spread of *Spartina* to Victorian estuarine environments; and
- The introduction of exotic organisms into Victorian marine waters.
- Habitat fragmentation as a threatening process for fauna in Victoria.

Several other PTPs may indirectly affect marine invertebrates, as they involve broad ecosystem-scale alteration to rivers and streams. These include alteration to natural flow regimes and

[195] Information from http://www.fisheries.nsw.gov.au/recreational/rfa/home_rfa.htm
[196] Accessible via: http://www.dms.dpc.vic.gov.au/. General information at http://www.nre.vic.gov.au/web/root/domino/cm_da/nrenpa.nsf/frameset/NRE+Plants+and+Animals?OpenDocument.
[197] http://www.nre.vic.gov.au/web/root/domino/cm_da/nrenpa.nsf/frameset/NRE+Plants+and+Animals?OpenDocument
[198] http://www.nre.vic.gov.au/web/root/domino/cm_da/NRECPA.nsf/3d08e37a810f38b94a256789000ee6bb/f8ad510c4f24133e4a256803000cc81d?OpenDocument

temperature regimes, and increase in sediment input and input of toxic substances into rivers and streams.

In addition to the *Flora and Fauna Guarantee Act*, the Department of Natural Resources and Environment (NRE) maintains lists of threatened fauna and flora, including threatened invertebrate fauna, whether terrestrial, freshwater or marine (Doeg 1999).

Species are listed according to the current IUCN threat categories and criteria, using information maintained on NRE databases, with assessments based solely on biological criteria and subject to peer review. Updated lists are released periodically. The lists have no real legal standing, but are put out as a guide to managers in the field and are designed to be taken into account during strategic planning and established planning processes, such as national park management plans, Regional Forest Agreements, local government planning schemes, catchment strategies and setting priorities for recovery actions (DNRE 2000). Some of the species on these lists are also covered by the provisions of the Wildlife Act 1975 and the Flora and Fauna Guarantee Act 1988 (FFG Act). The FFG Act Schedule of threatened species and communities (Schedule 2) only includes items nominated, assessed by the Scientific Advisory Committee and approved by the Minister. There can be a time lag between a species being recognised as threatened, being nominated and finally being included on the FFG Act Schedule (DNRE 2000).

Other relevant Acts in Victoria

Legislation	**Relevance**	**Provisions**
Catchment and Land Protection Act 1995	Management of activities in catchments (that could impact on downstream marine systems)	Sets up a framework for the integrated management and protection of catchments, including community participation in the management of land and water resources, and a system of controls on noxious weeds and pest animals. The Act also establishes the Victorian Catchment and Land Protection Council, Regional Catchment and Land Protection Boards and the Pest Animal Advisory Committee.
Environment Effects Act 1978		Provides for assessment, through Environmental Effects Statements, of public or private development proposals which have a significant impact on the environment
Environment Protection Act 1970	Control of threatening processes	Established the Environment Protection Authority, whose powers, duties and functions include managing waters, control of noise and control of pollution. Also State environment protection policies etc..
National Parks Act 1975	Including marine waters adjacent to national parks	Provides for the permanent reservation of national and other parks to protect indigenous species of flora and fauna
Planning and Environment Act 1987		Local government planning schemes
Coastal Management Act 1995		Established the Victorian Coastal Council, made up of community representatives and departmental nominees, to be the peak body for the strategic planning and management of the Victorian coast, and to provide advice on coastal issues to the Minister for Environment and Conservation. One of the major tasks of the Council has been the preparation of a Victorian Coastal Strategy (Victorian Coastal Council 1997).

Another avenue for marine invertebrate conservation in Victoria is under the *Fisheries Act 1995*, which allows the declaration of any aquatic taxon or community of flora or fauna (not only rare or endangered species) as 'protected aquatic biota'. Consequently permits are required to take, destroy or disturb that species or community. So far, no taxa have been declared (apart from those aquatic taxa automatically listed under the *Flora and Fauna Guarantee Act 1988*) (Doeg 1999).

8.2.5 Tasmania

Until 1995, the *National Parks and Wildlife Act 1970* provided for the establishment and maintenance of National Parks and reserves, and the listing of endangered wildlife. Under the Wildlife Regulations of this Act, the only invertebrates that were declared as wholly protected wildlife were cave invertebrates. Since the proclamation in November 1995 of the *Threatened Species Protection Act 1995,* invertebrates can be listed in the Schedules of this Act as Endangered, Vulnerable or Rare (Taylor and Bryant 1997).

The application of Tasmanian legislation to the coastal marine environment is determined by the Tasmanian State Coastal Policy 1996. This Policy applies to the Crown in all its capacities, in particular through the *State Policies and Projects Act 1993* and the *Land Use Planning and Approvals Act 1993*. Subject to contrary statutory provision, it also applies to statutory authorities.

Threatened Species Protection Act
Only species and subspecies can be considered under this act and there is currently no recognition of threatened communities. Undescribed species can be admitted to schedules provided voucher material is reliably archived. The Threatened Species Unit of the Parks and Wildlife Service administers much of the detail, including the preparation of recovery plans. A Scientific Advisory Committee (SAC) meets about six times a year and advises the Minister on matters of scientific relevance pertaining to the Act. Criteria for listing are agreed on a case-by-case basis by the SAC using modified IUCN criteria with consideration given to the nature of the threatening processes. Invertebrate determinations tend to emphasise population criteria rather than numbers of individuals (McQuillan 1999). Tasmania has listed the most invertebrate species of any State and this has not been without its detractors (Taylor and Bryant 1997).

The Act, including Schedules 3, 4 and 5 which contain the lists of endangered, vulnerable and rare taxa (respectively), is available electronically on the Tasmanian government website[199]. Two marine invertebrates (*Patiriella vivipara* Dartnall and *Marginaster littoralis* Dartnall are listed under Schedule 3, and one under Schedule 5 (*Smilasterias tasmaniae* O'Loughlin and O'Hara).

McQuillan (1999) outlined the tasks of the Scientific Advisory Committee (SAC) as to:

- Define "critical habitat" for various species, with an emphasis on mapping of species habitat;
- Review the schedules of species. Specialist advisory committees put forward nominations as interim lists in 1994 that formed the basis of the current schedules. As a result the invertebrate advisory committee nominated seven species as extinct, eight as endangered, 18 as vulnerable and 142 considered to be rare. However, some taxa such as marine invertebrates were under-represented in deliberations, and the schedules require updating as new information comes to hand.
- Invite and evaluate public nominations for listing or delisting of species.

[199] http://www.thelaw.tas.gov.au/view/83++1995+AT@EN+2000041000/

Other relevant Acts

Legislation	Relevance	Provisions
Living Marine Resources Management Act 1995	Fisheries management	Replaced the *Fisheries Act 1959*. An Act designed to promote the sustainable management of Tasmania's marine resources (including management plans relating to fish resources and protection of marine habitats). Administration of the Act is the responsibility of the Department of Primary Industries and Fisheries
Environmental Management and Pollution Control Act 1994	Control of threatening processes (pollution)	Replaced the *Environment Protection Act 1973* and provides for the protection of the environment.
Marine Farming Planning Act 1995	Aquaculture	Provides for the planning of marine waters for marine farming and the allocation of marine farming leases
National Parks and Wildlife Act 1970	Marine protected areas	Reserved "land" can include marine waters and the seabed

Macquarie Island

Macquarie Island was listed as a World Heritage Area on 3rd December 1997, - the second Tasmanian World Heritage Area. A management plan (DPWH 1991)**,** in accordance with the requirements of the Tasmanian *National Parks and Wildlife Act 1970*, took effect on 26 June 1991. Three zones have been established: one covering the ANARE station; a second covering the whole of the rest of the island, where limited activities may occur; and a third covering the outlying islands which are to be preserved in as near a pristine state as is possible. Daily administration is carried out by the ANARE station leader employed by the Antarctic Division (and made an honorary ranger under the National Parks and Wildlife Act 1970 for the duration of their stay in the reserve; DPWH 1991). Legislation relevant to the Island totals 18 Commonwealth Acts and seven Tasmanian State Government acts, listed in DASETT (1991).

8.2.6 South Australia

There is no specific threatened species legislation in South Australia, although it was recommended in a draft Endangered Species strategy released under a previous government (Greenslade 1999). Although South Australia does not have specific threatened species legislation, provisions for listing threatened species can be found in the *National Parks and Wildlife Act 1972*[200], with endangered, vulnerable and rare species being listed in Schedules 7, 8 and 9 respectively. Invertebrates cannot be listed as "protected animals" under this Act, as "animals" are defined as comprising only mammals, birds and reptiles. Under the Act, invertebrates and their habitats are legally protected in all reserves, including marine invertebrates in the several marine reserves along the coastline.

As in other States, the fisheries legislation (section 42 of the *Fisheries Act 1982*[201]) enables various categories of "fish" (including invertebrates) to be declared as "protected", with their collection prohibited; other commercially important species are subject to special management arrangements. Examples of the latter include abalones (*Haliotis* spp.), the Blue Swimmer Crab (*Portunus pelagicus*), the Southern Rock Lobster (*Jasus edwardsii*) and Southern Calamari (*Sepioteuthis australis*). The *Fisheries Act 1982* also has provisions for the declaration of aquatic reserves and marine parks.

[200] http://www.environment.sa.gov.au/dehaa/legislation.html
[201] http://www.austlii.edu.au/au/legis/sa/consol_act/fa1982110/

The *Environment Protection (Marine) Policy* was introduced in 1994, in recognition of the need to control marine pollution originating from land based activities, to ensure environmental monitoring and improvement programs were included as a condition of all discharge licenses. This Policy requires dischargers to demonstrate by March 2001 that they are complying with minimum standards. *Our Seas and Coasts - a marine and estuarine strategy for South Australia* was released in August 1998[202].

Other relevant Acts

Legislation	Relevance	Provisions
Coast Protection Act 1972	Coastal management	Makes provision for the conservation and protection of SA beaches and coast, including Coast Protection Districts, management plans, etc.
Environment Protection Act 1993	Control of threatening processes (pollution)	Established the Environment Protection Authority, and replaced the Clean Air Act 1984, the Environmental Protection Council Act 1972, the Marine Environment Protection Act 1990, the Noise Control Act 1977, and the Waste Management Act 1987, and amended others
Environmental Protection Council Act 1972		Established the Environment Protection Council to investigate, advise and report on matters including the conservation of flora and fauna

All of the Acts administered by the Department of Environment and Heritage can be accessed electronically via their web page[203].

Marine Protected Areas

South Australia created some of the first marine protected areas (MPAs) in Australia. Six relatively small (largest 36 km^2), aquatic reserves were established under the Fisheries Act 1971. Eight more were established over the next 15 years (some under the *Fisheries Act 1982*), and several others proposed by independent groups. The only large, multiple use MPAs created so far is the Great Australian Bight Marine Park, the only marine park in South Australia. It was established under three different Acts, and covers an area of more than 20,000 km^2, extending into Commonwealth waters[204]. There are also one MPA created under the *Historic Shipwrecks Act 1981*. Further information is available at http://www.environment.sa.gov.au/coasts/marine_paps.html.

8.2.7 Western Australia

In Western Australia the legislation most relevant to threatened invertebrates is the *Wildlife Conservation Act 1950*, which is currently administered by the Department of Conservation and Land Management (CALM).

A *Wetlands Conservation Policy* for Western Australia was launched in August 1997.

Details of this administrative function of the *Wildlife Conservation Act 1950* are given in CALM's Policy Statement No. 33 (CALM 1991), and more recently by Mawson and Majer (1999), with an

[202] Available via http://www.environment.sa.gov.au/coasts/pdfs/strategy.pdf
[203] http://www.environment.sa.gov.au/dehaa/legislation.html
[204] The Great Australian Bight Marine National Park was established under the *National Parks and Wildlife Act 1972* ; The Great Australian Bight Whale Sanctuary, established under the *Fisheries Act 1982*; The Great Australian Bight Marine Park (Commonwealth Waters), established under the National Parks and Wildlife Conservation Act 1975, "but is now managed under the *Environment Protection and Biodiversity Conservation Act 1999*".

overview by Horwitz (1999). Under the Act, all indigenous Western Australian animals (defined as any living thing that is not a human being or a plant) are protected unless declared otherwise by the Minister. Three schedules of the Act are used to identify different categories of taxa in need of special protection. Schedule 1 includes species "rare or likely to become extinct" (otherwise known as "threatened species"), Schedule 2 includes species presumed to be extinct and Schedule 4 lists species that, for other reasons, are in need of special protection.[205] Currently various invertebrates are listed under these Schedules, but none are marine. There are provisions under the Act for taxa to be listed and protected from "taking" under a closed season notice, and 38 invertebrates are currently listed, all non-marine.

Nominations for listing are considered by a threatened species scientific committee that was established in 1997. Criteria such as "adequacy of survey" and "taxonomy" are considered and ranking is according to IUCN criteria. The committee comprises mainly terrestrial ecologists, mostly with mammalian, avian or higher vascular plant expertise, only two (of nine) having any invertebrate expertise. Marine and estuarine expertise is not formally represented, and inland waters, fish, frogs, and almost all invertebrates are not represented (Horwitz 1999). The Act has been under review for six years and will be replaced by a *Biodiversity Conservation Bill*.

There is currently no legislation covering the conservation of threatened ecological communities. However, an informal, non-statutory process, including advice from a scientific advisory committee, the establishment of the threatened ecological communities database, and steps for assigning ecological communities to categories of threat, is now in place (CALM 1999).

Other relevant Acts

Legislation	Relevance	Provisions
Fisheries Act 1905	Fisheries management	Covers aquatic organisms, including all aquatic animals and plants. The definition of "fish" includes molluscs and crustaceans. Certain fish are protected from sale except under licence
Marine Reserves Act 1997	Marine protected areas	
Environment Protection Act 1986	Control of threatening processes (pollution)	Aims to enhance the quality of the environment and to control pollution

8.2.8 Northern Territory

The *Territory Parks and Wildlife Conservation Act 2000*[206] is administered by the Parks and Wildlife Commission of the Northern Territory. It provides for the conservation and management of wildlife, establishment of reserves, and the listing of protected and specially protected animals and plants (Schedule 7 of the *Territory Wildlife Regulations*[207] lists specially protected animals. The definition of "wildlife" in the Act has been amended in the latest revision so that it can now include all animals, including invertebrates. Invertebrates are thus not specifically excluded from protection under this Act as they were previously (Yen and Butcher 1997; Hutchings and Ponder 1999). To date, no marine invertebrates are listed under this modified legislation but six terrestrial arthropods are.

[205] The Wildlife Conservation (Specially Protected Fauna) Notice 1999, containing the Schedules of listed species, is available at http://www.calm.wa.gov.au/plants_animals/pdf_files/watscu_fauna_gazettal99.pdf
[206] http://notes.nt.gov.au/dcm/legislat/legislat.nsf
[207] http://notes.nt.gov.au/dcm/legislat/legislat.nsf

Other relevant Acts

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Legislation	Relevance	Provisions
Fisheries Act	Fisheries management	
Environmental Assessment Act 1982; *Environmental Offences and Penalties Act*	Control of threatening processes (pollution etc.)	Provides for the assessment of the environmental effects of development proposals and for the protection of the environment Establishes penalties for certain offences relating to the protection of the environment

8.2.9 Queensland

The *Nature Conservation Act 1992* enables any (including marine) native organisms to be listed and protected under the first five Schedules of this Act (presumed extinct, endangered, vulnerable, rare, common). These are listed in the *Nature Conservation (Wildlife) Regulation 1994*[208], and a summary is available on the Environmental Protection Agency's website[209]. Several invertebrates are listed, all butterflies. There are no provisions to list communities or threatening processes. There are currently no protocols for accepting public nominations; relevant experts review internal nominations assessed using IUCN criteria. After review, nominations are considered by the Scientific Advisory Committee that makes recommendations to the Minister (Driscoll 1999).

Other relevant Acts

Legislation	Relevance	Provisions
Marine Parks Act 1982		Provides for the setting apart of tidal lands and tidal waters as marine park
Environmental Protection Act 1994	Control of threatening processes (pollution etc.)	Replaced the Clean Air, Clean Waters, Noise Abatement, Litter, Contaminated Land and State Environment Acts.

There is also *A strategy for the conservation and management of Queensland wetlands* (including mangroves, saline coastal flats etc.) (QLD EPA 1999)[210].

The Great Barrier Reef

The Great Barrier Reef is administered in a complex arrangement by the Commonwealth (responsible for the development of the management plans) and the Queensland State Government (through the Queensland Parks and Wildlife Service QPWS - responsible for day to day management of the reef), via the Great Barrier Reef Marine Park Authority (GBRMPA[211]). The obligations of the Commonwealth and Queensland Governments in the protection and management of the Great Barrier Reef Marine Park are outlined in the Emerald Agreement of 1979 which states that the day-to-day management of the Marine Park should be undertaken by officers of the Queensland Parks and Wildlife Service, subject to Authority policy. Protection of the values of the

[208] Available via the Qld legislation website, on http://www.legislation.qld.gov.au/Legislation%20Docs/CurrentN.htm
[209] http://www.env.qld.gov.au/environment/plant/endangered/
[210] Available via http://www.env.qld.gov.au/cgi-bin/w3-msql/environment/environment/conservation/msqlwelcome.html?page=wl.html
[211] http://www.gbrmpa.gov.au/corp_site/index.html

Reef against illegal activities is also achieved through strategic alliances with the Queensland Boating and Fisheries Patrol (QBFP), Queensland Water Police, Coastwatch and the Australian Maritime Safety Authority (AMSA).

Table 8.2: A summary of the Commonwealth, State and Territory threatened species legislation relevant to marine invertebrates

	Commonwealth	**NSW**	**VIC**	**TAS**	**SA**	**WA**	**NT**	**QLD**
Principal legislation	*Environment Protection and Biodiversity Conservation Act 1999* (replaces the *Endangered Species Protection Act 1992*)	*Fisheries Management Act 1994*; *Fisheries Management Amendment Act 1997*	*Flora and Fauna Guarantee Act 1988*	*Threatened Species Protection Act 1995*	1. *National Parks and Wildlife Act 1972* 2. *Fisheries Act 1982*	*Wildlife Conservation Act 1950* (Act in review)	*Territory Parks and Wildlife Conservation Act 1980*	*Nature Conservation Act 1992*
Regulations	*Environment Protection and Biodiversity Conservation Regulations 2000*		*Flora And Fauna Guarantee Regulations 1990*	*Threatened Species Protection Regulations 1996*	1. *Wildlife Regulation 1990* 2. *Fisheries (General) Regulations 1984*	*Wildlife Conservation Regulations 1970, Wildlife Conservation (Specially Protected Fauna) Notice 1999*	*Territory Wildlife Regulations*	*Nature Conservation (Wildlife) Regulation 1994*
Agency Responsible	Environment Australia	NSW Fisheries	Department of Natural Resources and Environment	Parks and Wildlife Service (Department of Primary Industries, Water and Environment)	1. Parks and Wildlife (Dept of Environment and Natural Resources) 2. Fisheries (Primary Industries and Resources SA)	Department of Conservation and Land Management	Parks and Wildlife Commission of the NT	Environmental Protection Agency
Area covered	Terrestrial Aquatic	Aquatic (marine, estuarine or freshwater)	Terrestrial Aquatic	Terrestrial Aquatic	1. Terrestrial / Aquatic 2. Aquatic	Terrestrial Aquatic	Terrestrial Aquatic	Terrestrial Aquatic

	Commonwealth	NSW	VIC	TAS	SA	WA	NT	QLD
Groups covered	Any 'nationally threatened' non-human 'biological entities' (incl. plants, animals and communities)	'Fish' (= non-tetrapod aquatic animals at any stage of their life history, whether alive or dead); Marine vegetation	Vertebrates, Invertebrates, Plants (vascular and non-vascular)	Vertebrates, Invertebrates, Plants (vascular and non-vascular)	1. Mammals, birds, reptiles, Plants 2. Fish (defined as aquatic organisms of any species)	Indigenous animals (non human), Flora	Wildlife, which has been recently redefined to include invertebrates	Vertebrates, Invertebrates, Plants (vascular and non-vascular), Protists, Procaryotes, Viruses
Threat Categories	Extinct, Extinct in the wild, Critically endangered, Endangered, Vulnerable, Conservation dependent **species** (including **sub-species** or **populations**); Critically endangered, Endangered, Vulnerable **communities**	Endangered species, Endangered populations, Endangered ecological communities, Species presumed extinct, Vulnerable species	Threatened taxa, Threatened Communities	Endangered (extant), Presumed extinct, Vulnerable, Rare	1. Endangered, Vulnerable, Rare 2. Protected	All fauna is wholly protected. Threatened fauna categories include critically endangered, endangered and vulnerable. Fauna in need of special protection include: Rare or likely to become extinct; Presumed extinct; Otherwise specially protected	Protected, Specially Protected	Presumed Extinct, Endangered, Vulnerable, Rare
Other Categories	Key Threatening Processes	Key threatening processes	Potentially Threatening Processes	Critical habitats, threats etc. are not listed but can be considered during preparation of Recovery Plans and Threat Abatement Plans for listed taxa				Common wildlife, International wildlife, Prohibited wildlife, Critical habitats, Threatening processes

	Commonwealth	NSW	VIC	TAS	SA	WA	NT	QLD
Criteria used for status assessment	As defined in the Regulations, with consideration of specific Taxa Action Plans and Conservation Overviews. IUCN and Milsap criteria for ranking also considered where relevant	As defined in the Act (Section 220F), based on IUCN criteria	As defined in the Regulations, with more detailed criteria also considered, as published in Flora and Fauna Guarantee Information Paper No.1	As defined in the Act, based on IUCN. Guidelines for application of criteria prepared by SAC and published in Gazette. These are currently being reworked with a view to their relevance to invertebrates and marine life	1. Not strictly defined in Act. Species listed on advice from biologists within DENR, State Herbarium and SA Museum, with consideration of IUCN and CITES criteria. 2. Evaluated on individual species- and case-by-case basis with scientific, industry and policy input. IUCN, CITES and Federal lists considered	Defined in CALM Policy Statement 33 – Criteria for Threatened Fauna and Specially Protected Fauna. Ranking of threatened species based on IUCN criteria (CALM Policy Statement 50)	IUCN criteria considered. Commission intends to use "current" criteria as they develop, ie. IUCN and other States' criteria	As defined in the Act, with consideration of CITES lists. Changes moving towards IUCN criteria with consideration of modified Milsap ranking for fauna
Nominations	Any person	Any person	Any person	Any person				
Assessment of nominations / decisions on listing	Endangered Species Scientific Subcommittee advises Minister. Additions or deletions from lists published in Gazette	Fisheries Scientific Committee makes recommendations to the Minister	Scientific Advisory Committee makes recommendations to the Minister	Scientific Advisory Committee makes preliminary recommendation, then (following period of public comment) a final recommendation to the Minister	No formal committees	Threatened Species Scientific Committee responsible for listing and ranking within IUCN categories. Recommends to Minister	Parks and Wildlife Commission (which must have 3 scientists) makes recommendations to the Minister after public process	Scientific Advisory Committee advises Minister on listings after assessment of nominations
Location of lists	Schedules 1, 2, 3 of ESP Act 1992	Schedules 4, 5, 6 of FM Act 1994	Schedule 2 of the FFG Act 1988	Schedules 3, 4, 5 of TSP Act 1995	1. Schedules 7, 8, 9 of NPW Act 1972 2. Reg. 6, Part 3 – Protected fish	Schedules 1, 2, 4 in the WC (SPF) Notice 1999	Schedule 7 of the TW Regulations (lists specially protected animals)	Schedules 1, 2, 3 of the NC (W) Regulation 1994

8.2.10 Territories

Norfolk Island

Norfolk Island is administered in accordance with the *Norfolk Island Act 1979*, which conferred a measure of self-government on Norfolk Island as a territory under the authority of the Commonwealth. Wide powers are exercised by a nine-member Legislative Assembly, elected for a three year term. Generally, Commonwealth laws do not apply to Norfolk Island unless expressed to do so (DTRS 2000a). However, Commonwealth environmental legislation (such as the *Environment Protection and Biodiversity Conservation Act 1999*) applies to areas of Commonwealth land on the Island and to all marine waters (i.e., the EEZ, up to 200 nm around the Island). A Norfolk Island Conservator is appointed (within Environment Australia) to manage environmental issues arising in relation to these areas.

The Norfolk Island government also has its own environmental legislation, including the *Norfolk Island National Park and Botanic Garden Act 1984* and the *National Parks and Wildlife Conservation Act 1975*, and jointly manages, with the Commonwealth, some reserves on Commonwealth land. However, there are no declared protected areas in marine waters. An area 39 nm by 50 nm immediately surrounding the Island is known as the Norfolk Island Fishery or "box" where the Norfolk Island community enjoys exclusive rights to fish. The Norfolk Island Fisheries Consultative Committee (NIFCC) is responsible for advising the Australian Fisheries Management Authority (AFMA) on the management of commercial fishing around Norfolk Island. The NIFCC is chaired by AFMA and included representatives from the Norfolk Island Legislative Assembly, the Norfolk Island Fishing Club, the Administrator's Office and the Administration of Norfolk Island.

Christmas Island

Sovereignty of Christmas Island was transferred to the Commonwealth of Australia (from the UK) in 1958 under the *Christmas Island Act 1958*. Commonwealth legislation thus applies, and the national park (which includes much of the fringing reef, out to 50 m from shore) is managed by Parks Australia North (a branch within Environment Australia).

Australian Antarctic Territory and Heard and McDonald Islands

Administration of the Australian Antarctic Territory and the sub-Antarctic territory of Heard and McDonald Islands is the responsibility of the Australian Antarctic Division, within the Commonwealth portfolio of Environment and Heritage. Australia's international obligations in relation to use and protection of the Antarctic Environment are defined under several international agreements (*Antarctic Treaty 1959*, *Protocol to the Antarctic Treaty on Environmental Protection (Madrid Protocol) 1991*, *Agreed Measures for the Conservation of Antarctic Fauna and Flora*, *Convention on the Conservation of Antarctic Marine Living Resources 1980 (CCAMLR)*; see Table 8.1). Other relevant Commonwealth Acts, apart from those establishing Australia's obligations under international agreements, include the *Australian Antarctic Territory Acceptance Act 1933*, *Australian Antarctic Territory Act 1954*, and the *Heard Island and McDonald Islands Act 1953*. Bönner (1990) provides an overview of international agreements and how they relate to conservation in the Antarctic.

8.2.11 Legislation at the local level

Australia has a great diversity of local council regulations but local council by-laws and conservation regulations can play an important role in the conservation of invertebrates by maintaining and protecting habitat (Yen and Butcher 1997).

8.3 National policies and strategies

8.3.1 Ecologically Sustainable Development

The origins of the concept of sustainable development stem back to the release of the World Conservation Strategy in 1980, and perhaps more importantly, the 1987 report of the World Commission on Environment and Development, Our Common Future (the Brundtland Report). The Brundtland Report recognised that sustainable development means adopting lifestyles within the planet's ecological means, made it clear that the world's current pattern of economic growth is not sustainable, and argued that a new type of development is required to meet foreseeable human needs (Commonwealth of Australia 1992).

While there is no universally accepted definition of sustainable development, in 1990 the Commonwealth Government adopted the following definition for Ecologically Sustainable Development (ESD) in Australia:

> *'using, conserving and enhancing the community's resources so that ecological processes, on which life depends, are maintained, and the total quality of life, now and in the future, can be increased'.*

The IUCN/UNEP/WWF strategy for sustainable living, "Caring for the earth", defined it as:

> *"improving the quality of human life while living within the carrying capacity of supporting ecosystems" based on respect and care for each other and the Earth"* (IUCN/UNEP/WWF 1991).

In other words, a use is sustainable if it is within the resource's capacity for regeneration (IUCN/UNEP/WWF 1991). Yen and Butcher (1997) summarised sustainable development as development that should "not … be at the expense of other groups or later generations, nor threaten the survival of other species. Development must protect the structure, functions and diversity of the world's natural systems, on which species depend".

In 1990, the Commonwealth Government set out to systematically identify what needed to be done in Australia to embrace the principles of ESD by establishing Working Groups to look at sustainability issues in key industry sectors. These Working Groups presented reports in November 1991 covering agriculture, forest use, fisheries, manufacturing, mining, energy use, energy production, tourism and transport, and in 1992 further reports were presented on various intersectoral issues and greenhouse (Commonwealth of Australia 1991). Out of this process the *National Strategy for Ecologically Sustainable Development* (NSESD)[212] was developed, and was endorsed by Heads of Government in 1992. Its principles are now supposed to inform all other government policy.

National Strategy for Ecologically Sustainable Development

The Core Objectives as recognised in the NSESD are:

- To enhance individual and community well-being and welfare by following a path of economic development that safeguards the welfare of future generations;
- To provide for equity within and between generations; and
- To protect biological diversity and maintain essential ecological processes and life-support systems.

The Guiding Principles of the NSESD are:

- Decision making processes should effectively integrate both long and short-term economic, environmental, social and equity considerations;
- Where there are threats of serious or irreversible environmental damage, lack of full scientific certainty should not be used as a reason for postponing measures to prevent environmental degradation (the "precautionary principle");
- The global dimension of environmental impacts of actions and policies should be recognised and considered;
- The need to develop a strong, growing and diversified economy which can enhance the capacity for environmental protection should be recognised;
- The need to maintain and enhance international competitiveness in an environmentally sound manner should be recognised;
- Cost effective and flexible policy instruments should be adopted, such as improved valuation, pricing and incentive mechanisms; and
- Decisions and actions should provide for broad community involvement on issues that affect them.

Many of the Working Group recommendations will assist the conservation of marine invertebrates in an indirect, but very important way, such as reducing greenhouse gas emissions, reducing pollution, and the adoption of ecologically sustainable land and water management practices. Some of the areas dealt with, such as Fisheries Ecosystem Management (Part 2, Chapter 2) and Coastal Zone Management (Part 3, Chapter 17) are of more direct relevance. The requirement to incorporate ESD principles into decision-making has been included in some Commonwealth, State and Territory legislation, such as the Commonwealth's *Fisheries Management Act 1994* and *Environment Protection and Biodiversity Act 1999*.

[212] http://www.environment.gov.au/psg/igu/nsesd/index.html

8.3.2 Biological diversity

The internationally accepted importance of conserving biodiversity was recognised by the Convention on Biological Diversity (Section 8.1.3), and this is reflected in the *National Strategy for the Conservation of Australia's Biological Diversity* (Commonwealth of Australia 1996)[213]. This Strategy, which has been endorsed by all State, Territory and Commonwealth governments, has as its goal the protection of biological diversity and the maintenance of ecological processes and systems. It recognises that biodiversity conservation results in many benefits, that there is a need for more knowledge and better conservation practices, and that we share the earth with many other life forms that have intrinsic values.

National Strategy for the Conservation of Australia's Biological Diversity
The Strategy consists of the following sections:

- Conservation of biological diversity across Australia – conservation within and outside reserves; threatened species, etc;
- Integrating biological diversity conservation and natural resource management – sectoral issues for natural resource industries including fisheries, forestry, tourism etc.;
- Managing threatening processes – relevant processes covered include alien species, pollution control, and climate change;
- Improving our knowledge;
- Involving the community;
- Australia's international role; and
- Implementation.

Parts of the Strategy directly relevant to marine invertebrate conservation include recommendations calling for:

- All Governments to "establish and manage a comprehensive, adequate and representative system of protected areas covering Australia's biological diversity" and
- "The development and implementation of a marine conservation and management strategy including mechanisms such as zoning for minimising the adverse impacts of such activities as coastal development, land-based discharge of pollutants, shipping and the harvesting of marine resources".

The National Strategy for the Conservation of Australia's Biodiversity also committed each State and Territory to developing its own regional strategy. To date, the status of these are as follows:

- **New South Wales**: The NSW NPWS released a draft biodiversity strategy in 1997 for public comment and this was made government policy in March 1999[214]. However, this policy did not include marine life due to the government's preference that this component be developed by NSW Fisheries; a strategy for aquatic habitats is now being developed (Talbot 1999).

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[213] http://www.ea.gov.au/biodiversity/publications/strategy/index.html
[214] Available as a pdf file on: http://www.npws.nsw.gov.au/wildlife/biodiversity.html

- **Victoria**: the Victorian strategy for conserving biodiversity[215] consists of three documents – *Victoria's biodiversity – our living wealth* (describing in broad terms the State's ecosystems, flora and fauna), *Victoria's biodiversity – sustaining our living wealth* (describing how biodiversity conservation can be integrated into actions throughout the community) and *Victoria's biodiversity – directions in management* which documents the necessary methods including management approaches and biogeographical regionalisations.
- **Tasmania**: The State biodiversity strategy is known as the Nature Conservation Strategy[216], which is currently in draft form (June 2001).
- **South Australia**: There is no specific strategy in place[217].
- **Western Australia**: The 1992 Draft Nature Conservation Strategy is currently being reviewed.
- **Northern Territory**: The NT has produced a Conservation Strategy for the Northern Territory (Northern Territory Government 1994), the goals of which include the conservation of existing biological diversity and the sustainable utilisation of natural resources.
- **Queensland**: While the Queensland Government has ratified the National Strategy and the legislative framework supports biodiversity conservation and management, there is, as yet, no actual Queensland biodiversity strategy to provide an overall strategic framework. The Strategy for the Conservation and Management of Queensland's Wetlands performs such a function for wetlands.

8.3.3 Threatened species

Under the 1992 Intergovernmental Agreement on the Environment, the Australian and New Zealand Environment and Conservation Council (ANZECC) was charged with developing and reporting on a strategy for a national approach to the protection of rare, vulnerable and endangered species, taking into account the *Australian National Strategy for the Conservation of Species and Communities Threatened with Extinction*[218] prepared by the Endangered Species Advisory Committee (ESAC). This strategy was to be implemented during the next 10 years.

Endangered Species Advisory Committee

The purpose of the ESAC review was to:

- Identify the additional measures needed to ensure the survival of endangered and vulnerable Australian plants, animals and ecological communities;
- Define overall aims and objectives for a program to save Australia's endangered and vulnerable species and ecological communities; and

[215] Background information and electronic versions via http://www.nre.vic.gov.au/web/root/domino/cm_da/nrence.nsf/frameset/NRE+Conservation+and+Environment?OpenDocument

[216] http://www.nccnsw.org.au/member/cbn/projects/LifeLines7.3/State_TAS.html

[217] See http://www.environment.sa.gov.au/sustainability/ for summary of programs that include survey work and salinity programs.

[218] http://www.biodiversity.environment.gov.au/threaten/strategy/strategy.htm

- Outline the steps required to achieve the objectives, including the scope for Commonwealth action under its Endangered Species Program.

National approach to the protection of rare, vulnerable and endangered species

The rationale for the national endangered species strategy as a key element in the broader national biological diversity strategy is that it directs attention to the cases where there is immediate threat and forces attention to the wider environmental issues that have led to the decline.

The introductory sections present the case for action to conserve our endangered and vulnerable species and ecological communities, namely:

- The unique characteristics of Australian flora and fauna and the reasons for their conservation;
- The extinction of species in Australia and the main threatening processes;
- Responsibilities for conservation action and some recent developments that provide hope for the future.

The overall aims defined in the strategy are:

- To ensure that endangered and vulnerable species and ecological communities can survive and flourish;
- To ensure that endangered and vulnerable species and ecological communities retain their genetic diversity and potential for evolutionary development in their natural habitat; and
- To prevent further species and ecological communities from becoming endangered.

Objectives to achieve these aims cover:

- A national education program;
- Adequate resources for conservation agencies;
- Identification of endangered or vulnerable species and ecological communities;
- Increased research on threatening processes;
- Development and implementation of plans to mitigate the effects of threatening processes;
- Preparation and implementation of recovery plans for all endangered species and ecological communities and selected vulnerable species and ecological communities;
- Involvement of all levels of government;
- Provision of incentives and support for conservation on private and leasehold land;
- Introduction of appropriate legislation by the Commonwealth, States and Territories;
- Integration of endangered species conservation into economic planning and assessment procedures;
- Promotion and participation in international conservation efforts; and
- Measures to effectively implement the strategy.

8.3.4 Oceans policy

Australia's Oceans Policy[219] was launched on 23 December 1998, after a period of consultation following the release of *Australia's Oceans Policy - An Issues Paper*, for public comment in May 1998. The Policy, which covers all of Australia's EEZ, recognises that maintenance of healthy ocean ecosystems is fundamental and that a strong, competitive marine industry base is also important and depends on ensuring the long- term ecological sustainability of a wide range of ocean uses. Volume 1 deals with the context, general issues and principles, and some key initial actions (integrated planning and management; biodiversity conservation; pollution; tourism; community participation etc.), while Volume 2 deals with specific sectoral measures (Commonwealth of Australia 1998a, 1998b).

In December 1999, the National Oceans Office[220] was formed as an Executive Agency under the Commonwealth *Public Service Act 1999* to oversee implementation of the Plan. A key focus of the work of the NOO is the development of Regional Marine Plans (RMPs). Preparation of the first of these, for southeast Australia, is currently underway.

The Policy has been criticised by WWF Australia for focusing primarily on the development of economic opportunities and lacking a strong conservation vision (Moore 1998).

8.3.5 Marine Science and Technology Plan

Released in June 1999 as a companion document to the Oceans Policy, the *Marine Science and Technology Plan* provides a scientific underpinning for the implementation of the goals and initiatives of the Oceans Policy. It deals with issues such as marine industries, mineral exploration, defence and surveillance etc. as well as marine science, biodiversity and conservation.

There are three Programs:
- Understanding the marine environment;
- Using and caring for the marine environment;
- Infrastructure for understanding and utilising the marine environment.

Objectives relevant to marine invertebrate conservation

Program 1: Understanding the marine environment
This program has a number of objectives dealing with the need to increase our knowledge of various different aspects of the marine environment. The first 5 objectives, which relate to the geology, topography etc. of the seabed, major physical and oceanographic processes in the oceans, and their relation to climate, are indirectly relevant through improved knowledge of the ocean system as a whole. However, the objectives that most

[219] http://www.oceans.gov.au/oceans.jsp
[220] http://www.oceans.gov.au/

directly relate to the conservation of marine invertebrates, through increasing our knowledge of them and the systems of which they form a part, are:

- *Objective 6 - To understand marine biodiversity and biological processes in Australia's oceans;* and
- *Objective 7 - To understand the dynamics of Australia's marine habitats and ecosystems*

The immediate priorities within these objectives are the development of an integrated southern temperate research program and an integrated northern tropical research program. The aim of the **southern temperate program** would be to understand the nature and extent of marine biodiversity, the impacts of introduced marine organisms, and the oceanic and biological processes that sustain the various levels of biodiversity, at a regional level[221].

The aim of the **northern tropical program** would be to develop an understanding of marine resources and ecosystems, build basic knowledge for continuing success of commercial fishing, aquaculture and pearling industries, offshore petroleum and bilateral agreements with PNG and Indonesia.

Program 2: Using and Caring for the Marine Environment

This program deals with the management of human impacts on the marine environment, and includes objectives relating to specific sectors. Objectives 1-4 deal with the development and application of effective monitoring and assessment procedures and sustainable management practices; an improved understanding of the impact of land-based human activities on the marine environment; the provision of the scientific basis for sustainable multiple use management practices in the marine environment; and application of knowledge of the oceans' variability and change. Objectives 5-12 deal with the scientific research and engineering innovations required to underpin marine industries, including petroleum and minerals industries, wild harvest fisheries, aquaculture, shipping, marine tourism, and emerging industries such as those dealing with renewable energy. Objectives 13-15 relate to social and legal issues, including indigenous people, security and defence, and the implications of marine law and policy.

Program 3: Infrastructure for Understanding and Utilising the Marine Environment

This program deals with capacity building, including the provision of infrastructure and professional development, and the promotion of marine science and technology in the community.

Immediate priorities are:

- Improvement of the marine science and technology skills base;

[221] The rationale is that these areas contain some of the more important fisheries, tourism values and petroleum resources, and are most affected by mainland coastal development and aggregations of urban populations.

- Provision of physical infrastructure, in particular the refurbishment of the Research Vessels *Franklin* and *Southern Surveyo*r, and increasing the number of days they spend at sea as National Facilities; and the acquisition of other major facilities;
- Implementation of long-term marine observational programs; and
- Improvement of marine data management.

Undoubtedly the Plan contains a number of laudable objectives, many of which could be (directly or indirectly) of relevance to marine invertebrate conservation, largely through the emphasis on improving our knowledge and building a better skills base, infrastructure and community profile for marine science. However, the Plan has been criticised on a number of fronts, not least because there is no new funding associated with it for the implementation of the extra research (Luntz 1999), and because of its (perhaps undue) emphasis on particular marine industries. The main focus is on mapping of the continental shelf, with oil exploration likely to be the largest beneficiary (Luntz 1999).

8.3.6 Ocean Rescue 2000

The *Ocean Rescue 2000 Program* was announced in 1991 as a 10-year program to achieve conservation of Australia's marine environments. However, this program has been superseded by the Coasts and Clean Seas Program[222] under the National Heritage Trust (NHT), which has provided funding for similar and other marine conservation projects.

The Ocean Rescue 2000 program comprised six elements (Fien et al. 1999):

- The development of a nationwide representative system of marine protected areas;
- A state of the marine environment report (SOMER);
- A national marine education program;
- The development of a national marine conservation strategy;
- The marine and coastal community network; and
- The national marine information system.

Implementation of the program was a joint arrangement between the Australian Nature Conservation Agency (ANCA, now Environment Australia, EA), the Department of Environment, Sport and Territories (DEST) and the Great Barrier Reef Marine Park Authority (GBRMPA), in collaboration with State and Northern Territory agencies.

Marine and Coastal Community Network

The *Marine and Coastal Community Network*[223] was developed under the Ocean Rescue 2000 program (and is now funded under Coasts and Clean Seas). It is a national, non-government community-based network which aims to encourage and facilitate community support for the conservation and sustainable use of Australia's marine and coastal environments.

[222] http://www.ea.gov.au/coasts/ccs/
[223] http://www.mccn.org.au/

The Network has the following functions:

- To identify and facilitate liaison amongst community, industry and other user groups and individuals with interests in the marine and coastal environments; and
- To provide information to all spheres of government and facilitate community consultation and participation in marine and coastal initiatives.

A non-government organisation, the Australian Marine Conservation Society[224] (formally the Australian Littoral Society), is the national coordinator of the Network. The structure of the Network is based on the existing *National Threatened Species Network*[225], which was also established by ANCA.

State of the Marine Environment Reporting

Information about the State of the Marine Environment reporting is available on the Environment Australia website[226].

National Marine Education Program

The *National Marine Education Program* was coordinated by GBRMPA, with input from EA. The series of three TV community service announcements ("*Our Oceans - Our Heritage"*), produced by ANCA in collaboration with GBRMPA, were screened nationally during 1993 (Fien et al. 1999).

National Marine Information System

The *National Marine Information System* (NatMIS), now the Coasts and Oceans Information Centre is being developed as a comprehensive computerised scientific information base which includes data on various aspects of the marine environment. It is being developed by the Environmental Resources Information Network (ERIN) and has been supported by Ocean Rescue 2000 and ERIN funding. NatMIS has been used to help develop a representative system of marine protected areas, and to assist in monitoring the condition and biodiversity of marine environments. The Marine and Estuarine Protected Areas Inventory has been updated and loaded as one data layer onto NatMIS (Fien et al. 1999).

8.3.7 Water Quality

National Principles for the Provision of Water for Ecosystems 1996 and National Water Quality Management Strategy 1992.

Background.

A revised draft of the Australian and New Zealand Guidelines for Fresh and Marine Water Quality[227] was released for public comment on 19 July 1999. The Guidelines

224 http://www.amcs.org.au

225 http://www.nccnsw.org.au/member/tsn/

226 http://www.ea.gov.au/soe/coasts/index.html

represent a major shift in approach over previous Guidelines published in Australia and abroad, and feature:

- For the first time, recognition of aquaculture and harvesting of aquatic food species as a separate environmental value to be protected.
- For aquatic ecosystems:
    - For the first time, guidelines for sediment quality and detailed description of biological assessment.
    - Flexibility to refine the default guideline values for specific sites using a risk-based approach.
    - Focus on issues rather than levels of specific contaminants.
    - For physical and chemical stressors, a greater range of ecosystem types represented, as well as an introduction to environmental flows..

8.4 Main issues and recommended actions

Issues

Despite the existence of international programs and several substantial national policies concerned with marine conservation, there is a considerable disparity in the approaches to marine conservation, and the recognition of invertebrates, in different Australian jurisdictions. This applies to:

- The legislative recognition of marine invertebrates;
- The legislative framework in which conservation measures applicable to marine invertebrates can be implemented.
- The agencies responsible for the conservation of marine environments and the organisms that live in them.

The existence of multiple agencies responsible for the administration of various aspects of the marine environment results in complexity, inefficiency and lack of coordination and some rationalisation would be desirable in most jurisdictions.

- There is no separation of the agencies responsible for conservation and exploitation in the marine environment in some jurisdictions, with the likelihood of conflicts of interest arising.
- The separation of responsibilities into marine and non-marine, or aquatic and terrestrial, habitats by way of legislation and/or agencies, can result in the vulnerable transitional habitats (that contain unique biota) being ignored.

There is often an inconsistent approach to marine conservation. For example fishing is often allowed in marine reserves but there are blanket bans on harvesting or collecting even superabundant, or the dead remains of, marine invertebrates.

Collecting (of shells etc.) by responsible collectors, interested children etc. should not be generally discouraged (demonstrated local or specific threats to the habitat or fauna

[227] http://www.ea.gov.au/water/quality/

should be managed individually) as such activities are a means of gaining interest in invertebrates and a potential source of valuable information.

- For example, dead (empty) strandline shells/tests etc., should **not** be treated identically to living specimens (as they are in some jurisdictions) - this resulting in a potential loss of information and an educational tool.

Recommended actions

We recommend **the adoption of a uniform national approach** that will result in: A consistent or coordinated approach to marine biodiversity conservation and relevant legislation in Australia.

The recognition of marine invertebrate taxa as fauna of equal status to marine vertebrates.

The emphasis in legislation should be on habitat conservation and reduction in threatening processes, with the focus on individual taxa, while being important in some circumstances, not the main thrust for marine invertebrate conservation.

The definitions employed in legislation need to be carefully framed to best achieve the desired conservation.

- More flexible approaches to collecting and harvesting are required; such as
- Imposing bag limits for specific taxa as an alternative strategy to blanket bans.

CHAPTER 9 – CHANGING ATTITUDES: EDUCATION AND PUBLIC INVOLVEMENT

9.1 Attitudes to invertebrate conservation - perceptions and semantics

There is a general awareness that there is a strong bias towards larger vertebrates (especially mammals and birds) in taxon-oriented conservation (e.g., Wilson and Peter 1988; Ponder 1992; Czechura 1994; Yen and Butcher 1997; Horwitz et al. 1999), and towards terrestrial habitats in systems-based conservation (Norse 1996; Norse 1997). This is translated into a grossly disproportionate resource allocation, heightening concerns amongst invertebrate biologists for the long-term viability of many invertebrate taxa and communities. There is an urgent need to increase the general awareness and interest by the public in invertebrate conservation, as highlighted by a number of writers (e.g., Kellert and Clark 1991; Czechura 1994; Meehan 1995; Yen and Butcher 1997; Horwitz et al. 1999; Lunney 1999). However, as pointed out by Lunney (1999), much environmental debate is not taxon driven but is focused on the conservation of particular areas. In these instances, it is important that any contribution that invertebrates can make be brought to the fore but, for many reasons, the icons will probably continue to mainly be the megafauna or flora in the terrestrial arena.

However, in many respects, the problems encountered with negative perceptions, low profile and lack of interest in invertebrate conservation in the terrestrial environment (Smith 1999b), are probably not as apparent in relation to marine invertebrates - although this remains to be tested. We suggest that marine invertebrates probably have a better public profile than terrestrial invertebrates because of the high levels of public, political, economic and scientific interest in coral reefs and general familiarity with larger, common invertebrates (whelks, starfish, crabs etc.). Many of the commonly encountered marine invertebrates are large, often colourful or otherwise attractive, mostly harmless, and a "purpose" or "use" can often be readily identified (e.g., food, bait, or food for fishes). In addition, environments in the marine sphere are usually not privately owned so there are fewer potential serious conflict situations.

While this view is less pessimistic than that regarding non-marine invertebrates (e.g., Yen and Butcher 1997), there are still serious problems in terms of perceptions and attitudes. As discussed in Chapter 3, the greatest impediments to marine invertebrate conservation are lack of knowledge and awareness of the situation, adverse public perceptions and lack of interest in marine invertebrates. As well as the direct effects, such as the low level of government and private sector funding for conservation-related research or infrastructure, there is also a general lack of political will to enforce conservation measures for these organisms, and a strong bias in resource provision to more “popular” and “appealing” organisms such as the higher vertebrates.

Public support is a vital part of successful conservation (e.g., Horwitz et al. 1999; Lunney 1999) and the research base providing the necessary information. It is expressed through political canvassing, financial support, practical assistance and access to areas in private ownership, as well as the provision of specimens or information (Czechura 1994).

Perceptions

Perceptual issues common to the promotion of conservation in Australia (Commonwealth of Australia 1984; Yen and Butcher 1997) include:

- A belief in unlimited capacity (the size of Australia or the oceans);
- The lack of recognition that there are limits to the capacity of life support systems' to withstand human impacts;
- A belief that conservation does not apply to areas outside of national parks and that national parks are an adequate method of conserving the biota; and
- The belief that humans are superior to all other life (encompassing the right to do as we wish with other species and the environment).

Despite these negative perceptions, more Australians are now aware of environmental issues and support environmental causes than previously. In addition, the environment has become a major political issue in recent years, locally, nationally and internationally. The stakeholder groups involved in conservation infrastructure and support range across the entire community (Figure 9.1).

Brailovskaya (1998) noted the public perceptions of marine “wildlife” in the USA, where, “unlike terrestrial wildlife, most commonly known marine species in New England are usually considered food”. A survey of residents in Connecticut (Kellert 1993) showed that most people expressed feelings of aversion, dislike or fear toward most invertebrates. More positive attitudes were expressed when the taxa possessed aesthetic or practical value. The general dislike was mainly focused on terrestrial invertebrates (e.g., insects, spiders etc) but also mentioned octopuses etc. (Kellert 1993). These perceptions resulted from:

- The small size of most species;
- Their high diversity and abundance;
- Adverse impressions caused by a small number of pest species;
- Fear; and
- The idea that they were a low form of life.

Similarly, Smith (1999b) discussed how the prejudices of the general public, which lead to invertebrates being widely ignored and despised, make it difficult for anyone attempting to use the electronic media to push the case for invertebrate conservation.

However, these negative impressions are all, in large part, false. While the majority of species are small, many others are large (e.g., giant clams, crayfish, squids). Together, many tiny colonial animals can form massive structures with the coral reefs of eastern Australia forming the largest living structures in the world. Again, while many species are very abundant, others are long-lived and have low reproductive rates. Many invertebrates are also extremely specialised and highly susceptible to habitat modification or destruction.

While many terrestrial invertebrates are considered nuisances or pests, or are parasites or vectors for disease, this is less so in the marine environment. The fear of insects and spiders is not an issue in the marine environment, but some marine taxa inspire these reactions (stingers - such as the Box Jellyfish, and poisonous species, such as Blue Ringed Octopus, etc.).

The term "invertebrates" is a catchall for all animal groups other than the vertebrates (see 9.1.1). It includes many groups that have incredibly complex organisation and, in some cases, highly developed nervous systems, eyesight, social structure and learning capacity. Nevertheless, the bias towards vertebrates is still paramount. Lunney (1990) reported that 77% of the readers of the "Australian Zoologist" preferred articles on terrestrial vertebrates and only 6% preferred invertebrates (the least preferred group) and, although 77% of readers were interested in conservation zoology, only 10% were interested in invertebrates. This same bias is also almost certainly reflected in the interests of the managers and researchers in conservation agencies, as well as in the universities where training of future staff occurs (Ponder 1992; Yen and Butcher 1997). This is also reflected in the lack of media attention given to invertebrates except for "nasties" (Smith 1999b), especially by television.

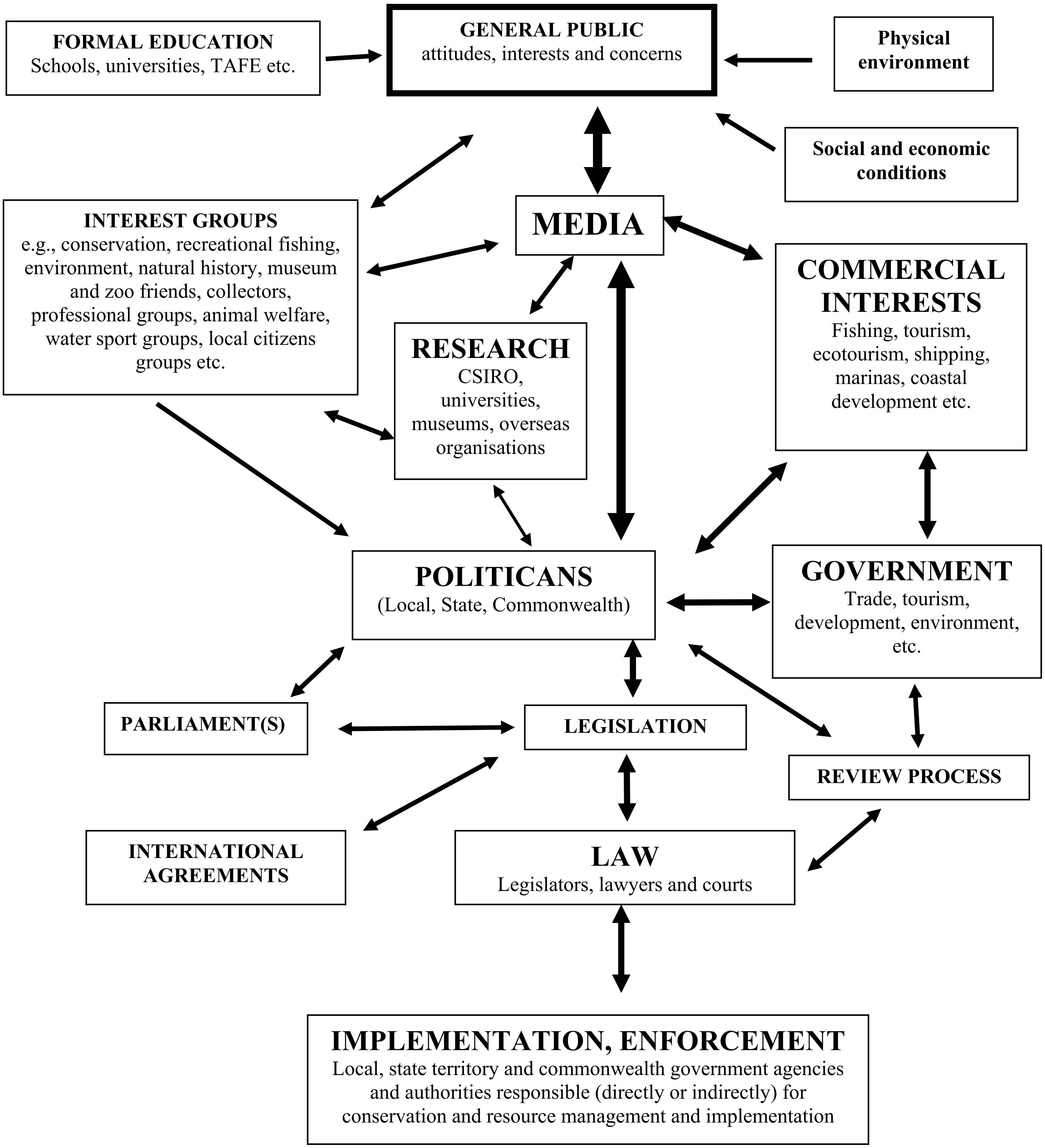
Yen and Butcher (1997) stress that there is a need to undertake market surveys of attitudes towards invertebrates before major programs promoting invertebrate conservation are developed in Australia. They argue against the use of survey results from overseas as there may be cultural differences resulting in different attitudes, and these attitudes are likely to differ across the multicultural landscape of Australian society. The attitudes of the indigenous population is critical because, although Aboriginal and Torres Strait Islanders make up only about 1.5% of the population, they are managers to a large proportion of Australia (Aslin and Norton 1995), and are well represented along the coast especially in northern Australia. We concur that surveys of this kind would provide useful information. Because the unsatisfactory term "invertebrates" is not inclusive of a group, but is a catch-all for all non-vertebrate animals ranging from sponges and jellyfish to insects, starfish and squid, surveys should be designed to address attitudes regarding specific subsets of invertebrates rather than in the overall sense.

9.1.1 Abandon the use of the word "Invertebrates"?

One of the problems for the conservation of invertebrates is the use of the word "invertebrates" in a catch-all fashion (Lunney and Ponder 1999 - see also Section 1.1.2). The word means very different things to different scientists, let alone members of the public. For example, arthropods (or insects in particular) are used as a surrogate for all invertebrates in many texts advocating invertebrate conservation (e.g., see Yen and Butcher 1997 for many examples of this) or in terrestrial biodiversity studies. This is partly because entomologists have been amongst the most vocal of those advocating "invertebrate" conservation and comprise by far the largest group of invertebrate workers. However, this view of invertebrate conservation is extremely lop-sided and potentially just as detrimental to the overall cause as any other biased approach. It would be more honest, understandable, and probably more "saleable", to clearly identify the name of the taxon - or the ecosystem - that is the focus of concern or interest. Thus, in the case of insects, we should talk about the conservation of particular endangered butterflies, or native bees, without any implication that these extremely worthwhile projects are addressing "insect conservation" in any general sense, or, worse, "invertebrate conservation". Such perceptions are all important - few would suggest that koala conservation addresses the conservation concerns of other Australian mammals except, perhaps, in an incidental way, for a few co-existing taxa.

This issue is an important one when "selling" invertebrate conservation. For instance, programs addressing the conservation of intertidal "invertebrates" may well do better to focus on the two or three groups with which the public are familiar (decapod crustaceans (crabs), echinoderms (sea eggs and starfish), molluscs (snails, oysters etc.) and, in the tropics, corals). Looked at in another way, the conservation concerns regarding sponges are totally different than those concerning squids or butterflies. It is clearly ridiculous to continue to lump these hugely different groups in such a meaningless and artificial way and we use the term in this report only because the brief demands it. At the same time, the conservation of some habitats (such as coral reefs) may be best addressed in terms of the suite of species – not only "invertebrates" but also the fish and marine plants that occupy them. These ideas could be tested in public surveys, such as suggested below and by Yen and Butcher (1997).

Figure 9.1: A diagrammatic representation of some of the complex interconnections between public attitudes, government, agencies and education, with particular reference to conservation attitudes and outcomes.



9.2 Education

Education is an essential and integral component of improving the conservation status of marine invertebrates, both by the provision of information as well as a concerted effort to promote the values of the various groups of invertebrates and the conservation issues concerning them. While so doing will increase awareness and sympathy among the general community, this must also be translated into political awareness to have a substantial impact. In general, the lack of public awareness is probably largely due to a lack of readily accessible available information (e.g., contrast the availability of multiple choices of handbooks and guides to mammals and birds to most marine invertebrate groups) and the minimal attention paid to them in the formal education system.

What needs to be done

Education regarding marine invertebrates should be improved at all levels within the community through:

- The provision of better and more accessible information to the general community (e.g., field guides, popular books, exhibitions, web-based information, museum reference centres, documentaries, magazine articles, etc.);
- Increasing the level of content in appropriate courses and curricula in formal educational institutions at all levels.

There is a need to educate people about the tremendous level of diversity represented by marine "invertebrates", which comprise many different phyla (c.f. the vertebrates, which belong to a single phylum).

Selling points

A major hurdle in obtaining public interest in issues concerning marine invertebrates is undoubtedly the general lack of knowledge and awareness of these animals in the general community. However, some significant and positive starting points and existing knowledge can be built upon to develop greater levels of awareness and interest. These include:

- Existing high levels of significant interest, knowledge or concern (coral reefs are the most obvious example, although many in the general community are probably unaware that corals are animals).
- The beauty of many marine invertebrates (corals, many shells, nudibranch slugs, some starfish, etc.) gives them a significant advantage over most terrestrial invertebrates (with some obvious exceptions, particularly butterflies). These positive images can be used to arouse more general interest;
- Many already have a variety of well-known uses and values (e.g., seafood, recreational "users" - bait, scuba divers, shell collectors, etc.); and
- The majority are seen as benign rather than harmful (unlike many terrestrial invertebrates; although there are also obvious exceptions such as marine stingers, borers and fouling organisms).

9.2.1 Formal education

There is a need for more emphasis on invertebrates and their critical role in ecosystems and life-support systems, as well as a promotion of their conservation needs, at all levels of education in Australia. This not only involves changing the emphasis in curricula but also educating (training) teachers at all levels (Yen and Butcher 1997).

Invertebrates should be an essential and integral part of the natural science and environmental component of any curriculum and will result in a greater appreciation for their importance. It should include:

- Learning about the enormous taxonomic, structural and biological diversity of invertebrates;
- Their critical importance in the functioning of ecosystems;
- Their critical importance in life support systems; and
- Their conservation needs.

The fascination of children with invertebrates is legendary but this interest is, it seems, not fostered but rather discouraged, or channelled into other avenues during formal education and exposure to other external influences (parents, media, etc.). Programs at primary school in which both children and parents are involved could be effective in overcoming some of these problems and may assist in the broader promotion of invertebrate conservation (Yen and Butcher 1997). Casual observation suggests that there has been a decreasing emphasis on invertebrates in biological courses at most Australian universities over the last two decades. This relative lack of interest at the university level is probably also influencing the lack of emphasis on invertebrates in schools (e.g., Haemig 1990).

Invertebrates in general, and marine invertebrates in particular, offer many opportunities for direct student involvement with animals. They are readily available, often common and many can be kept in aquaria for close observation. Thus, they can serve as models for observing life functions, as well as in demonstrating first hand the diversity of form in the animal kingdom. What a wonderful lesson in contrasting a starfish, snail, crab and polychaete worm!

The provision of more teaching aids (e.g., posters, books, wall charts, videos, CD ROMs, web pages) is required to complement and facilitate any increased emphasis on invertebrates in formal education programs.

Field studies with children and older students can be very important in generating interest, and while this can be hampered by the lack of field guides, this is probably only an important factor at higher levels (senior high school, universities). Field studies of marine invertebrates are obviously difficult, if not impossible, for students in inland areas, but since most of Australia's population lives near the coast, the majority of schools should have some degree of access to these environments.

The attitude and training of teachers at all levels (kindergarten to university) is critical to how invertebrates are perceived. This in turn is reflected in attitudes relating to the importance of invertebrate conservation. The organisation Marine Education Society of Australasia (MESA)[228] aims to promote appreciation and understanding of the marine environment through excellence in education, and is involved in training teachers about the marine environment.

We endorse Yen and Butcher's (1997) strong advocacy of courses on invertebrates for teachers and trainee teachers. They suggest that these courses could be run at nominated institutions, or as part of field studies centres. While there is no formal field studies network in Australia, facilities have been established by individual schools, education groups, national parks, and universities. These facilities are, of course, not all oriented towards marine studies. Certainly the Adult Education Programs in Marine Ecology, run by groups like WEA and Sydney University, have been very successful.

Field studies would be enhanced by the provision of local guides to the fauna, such as those established by the Field Studies Council in the UK who initiated AIDGAP (Aids to Identification in Difficult Groups of Animals and Plants) program (Yen and Butcher 1997).

There is also a marked reduction at the university level in invertebrate expertise, whole organism biology, taxonomy and systematics (Richardson and McKenzie 1992; see also Section 3.4 and Chapter 7). The reduction in these areas of expertise equates with undergraduates and postgraduates not receiving training in areas needed to study invertebrate diversity, particularly systematics (Yen and Butcher 1997).

The promotion of invertebrates and their conservation should also be pursued outside the mainstream formal education system in areas such as Adult Education courses and TAFE colleges.

9.2.2 Facilitating public access to information

Czechura (1994) highlights some issues regarding the delivery of invertebrate information to the general public:

- Need to target regional or perceptual groups of animals (e.g., guide to mangrove animals; the rocky shore animals of temperate Australia) as well as standard taxonomic approaches;
- The lack of common names for many taxa needs to be addressed;
- Pejorative images of invertebrates need to be countered; and
- In a tight economy, wider cooperation and sharing of resources will be necessary to meet aims.

A multifaceted approach is required, with the provision of field guides and non-technical references, museum displays and exhibitions, media articles and documentaries, and web-based delivery. It is in this later area (the web) that the greatest growth will undoubtedly

[228] http://www.mesa.edu.au/

occur in the next decade or so. It is probably the most cost-effective means of reaching a large audience and has endless possibilities - picture galleries, fact sheets, interactive keys, local or regional information etc.

There is a particular need to provide basic information that will enable interested individuals, community groups, councils, etc. to find out what marine invertebrates are living in their estuary or on their coast. This can be best achieved by the provision of online facilities such as ABIF (see Section 7.5.2) and the development of virtual zoological collection databases (see Section 7.2.3).

The provision of national or regional guides that facilitate the identification of marine invertebrates should be a priority. This objective can be achieved through handbooks, interactive keys etc. for particular taxonomic groups, locations or habitats.

9.3 Promoting marine invertebrate conservation

Effective conservation programs for marine invertebrates will require the development of awareness of the existence and importance of these animals, an understanding of the need for their conservation and the protection of their habitats, followed by an active involvement by the community in their conservation (Yen and Butcher 1997).

The key to invertebrate conservation is by developing a change in attitude towards one that is sympathetic and understanding at the public, political and administrative levels (Yen and Butcher 1997). This can be achieved, for instance, by emphasising the importance of marine invertebrates economically, as food for vertebrates, and their potential value as environmental indicators.

Education (in the broadest sense) is a fundamental component, and we generally concur with Yen and Butcher (1997) that this *should* include a major invertebrate awareness campaign with different approaches targeting different audiences. Yen and Butcher (1997) suggest that this would include media publicity, publications, workshops, conferences, as well as more collaboration between invertebrate biologists and biologists in other fields. While we agree with this general approach in principle, we question whether it is likely to be an achievable goal or effective (i.e., influencing a wide section of the community) unless it is driven by committed, well-resourced groups with particular, clearly defined objectives and substantial media access. Given the large range of diversity of invertebrates and the multiplicity of issues, especially in the marine sphere, there would probably have to be several campaigns running in parallel, some of which may possibly even be in conflict.

Encouraging the consideration of invertebrates in local conservation issues or planning, the development of material that will empower groups or individuals to be advocates for invertebrate causes and issues, and the development of tools that will assist in educational arenas, are, on the other hand, more realistic goals that should have a considerable impact over time.

9.3.1 Involving the community

Non-marine invertebrates have a major advantage over many vertebrates in that they are present, and therefore accessible, in all environments naturally inhabited by human beings (Yen and Butcher 1997). However, marine invertebrates are found in all marine habitats and even the common, widely distributed species can be used to advantage in programs aiming to raise the profile and improve the image of invertebrates. Such programs must first focus on raising the awareness and understanding of invertebrates, focusing on their intrinsic beauty, and of their positive and essential roles (Yen and Butcher 1997) in the ecology of marine systems.

Utilising marine invertebrates as educational tools

Marine invertebrates can be used in teaching about biodiversity, ecology, etc. by parents and teachers, especially as specimens of some taxa can be collected in beach drift without killing living specimens. Once there is a greater awareness, by teachers and students alike, of the diversity of life in particular habitats, these habitats should gain a greater intrinsic value. Similarly, learning about sustainable development, the conservation value of retaining good quality habitat and the value of appropriate management of these habitats could be enhanced through learning about invertebrates.

Involvement in conservation measures

Adoption of conservation measures should, ideally, be in a cooperative arrangement with local community groups. Failure to do this will likely result in failure in meeting the conservation objectives through lack of resources to maintain adequate control. No matter what level of funding is received, the success of any conservation program will rely primarily on its gaining public acceptance (Yen and Butcher 1997).

In some cases, individuals may fear that conservation measures may interfere with their traditional access to the sea or shore (e.g., food or bait collecting, recreational fishing, traditional access by indigenous Australians). For some, their livelihoods may depend on this access (e.g., those involved in aquaculture or commercial fishing). Such issues need to be addressed in any conservation plan. This strategy has certainly been adopted by the Great Barrier Marine Park Authority and by the Solitary Marine National Park by heavily involving all users in an extensive public participation program. Unless the users of such parks understand the rationale of zoning and feel that it is their plan, policing and law enforcement becomes almost impossible. Marine parks must almost of necessity (because of their size) adopt this approach, rather than relying on policing officers. With this in mind, prohibition is more likely to cause problems compared to offering involvement in conservation of threatened species (Yen and Butcher 1997) and the use of incentives for conservation is the best approach (Carlton 1986). Prohibition also reduces the likely level of involvement in the area concerned by school groups or amateur naturalists, etc. that would benefit from collecting a few specimens for identification or future reference. This in turn could reduce the information available about the area.

The farming of some species of Giant Clams is now being undertaken, a group containing some species whose wild populations are very vulnerable. This raises the possibility of economic return from rearing other rare or threatened marine invertebrates that have value either as food, as ornaments or as items for collectors.

Development of resources

The question of what resources, besides financial support, are required to run such programs has been addressed by Yen and Butcher (1997). The suggest that the programs should concentrate on:

- The beauty and diversity of Australia's native invertebrate fauna;
- The essential and beneficial roles played by the native invertebrate fauna;
- The importance of conserving invertebrates; and
- The small number of detrimental invertebrate species.

This will involve development of:

- Promotional material (posters, books, documentaries, web sites, etc.);
- Educational material (publications, electronic media, lectures, web sites etc.)
- Local action material (material aimed at local or regional issues or communities).

This promotional activity could be undertaken across a wide cross section of the community including individuals, local action groups, societies, government agencies, educational institutions, scientists and international organisations.

9.3.2 Involving the establishment

To be effective, conservation strategies must not only have grass-roots support, but must also have the support of the conservation agencies, scientists and politicians at all levels of government.

Conservation administrators

Yen and Butcher (1997) point out that many conservation administrators have little specialised invertebrate training, most having a background in vertebrate or plant biology. Thus they may not be aware of the needs and issues relating to invertebrate conservation. For example, many will see invertebrate conservation as a nearly impossible task because they assume that they may require the individual species management techniques often applied to vertebrates (Yen and Butcher 1997) (see also Chapter 4). These attitudes can be changed through the provision of information via, for example, workshops, conferences and publications. Yen and Butcher (1997) further advocate the employment of an "invertebrate biologist as an invertebrate policy person in each conservation department" as a way of helping to address this problem. Certainly, managers should seek the provision of relevant expertise relating to invertebrates but we question whether a single individual would be a suitable surrogate for such skills given the enormous diversity of invertebrates, their habitats and their biological attributes.

Scientists and scientific societies

While scientists, particularly biologists, have a more positive attitude towards "invertebrates" than the general community (Kellert 1993), many do not understand the need for their conservation, although this could be rectified by conferences, seminars, publications etc., as well as through collaborative research projects involving invertebrates (Yen and Butcher 1997).

Scientific societies can contribute to conservation by making submissions to government enquires, commenting on legislation, submissions, proposals etc. A list of the societies that have a general interest in marine issues is given in Appendix 2.

Scientists themselves have different ideas regarding invertebrate conservation and some areas are more heavily promoted than others. An analysis of the published proceedings of the three meetings on conservation and biodiversity of invertebrates held in Australia (Ingram et al. 1994; Yen and New 1997; Ponder and Lunney 1999) is presented in Figure 9.2. These show a heavy emphasis towards terrestrial systems with only 9% of the papers dealing with marine animals or systems (two of the 14 on supralittoral faunas). Similarly, of the particular groups focused on (see Figure 9.3), only 27.7% dealt with non-arthropod groups and 45.5% of the arthropod papers (35.7% of the total) exclusively dealt with insects. About 8% of those for which a taxon could be identified dealt with marine invertebrates. This may be explained in part by marine conservation issues being pursued in other fora (e.g., AMSA, Australian Coral Reef Society, etc.) there are also other avenues for non-marine conservation issues to be pursued (Entomological Society, general conservation meetings, etc.). We suggest that the proportions of the papers in these three meetings probably fairly reflect the distribution of effort in invertebrate conservation in Australia, with the notable exception of coral reef studies. There was an even more marked imbalance in the papers presented at the 1998 International meeting of the Society for Conservation Biology at Macquarie University, Sydney (Fig. 9.4).

Figure 9.2: The proportions of the papers by environment in the proceedings of three meetings on the conservation and biodiversity of invertebrates in Australia. General papers refer to those dealing with non-specific environments.

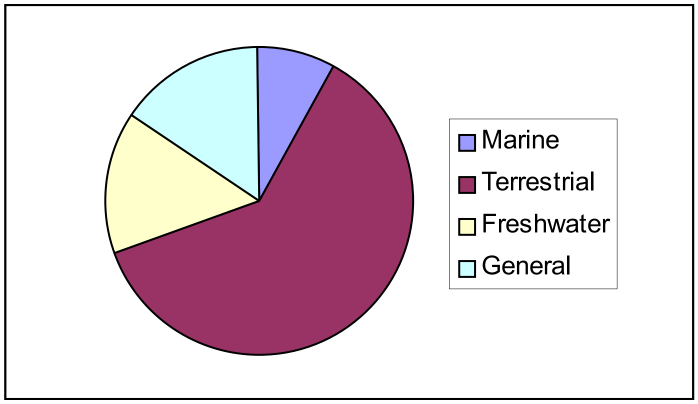


Figure 9.3: The proportions of the papers by taxon-group in the proceedings of three meetings on the conservation and biodiversity of invertebrates in Australia. General papers not focusing on identifiable taxa are not included. NM arthropods = non-marine arthropods, excluding those papers dealing specifically with insects, spiders or crustaceans. General marine - one paper dealt with marine invertebrates in general, similarly for General FW (= freshwater). Other Marine Inverts includes sponges (1 paper), bryozoans (1 paper), echinoderms (2 papers) and sessile marine invertebrates (mainly tunicates) (1 paper).

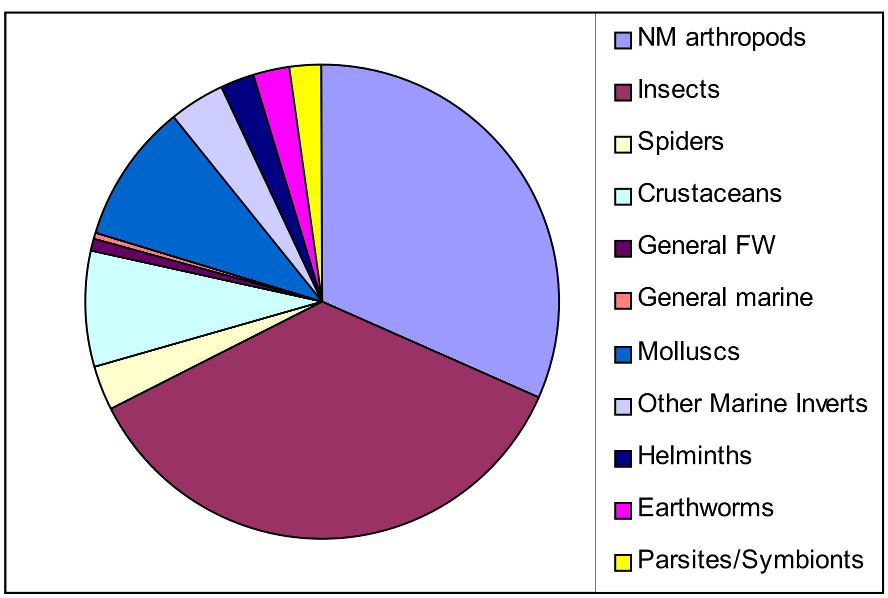
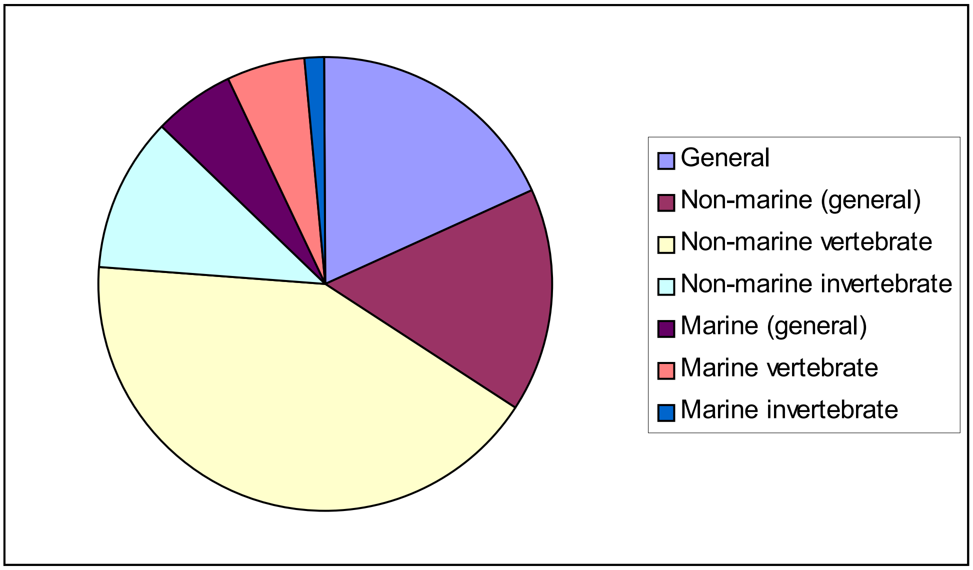


Figure 9.4: The proportions of papers by taxon and environment in the 1998 International meeting of the Society for Conservation Biology at Macquarie University, Sydney. General papers refer to those that did not specify taxon group or environment; marine (general) and non-marine (general) papers specified environment but not taxon group. Non-marine includes both freshwater and terrestrial.



9.3.3 Societies

A list of the societies and conservation groups involved directly or indirectly in marine conservation is given in Appendix 2.

9.3.4 Non-government conservation groups

Many non-government conservation organisations are demonstrating an increasing interest in broader conservation issues in recent years, although the interests of some are still predominantly focused on terrestrial vertebrates and forests, both extremely worthwhile causes in their own right. While such groups rarely, if at all, have invertebrates as a central concern, marine issues are increasingly in the forefront - particularly as regards marine pollution, whaling, overexploitation, shoreline development etc.

The importance of the involvement of non-government groups in conservation activities cannot be overrated. They play a major role in influencing public opinion (an hence political opinion), not only by acting as pressure groups, but by assisting in gathering critical information, providing on-ground support for conservation initiatives, including direct action where deemed necessary.

We agree with Yen and Butcher (1997) that educational material needs to be prepared for conservation groups on the value of invertebrate conservation. This may be an even easier message to sell for marine invertebrates than for their terrestrial counterparts, as they are a much more conspicuous component of most marine ecosystems. Certainly, the success of marine conservation in Australia will depend largely on the level of commitment of such groups to this cause.

Partnerships between community organisations and scientific organisations can be particularly effective and, potentially, have considerable influence. Allen (1999), for example, described how the partnership between the Marine and Coastal Community Network, the Threatened Species Network and the Community Biodiversity Network very effective promote marine invertebrate conservation.

The role of the media

The media probably has the greatest influence on public opinion and could be used to a much greater extent to promote positive invertebrate information and conservation (e.g., Smith 1999b), rather than focusing on negative images involving poisonous or otherwise harmful animals or trivialising invertebrate stories. This will not be an easy task (Yen and Butcher 1997), although there have been numerous popular documentaries dealing with marine systems and their conservation – especially relating to coral reefs. The positive impact of excellent documentaries dealing in large part with invertebrates in recent years is a welcome trend that needs to be encouraged, and built on.

9.4 Recommended actions

Increase the level of interest and involvement in marine invertebrates, and increasing knowledge on them, by the establishment of a field studies network in Australia.

- Establishment of a number of such facilities in coastal locations so that a wide section of the community around Australia can participate in marine studies courses.
    - Available facilities run by schools, education groups, national parks, and universities could be used as a starting point.

Influence attitudes to invertebrate conservation by:

- Encouraging the consideration of invertebrates in local conservation issues or planning, in conjunction with vertebrates etc. (Lunney 1999); and
- Develop material that will empower groups or individuals to be advocates for invertebrate causes and issues.

Encourage a positive attitude about invertebrates, their roles and uses, as well as their intrinsic values.

- Attitude surveys should be conducted to assist in the development of campaigns to promote awareness of marine invertebrates (and invertebrates in general) and of the need for their conservation; and

- Avoid the use of the word "invertebrates" because it encompasses such a huge, totally artificial grouping of animals with vastly different morphology, biology and ecology. Instead, we recommend that, wherever possible, specific, readily identifiable taxon or ecosystem labels be used.

Develop tools that will assist in educational arenas and for the public good.

- Include more content on marine invertebrates, and their importance for our wellbeing, in curricula;
- Train teachers and teacher trainees in issues relating to marine invertebrates; and
- Develop resources that will provide information to a wide range of users in the community. Much of this could be web-based but the provision of printed information should not be neglected.

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APPENDICES

APPENDIX 1. MUSEUM COLLECTION DATA

To get some idea of the size and content of Australian marine invertebrate collections, questionnaires were sent out in 1999 to all the State museums (Australian Museum, Museum Victoria, Queen Victoria Museum, Tasmanian Museum and Art Gallery (TMAG), South Australian Museum, Western Australian Museum, Museum and Art Gallery of the Northern Territory, Queensland Museum and Museum of Tropical Queensland). Questions were asked about the number of lots held for each taxon (see Figures 1 and 2) and the quality of geographic coverage of the state for each taxon (see Table 1). It should be noted that much of the information given is as provided in 1999-2000. Exact figures are impossible, and these data were often either estimated or 'guesstimated'. In addition, the quality of the geographic coverage is a subjective question and is therefore not easily compared between museums. As well as information about the collections, questions were asked about the number of staff available for marine invertebrates and the database system used (see Table 2), and the amount of unsorted, unregistered or unidentified material held in the collection that was not included in the answers to the questionnaire (see Table 3).

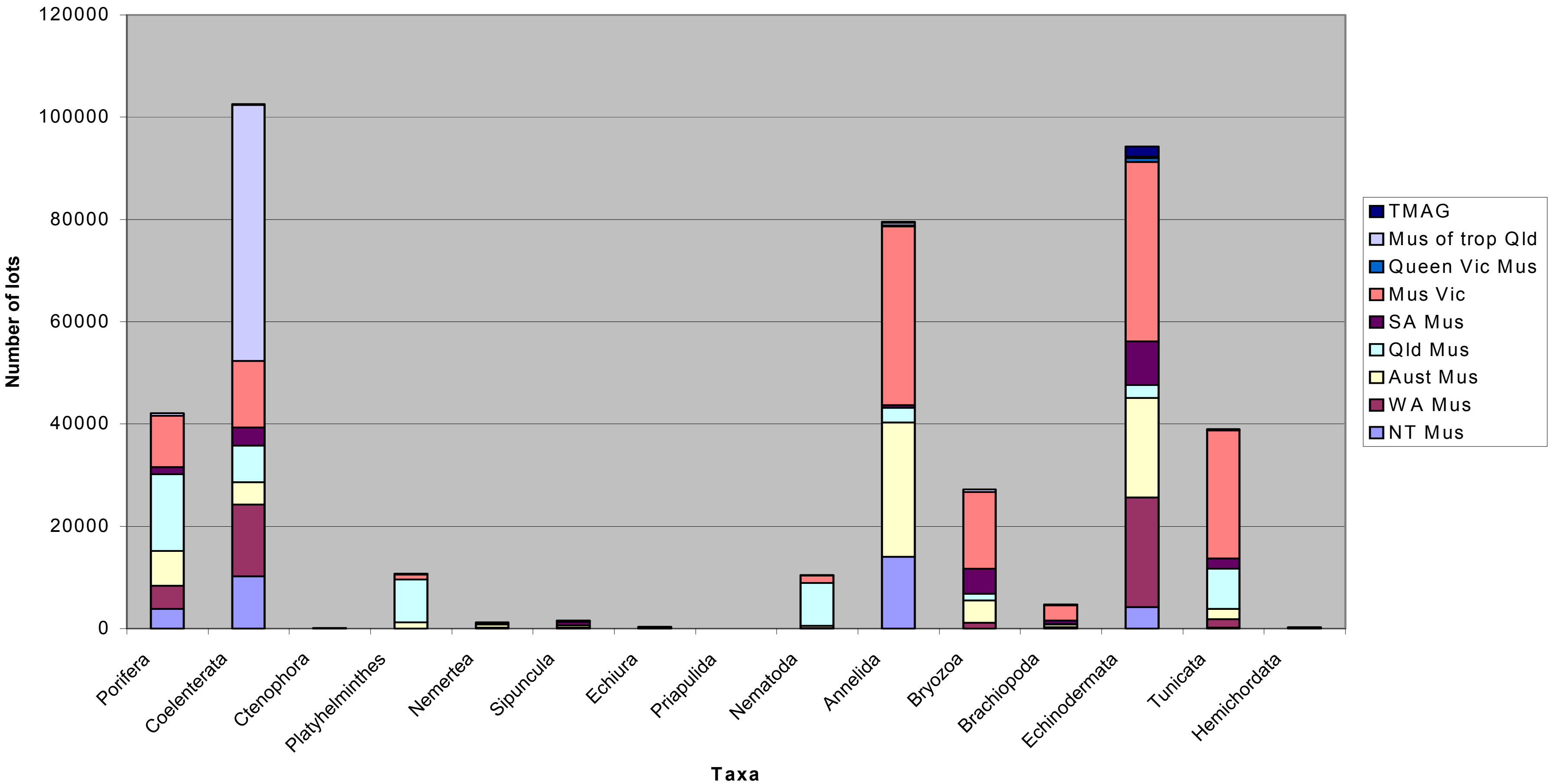

Figure 1: Approximate number of lots held by each museum for all marine invertebrate phyla, excluding Mollusca and Arthropoda. Only the marine members of each phylum have been recorded.

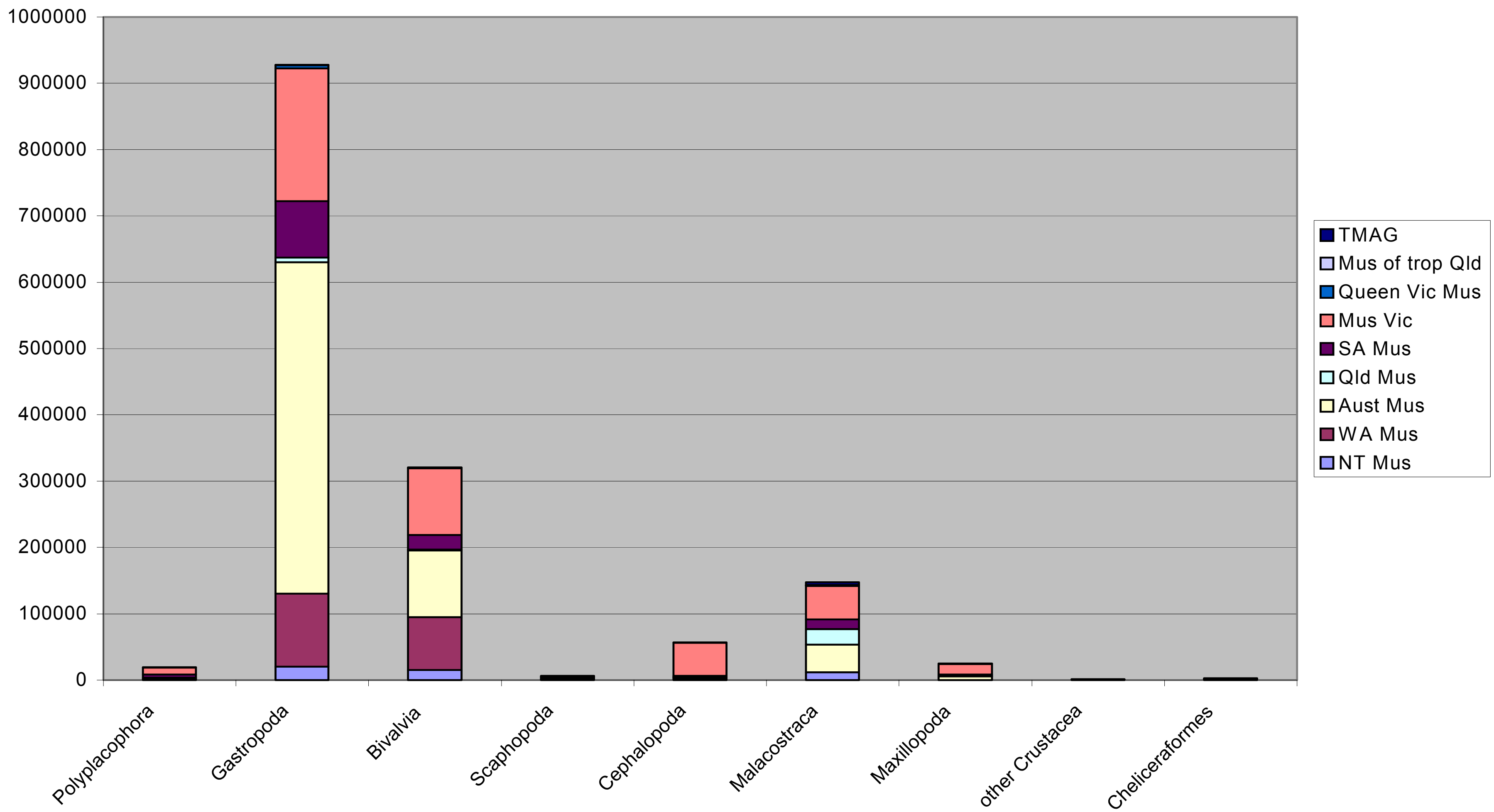

Figure 2: Approximate number of lots held by each museum for marine members of Mollusca and Arthropoda.

Table 1: The quality of the geographic coverage for each museum.

	# Lots	NSW	Vic	Tas	SA	WA (south)	WA (north)	NT	QLD	Christmas / Cocos Islands	Coral Sea	Tasman Sea	Sub-Antarctic Islands	Aust Antarctic Territory
Australian Museum	719,447													
Museum Victoria	566,156													
Queen Victoria Museum	9,225													
Tasmanian Museum and Art Gallery	7,859													
South Australian Museum	156,376													
Western Australian Museum														
Museum and Art Gallery of the Northern Territory	103,927													
Queensland Museum	87,957													
Museum of Tropical Queensland	54,910													

KEY:							
	0 (none)	1.5	1.0 (poor)	1.5	2.0 (adequate)	2.5	3.0 (good)

Table 2: The number of staff working on marine invertebrates in each museum. Other assistance can include volunteers, students and honorary workers. Numbers in brackets indicate the number of those staff members that are ecologists.

Museum	Research	Collection management	Other assistance	Database used
Australian Museum	8.8 (1.5)	6	4.2	Texpress
Museum Victoria	4 (2)	4	6 to 7	Texpress, soon to transfer to KEMu
Queen Victoria Museum	0	0	0.25	Texpresss
Tasmanian Museum and Art Gallery	1.25	0	2	Macintosh FileMaker Pro (Microsoft)
South Australian Museum	1	2	4	NONE
Western Australian Museum	0.33	0.83	0.25	Foxpro
Museum and Art Gallery of the Northern Territory	3	2	1	Filemaker Pro 5.0
Queensland Museum	11	5	10	R-Base
Museum of Tropical Queensland	2	1.5	2.3	R-base

Table 3: The approximate amount of material held by each museum that is unidentified, unsorted or unregistered and therefore not included in Figures 1 and 2 and Table 1.

Museum	Approx. amount of material	Content of unsorted, unidentified or unregistered material
Australian Museum	large unidentified collections both sorted and unsorted	Large unsorted, unidentified collections from: • NT, Vic, WA, NSW, Qld, Sub Antarctic Islands, Lord Howe Island, One Tree Island; • Elizabeth and Middleton Reefs, Tasman Sea, Torres Strait; • Lizard Island, far north Qld, Eurobodalla; • Plankton collections; • Cruises made by 'Franklin', 'Kimbla' ships from the east coast of Australia (off the shelf); • Cruises made by 'Kapala' off NSW coast; • Cruises by HMAS Cook in >2500m off Sydney. Large numbers of jars of sorted, unidentified material through entire collection (all phyla) - not accessible for researchers because not identified (so not registered / recorded) to family level, which is the basic level for invertebrate researchers to access and use
Museum Victoria	5000 unsorted	Collections from benthic habitats frm bays and shelves in Vic (1000s of lots) shallow water collections from algal habitats in Vic, Tas, SA, WA (100s of lots) continental slope of NSW-Tas (100s of lots) sedimentary habitats in Qld, northern WA (few 100s lots) Few samples from Antarctica
Queen Victoria Museum	small amounts unsorted	Small amount of unsorted - most is sorted to at least major groups.
Tasmanian Museum and Art Gallery	some unregistered	CSIRO marine has been given batches up to 200 reg numbers in all marine phyla for seamount etc. material. This is yet to arrive. More is also expected, but not with registration numbers yet. Many collections of molluscs non-registered.
South Australian Museum		
Western Australian Museum	over 2000 unsorted	10 20L polydrums, and 5 nallybins - well over 2000 specimens
Museum and Art Gallery of the Northern Territory	12500 both sorted and unsorted	Extensive (sorted, partly sorted and unsorted) material from the following field expeditions: NABALCO Gove survey (R. Hanley) Beagle Gulf Epibenthic Surney (NT Parks) Darwin Harbour survey (R. Hanley) Port of Darwin Survey (CSIRO and NT Museum) Ashmore Reef Collections (back to 1988)
Queensland	6000 sorted, 1000 unsorted	3000 both sorted and registered

Museum		1000 unsorted/unregistered Monogenean slide collection (Klaus Rhode) - ~700 slides ~3000 lots of bulk preserved material coll. in the last 20 yrs
Museum of Tropical Queenland	some unsorted	Cross-shelf transects of soft-sediment epibenthos from central Qld shelf Collections of deep-sea benthos from Coral Sea (Townsville to Cape York) and to depths of 3.5 km Shelf soft sediment echinoderm material (fully identified but not yet deposited in museum)

APPENDIX 2. GENERAL-INTEREST MARINE SOCIETIES AND MARINE CONSERVATION ORGANISATIONS AND GROUPS

Australian Marine Sciences Association (AMSA) - a professional scientific body representing marine science. Acts as a lobby group as well as supporting students and runs an annual scientific conference. http://www.uq.edu.au/amsa

Australian Coral Reef Society (formerly the Great Barrier Reef Committee) - a professional scientific society representing coral reef scientists. Prepares Handbooks, funds student research, and runs an Annual Scientific Conference and also provides input into government submissions on matters dealing with Australian coral reefs. http://www.tesag.jcu.edu.au/ACRS

Coast and Wetlands - a body of scientists and people interested in coastal wetlands primarily along the NSW coast of Australia. Acts as a lobby group and prepares submissions on matters relating to wetland conservation and management.

Queensland Littoral Society - a body of scientists and people interested in coastal wetlands primarily along the Qld coast. Acts as a lobby group and prepares submissions on matters relating to wetland conservation and management, and in raising public awareness of the value of wetlands.

Marine and Coastal Community Network - a consortium of marine conservation societies which acts as a clearing house in terms of providing local groups with information, access to specialists and develops liaisons between these groups through a newsletter and electronically. Also attempts to increase public awareness of marine conservation issues (Government-funded).
http://www.ozemail.com.au/~mccnet

Marine Education Society of Australasia - acts on behalf of its members to promote appreciation and understanding of the marine environment through excellence in education. MESA provides a national forum and advocacy on marine education and encourages international initiatives in this field.
http://edx1.educ.monash.edu.au/peninsula/seaweek/mesa.htm
Contacts: The Marine Education Society of Australasia, PO Box 616, Indooroopilly, Qld 4068, Australia, Tel: 07 3378 0128 / Fax: 07 3378 0128

Australian Marine Conservation Society.
http://www.amcs.org.au

Community Biodiversity Network
http://www.cbn.org.au

Marine Life Society of South Australia
http://www.mlssa.asn.au/

Marine Studies Group, Victoria. Now part of the Field Naturalists Club of Victoria.

Malacological Society of Australasia - devoted to the study of molluscs (htttp://www.austmus.gov.au/malsoc) and publish *Molluscan Research*. There are also several "shell clubs".