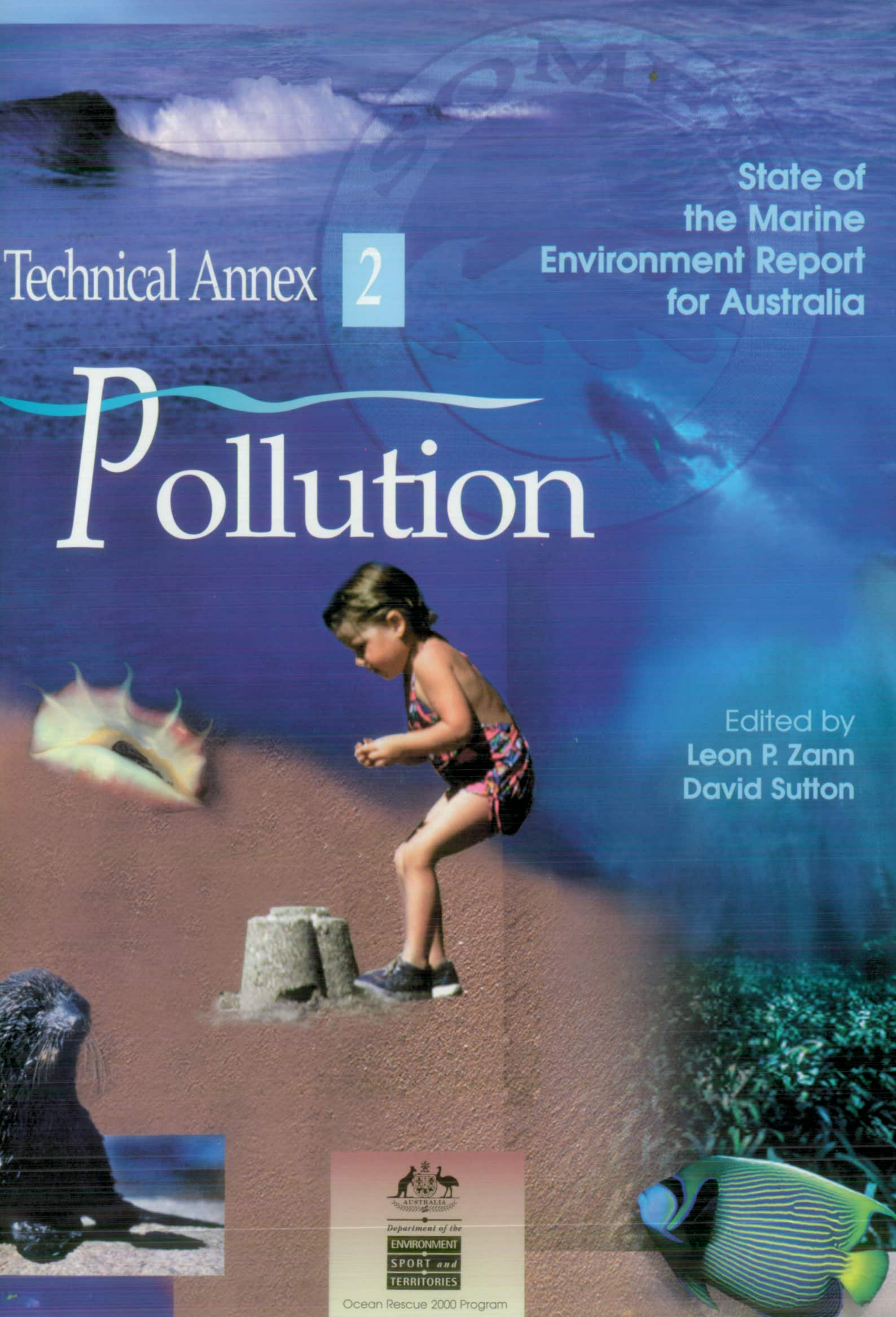
State of the Marine Environment Report for Australia

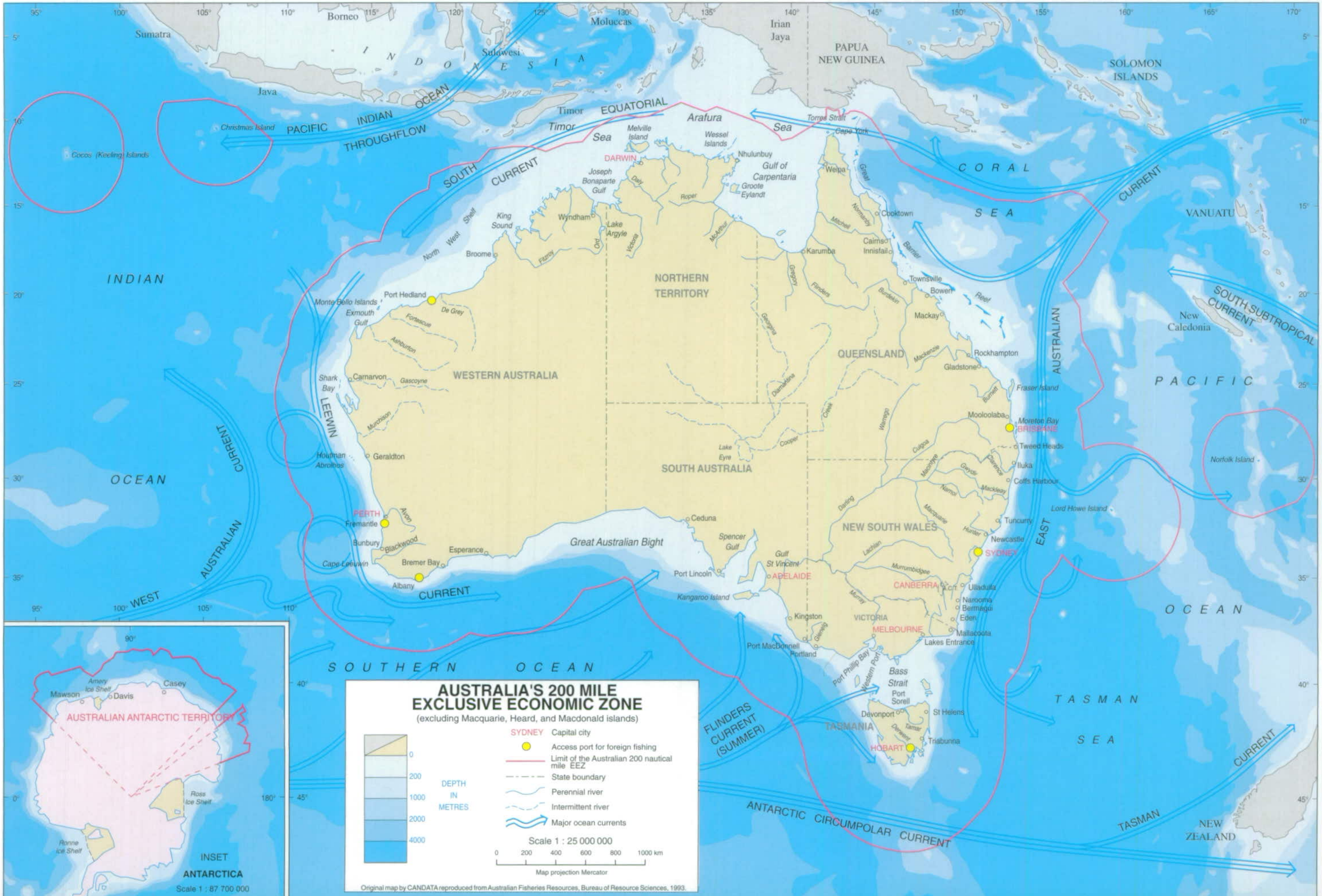
Technical Annex 2

Pollution



Edited by
Leon P. Zann
David Sutton

AUSTRALIA
Department of the
ENVIRONMENT
SPORT and
TERRITORIES
Ocean Rescue 2000 Program



AUSTRALIA'S 200 MILE
EXCLUSIVE ECONOMIC ZONE
(excluding Macquarie, Heard, and Macdonald islands)
SYDNEY Capital city
Access port for foreign fishing
Limit of the Australian 200 nautical mile EEZ
State boundary
Perennial river
Intermittent river
Major ocean currents
DEPTH IN METRES
0
200
1000
2000
4000
Scale 1 : 25 000 000
0 200 400 600 800 1000 km
Map projection Mercator
Original map by CANDATA reproduced from Australian Fisheries Resources, Bureau of Resource Sciences, 1993.
INSET
ANTARCTICA
Scale 1 : 87 700 000
AUSTRALIAN ANTARCTIC TERRITORY
Mawson
Amery Ice Shelf
Davis
Casey
Ross Ice Shelf
Ronne Ice Shelf
INDIAN OCEAN
SOUTHERN OCEAN
PACIFIC OCEAN
CORAL SEA
TASMAN SEA
Timor Sea
Arafura Sea
Gulf of Carpentaria
Great Australian Bight
INDONESIA
PAPUA NEW GUINEA
Irian Jaya
SOLOMON ISLANDS
VANUATU
New Caledonia
NEW ZEALAND
Sumatra
Java
Borneo
Sulawesi
Moluccas
Timor
Christmas Island
Cocos (Keeling) Islands
Norfolk Island
Lord Howe Island
PACIFIC INDIAN OCEAN THROUGHFLOW
EQUATORIAL
SOUTH CURRENT
LEEWIN CURRENT
WEST AUSTRALIAN CURRENT
EAST AUSTRALIAN CURRENT
SOUTH SUBTROPICAL CURRENT
FLINDERS CURRENT (SUMMER)
ANTARCTIC CIRCUMPOLAR CURRENT
TASMAN CURRENT
WESTERN AUSTRALIA
NORTHERN TERRITORY
SOUTH AUSTRALIA
QUEENSLAND
NEW SOUTH WALES
VICTORIA
TASMANIA
DARWIN
PERTH
ADELAIDE
MELBOURNE
CANBERRA
SYDNEY
BRISBANE
HOBART
Port Hedland
Broome
Wyndham
Carnarvon
Geraldton
Fremantle
Bunbury
Albany
Esperance
Bremer Bay
Ceduna
Port Lincoln
Kingston
Portland
Port MacDonnell
Lakes Entrance
Mallacoota
Eden
Bermagui
Narooma
Ulladulla
Newcastle
Tuncurry
Coffs Harbour
Iluka
Tweed Heads
Mooloolaba
Moreton Bay
Gladstone
Rockhampton
Mackay
Bowen
Townsville
Innisfail
Cairns
Cooktown
Weipa
Karumba
Nhulunbuy
Devonport
St Helens
Triabunna
Bass Strait
Port Sorell
Torres Strait
Cape York
Melville Island
Wessel Islands
Groote Eylandt
Joseph Bonaparte Gulf
King Sound
Lake Argyle
Lake Eyre
Spencer Gulf
Gulf St Vincent
Kangaroo Island
Port Phillip Bay
Western Port
Shark Bay
Exmouth Gulf
Monte Bello Islands
Houtman Abrolhos
Cape Leeuwin
Fraser Island
North West Shelf
Great Barrier Reef

The State of the Marine Environment Report for Australia Technical Annex: 2

Pollution

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

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MARINE PARK AUTHORITY

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AUSTRALIA

Department of the
ENVIRONMENT
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Introduction

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

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

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

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

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

Ocean Rescue 2000 program

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

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

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

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

The State of the Marine Environment Report

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

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

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

The SOMER Process

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

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

Much of the information collected for SOMER is unpublished. Because of the scientific value of this information, a range of papers is being published. SOMER identified declining water quality, and particularly elevated nutrients and sediments, as one of the major issues in Australia's marine environment. This volume contains eight papers on aspects of marine pollution.

Leon P. Zann
SOMER Coordinator

SOMER process

topics identified by workshop

expert scientists commissioned to produce **technical papers**

papers peer reviewed

Selected papers published: **SOMER Technical Annexes 1 & 2**

technical papers, condensed & management implications identified

reviewed by authors, SOMER Advisory Committee and relevant govt agencies

combined to produce technical report: **SOMER Technical Summary**

major findings and issues identified & collated

reviewed by SOMER Advisory Committee and relevant govt agencies

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

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

applications of SOMER:

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

Acknowledgments:

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

Contents

The problems of nutrients and eutrophication in the Australian marine environment

Jon Brodie

Great Barrier Reef Marine Park Authority
Townsville Qld

Introduction

Eutrophication results from the supply of excessive plant nutrient substances to an aquatic ecosystem leading to enhanced plant growth or to a change in the composition of plant and other species. Coastal eutrophication is recognised as a worldwide and growing problem in areas affected by agricultural and urban run-off (GESAMP 1990; Nixon 1990; Smayda 1990; Rosenberg 1985). Some of the problems in Australian fresh and marine waters have been summarised in recent publications (e.g. Brodie et al. 1990; AEC 1987; Cullen 1986). The principal nutrients associated with eutrophication are nitrogen (N) and phosphorus (P) but others such as organic carbon, silicon, iron, molybdenum and manganese may play a supplementary role. On a global scale, it is now estimated that the input of nutrients to the oceans from human sources via rivers is equal to, or greater than, the natural input (Windom 1992).

The growth of aquatic plants is regulated by limiting biological, chemical and physical factors. Of the chemical factors, growth nutrients such as N or P may be present or available in the environment in limited amounts. The ratio of nutrient elements in the environment is also important, as plants need and use nutrients in a ratio linked to the composition of their own biomass. Thus it has been postulated that marine phytoplankton require carbon (C), N and P in the ratio 106:16:1 (Redfield 1934,1958) - the Redfield Ratio - which corresponds to the elemental ratio of these elements in phytoplankton. However it is now clear that requirements vary among species though there are some common features in their requirements (Hecky & Killam 1988; Howarth 1988). The supply of silicon is also particularly significant in coastal phytoplankton dynamics, because of its importance as a structural component of diatoms. The ratio of silicon to P is often grossly changed by anthropogenic inputs to coastal areas, and this can lead to phytoplankton community shifts from diatoms to flagellates (Smayda 1990).

Seagrasses and marine macroalgae have different C:N:P ratios (Atkinson & Smith 1983) and changes in the supply of N or P may change conditions to favour the growth of one plant at the expense of others. Because of differences in the supply of a limiting nutrient, increased loading of that nutrient may produce very different responses in different ecosystems. In the case of Port Phillip Bay in Victoria it has been shown that the seagrass, *Herterozostera tasmanica*, has responded positively to N inputs because its growth was limited by N concentrations in the sediment porewater. In contrast, in Western Port, where sediment porewater N concentrations are much higher, the seagrass showed little response to N enrichment (Bulthius et al. 1992).

The idea of nutrient limitation in aquatic environments has a long history (see for example Hecky & Killam 1988), particularly in freshwater lakes. Extensive research in recent decades, including artificial fertilisation of a set of lakes in Canada (Schindler 1975), has clarified the role of P limitation in lake ecosystems. This has led to an understanding of P loading (Vollenweidel 1976) and an ability to make recommendations and predictions regarding the rehabilitation of eutrophic lakes. It was also clearly shown that lakes could eutrophy naturally and that anthropogenic influences can accelerate this process to decades rather than thousands of years. This idea of 'cultural' eutrophication enhancing an existing phenomenon has also been recognised in estuarine environments. Estuaries of south-west Australia are in a process of slow natural eutrophication and anthropogenic impacts may be accelerating this process dramatically (Hodgkin 1988).

In the marine environment the question of nutrient limitation of primary production is controversial (see Howarth 1988). While N has been commonly regarded as the limiting element in many marine ecosystems (Ryther & Dunstan 1971), P may be limiting in some systems (Smith

1984) and various combinations of simultaneous or alternating N or P limitation have been reported. In diatom populations, where silicon may also be a limiting nutrient, enhanced diatom growth due to anthropogenic inputs of this element is not likely as silicon is not a major component of wastewaters.

The principal nutrients involved in eutrophication in coastal waters, N and P, may exist in a range of forms or species. Inputs of N and P into the marine environment normally occur as a mixture of a number of chemical forms which vary considerably in many important properties relevant to eutrophication including bio-availability, mobility and stability. The principal forms of N include the dissolved inorganic species, nitrate, nitrite and ammonium/ammonia; dissolved organic N including simple, identifiable compounds such as amino acids and urea as well as complex, high molecular weight, poorly characterised material ; and particulate N. Additionally, dissolved nitrogenous gases may also play a role in marine N cycles: N gas is available in unlimited quantities from the atmosphere, and can be used by N-fixing organisms.

Phosphorus may occur as dissolved inorganic P (primarily orthophosphate but also as polymeric forms), dissolved organic P, particulate organic P, and particulate inorganic P. The latter may include, especially in river run-off, P mineral substances which are not available for uptake by marine organisms.

N and P are regarded as the major environmental nutrients and those we know most about. Carbon is generally regarded as nonlimiting. Other nutrient elements, especially micronutrients, have been less studied and little is known of environmental responses to their concentration or their distribution and availability in the marine environment.

Sources

Anthropogenic or anthropogenically-enhanced sources of nutrients to the marine environment are soil erosion, fertiliser run-off, sewage discharge, rainfall, intensive animal production including aquaculture, and industrial discharges.

Phosphorus

The major anthropogenic inputs of P to the marine environment are agricultural run-off (diffuse inputs) and sewage discharges (point source inputs). The principal agricultural inputs derive primarily from the use of fertilisers and secondly from increased mobilisation of natural P from increased soil erosion. Sewage P sources are primarily human waste and P-containing detergents. In 1983 it was estimated that 5000 tonnes of P compounds were added directly to Australian coastal waters from marine sewage outfalls while coastal flowing rivers added 34 000 tonnes from diffuse sources and 1100 tonnes from sewage (Garman 1983). It has been estimated that in excess of 85% of all P entering Australian waters originates from diffuse sources (AEC 1987). However in low flow conditions in rivers, sewage may be the major source of nutrients. This has been clearly shown in studies on the sources of nutrients causing blue-green algal blooms in the Murray-Darling river system (GHD 1992), where sewage inputs predominate in dry conditions (when the blooms form) and diffuse inputs dominate during the wetter periods. Information on the relative magnitude of different sources of nutrients within catchments can be found in Rosich and Cullen (1982).

Phosphorus in soil is derived from weathering of phosphatic minerals, such as apatite, in the parent rock. Australian soils are, in general, extremely poor in P compared to soils found in North America and Europe (McLaughlin et al. 1992). It may thus be postulated that Australian coastal ecosystems have also developed in a low P environment. Phosphorus from soil inputs to the marine environment may not be completely available for uptake by marine plants and the phosphate buffer mechanism (Froelich 1988) exerts some control over available dissolved P concentrations.

An increase in use of P in fertilisers in Australia from 1951 is shown in Figure 1. In some

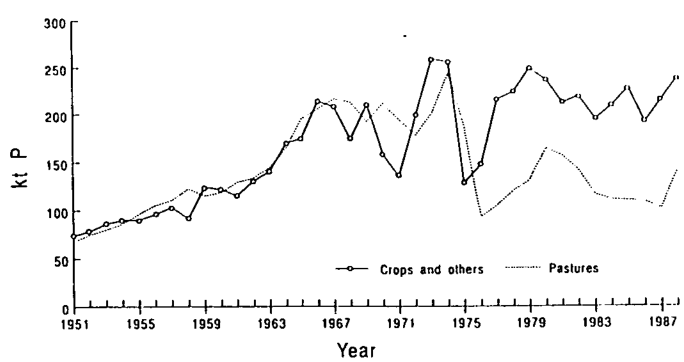


Figure 1: Consumption of P (expressed as elemental P) in Australia between 1951 and 1988. Adapted from McLaughlin et al. (1992)

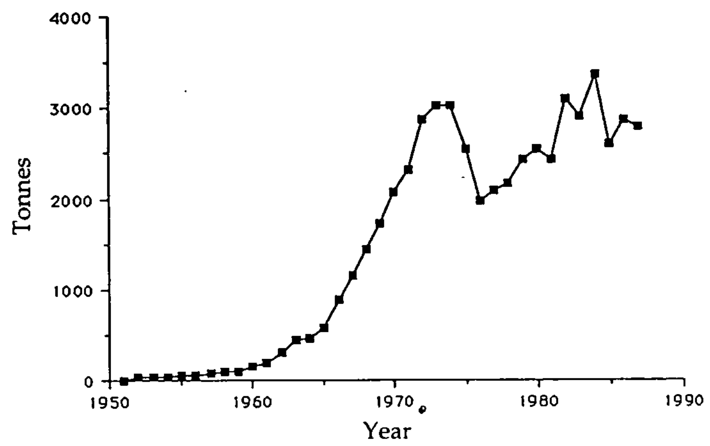


Figure 2: Annual fertiliser use in Atherton Shire (based on ABS data: total of all types). Adapted from Valentine (1988)

catchments, similar or more dramatic increases in the use of fertilisers in this period have been documented. For example Figure 2 shows the rise in total fertiliser use in Atherton Shire on the Barron River catchment (Qld) where, in offshore areas, potential problems with reef degradation due to enhanced nutrient loading have been recognised (Rasmussen & Cuff 1990). Figure 3 shows a similar rise in superphosphate usage in the Swan coastal plain catchments (WA), where major eutrophication problems have been experienced in the Peel-Harvey system. The sources of fertiliser P are almost entirely from imported P fertilisers or phosphate rock, the latter having been processed into fertilisers in Australia.

Near major urban areas sewage input of P may exceed that from other sources. Moss et al. (1992) have shown that along the Queensland east coast, agricultural inputs of P (and N) are far greater than those from sewage except in south-east Queensland around the heavily urbanised areas of Brisbane, the Gold Coast and the Sunshine Coast. For the rest of the coast (Cape York to Fraser Island) the ratio of agricultural to sewage inputs is 30:1 while for the south-east the ratio is 0.6:1 (Moss et al. 1992).

Nitrogen

The principal anthropogenic sources of N which may reach the coastal zone are also agricultural run-off and sewage discharges. Nitrogen fertiliser usage has been growing rapidly over the last three decades (Figure 4). In 1987 approximately 371 000 tonnes were used in agriculture. The largest crop use of N was for wheat and sugarcane. Table 1 (over) shows a breakdown of usage versus crop type.

Other significant agricultural sources of N include discharges from intensive animal industries such as piggeries and beef feedlots (see review by Bowmer and Laut 1992). Urban stormwater containing garden fertiliser, animal faeces, septic system leachate and sewerage system overflows may also be significant localised sources of N and other nutrients. Typical stormwater flows can contain concentrations of total N and P of the order of 2 mg/L-N and 1 mg/L-P (GHD 1981). A Victorian study (Weeks 1982) showed that the total flux of nutrients in stormwater from urban catchments may equal that in sewage discharges for the same population. Direct industrial discharges of wastewater may also contain considerable amounts of N (e.g., from a nickel refinery; Carey et al. 1982) but the discharge of such wastes has generally been

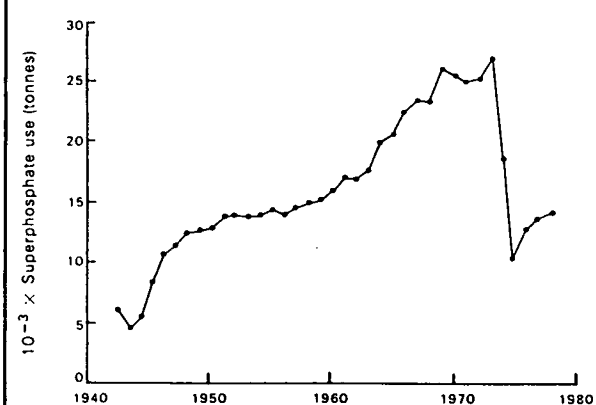


Figure 3: Superphosphate fertiliser usage for coastal plain catchments adjacent to the Peel -Harvey system. Adapted from Birch (1982)

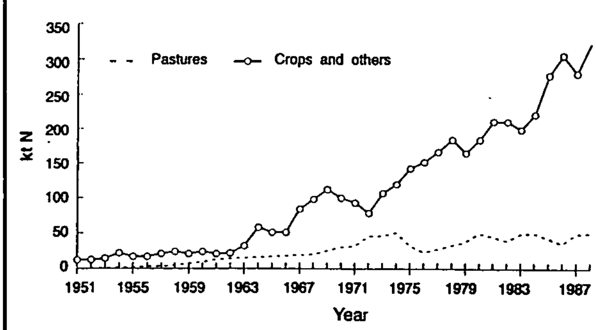


Figure 4: Consumption of N (expressed as elemental N) in Australia between 1951 and 1988 (Australian Fertiliser Manufacturers' Committee data). Adapted from McLaughlin et al. (1992)

Table 1: Estimates of nitrogen use (kt) in Australian agriculture in the 1987 crop year

Crop	Qld	NSW	Vic	Tas	SA	WA	Total	%
Sugar	55	3	0	0	0	0	58	15.6
Wheat	14	20	11	0	11	63	119	32.1
Barley, oats	6	5	5	0	8	14	38	10.2
Sorghum, maize, rice	19	16	0	0	0	3	38	10.2
Other crops	20	6	0	0	1	0	27	7.3
Fruit and vines	6	8	1	1	3	3	22	5.9
Vegetables	8	6	3	1	1	2	21	5.7
Pastures and fodders	17	6	9	1	3	8	44	11.9
Miscellaneous	1	2	1	0	0	0	4	1.1
Total	146	72	30	3	27	93	371	
%	39.3	19.4	8.1	0.8	7.3	25.1		

Source: Bellingham 1989

reduced through compliance regulations in recent years.

In catchments with a substantial industrial coverage, N can also be exported to coastal waters via N-enriched rainfall (Paerl 1985). This has been shown to be an important proportion of the N flux in some overseas coastal waters (Hinga et al. 1991; Paerl 1985) and may be significant in understanding nutrient limitation in some oceanic areas (Fanning 1989). There have been few published studies of such potential inputs in Australia except in the context of acid rain (AEC 1990) but around the major industrial cities of Newcastle, Sydney, Melbourne, Wollongong and Geelong some input of this type may be significant. Inputs of N, from wash-out of N oxides to Port Phillip Bay, have been estimated to be of the order of 500 tonnes per year, about 10% of the total load (Carnovale & Saunders 1987). Other estuarine systems near our major cities could be expected to receive atmospheric loads of similar magnitudes.

Soil erosion

Estimates of increased soil erosion and hence increased inputs of natural soil nutrients are preliminary at present. On an Australia wide basis it is considered that in the arid zone, 880 000 km^2 of the area used for grazing (3.36 million km^2) is experiencing enhanced soil erosion (Woods 1983). For eastern Queensland catchments the estimates of Moss et al. (1992), which are derived from catchment modelling studies refined by results from a few well monitored catchments, show that approximately four times as much sediment, N and P now enter the marine environment than before the introduction of western agriculture. Much of the increased nutrient load is postulated to be natural soil nutrients mobilised by erosion from grazing (see Figures 5 & 6). As there are considerable problems in the estimation of P losses from erosion using areal land-use figures (McLaughlin et al. 1992), the estimates of Moss et al. (1992) must be viewed as indicative.

Deforestation has been a recurring theme in Australia's development, the prime purpose being for agricultural development rather than forestry. Even in recent times clearing has continued, the most notable example being the extensive clearance of the brigalow (*Acacia harpophylla*) belt in Queensland (Bailey 1984). Several million hectares were cleared for cropping and grazing land in the Fitzroy River catchment from the mid 1960s till the early 1970s (Webb 1984). Deleterious effects on soil erosion and salinisation have been described by Webb (1984) and Johnson (1985). Effects extend downstream to the marine environment in the form of increased sediment and nutrient fluxes to the coastal zone (Byron et al. 1992; Moss et al. 1992). Table 2, adapted from Bucher and Saenger (1991), documents Australian catchments rated on the extent of clearance and shows the larger extent of clearance in the smaller south-eastern states with their greater population densities compared to Qld, WA and the NT.

Intensive logging practices, such as clear-felling, can produce major changes in river condition (including elevated concentrations of dissolved salts, suspended solids and nutrients) especially during peak flow periods (Campbell & Doeg 1989). The major downstream effect is increased sedimentation and this has been documented in the Eden area of south-eastern NSW (Olive & Rieger 1988). This study also highlighted the difficulty of detecting such changes in such a highly variable rainfall and stream flow regime.

Sewage

Most sewage effluent discharged into coastal waters has been treated to a secondary level and as such contains most of the nutrients originally present in the raw sewage. Secondary effluent typically contains 30-60 mg/L N and 6-12 mg/L P. Other plant growth stimulating substances are commonly found in sewage effluents. Many secondary treatment plants achieve substantial reduction in N content in their effluent by aeration control. Nitrogen concentrations in the effluent may be reduced to below 15 mg/L in this way ; for example, the Coombabah plant on the Gold Coast, as described in Moss (1992). Approximately 10 000 tonnes of P and 100 000 tonnes of N are produced in sewerage effluent in Australia annually. As the majority of the Australian population lives near the coast much of this effluent will enter coastal waters.

In some cases effluent is reused for industrial water e.g. in the lower Hunter region (NSW) and in Kwinana (WA) (Hanrahan & Gale 1989); for irrigation on golf courses (e.g. Townsville and Cairns); for horticulture near Melbourne (Hanrahan & Gale 1989) as well as for agriculture on the Werribee Sewage Farm, Melbourne. Reuse for irrigation on golf courses and on gardens is mandatory on resort islands in the Great Barrier Reef (GBR) Marine Park (Brodie 1992). Reuse schemes have also been investigated in the Tuggerah Lakes (Wyong), Coffs Harbour and Lake Macquarie areas of NSW, but considerable problems in their application have prevented implementation (Hanrahan & Gale 1989). An investigation into large scale reuse for irrigation of plantation forests near Adelaide is under way (Anon 1991). Tertiary treatment plants which reduce nutrients are now being introduced in many inland situations but as yet only a few operate in coastal regions.

Processes

There are significant differences in the fate and behaviour of N and P in the marine environment. A number of micro-organisms have the ability to either fix atmospheric N or denitrify N compounds (principally nitrate) to N gas. Thus N can be derived from, or lost to, the enormous pool of atmospheric N. Phosphorus does not undergo such transformations but can be locked up in soils or sediments by adsorption and binding to the soil particles and become, in a practical sense, removed from the ecosystem. Nitrogen is less readily lost in this way.

Particle-bound P is not easily leached by percolating water but can be mobilised by the movement of the soil itself - as when erosion occurs. Much of the P lost from agricultural lands is transported in particulate form. In the South Pine River in south-east Queensland, Cosser (1989) found during a 2 year study that over 77% of P was transported as particulates. He also noted that 80% of this P flux occurred during 2.8% (only 20 days) of the total time, during periods of intense rainfall and subsequent storm flow in the river. Thus control of soil erosion may also have major impacts on P transport to the coastal environment. It is estimated that the practice of green cane harvesting of sugarcane followed by the use of cane trash as a soil 'blanket' reduces annual soil erosion to 5 tonnes per hectare. This compares to traditional burning and tillage practices which yield average annual soil loses of about 150 tonnes per hectare. This method has a similar lowering effect on P losses (Prove & Hicks 1991). North of Townsville over 70% of cane is now harvested by the green cane method (Prove & Hicks 1991) and this is believed to have led to a significant decrease in P loading to the adjacent coastal seas (Kuhn 1990).

Sandy soils may behave somewhat differently than most other soils in that they can export a relatively high proportion of phosphate not bound to particulates. This was the situation found for the sandy, coastal plain catchments of the Peel-Harvey system (Hodgkin et al. 1985; Birch 1982).

The proportion of sediment-bound P which is available for uptake by marine organisms has been the subject of some debate but it is known that P can be desorbed from sediment particles in

Table 2: Distribution of catchment clearance[a]

State/Territory	% of Catchment Cleared of Natural Vegetation					Total
	<25%	25-50%	50-75%	>75%	Insufficient Information	
Queensland	170	17	4	3	113	307
New South Wales	20	31	25	5	-	81
Victoria	8	6	6	15	-	35
Tasmania	17	-	-	-	46	63
South Australia	-	1	1	12	1	15
Western Australia	125	2	3	1	14	145
Northern Territory	136	1	-	-	-	137
Total	476	58	39	36	174	783

a Number of catchments in each category
Source: Bucher and Saenger 1991

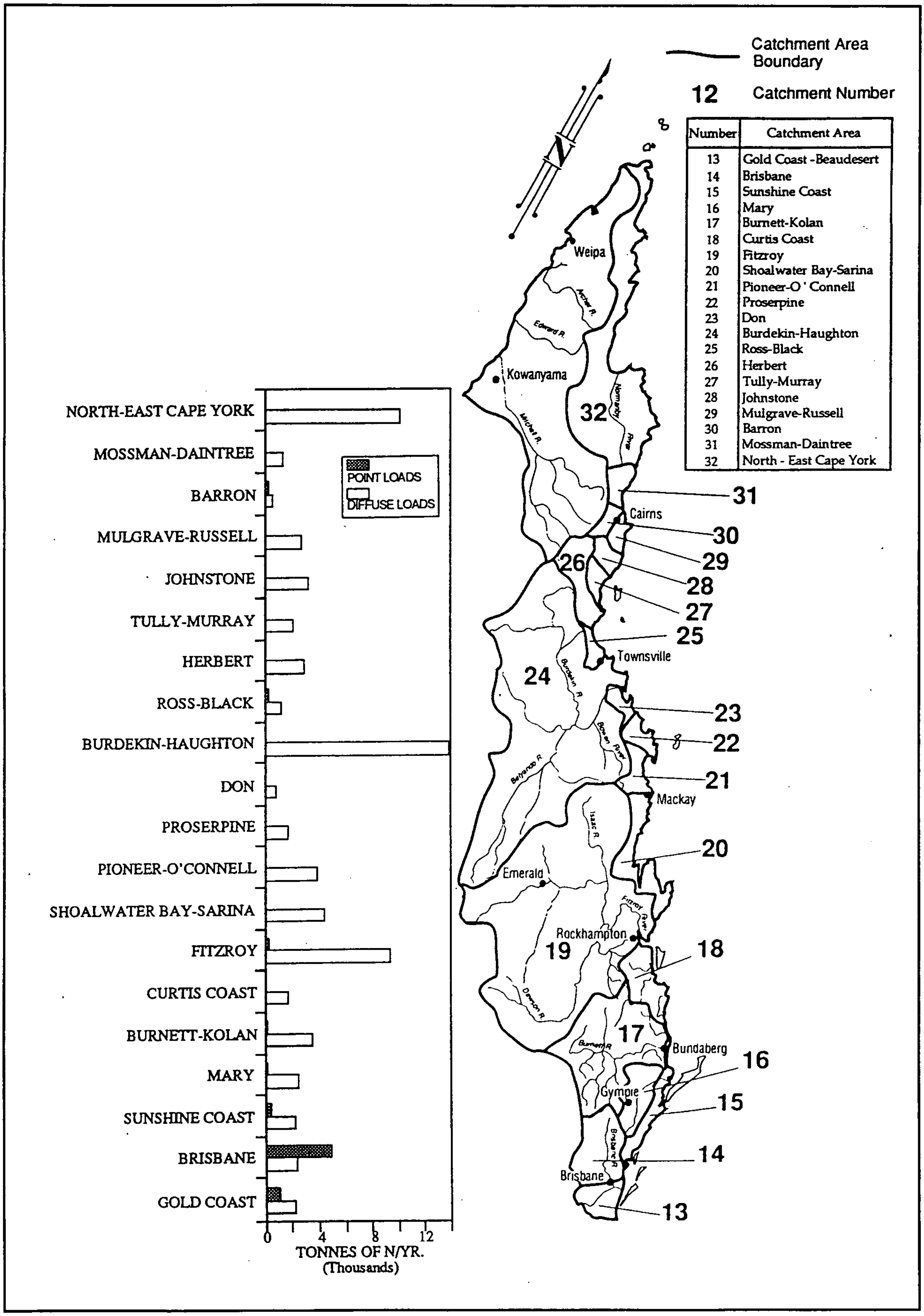
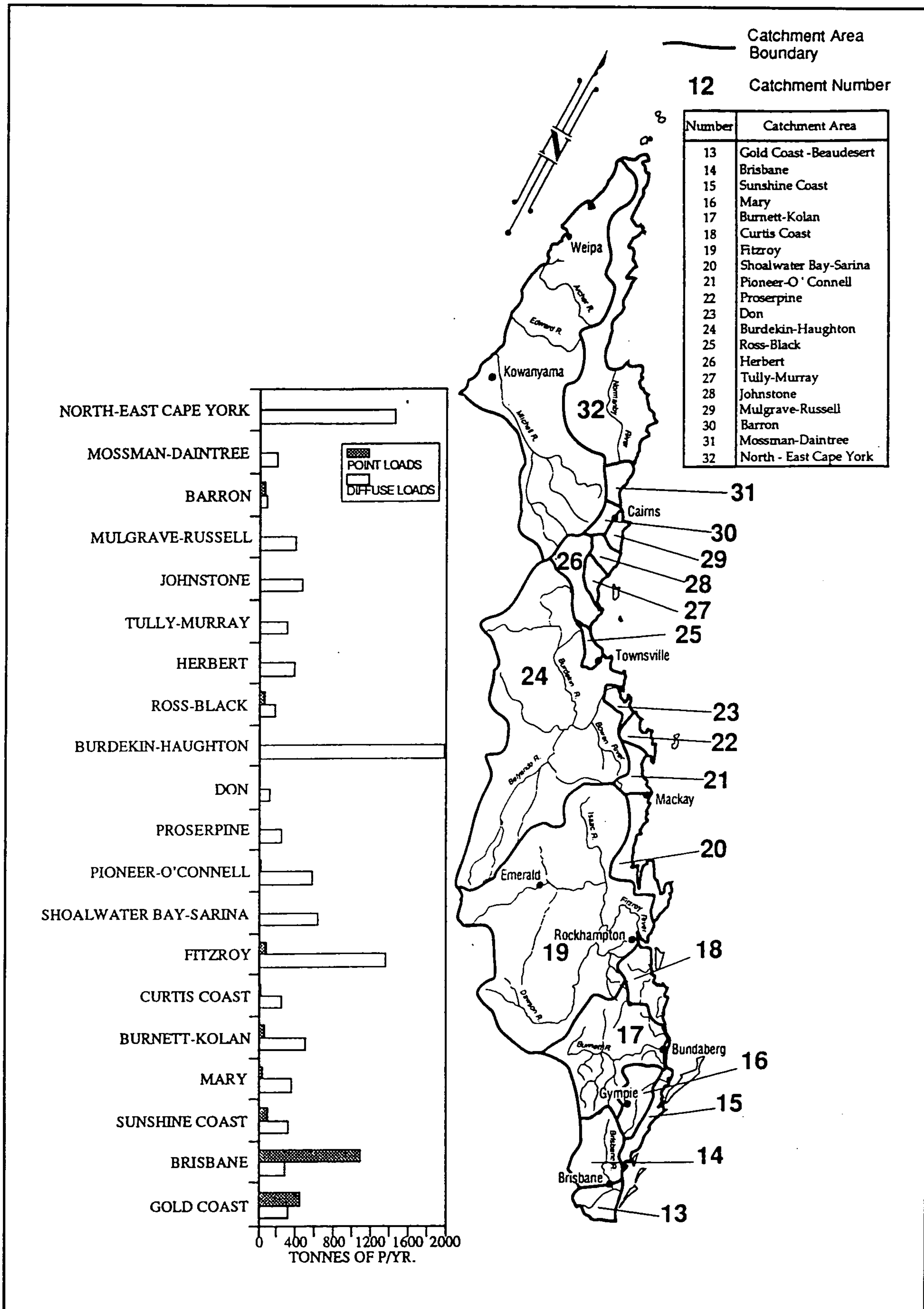


Figure 5: Comparison of magnitude of point and diffuse catchment sources of N. From Moss et al. (1992)



Number	Catchment Area
13	Gold Coast - Beaudesert
14	Brisbane
15	Sunshine Coast
16	Mary
17	Burnett-Kolan
18	Curtis Coast
19	Fitzroy
20	Shoalwater Bay-Sarina
21	Pioneer-O'Connell
22	Proserpine
23	Don
24	Burdekin-Haughton
25	Ross-Black
26	Herbert
27	Tully-Murray
28	Johnstone
29	Mulgrave-Russell
30	Barron
31	Mossman-Daintree
32	North - East Cape York

Figure 6: Comparison of magnitude of point and diffuse catchment sources of P. From Moss et al. (1992)

the mixing zone between freshwater and seawater, where salinity, pH and redox conditions change. A number of studies both in Australia (Carpenter & David Smith 1985) and overseas (Fox 1989) have demonstrated this process.

Nitrogen, on the other hand, is not as strongly bound to soil particles and is more readily lost to percolating water by leaching. Especially in the inorganic forms of nitrate and ammonium, N is very soluble in water and is easily moved by both overland and groundwater flows. Control of soil erosion is thus less effective in controlling N losses.

A significant factor in the transport of nutrients and sediment in river systems is the overwhelming contribution of the wet season (in climates with distinct wet and dry seasons), and storm flow. The problems in estimating fluxes under these conditions have been quantified in a number of overseas studies (Walling & Webb 1985) and is most difficult in semiarid climates with monsoonal rainfall regimes. Much of tropical Australia fits this latter category and it is recognised that Australian rivers have more variable annual river flows and annual floods than the rest of the world's continents (Finlayson & McMahon 1988). Major northern Australian rivers such as the Fitzroy (Brodie & Mitchell 1992), Burdekin, Tully and Johnstone (Arakel et al. 1989; Mitchell 1987) transport almost their entire flux of nutrients to the coastal zone during storm flow (normally cyclonic). Measurement of these fluxes is difficult as the episodes may only last a few days per year and access is limited.

Another natural source of N to coastal waters is N fixation by blue-green algae, particularly the pelagic *Trichodesmium*. *Trichodesmium* occurs widely in Australian tropical and subtropical waters (Creagh 1985) but its significance is unclear. Published N fixation rates vary 20-fold (Carpenter & Capone 1992) and the concentration of *Trichodesmium* in Australian waters is poorly known. It has been suggested that it is now more common in the GBR due to enhanced P inputs (Bell 1991) but there is limited evidence to support this.

Concentrations in the marine environment

The nutrient status of Australian marine waters is a function of hydrodynamics i.e. flushing times and the presence of adjacent sources of nutrient input e.g. currents, upwellings or sewage discharges. As a result of anthropogenic inputs most of the enclosed or semi-enclosed coastal water bodies in the southern half of the continent near major river estuaries, urban and industrial areas have elevated nutrient status and are eutrophic or showing signs of incipient eutrophication.

In general, oceanic waters surrounding Australian have low nutrient concentrations (Jeffrey et al. 1990). While upwelling systems occur in a number of areas they are limited and intermittent. The system on the western coast of Australia (Jeffrey et al. 1990) is far weaker than those characterising the continental western coasts of the Americas and Africa (Pickard & Emery 1990), being blocked by the southward-flowing Leeuwin current. In Bass Strait, nutrient concentrations are low throughout the year but show evidence of at least weak upwelling on the eastern boundary of nutrient rich sub-Antarctic water (Gibbs et al. 1986). Along the eastern coast of Tas. and Vic., and the coast of NSW, some evidence of intrusion of subsurface waters has been found. This is considered to be an important source of nutrients for these coastal areas (Rochford 1984). Episodic upwelling also occurs from the Coral Sea into the GBR lagoon. This probably forms an important nutrient supply for the GBR (Furnas & Mitchell 1986; Andrews & Gentian 1982).

For many water bodies the total flux of nutrients to the system may be a better indicator and predictor of eutrophication than nutrient concentrations. While the technique has been extensively used and refined for freshwater lakes, its use in the marine environment is often much more difficult to apply due to the difficulty in quantifying natural fluxes.

Australia is a continent of low surface run-off to its coastal seas and has no large river systems in terms of discharge. The larger rivers in terms of catchment size and length (the Coopers Creek/Thomson and the Warburton Creek/Diamantina systems) either do not discharge to the sea at all, or have a low discharge (the Murray-Darling). By comparison the Fly River, which partially discharges into Australian coastal waters from Papua New Guinea, yields a volume of water and sediment similar to that of all Australian rivers combined (see Figure 7; from Harris 1991). Thus Australian coastal regions are not naturally exposed to continuous high levels of terrestrial run-off. Australia also has few large coastal embayments or seas (e.g. Gulf of Carpentaria) comparable to Chesapeake Bay, Puget Sound, the Red Sea, the Baltic Sea or the

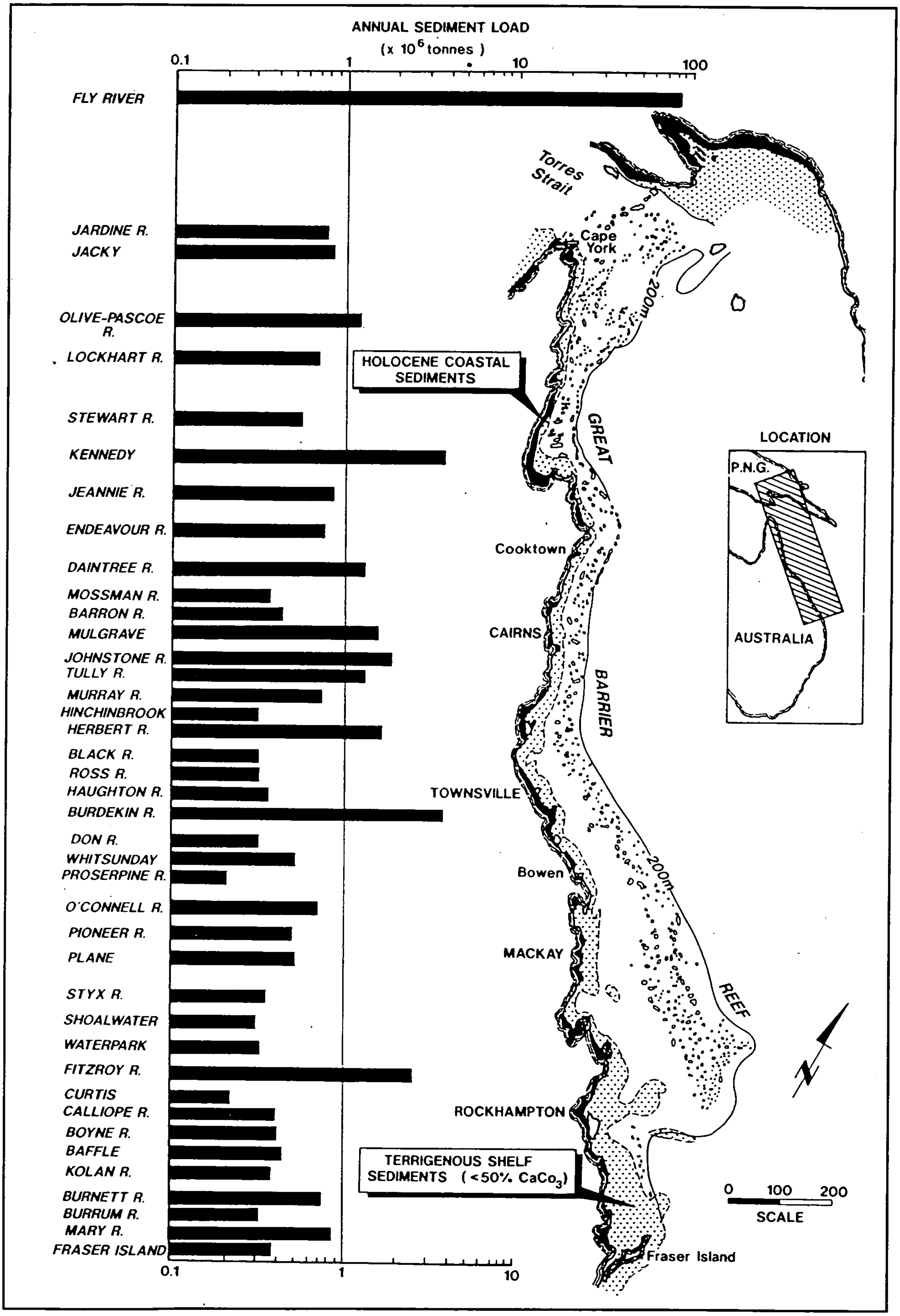


Figure 7: Sediment discharge of rivers entering the Great Barrier Reef Lagoon and location of terrigenous shelf and Holocene coastal sediments. From Harris (1991)

Table 3: Nutrient concentrations at which observable increases in plant growth have occurred in various water bodies[a]

Waterbody/Area	Total Nitrogen (μg/L)	Total Phosphorus (μg/L)
Hawkesbury Nepean	650	55
Peel/Harvey	150	25
Lake Burley Griffin	90[b]	60
Lake Macquarie	600	60
Murray River	550	40
Kosciusko National Park	360	40

a It must be stressed that these are approximate thresholds only. These waterbodies show a gradation of effects both with time and location and hence it is not possible to be definitive as to exactly where and when eutrophication is apparent.
b Only oxidised forms of nitrogen were measured.
Source: AEC 1987

Gulf of Thailand, where the discharge of a large river system might be retained.

Much of our knowledge of the general nutrient status of Australian coastal waters has come from the work of the CSIRO and recently, in northern waters, from the Australian Institute of Marine Science (AIMS).

Substantial data sets describe concentrations of N, P and silicon species in coastal and shelf waters; for example Bass Strait (Gibbs et al. 1986), the east coast (Rochford 1984), the GBR lagoon (Furnas 1991;Andrews 1983; Revelante & Gilmartin 1982) and the North West Shelf (Mackey 1984; Rochford 1977).

The concentrations of N and/or P which indicate, or indeed cause, eutrophication have been long debated. Obviously these critical concentrations depend on the ecosystem in question, with some able to better cope with nutrient stress than others. Table 3 from the Australian Environment Council's review of nutrients in Australian waters (AEC 1987) lists the nutrient concentrations at which 'observable increases in plant growth have occurred in various water bodies' (both fresh and marine). Of note is the similarity in the critical concentrations between the diverse water bodies, from saline coastal lakes and estuaries to large rivers and inland artificial impoundments. The report concludes that 'visual evidence of eutrophication is likely to occur if total N concentration is within or exceeds the range of 400-600 μg/L and/or total P concentration is within or exceeds the range of 40-60 μg/L'.

Environmental impacts

Generally eutrophication in coastal ecosystems is derived from increased concentrations of N and P enhancing plant growth. In any locality the species which best respond to enhanced nutrients depends on the interaction between physiological limitation factors and competition between plant types. Thus while in isolation, coral zooxanthellae, seagrasses, phytoplankton and benthic algae may all respond positively to an increased nutrient supply, in mixed communities the response of one may dominate. While low levels of nutrient enhancement may promote seagrass growth (Lukatelich et al. 1987), at higher levels epiphytic algal overgrowth occurs, reducing light and leading to seagrass demise. This has been reported in Cockburn Sound, WA (Cambridge & McComb 1984) and Gulf St Vincent , SA (Neverauskas 1987). In some situations secondary effects may dominate the benthic response, such as filter feeders (barnacles, tube worms, sponges) expanding their coverage in response to increased phytoplankton concentrations. However, light limitation due to high turbidity, may inhibit plant growth even though nutrient levels are high, as noted in the Brisbane River (Moss 1987) and as suggested to occur in the Georges River, NSW (Heath et al. 1980) and Lake Bonney, SA.

Eutrophication will often progress through a sequence of stages, characterised in the global State of the Marine Environment Report as an idealised progression involving: '(a) enhanced primary productivity, (b) changes in plant species composition, (c) very dense blooms, often toxic, (d) anoxic conditions, (e) adverse effects on fish and invertebrates, (f) impact on amenity, (g) changes in structure of benthic communities' (GESAMP 1990). Not all these stages will always be present or evident.

Overseas, coastal eutrophication has been most evident in enclosed and semi-enclosed seas and estuaries. The most prominent examples include the northern Adriatic Sea where annual noxious algal blooms now occur (Justic 1987); the Baltic Sea with anoxia problems (Larsson et al. 1985); the North Sea with increasingly regular algal blooms including the toxic species *Chrysochromulina polylepis* (Underdal et al. 1989) and the nuisance alga *Phaeocystis pouchetti* (Lancelot et al. 1987); the Black Sea with increasing anoxia problems, loss of fisheries and blooms of introduced species (Mee 1992); the Inland Sea of Japan (Seto Sea) (Yasui & Kobayaski 1991); and Chesapeake Bay with loss of benthic fauna and fisheries (Officer et al. 1984).

In addition to large nutrient fluxes, the principal factors implicated in eutrophication in many of these overseas localities is long water residence times (i.e. poor flushing) promoted by the enclosed nature of the water body. This has the effect of allowing build-up of nutrients during periods of high input. Many Australian coastal water bodies of similar long residence times and high nutrient inputs are showing analogous eutrophication.

Coastal lagoons represent 11.4% of the Australian coastline (Cromwell 1971, cited in King & Hodgson 1986). In all areas affected by urban and agricultural run-off, these lagoons and lakes are showing stages of incipient eutrophication as a result of human activity. This is generally true for that part of the southern mainland Australian coastline stretching from Perth to Brisbane. Prominent examples include the Peel-Harvey system, Gippsland Lakes and many of the NSW coastal lakes. In addition, many of the embayments and estuaries in this region are affected, including Cockburn Sound, Gulf St Vincent, Western Port, Botany Bay and Moreton Bay. For the few very large systems in Australia which could be threatened by eutrophication (the GBR lagoon and the Gulf of Carpentaria) convincing evidence of effects is not available, although claims have been made that a problem exists in the GBR (Bell 1992).

Eutrophication in enclosed coastal waters is comparable to that commonly seen in recent years in Australia's inland water bodies, particularly the Murray-Darling system (GHD 1992). This has led to blue-green algal blooms (frequently toxic) with consequential effects on human and stock drinking water supplies and in-river ecological effects. The causes of the eutrophication (sewage and agricultural run-off), are the same as for coastal regions with the added problem in rivers of a lack of water flow at the end of the dry season, which leads to enhanced nutrient concentrations.

Loss of seagrass beds

One of the most common features in the record of effects of Australian coastal eutrophication has been a loss of seagrass beds. Shepherd et al. (1989) reviewed twelve well described cases of major seagrass loss following enhanced nutrient or sediment supply. Walker and McComb (Table 1; 1993), ascribed eutrophication as a cause of the losses in most of the cases.

In Cockburn Sound (WA), there has been increasing urban and industrial development of the adjacent coast from 1954, and increased inputs of industrial discharges and sewage during the 1960s (Cambridge et al. 1986), led to a 97% loss (3300 hectares) of seagrass beds by 1978 (Figure 8). The loss has been attributed to overgrowth by epiphytic algae (Cambridge et al. 1986; Silberstein et al. 1986) leading to substantial reductions in ambient light reaching leaf surfaces. The decline in the seagrass coverage was closely correlated with increasing N loadings to the Sound. Recent reductions in nutrient loadings have occurred, including a new sewage outfall outside the Sound (1984) and better quality industrial discharges (1982). These management actions have led to a halt in the decline of the seagrass beds but only minimal recovery and regrowth (Hillman 1986).

In Gulf St Vincent (SA), the principal Adelaide sewage outfalls at Glenelg and Bolivar, and the sewage sludge outfall at Semaphore have caused

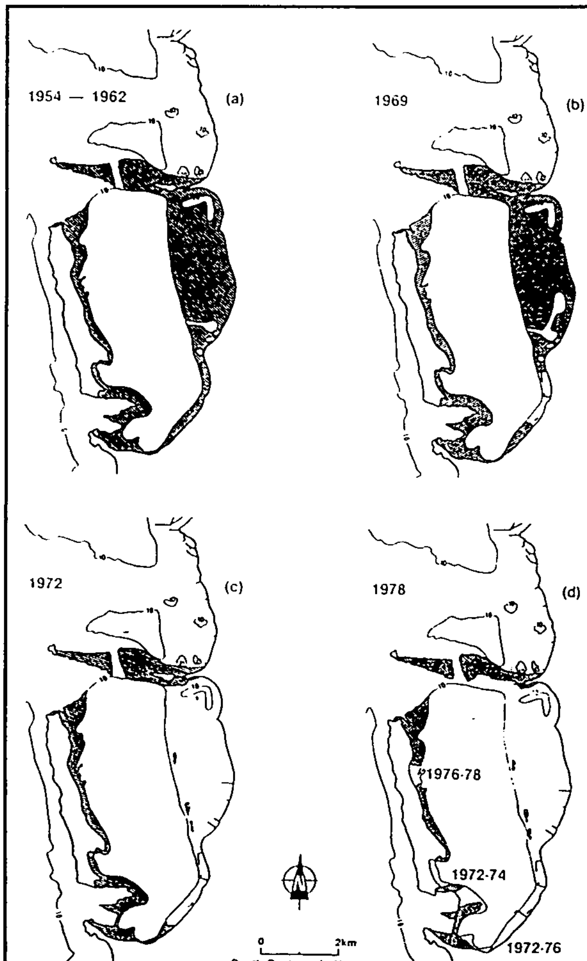


Figure 8: Seagrass loss in Cockburn Sound. Each map shows Cockburn Sound surrounded by the coast of the mainland to the right, and Garden Island to the left. The 10 m contour line is indicated. The shading shows the area of seagrass meadow present at different times. From Cambridge and McComb (1984)

substantial losses of seagrass in their vicinity (Figure 9; Shepherd et al. 1989; Neverauskas 1987). The total loss of seagrass area is estimated to be over 5000 hectares. As in Cockburn Sound, the principal cause of the loss has been epiphytic overgrowth leading to light loss. This has been accompanied by shifts in species composition (Neverauskas 1987).

Western Port (Vic.) has suffered almost complete destruction of the extensive seagrass beds present before 1973 (Coleman 1982; Bulthius 1981). By 1984 the area of macrobenthic plants in the Bay had declined from the 25 000 hectares of 1973 to 7200 hectares (Bulthius, unpublished, in Shepherd et al. 1989). Remaining areas of seagrass have also declined in biomass, with an 85% decrease in the standing crop between 1975 and 1984. The causal factors are not clearly identified but general development in the catchment leading to increased sedimentation, turbidity and nutrient levels in the Bay has been implicated. The principal cause appears to have been increased sedimentation combined with the stress of higher temperatures on the most severely affected intertidal beds (Bulthius 1983a, 1983b). The loss of seagrass appears to be closely correlated with an 80% reduction in the catch of King George Whiting.

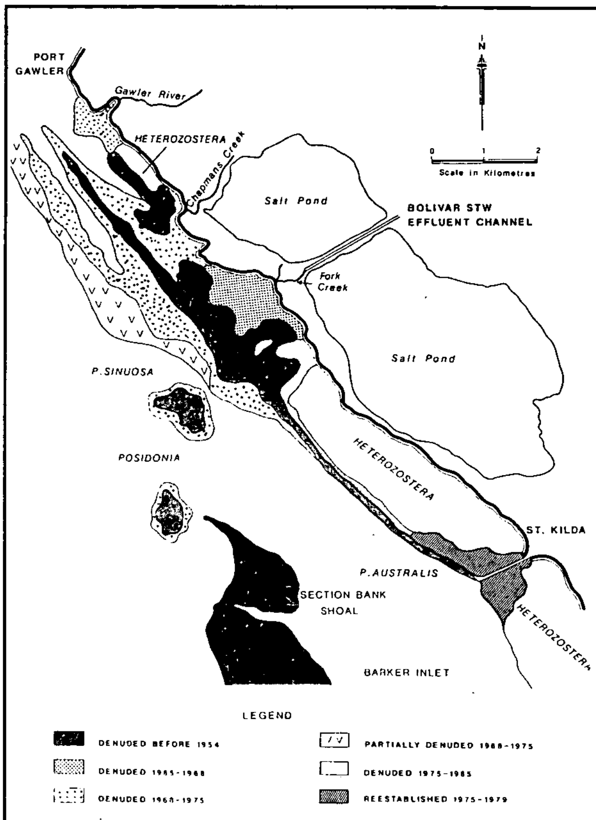


Figure 9: Time course of seagrass loss from Barker Inlet to Gawler River from 1954 to 1985. From Shepherd et al. (1989)

In NSW a large number of bays, lagoons and river estuaries have suffered loss of seagrass areas in recent years. Anthropogenic factors have been implicated in many cases but as King and Hodgson (1986) note, the natural high variability in seagrass area and growth makes monitoring and detecting anthropogenic changes difficult. Changes in techniques and methodologies over the years have made comparisons difficult.

In the coastal embayments of NSW, substantial changes in the area of seagrass beds have been documented in Lake Illawarra (King 1988a; Yassini 1985); Botany Bay (Larkum & West 1990); Tuggerah Lakes (King & Hodgson 1986; King & Holland 1986); Lake Macquarie (Simmons & Trengrove 1989; King & Hodgson 1986) and the estuaries of the Georges, Clarence and Tweed Rivers (Shepherd et al. 1989). Some doubt as to the extent of losses due to anthropogenic causes versus natural change exists for Lake Illawarra (King 1988a) and the causes of loss in Botany Bay are also not clear (McGuinness 1988). For Lake Macquarie and the Tuggerah Lakes, catchment urbanisation (particularly stormwater run-off and point source discharges), combined with agricultural run-off have led to increased nutrient loadings. As a result, increases in phytoplankton, turbidity and macroalgae have occurred, along with decreased clarity and losses of seagrass areas. The estimated losses in seagrass areas are 700 hectares for Lake Macquarie and 1300 hectares for the Tuggerah Lakes (Shepherd et al. 1989). Studies of the Clarence River estuary have shown that the present seagrass beds (*Zostera* spp.) only occur over 20% of the area described in the 1940s (NSW Government 1992).

Other systems showing loss of seagrass possibly correlated with terrestrial run-off are Princess Royal Harbour and Oyster Harbour near Albany, WA (Bastyan 1986); Port Lincoln, SA (Shepherd 1975); Peel-Harvey Inlet, WA (discussed elsewhere in this review) and possibly Moreton Bay, Qld (Kirkman 1978). In northern Australia cyclonic freshwater inundation may lead to massive seagrass loss but subsequent recovery e.g. in Cleveland Bay in 1971 associated with Cyclone Althea (Pringle 1989) and Hervey Bay associated with Cyclone Fran in 1992 (Preen 1993). The losses are probably associated with prolonged freshwater stress and loss of light due to turbidity. In the recent event in Hervey Bay, over 1000 km^2 of seagrass have been lost resulting in significant mortality and migration of the dugong population and reduction in commercial prawn and fish catches. The loss/recovery cycle in tropical Australia may be contrasted to southern Australia, where seagrass loss from eutrophication may be more gradual but recovery has not been recorded. After physical damage to *Posidonia* beds recovery is also slow as shown in the lack of regrowth in areas of seagrass in Jervis Bay, denuded in the 1960s by military bombing practice (Beckmann 1991).

Low levels of nutrient enhancement may favour the growth of seagrass beds. For example, at Wilson Inlet (WA), where enhanced seagrass growth has been documented, catchment nutrient loads were elevated above natural loads, but were much lower than in the Peel-Harvey system, and macroalgae were less prominent (Lukatelich et al. 1987). At Green Island (north Qld) the sewage discharge from a primary treatment plant was trapped by prevailing wind and tidal conditions in a hydrodynamic retention area to the north of the island. The extensive expansion of the seagrass beds which occurred (Van Woesik 1989) has been generally attributed to the effects of the outfall, although it may also have been promoted by changes in sediment composition due to nearby channel dredging.

Nutrients may also enhance mangrove growth. In areas where increasing siltation has provided 'new' habitat for mangroves at the expense of seagrass beds, extra nutrients can lead to vigorous mangrove growth. This has occurred in a number of estuaries in the Sydney area (Dunstan 1990) including Botany Bay (Georges River), Brisbane Waters, and the Lane Cove (McLaughlin 1987) and Cooks Rivers. At high levels of nutrient enrichment the mangroves may be affected by excessive algal growth. This has been the case in Barker Inlet and Chapman Creek near Adelaide, where discharges from the Bolivar sewage treatment plant has caused increased growth of the green alga *Ulva*. Drifts of the alga directly shade or smother newly established mangrove seedlings. Seedling mortality up to ten times levels in unimpacted areas has been observed (Edyvane 1991).

Algal blooms and red tides

The most visible indicator of coastal eutrophication is extensive blooms of phytoplankton and/or benthic macroalgae. In many overseas localities eutrophication in large water bodies has been characterised by blooms of planktonic species such as *Phaeocystis* (the North Sea; Davidson & Marchant 1992; Lancelot et al. 1987) and *Noctiluca miliaris* (the Black Sea; Mee 1992), and mucus aggregates (the northern Adriatic Sea; Stachowitsch et al. 1990). Some of the algae are toxic and may cause fish kills while others are aesthetically 'nuisance' algae, causing spoiling of beaches, offensive odours and slimy water. Many cause secondary blooms of undesirable fauna, such as the ctenophore *Mnemiopsis leidyi* (the Black Sea; Mee 1992) which has reached bloom biomass densities of 1 kg m^{-2}.

In Australia a large number of estuaries, bays and coastal lakes have begun to experience algal blooms in the last thirty years . The best known examples of phytoplankton and macroalgal blooms come from the Peel-Harvey system in WA, while the effects of toxic algae are best known from Tasmania and Victoria, where the closure of shellfish beds has resulted from blooms of an algae suspected to have been introduced into Australia in ballast water (Hallegraeff & Bolch 1992).

The Peel-Harvey system provides the best example of an Australian marine eutrophic system and the range of impacts caused. Located south of Perth in WA, it consists of two connected coastal lagoons which form the estuary of three small rivers: the Murray, the Harvey and the Serpentine (Figure 10). The catchment has been

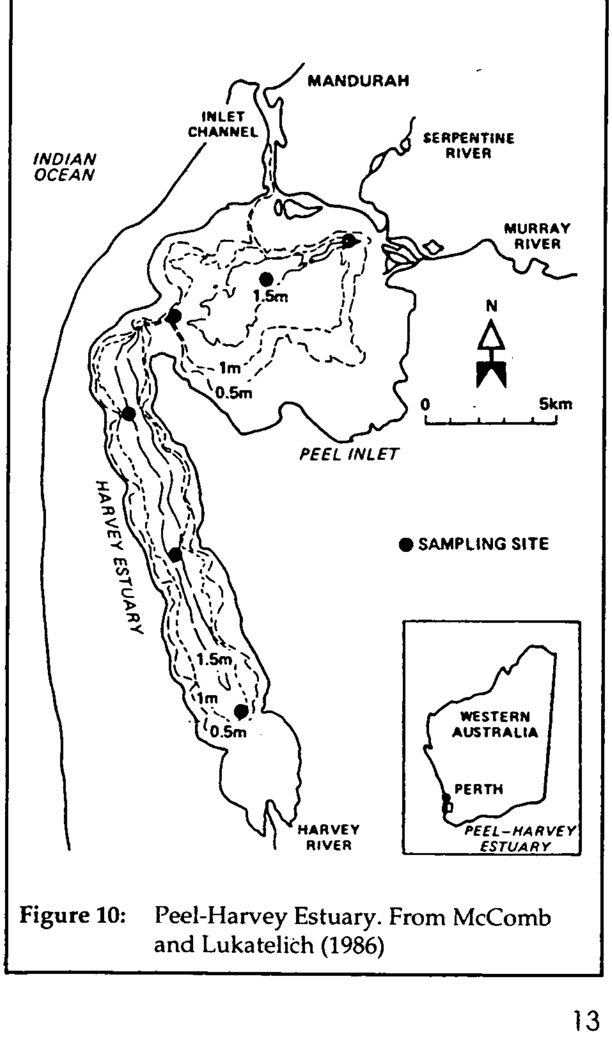


Figure 10: Peel-Harvey Estuary. From McComb and Lukatelich (1986)

extensively modified for agricultural uses, mainly beef, sheep and dairy production (Yeates et al. 1984), and extensive use of phosphatic fertilisers has occurred over the last forty years. Phosphorus input, in particular, has risen by factors of nine times for the Serpentine River and fifty times for the Murray River (Hodgkin et al. 1981) in the period between 1949-56 and 1972-78.

The two lagoons show distinctly different eutrophic responses. In Peel Inlet the main response has been extensive growth of the macroalga *Cladophora* in the early years of the problem, changing more recently to *Chaetomorpha* and *Ulva* as dominant species (Lavery et al. 1991). The Harvey Estuary, in contrast, has little growth of macroalgae but massive blooms of the blue-green alga *Nodularia spumigena* (McComb et al. 1981). The macroalgae in the Peel Inlet break free from the lagoon floor, accumulate on the beaches and rot, producing nauseating odours. They have been mechanically harvested offshore and regularly removed from beaches, but this has only been partially successful in keeping beaches near populated areas free from the rotting algae (Hillman et al. 1990). In the Harvey estuary the growth of macroalgae is inhibited by light attenuation due to high water turbidity. This seems to be due to the geographic orientation of the estuary (Figure 10), which lies with its long axis parallel to the prevailing winds and hence has high wind-resuspension rates (Gabrielson & Lukatelich 1985). As a result, while not having macroalgal problems, the Harvey estuary experiences massive blooms of *Nodularia spumigena* in late spring and early summer. With the exception of 1979 and 1987, the blooms have occurred each year since 1978 (Hillman et al. 1990) and are closely correlated with the freshwater inflow from the Harvey River, with its associated P load (McComb & Lukatelich 1986). Figure 11 shows the relationship between *Nodularia* chlorophyll *a* and P load for the period 1977 to 1983 (modified from McComb & Lukatelich, 1986). *Nodularia* also has a nauseating smell, especially when it accumulates and rots on the shores, and has been blamed for sickness in local residents (Hodgkin et al. 1985).

Dense blooms of *Nodularia spumigena* have affected fish and crab populations in the Harvey estuary. This has been shown by reduced commercial fish catches and reduced fish numbers in areas with high chlorophyll *a* levels, indicative of high *Nodularia* density (Lenanton et al. 1985; Potter et al. 1983). However increased macroalgal biomass in the Peel has led to an increase in commercial fish catch (Lenanton et al. 1984).

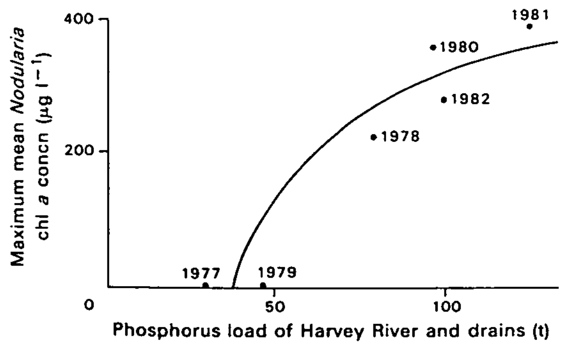


Figure 11: Relationship between maximum average concentration of chlorophyll *a* in Harvey Estuary in summer, and the amount of phosphorus entering the estuary in winter from the Harvey River and associated drains. From McComb and Lukatelich (1986)

The south-east coast of Australia, from Port Phillip Bay to Fraser Island, has many coastal lagoons, lakes and bays similar in a geomorphological sense to the Peel-Harvey system. They have narrow connections to the sea, are fed by rivers flowing from small to moderate sized catchments and are shallow with low tidal influences and long water residency. They vary in salinity regime, size, riverine inflow, rainfall and flushing period and any wetlands associated with them reflect these physical differences (Bucher & Saenger 1991). Those water bodies that have documented problems with algal blooms are listed in Table 4. From this it is clear that the problem occurs throughout Australia, especially near centres of high population and agricultural development.

The largest of the relatively enclosed systems is Port Phillip Bay. At various times over the last two decades concern has been expressed that the Bay was becoming eutrophic. Monitoring and research studies have attempted to resolve the issue. Those of the 1970s established only minor changes to water quality and benthic communities (Axelrad et al. 1981; Brown et al. 1980; EPA 1979; Poore & Rainer 1979). However continued concern about the effects on the Bay of urban development in the catchments and the long-term options for disposal of treated sewage effluents has led to further studies and monitoring by a variety of organisations, particularly the Victorian EPA (e.g. Lukatelich 1990) and Melbourne Water (e.g. Bremner et al. 1989), culminating in a recently commissioned large-scale study coordinated by the CSIRO (1992).

In NSW nuisance algal blooms have been recorded over a number of years in many coastal water bodies. Cheng (1981) lists Tuggerah Lakes, Narrabeen Lagoon, Dee Why Lagoon, Harbord Lagoon, Avoca Lake and Lake Illawarra as having major blooms while also noting that lagoons with less populated and developed catchments such as the Myall Lakes and Smith Lake remain relatively unspoiled.

In the Tuggerah Lakes the nuisance algae are mostly green macroalga of the genera *Enteromorpha, Chaetomorpha* and *Rhizoclonium* (King 1988b). The lake area is 80 km^2, the catchment area 670 km^2 while only a 1% exchange of water with the ocean occurs on each tidal cycle under normal entrance conditions (Anon 1990). The catchment is significantly urbanised and falls entirely within the Wyong Shire Council area. There has been a 7-8% annual population growth in the Shire between 1973 and 1990 (Anon 1990). Considerable areas within the catchment are used for citrus and other crops while a large coal fired power station (Munmorah - 1400 MW) uses lake water for cooling purposes.

Table 4: Australian coastal areas showing eutrophication

Locality	Effects
Swan River Estuary	Phytoplankton blooms
Peel-Harvey Estuary	Phytoplankton, macroalgal blooms
Cockburn Sound	Seagrass loss
Wilson Inlet	Minor enhanced seagrass growth
Albany Harbour	Seagrass loss
Port Lincoln	Seagrass loss
Gulf St Vincent	Major seagrass loss, toxic algae
Port Phillip Bay	Macrophyte growth, toxic algae
Western Port	Major seagrass loss
Gippsland Lakes	Phytoplankton blooms
Derwent Estuary	Phytoplankton blooms, toxic algae
Huon Estuary	Phytoplankton blooms
Lake Illawarra	Seagrass loss, macrophyte growth
Botany Bay	Seagrass loss
Avoca Lagoon	Phytoplankton blooms
Harbord Lagoon	Phytoplankton blooms
Tuggerah Lakes	Seagrass loss, macrophyte growth
Lake Macquarie	Phytoplankton blooms
Clarence Estuary	Seagrass loss
Tweed Estuary	Minor seagrass loss
Moreton Bay	Phytoplankton blooms
Great Barrier Reef Lagoon	Macrophyte growth, coral in decline

Potential problems in the lake system were recognised as early as 1969 (Higginson 1970). Controversy exists over whether hot water discharges from the power stations or nutrient enhancement are responsible for algal blooms, with the evidence favouring nutrient enhancement (King 1988b). As in the Peel-Harvey system the excessive growth, accumulation, and subsequent decay of algae, producing offensive odours, is the major focus of concern.

Considerable changes in the floristic composition of Lake Illawarra appears to have taken place since the early 1960s, in which time a number of studies of the lake occurred (King 1988a; King & Barclay 1986; West 1985; Evans & Gibbs 1981; Harris et al. 1980; Harris 1976; Higginson 1968). King (1988a) noted that this temporal variation is common in lakes and drew attention to the problems of detecting anthropogenically-induced changes against an unknown natural background variation. However Lake Illawarra is regarded as having undergone significant eutrophication, manifested mainly in the form of excessive filamentous algal growth (LIMC 1986).

Other Australian coastal systems to have experienced blooms of either benthic macroalgae or phytoplankton include Moreton Bay where red tides were followed by fish kills (Moss 1987), Sydney Harbour (Revelante & Gilmartin 1978), Gippsland Lakes (Poore 1982), the Hawkesbury River and estuary, the Derwent River estuary (Coleman 1983), the Huon River estuary, the Swan River estuary (John 1987; Hodgkin & Vicker 1987 in John 1987), the Tweed Estuary (Anon 1986) and Orielton Lagoon (Buttermore 1977). Other systems such as Lake Macquarie have experienced a slow decline in water clarity with increasing nutrient concentrations (SPCC 1983), the latter attributed to rising nutrient input from an increasingly urbanised catchment (Simmons & Trengrove 1989). Even the oceanographically dynamic seas off Sydney have very recently (January 1993) experienced widespread red algal blooms extending from Wollongong to the Hawkesbury estuary. Whether the recent relocation of the principal Sydney ocean sewage outfalls offshore into deeper waters will contribute to changes in the phytoplankton community remains to be seen.

Toxic red tides

An increasing problem in coastal waters worldwide is the occurrence of toxic phytoplankton blooms (Smayda 1990). These blooms, intensified by eutrophic coastal conditions, cause finfish kills and shellfish to become toxic. In Tasmanian waters, blooms of the dinoflagellate *Gymnodinium catenatum* have occurred since 1986 (Hallegraeff et al. 1989), and caused temporary closures of commercial shellfish beds. From distributional evidence and the absence of the organism in sediment records, it has been suggested that *G. catenatum* was introduced into Tas. via ships' ballast water (Hallegraeff & Bolch 1992). In SA the toxic dinoflagellate *Alexandrium minutum* now blooms annually in the Port River, near Adelaide (Cannon 1990; Hallegraeff et al. 1988), causing toxicity in shellfish (Oshima et al. 1989). This organism is also thought to be introduced. Toxicity in wild mussels has resulted from blooms of *Alexandrium catenella* in Port Phillip Bay, but effects on shellfish farms have been minor (Hallegraeff et al. 1991). The incidence of harmful algal blooms in the Australian region has recently been reviewed by Hallegraeff (1993a).

The increasing incidence of 'toxic red tides' around the Australian coast has led to claims that these will now be a regular event and that their principal cause is the discharge of sewage effluents into the ocean (Illert 1993). While algal research workers discount the more tenuous links of the 'ecological holocaust' claim, they acknowledge the continuing discharge of poorly treated sewage effluents into the oceans is undesirable (Hallegraeff 1993b).

Long-term effects on benthos near sewage outfalls

Ocean sewage outfalls are best located in deep water, on open, hydrodynamically active coasts to maximise dilution and dispersion. In Australia many large outfalls have been relocated to such positions in recent years, having previously discharged into rivers, coastal lakes and bays or onto the ocean shoreline. Examples include Devonport, Tas. (Wallis & Holmes 1987); Cape Peron, WA (Chalmer & Edmonds 1986); Geelong, Vic. (McLearie & Barkley 1987); Latrobe Valley, Vic. (Sampson & Howard 1987); Sydney (North Head, Bondi and Malabar; Fagan et al. 1993); Lake Macquarie and Tuggerah Lakes (Norah Head) in NSW, and eastern Melbourne (Cape Schanck) (Brown et al. 1990). Problems of eutrophication from well-placed shoreline ocean outfalls on active coasts, some of which have been discharging for many decades, appear to have been very localised in the examples which have been studied.

Monitoring of the Cape Schanck outfall from before the commencement of discharge (1975) until 1988 has indicated that effects on intertidal macroalgal communities were restricted to within one kilometre of the outfall (Brown et al. 1990; Manning 1979). Studies of algal flora and intertidal invertebrates adjacent to the Sydney shoreline outfalls (now moved offshore) also revealed very localised effects, with green algal mats predominating at distances of up to 0.5 km from the outfall, but little measurable effect at distances over one kilometre (Fairweather 1990; May 1985).

The Great Barrier Reef

The GBR is the only Australian marine ecosystem which is comparable in size to areas overseas where large scale eutrophication has occurred (such as the Black and Baltic Seas), and which has been suggested as becoming eutrophic (Bell 1992, 1991). Bell's evidence is drawn mainly from comparison of phytoplankton records from near Low Isles (Qld) in 1928/29 (Marshall 1933) with others collected more recently off Townsville in the late 1970s and early 1980s (Revelante & Gilmartin 1982; Walker & O'Donnell 1981). While the conclusions have a limited scientific and statistical weight, the claims have promoted debate as to the possible eutrophication of the GBR (Bell & Gabric 1991; Kinsey 1991a; Walker 1991).

Queensland coastal catchments have been extensively modified since European settlement by forestry, urbanisation and agriculture, particularly sugarcane cultivation and beef grazing. A recent report (Moss et al. 1992), using catchment modelling calibrated with known run-off data, has estimated that 15 million tonnes of sediment, 77 thousand tonnes of N and 11 thousand tonnes of P are exported to the Queensland eastern coastal zone via river discharges. This is approximately four times the load estimated for the pre-European settlement period. The load in the GBR region is mostly from agricultural run-off, with sewage a secondary and minor contribution. The relative increase in nutrient load in recent times is comparable to that calculated for river inputs to the Black and North Seas, where eutrophication has occurred. Coral reefs, which normally exist in very low nutrient conditions (Kinsey 1991b), are particularly sensitive to nutrients and the algal and filter-feeder overgrowth which occurs in conditions of elevated nutrient supply.

Evidence that a eutrophication problem already exists on the GBR is patchy. Anecdotal evidence from local residents and long-term regular visitors suggests that many reefs were in 'better' condition in past decades than at present. This is supported to a limited extent by the sparse historical photographic records of reefs in locations such as Magnetic Island, Low Isles and the Whitsunday Islands. The claim in most of these cases is that the reefs in question are now far more dominated by algae than in the past. More compelling scientific evidence comes from coral cores taken from reefs offshore from Cairns. The corals from which the cores were taken range up to more than a century old. The yearly growth bands, which provide a record of the environmental conditions at the time the band was growing, have been interpreted as suggesting that significant changes in the growth of coral in this area began to occur about fifty years ago. This correlates with the introduction of the intensive agricultural use of fertilisers in the Barron River catchment on the adjacent coast (Rasmussen & Cuff 1990). The available evidence from nutrient and plankton sampling in the GBR lagoon has not indicated regional or temporal increases in nutrient concentrations (Furnas 1991) but it is acknowledged that systematic monitoring data is scant (Brodie & Furnas 1992).

The difficulty of determining whether the GBR is becoming eutrophic in the face of a limited historical data record and naturally variable environmental conditions reflects similar difficulties experienced in other Australian systems, such as Botany Bay (McGuinness 1988) and NSW coastal lakes (King 1988a). The development of scientific tools to clearly separate natural from anthropogenic changes, and to provide high levels of certainty about historical environmental conditions is a priority for the measurement of long-term degradation.

Monitoring

In Australia, monitoring of nutrient concentrations, nutrient enrichment and eutrophication can be conveniently divided into a few broad categories depending on time and space scales implicit in the objectives of each monitoring initiative. These are: long-term monitoring to derive ambient conditions and detect trends; monitoring associated with known or suspected eutrophic systems; monitoring associated with local developments and discharges; and monitoring associated with ecological process studies. Australian marine monitoring programs have been briefly considered by Williams and Gilmour (1986).

Long-term monitoring

Few long-term records of nutrient or phytoplankton concentrations, taken on a regular basis at consistent sites in Australian coastal waters, are available. For example, Table 5 shows a number of oceanographic stations sampled by CSIRO over long periods. The logistical problems of collecting such data are formidable, many of the samples in this data set being sampled by lighthouse keepers and island research station staff. Most of the stations are no longer sampled and at present there is no national coastal water quality monitoring program. Generally this important issue is one that 'slips through the cracks' of agency responsibility in the

Table 5: CSIRO Coastal Station Network - coastal water quality monitoring[a]

Station (position)	Attempted frequency (per month)	Depth range (m)	Parameters [b]	Date Commenced	Date Finished
Heron Island	1 - 2	0 - 50	T,S,NO_3,SiO_4	Dec 1976	1989
Norfolk Island	1 - 2	0 - 50	T,S,NO_3,SiO_4	Dec 1977	1989
Lord Howe Island	1 - 2	0 - 50	T,S,NO_3,SiO_4	April 1976	1989
Port Hacking 50m	1 - 4	0 - 50	T,S,NO_3,SiO_4 some DO,PO_4	1942	Active
Port Hacking 100m	1 - 4	0 - 100	T,S,NO_3,SiO_4 some DO,PO_4	1953	Active
Maria Island	1	0 - 50	T,S,NO_3,SiO_4	1944	Active
Rottnest Island	1 - 2	0 - 50	T,S,NO_3,SiO_4	1970	Active
Booby Island	1 - 2	0 - 10	T,S,NO_3,SiO_4	June 1977	1984
Lizard Island	1 - 2	0 - 25	T,S,NO_3,SiO_4	Aug 1974	1984
Geraldton	1 - 2	0 - 40	T,S,NO_3,SiO_4	Dec 1978	1985
Low Isles	1 - 2	0 - 10	T,S,NO_3,SiO_4	June 1977	July 1982
Port Macdonell	1	0 - 50	T,S,NO_3,SiO_4 some DO	1973	1981
Eden	1 - 2	0 - 50	T,S,NO_3,SiO_4	1974	1986

a CSIRO have long term records of basic hydrological parameters for Australian coastal waters, approximately 100 000 data points. The data is generally recorded at 6 depths.
b Abbreviations: T - Temperature, S - Salinity, NO_3 - Nitrate, SiO_4 - Silicate, PO_4 - Phosphate, DO - Dissolved Oxygen.
Source: CSIRO, Division of Oceanography, Tasmania

Commonwealth and State sectors. Currently the issue of long-term monitoring is under discussion by ANZECC.

Some long-term records of water quality or benthic conditions have been kept by individuals or small research groups, but these are site specific and the information gathered has not always been published. A good example is the unpublished thirty year record of benthic cover at fixed quadrats collected by Connel at Heron Island . Another data set, collected by the Queensland Department of Environment and Heritage over a 14 year period from 1979 to the present, consists of monthly chlorophyll concentrations from four stations stretching from inside the Southport Broadwater (Qld) to 5 km out to sea (Moss 1992). Another long-term monitoring data set, from Port Phillip Bay, also includes chlorophyll *a* data (Brown 1989).

The use of remote sensing for monitoring the nutrient status of coastal waters has been investigated. Satellite sensors measuring chlorophyll by colour has been the method of choice (e.g. Gabric et al. 1990), using various platforms such as the Coastal Zone Colour Scanner (CZCS - now defunct), Advanced Very High Resolution Radiometer (AVHRR), Landsat and SPOT. Although CZCS worked well in oceanic and clear water conditions, turbidity and coloured humic material severely limited its use in coastal waters. The other platforms still operating also have serious limitations (Muller-Karger 1992) such as poor spatial resolution (AVHRR) or high cost and poor spectral resolution (Landsat and SPOT). Aircraft sensors can also be used and CSIRO is pioneering this technique in Australia. Aircraft platforms can avoid the problems of cloud interference experienced with satellite sensors but may also be very expensive to operate.

Monitoring eutrophic and potentially eutrophic systems

Many of the coastal systems known to be, or suspected of being, either in an advanced state of eutrophication or becoming eutrophic, have had their status monitored. Examples include the Peel-Harvey system, eastern Gulf St Vincent, Cockburn Sound, Lake Illawarra, Lake Macquarie and the Tuggerah Lakes. State environment authorities often have programs at a regional scale for coastal water bodies. For example the Victorian EPA operates long-term monitoring in Port Phillip Bay, Western Port and the Gippsland Lakes. Major new programs commenced in recent years include an intensive research and monitoring program in Port Phillip Bay (CSIRO 1992) and a program incorporating monitoring of fish, benthos, water quality and sediment quality in the GBR region (Brodie & Furnas 1992; P. Moran, AIMS, pers. comm.).

Development, discharge and compliance monitoring

Monitoring programs, usually imposed as a condition of approval by the responsible environmental authority, are often associated with developments such as marinas and harbours, or discharges such as sewage effluent, industrial wastewaters and power station cooling waters. Examples of this type of monitoring include the CSIRO monitoring in Jervis Bay as part of the environmental impact assessment (EIA) for a naval development (Ward & Jacoby 1992); monitoring of the ocean environment off Sydney after the installation of deep-water, offshore sewage outfalls (Fagan et al. 1993); monitoring of thermal water discharges from power stations discharging into NSW coastal lakes (e.g. King & Hodgson 1986); monitoring of the effects of building and operating a marina adjacent to a coral reef (Brodie et al. 1992); and monitoring a nickel refinery wastewater outfall discharging ammonia-rich wastes (Carey et al. 1982).

Many of the monitoring programs of this type have a very limited scope and the results are not widely disseminated or part of the public domain. In addition most environmental management authorities do not require such monitoring to be carried out by independent researchers, the work often being carried out by the developer or discharger, or consultants reporting directly to them. Thus the results obtained, and their interpretation, are often open to criticism of bias or distortion. On the other hand many such programs have resulted in data sets generating valuable ecological information. Examples would include the work of Brown et al. (1990) on the long-term variation in algal intertidal flora both close to and some distance from the Cape Schanck sewage discharge, and studies investigating long-term changes in benthic flora in NSW coastal lakes receiving cooling water discharges from coal-fired power stations (King & Hodgson 1986).

Ecological and oceanographic monitoring

A large proportion of the published studies which could be described as monitoring have been associated with studies of ecological processes. Monitoring is an integral part of the information required to understand temporal and spatial patterns in populations of biota, geochemical fluxes and oceanographic forcing in ecosystems.

Two organisations which have been heavily involved in biological and chemical oceanographic studies are CSIRO, some of their work already having been described in this report (e.g. Mackey 1984; Rochford 1984), and the AIMS. AIMS has worked principally in the GBR region and their biological oceanography group has published records of nutrient concentrations and nutrient processes over the last 15 years (e.g. Furnas 1991; Andrews 1983).

Many marine ecological studies of nutrient/ecosystem interactions included a monitoring component and these have been carried out at many locations around Australia. A few examples include an examination of long-term variations in subtidal algal floras (Jervis Bay and Ulladulla; May 1985, 1981) and macrobenthos (Hawkesbury Estuary; Jones, 1987); comparative studies of communities in different parts of the coast (e.g. zooplankton in Port Phillip Bay and Western Port; Kimmerer & McKinnon 1985) and of macrobenthic fauna of seagrass beds in a number of bays in NSW (Collett et al. 1984); and ecosystem studies in the extensive seagrass beds of Shark Bay, WA (Walker 1989).

Information Systems

Given the lack of any national water quality monitoring programs or national coordination of the monitoring which does occur (note this is now being addressed under the aegis of the National Water Quality Management Strategy), it is not surprising that there is no central database of Australian monitoring results. The Environmental Resources Information Network (ERIN; a subunit of the Commonwealth Department of the Arts, Sport, Environment and Territories), is in the process of setting up a National Marine Information System (NMIS or NATMIS) which should eventually fulfil this need. The CSIRO, through its Coastal and Marine Resource Information System (CAMRIS), is also attempting to coordinate the storage of its national data holdings and further develop applications as part of the CSIRO Coastal Zone Program, as well as link with ERIN and the National Resource Information Centre (of the Department of Primary Industries and Energy) initiatives for database access.

Another activity which may improve the coordination of knowledge of the state of coastal areas is the estuary inventory of Bucher and Saenger (1991, 1989). This inventory 'was compiled to gain a national perspective on the ecological status of estuaries and to identify research and management priorities'. Similar efforts at a State level have previously been made, such as the NSW estuarine inventory of West et al. (1985).

Modelling

The complexity in processes and interactions of coastal ecosystems makes modelling an attractive option. Models can be used to integrate information on physical, chemical and biological processes to assist understanding of the system and to predict spatial and temporal change in parameters of interest to environmental management. Models may be numerical, physical or prototype (NSW Government 1992). They are widely used to estimate pollutant loadings (e.g. Moss et al. 1992), to simulate ecosystem response to increased nutrient loadings (e.g. the Coastal Ocean Ecology Model used in WA; Van Senden & Button 1992), and to provide managers with tools for scenario-building and assessing the implementation of environmental management strategies (e.g. the Port Phillip Bay study; CSIRO 1992). The use of estuarine models in Australia was reviewed by Beer (1983) who considered examples from Western Port, Blackwood River, Peel Inlet, Port Hacking and Gippsland Lakes. Shortcomings in their use were noted but the success of the Peel Inlet example highlighted.

Management

Management of the marine environment varies among states. Systems such as 'assimilative capacity' and 'ecosystem approach' have been published as an overall philosophy in some states (Pearce 1991), but such unifying ideas tend to change with time, circumstances and governments. The current inquiry by the Resource Assessment Commission into the management of the coastal zone in Australia may lead to a greater unity of approach.

Management of nutrient loading to coastal environments, and any subsequent problems of eutrophication, is based on the reduction of nutrient-rich effluents from the land or better dispersion of existing discharges. This may be accomplished by prevention of nutrient generation processes such as control of soil erosion, changes in the use or nature of fertilisers, reuse rather than discharge of nutrient-rich effluents, diversion of discharges into less sensitive or better flushed environments, engineering works to improve flushing, and nutrient removal from effluents. Rehabilitation of existing eutrophic systems has also been attempted in some places, using methods such as replanting of seagrasses (Hillman 1986) and removal of nutrient-rich sediments. Some Australian examples of management to ameliorate eutrophication in coastal systems are listed below.

Engineering and land management

Peel-Harvey

To manage and remedy the previously-described problems of eutrophication in the Peel-Harvey system, a three-fold strategy is being implemented (Gorham et al. 1988) involving:

1. catchment management using controls over fertiliser usage, increased use of slow release fertilisers and controls over land clearance;
2. continuation of mechanical algal clearance;
3. cutting a channel from the Harvey Estuary to the ocean (the Dawesville Channel) to improve flushing in the system (Figure 12).

The introduction of slow-release fertilisers, particularly New Coastal Superphosphate, is expected to have a major impact on the amount of P leached from the Peel-Harvey catchment (Yeates et al. 1984). New Coastal Superphosphate is a mixture of superphosphate, rock phosphate (the slow release component) and sulphur and supplies adequate P and sulphur to previously fertilised soils without excessive leaching. Controls over water movement by building locks, tree planting and creation of small wetlands on drainage lines is also being used to slow the movement of drainage water to the estuaries (Cribb 1993).

The Dawesville Channel will be 1.5 kilometres long, involve the excavation of 4.5 million cubic metres of material and cost $56 million. Tidal flushing combined with reduced residence time for the estuarine water is expected to reduce nutrient buildup and decrease the incidence of algal blooms. The channel will also increase tidal ranges in the estuaries and shift the estuarine conditions towards a more marine state, a change causing some concern in the area (Cribb 1993). The channel is planned to be completed by 1996.

Tuggerah Lakes

The restoration program for the Tuggerah Lakes also depends on a combination of rehabilitation strategies. The three identified objectives are to reduce sediment and nutrient inputs to the lakes, to remove sediment and nutrients from the lakes and to improve tidal exchange (Anon 1990). These objectives are to be achieved by:

1. construction of sediment traps and nutrient filters on streams and drains discharging into the lakes;
2. clearing aquatic plant accumulations from the lake beaches;
3. removing silt and aquatic plants from the inshore lake bed;
4. deepening channels to improve water circulation and navigational access;
5. implementing measures to improve tidal flushing of the lakes. (The principal measure will involve some form of entrance deepening);
6. catchment management measures including sewerage upgrades, erosion control, stormwater treatment, stream bank stabilisation and wetland preservation.

The restoration program is planned to take four years at an approximate cost of $10 million from NSW Government funds. This is in addition to spending some $3.2 million on lake rehabilitation over the period from 1980 to 1988, funded from a variety of sources, but in particular the Wyong Shire Council (Anon 1990).

Catchment management

The principal tool now being used in Australia for land use management, of relevance to nutrient run-off into the coastal zone, is the various forms of catchment management (known in NSW as Total Catchment Management or in Queensland and Victoria as Integrated Catchment Management). These schemes bring together local landholders, state and local government organisations and other interested parties in an attempt to manage catchments as a whole. Some of the methods relevant to nutrient run-off and used within catchments as components of management strategies include minimum tillage and stubble retention agricultural systems, revegetation of stream banks, buffer strips along stream banks, measures to prevent erosion along roadways and during road construction, sewage system upgrades, preservation of wetlands as sediment and nutrient 'filters', controlling stocking rates, contour cultivation; better fertiliser management including timing of use, subsoil injection and the use of slow-release types; and minimising erosion during urban development.

Sewage nutrient discharge minimisation

A number of strategies have been suggested for the minimisation of nutrient inputs to coastal areas from sewage discharge, and most of them are being implemented or trialed somewhere in Australia. They can be grouped into minimisation of nutrients entering the sewage system, better dilution/dispersion methods, reuse of effluents, and nutrient reduction before discharge.

Phosphate from detergents is a major component of the P content of sewage, estimates suggesting it may contribute 65% of the load in domestic effluent (Nicholls et al. 1977). A ban on phosphates in detergents was a major plank of the USEPA's policy to reduce nutrient loading to the US Great Lakes in the 1970s, and is believed to have been relatively successful. It has been suggested that similar bans may be desirable in the GBR region (Chiswell & Hammock 1991; Bell 1989). Recently released reports of the task force examining the blue-green algal problems of the Murray-Darling system have also contained similar recommendations. However reduction in P loading will not be as effective in N-limited systems as it is in P-limited lake and river systems. The replacement of phosphate in detergents may also lead to other coastal marine problems if the replacement is a N compound (Ryther & Dunstan 1971) or zeolite/polycarboxylic acids which have caused slime formation in the Adriatic Sea (MacKenzie 1993).

As previously mentioned, the relocation of sewage outfalls into waters with better dilution and dispersion characteristics has been a common management response to sewage problems in recent years. The schemes usually involve moving outfalls from discharging into rivers, lakes or shorelines to discharge into deep offshore conditions. Typical examples are the Latrobe Valley scheme (Sampson & Howard 1987), Lake Macquarie and Tuggerah Lakes schemes and the Sydney offshore outfall program (Fagan et al. 1993).

Tertiary treatment of sewage to reduce N and P loads in the effluent is not commonly practiced in Australia, but with the outbreak of blue-green algal problems in Australian river systems such measures will now be implemented in many inland sewage plants. Some coastal facilities are already running nutrient reduction methods. Examples include Port Macquarie, where N stripping occurs (AEC 1987) and a number of island resorts in the Great Barrier Reef Marine Park which have N stripping (Radisson Long Island), P stripping (Daydream Island) or both (Green and Lindeman Islands). Most extended aeration sewage treatment plants can be operated to reduce N levels to less than 15 mg L^{-1} and biological P removal is also possible in such plants. A review of the use of nutrient reduction in Australian sewage plants and the methodology available can be found in the Australian Environment Council publication 'Nutrients in Australian Waters' (AEC 1987).

The increasing practice of reuse of effluents for land irrigation has already been discussed in this report. Other schemes for land use of sewage include Werribee near Melbourne where land irrigation of sewage has been practiced since 1897. A final effluent equivalent to secondary treatment is discharged into Port Phillip Bay (Bremner & Chiffings 1991). The scheme has been successful with only minor undesirable effects observed in Port Phillip Bay (Axelrad et al. 1981), but an intensive study of the Bay has now commenced to check on its environmental condition in relation to inputs from Werribee, the Yarra River and other catchment sources.

Reuse of effluent for residential nonpotable application is growing overseas but is not permitted in Australia under current regulatory practices (Wilkins & Anderson 1991). A pilot project to assess operating requirements, risk and community acceptance was run at Shoalhaven Heads (NSW) from 1989 to 1991, the water being mainly used on gardens. The engineers operating the scheme concluded that such reuse can be made viable without compromising public health (Wilkins & Anderson 1991).

The National Health and Medical Research Council and the Australian Water Resources Council have now developed guidelines for the use of reclaimed water in Australia (Anon 1987).

Rehabilitation

Some research on the re-establishment of seagrass beds in Cockburn Sound has been done and is continuing (Hillman 1986; Nelson, pers. comm.). Results have been disappointing so far, with sediment instability a major factor contributing to failure to revegetate. Methods using artificial seagrass mats are now being investigated.

Costs

The effects of eutrophication have been estimated to cost Australia from $10-50 million per annum (Garman 1983). While it is difficult to make estimates of such costs, especially indirect ones from secondary effects, some published cost elements are described below. These appear to be in excess of Garman's estimates, perhaps reflecting the increasing costs involved since those estimates were made.

Structural measures required to reduce the effects of algal growth are estimated to have cost $170m in Cockburn Sound and $50m in the Peel-Harvey system. The smell and appearance of a persistent toxic bloom in the summer of 1987/88 in the Gippsland Lakes area is estimated to have cost $6.5m in lost tourist revenue. Economic loss due to reduction in fisheries catches are more difficult to quantify. In some instances catches have been claimed to have been enhanced while losses have been reported in other nearby areas e.g. enhancement in the Peel Inlet versus losses in the Harvey Estuary (Lennaton et al. 1985; Lennaton et al. 1984).

The costs of preventative measures for marine pollution may be very large. In Sydney the renovation of the inadequate sewage system is being partially financed through a household levy (The Special Environment Levy) of $80/year, which is planned to raise $485 million over 5 years. The total Sydney Clean Waterways Programme is budgeted for $7.1 billion over 20 years.

Summary and comments

Eutrophication in the Australian marine environment is characterised by those factors which make the Australian situation unique as well as those features common to the increasing worldwide incidence of coastal eutrophication. Australia has few rivers which have sizeable discharge and few large semi-enclosed coastal water bodies. However it does have a very large number of smaller enclosed and semi-enclosed coastal lakes, lagoons and bays, and a population highly concentrated (almost 80%) in the coastal zone. Most of our coastal catchments have been extensively modified by agriculture, forestry and urbanisation and the introduction of intensive agriculture in them has been a feature of recent decades. Australian soils are nutrient poor and nutrient run-off to the coastal zone was relatively low before European modification of the catchments. There are few large scale ocean upwelling systems on the coast and in combination with the low run-off, Australian coastal waters are naturally nutrient poor and relatively unproductive. They are thus particularly susceptible to nutrient pollution. The evidence presented in this review documents that almost all coastal water bodies of long residence times in the settled part of the Australian coast (from Cairns south around the coast to Spencer Gulf and the south-west of WA) have some effects of enhanced eutrophication (Table 4). With increasing urbanisation of the coast (notably north and south of Perth, the northern and southern NSW coast, the Sunshine and Gold Coasts and Hervey Bay, and north of Cairns) and further agricultural development on the coastal catchments, the problems will continue. Management by way of Integrated Catchment Management, sewage effluent reuse and system rehabilitation, while already initiated in some places, must continue on a national scale. A nationally coordinated approach to monitoring and management, as is being pursued through initiatives such as the National Water Quality Management Strategy, will assist in this and benefit Australia.

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Human health risk from micro-organisms in the Australian marine environment

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Introduction

The purpose of this review is to summarise present understanding of the micro-organisms of public health concern in the Australian marine environment. Human health aspects relating to chemicals released into the Australian marine environment are covered elsewhere in the SOMER documentation (Philip 1994). Guidelines for exposure to micro-organisms, as with other health-related contaminants, are categorised by water use. This review covers waters used for primary contact recreation (e.g. swimming/immersion in water), secondary contact recreation (e.g. boating) and fisheries.

From a human health perspective, most of Australia's coastal marine environment is distant from the focus of concern, that is regions of urban development. While little is known about the disease-causing micro-organisms (pathogens) in most of our marine environment, overseas experience points to two microbial groups of significance: 1) those introduced from animal and human wastes and 2) indigenous pathogens.

Both groups contain opportunistic pathogens, which only present a health hazard under certain conditions. For example, if an individuals' normal defence mechanisms are compromised, such as by a break in the skin or by the presence of immunosuppressive agents, or if pathogen(s) grow to high densities in the presence of increased organic wastes. The important routes of pathogen uptake are ingestion, aerosol/liquid inhalation and through breaks in the skin. Doses required to cause infection vary widely with micro-organism and status of individuals. However, in general, 10-100 viruses, 1-10 000 parasites and 1000-1 000 000 bacteria are required for infection.

Introduced pathogens (from faecal material) make their way to the marine environment from a number of sources including sewage, septic seepage, boats, stormwater and diffuse run-off via estuaries and major rivers/harbours. These protozoan, helminth, bacterial and viral pathogens may arrive freely suspended, but they are more likely to be associated with particulate matter. Considerable protection from die-off/predation may be conveyed to pathogens which associate with particulate material, and this may present problems in enumeration. Hence, caution is needed in interpreting and comparing quantitative results presented in the literature. Nonetheless, while predicting die-off is a complex problem involving many factors, dilution has the major impact.

In Australia, faecal pollution in seawater is inferred from the presence of certain indicator bacteria, primarily faecal coliforms and/or faecal streptococci (includes the enterococci). However, epidemiological studies of waterborne illness indicate that the common aetiological agents are more likely to be viruses and parasitic protozoa than bacteria (Moore et al. 1994; Seyfried et al. 1985; Cabelli et al. 1982). Furthermore, the recent literature illustrates the poor correlations between waterborne human viruses and faecal coliforms in marine waters (Deuter et al. 1991). This lack of a relationship relates in part to the sporadic presence of pathogens in sewage, which reflects the incidence of illness in the population. In addition, there are problems in sampling (Fleisher 1990), and many pathogens survive longer than the faecal indicator bacteria determined by culturing methods (Evison 1988), with viruses present when indicator bacteria are absent (Hughes et al. 1991). A further complication is the occurrence of nonculturable but viable indicator bacteria which are not enumerated by standard methods and result in further underestimation of pathogen presence (Byrd et al. 1991; Green et al. 1991).

Therefore, this review covers recent literature on the prevalence of a range of key micro-organisms and points to research and management's responsibilities. Secondary public health risks, such as the transfer of antibiotic resistance genes from sewage bacteria to the sediment microbiota and shellfishes (Belliveau et al. 1991; Avilés et al. 1993) are not examined. Also, diseases such as dengue fever and malaria, which are transmitted by insects associated with water, are not considered in this review.

Important pathogens in the marine environment

In Australia over 80% of the population reside in large coastal cities with aging sewerage systems. Sewage discharged from sewerage treatment works to estuarine and coastal waters receives at least primary treatment. As it is currently considered impractical to monitor for the range of pathogens possibly present in seawater, indicator organisms have been chosen (ANZECC 1992).

There is, however, no single suitable group of indicator micro-organisms available (Elliot & Colwell 1985), although the bacterial groups of faecal coliforms and faecal streptococci correlate reasonably well with some of the bacterial pathogens, such as salmonellae (Moriñingo et al. 1992). Furthermore, extension of indicator behaviour, as determined in extensive temperate water studies, to tropical waters must be questioned (McNeill 1992).

Factors affecting faecal coliform decay have been applied to complex models (Canale et al. 1993) in an attempt to predict health risk. Such an approach is doomed to failure, as a range of pathogens in sewage effluent survive for considerably longer periods in marine waters than the indicator bacteria (Tables 1 & 2). In addition, the ratio of faecal coliforms to streptococci is of no use in identifying pollution sources (Pourcher et al. 1991), although the use of strain specific molecular probes may prove useful in the not too distant future.

It is important to note when reviewing the literature that though 99.9% reductions in pathogens may at first appear satisfactory, this is often not the case. For example, infective doses of viruses or protozoa (possibly as low as 1-50 particles) will still be present after 99.9% removal from raw sewage, *viz.* secondary treated sewage could represent a significant health risk if further disinfection/dilution is not guaranteed.

Survival of introduced pathogens

Once an enteric pathogen of human or animal origin enters a marine or estuarine environment, there are a number of factors affecting its fate. These include sedimentation, predation (by copepods and protozoa), parasitism, inactivation by sunlight, temperature, osmotic stress, or toxic chemicals (Kueh et al. 1991). While dilution is clearly a major factor reducing the likelihood of infection, there is a range of interacting factors, such as presence of nutrients and suspended solids, that influence the survival of pathogens in seawater (McNeill 1992).

An indication of mortality rates, in terms of T_{90} or $T_{99.9}$ values (times for 1 and 3 log reduction in numbers, respectively), for selected indicators and pathogens is given in Table 2. Cooler waters and lack of sunlight increase survival. For some bacteria there appears to be a linear relationship between temperature and log T_{90}, and between light intensity and T_{90} (Evison 1988). Hence, release of subsurface sewage plume material not only substantially reduces harmful light effects but also allows for adaptation to higher salt concentrations with less cell stress, pushing T_{90}'s out to several tens of hours for bacteria (Pommepuy et al. 1992) and many months for enteric viruses (Gerba & Goyal 1988).

Viral pathogens

There is a paucity of data on viral illness associated with direct contact (swimming) in polluted waters. This is a result of the fact that of the over 120 classified enteric viruses, most are very difficult to isolate from the aquatic environment and many are simply nonculturable (USEPA 1985). Nevertheless, enteroviruses (polioviruses, coxsackie viruses, echoviruses) hepatitis A & E, adenoviruses, rotaviruses, and

Table 1: Key faecal micro-organisms in sewage and typical % removals by various treatment processes

Source	*Escherichia coli*	*Salmonella / Campylobacter*	Enteric Viruses	*Giardia* Cysts
Raw Sewage (L^{-1})	10^8-10^9	40 000	100-15 000	5200-22 700
% Removal by:				
Primary treatment	50-90, 27-96	50-90, 15	0-30	55
Secondary treatment	91-99	96-99	30-75, 76-99	99
Tertiary (ponds/chlorine)	99.99-99.99999	99.99-100	99.8-99.99	99.8

Source: McNeill (1985, pp. 48, 77), Höller (1988) and Yanko (1993) for bacteria and viruses; *Giardia* data unpublished (Ashbolt).

Table 2: Major potential pathogens/indicators in the marine environment

Group of Organism	Source(s)	Symptom(s)	Marine Survival
Viruses			$T_{99.9}$
Adenovirus	animal/human faeces	C Co F G H R	50 d
Astrovirus	human faeces	G	Unknown
Calicivirus (inc. Norwalk)	human faeces	G	Unknown
Coronavirus	human faeces	G	Unknown
Coxsackie A&B	human faeces	B C D E-M F H R S	2 d - 46 wk
Echovirus	human faeces	C E-M F G R P S	2 d - 46 wk
Hepatitis A	human faeces	H	> 24 d
Poliovirus	human faeces	C F E-M P R	2-130 d
Reovirus	animal/human faeces	None known	> 4 d
Rotovirus	animal/human faeces	G	2-34 d
Bacteria			T_{90}
Aeromonas spp.	animal/human faeces	G S W	'indigenous'
Campylobacter jejuni	animal/human faeces	G-F	Poor
Enterotoxigenic *Escherichia coli*	animal/human faeces	G	5 h-2 d
Faecal coliforms	animal/human faeces	indicator organism	2 h -2 d
Faecal streptococci	animal/human faeces	indicator organism	2 h - 12 d
Mycobacterium marinum	seawater	S W	'indigenous'
Salmonella spp.	animal/human faeces	G-F	12h - 5d
Shigella spp.	animal/human faeces	Bloody diarrhoea	<15->70 d
Vibrio spp.	seawater, faeces	G W	'indigenous'/<6d
Yersinia enterocolitica	animal/human faeces	Appendicitis-like G	days-weeks
Protozoa			
Cryptosporidium parvum	animal/human faeces	Watery diarrhoea F	Unknown
Entamoeba histolytica	faeces	G/dysentery	Unknown
Giardia intestinalis	animal/human faeces	Diarrhoea/bloating	Unknown
Helminths			
Ascaris spp.	animal/human faeces	Roundworm	Unknown
Taenia spp.	animal/human faeces	Tapeworm	Unknown
Dinoflagellates			
Alexandrium spp.	ballast/seawater	PSP	'indigenous'
Gambierdiscus toxicus	seawater	ciguatera shellfish poisoning	'indigenous'
Gymnodinium spp.	ballast/seawater	PSP	'indigenous'

Source: Chung & Sobsey 1993; Hallegraeff 1992; Evison 1988; McNeill 1985; Akin et al. 1976; Blawat et al. 1976.
Key: C-carditis, Co-conjunctivitis, F-fever, D-diabetes, E-M-encephalitis-meningitis, G-gastroenteritis, G-F-gastro+fever, H-hepatitis, P-paralysis, PSP-paralytic shellfish poisoning, R-respiratory infection, S-skin infection, W-wound infection.
T_{90} or $T_{99.9}$: times for 1 or 3 log reduction in numbers respectively at 10-25°C.

caliciviruses (including Norwalk and small round structured viruses) have all been associated with swimming-related illness (Dufour 1986).

Seasonality plays a role in the presence of enteric viruses in the community and sewage effluent. Therefore, it may be necessary to detect a range of viruses to determine viral risk in receiving waters. Identification of many of these nonculturable viruses requires molecular methods, which in the case of enteroviruses appear to largely identify infective viruses (Enriquez et al. 1993).

Viruses which specifically infect bacteria (bacteriophages) are useful surrogates for human enteric viruses in survival studies (Elliot & Colwell 1985). Bacteriophages are readily enumerated by plaque assays on bacterial host cells and can serve as useful tracers of sewage effluent, as demonstrated off Geelong, Victoria (Richardson et al. 1993). However, it is important to note that coliphage presence in sewage is related to initial coliform concentration, not the concentration/presence of enteric viruses (Grimes et al. 1986).

To date, only the enterovirus group is monitored in beach waters (in 10 L volumes) under a European directive (Council of the European Communities 1976) and proposed guidelines in some American states for recreational waters (Bitton 1980). As a consequence, most information for enteric viruses relates to the enterovirus group. A problem with this approach is that the majority of culturable enteric viruses in marine waters are not enteroviruses, but generally reoviruses (Grabow et al. 1989; Grohmann et al. 1993). Hence, as discussed in the Sydney region studies (Grohmann et al. 1993), it is important to focus on a range of viral groups, else risk in considerably underestimating enteric virus risk.

The majority of shellfish-associated illness reported in the US is of unknown aetiology, with the largest group identified being due to nonculturable Hepatitis A (20% of illnesses) and a small percentage due to nonculturable Norwalk viruses (Craun 1986). In contrast, Norwalk-like viruses have been the main cause of viral food poisoning from sewage contaminated oysters in Australia (Grohmann et al. 1980). Based on a virus prevalence of 19% in shellfish, Rose and Sobsey (1993) estimated that there was a one in a hundred chance of becoming infected with an enteric virus by consuming raw shellfish. While such an analysis lacks sufficient viral prevalence data, reliance on bacteriological standards appears to be quite deficient in protecting human health. For example, a multi-state outbreak of hepatitis A resulted from eating raw oysters in waters meeting bacteriological standards (Desenclos et al. 1991).

Viruses in waters generally adsorb to solids which protect them from inactivation by biological, chemical and physical factors (USEPA 1985). Marine sediments, for example can adsorb more than 99% of a poliovirus suspension (containing 10^8 plaque forming units per mL) and may contain 10-10 000 times the concentration of viruses in overlying water (Schaiberger et al. 1982; LaBelle & Gerba 1979). Hence, surface water sampling alone may not give a true indication of the potential viral hazard (Rao et al. 1984). Such sediment-bound viruses can also be taken up by shellfish, thus allowing their bio-accumulation in marine life near sewage outfalls (Lewis et al. 1986).

Virus survival in this adsorbed state is of particular interest, as it is well known that virus removal from shellfish by seawater flushing (depuration) occurs at a significantly slower rate than for bacteria (Lewis et al. 1986). Furthermore, sediment-associated viruses are known to maintain their infectivity (Taylor et al. 1980).

Due to considerable variability in enumerating enteric viruses, results are generally discussed on a presence/absence basis. For example, based on the European directive of no enterovirus in 10 L, studies on selected beach waters indicated 27-35% of southern Welsh, 100% of Yorkshire, 29% of English and 46% of Northern Ireland beaches failed the enterovirus directive over various periods in the last seven years (Deuter et al. 1991; Hughes et al. 1992). In Sydney, tests on 100 L samples indicated enteric viruses (entero-, adeno- and reo-viruses) were present in about 20% of city beach waters (Grohmann et al. 1993) prior to commissioning three deepwater ocean outfalls (discharging primary treated sewage effluent). In the two years since commissioning the deepwater outfalls, only two beach water samples (of over 300) cultured positive for an enteric virus (Grohmann, unpublished).

Bacterial pathogens and environmental strains

In most western countries, including Australia, gastrointestinal illnesses due to *Campylobacter* species outnumber those due to any other identified pathogen. However, illness from these bacteria will almost solely be due to food or person-to-person contact (Cohen & Gangarosa 1978). Of the introduced bacterial pathogens (Table 2) salmonellae and shigellas probably survive the longest in marine waters. Nevertheless, the most recent outbreak of swimming-associated *Salmonella* illness world-wide was typhoid fever (*Salmonella typhi*) which occurred in Western Australia in 1958 (Anon. 1961). *Shigella* gastroenteritis has only been implicated from swimming in fresh waters (Herwaldt et al. 1991).

The causative agent of cholera, *Vibrio cholerae* 01 (and recently type 0139 in Asia) is endemic in various countries, but not Australia. Nevertheless, pandemic strains of cholera could be transported to Australian marine waters in ships ballast water. Whereas non01 strains are not uncommon in warm Australian fresh waters, 01 types have only rarely been isolated from fresh waters (Desmarchelier 1989). This contrasts to the other pathogenic vibrios, *V. parahaemolyticus* and *V. vulnificus* which are endemic to temperate Australian seawaters and are two of the most common causes of bacterial food poisoning from shellfish. Furthermore, these endemic vibrios (*V. alginolyticus*, *V. parahaemolyticus*, *V. vulnificus* or *V. mimicus*) have been reported in Australia and overseas as the cause of swimmer's ear following contact with warm seawater, whereas *Pseudomonas aeruginosa* is the causative agent following freshwater contact (Dufour 1986). However, the significance of *P. aeruginosa* in seawater, as a causative agent of swimmer illness, is still being debated and some recommend monitoring due to its possible presence in the absence of faecal coliforms (Mates 1992). Similarly, *Staphylococcus aureus* has been shown to be a useful indicator for ear, respiratory and total illness developed by bathers at densely populated bathing beaches (Cheung et al. 1991). In contrast to these overseas findings, slight to heavily polluted bathing beaches in Sydney contained very low densities of *P. aeruginosa* and *S. aureus* (Ashbolt et al. 1993).

Indigenous marine vibrios are also the major aetiological agent in wound infections observed worldwide (Dufour 1986). It is not known if there is a relationship between pollution and occurrence of pathogenic vibrios or other wound bacteria, such as *Mycobacterium marinum*. The latter is also indigenous to seawater, and infection is so common that it is often considered an occupational hazard among commercial fishers (Dufour 1986).

Pneumonia, resulting from the inhalation of seawater containing *Pseudomonas putrefaciens*, *Staphylococcus aureus* or *Aeromonas hydrophila* have been associated with near-drowning accidents (Dufour 1986). Whereas the low densities of *S. aureus* are unlikely to be a problem in Sydney waters, *Aeromonas* and *Vibrio* spp. are present at concentrations which could cause wound infections or septicemia under certain conditions (Kueh et al. 1992).

Examination of survival characteristics of some bacteria in the laboratory has shown that traditional cultural methods do not detect all viable bacteria present. The portion of the viable population which do not grow in culture have been termed 'viable but nonculturable' or somnicells (Byrd et al. 1991). Somnicells have been reported for a range of pathogens, and such cells may retain their ability to cause disease (references cited in Turpin et al. [1993]).

Lack of correlation between pathogens and indicator bacteria may in part be due to light stress induced somnicell formation in *Escherichia coli* and *Enterococcus faecalis* (Byrd et al. 1991; Barcina et al. 1990). Furthermore, Lewis et al. (1991) suggest that *E. coli* in seawater (in the dark) may be underestimated by up to 100-fold by relying on culturing methods.

Parasitic protozoa and helminths

For the general population, cysts of *Giardia*, and oocysts of *Cryptosporidium* are probably the most important parasites in Australian sewage. In contrast there is an absence of data on the health significance of helminths in marine waters.

Though no relevant survival data for *Giardia* or *Cryptosporidium* was available for this review, artificially added cysts may remain viable for weeks in freshwater (De Regnier et al. 1989), and high numbers of cysts (10-100/L) have been detected in marine waters off San Juan, Puerto Rico (Correa et al. 1990); in Victoria Harbour, Hong Kong (Hutton, unpublished); and oocysts (up to 400/L) have been found in the Georges River, Sydney (Ferguson et al. 1994). Given that only a few parasite cysts cause infection and that both have been contracted during swimming (Sorvillo et al. 1992), the prevalence of viable parasites in environmental waters requires investigation.

Airborne pathogens

Cases of illness transmitted through inhalation of aerosols from contaminated waste water should not be ruled out, particularly near a surface sewage/contaminated stormwater plume. Aerosols produced in surf can contain a 200-fold increase in viruses per ml than that present in the seawater (Baylor et al. 1977). Also, bacterial aerosols have been reported to cause skin sensitivity in bathers (Gruft et al. 1975).

Environmental algae

Harmful algal blooms that have occurred in Australia were reviewed by Hallegraeff (1992) and are summarised below. Many Australian red tide species (primarily *Trichodesmium erythraeum*), although possibly more common in recent years in waters north from Sydney or Perth, have not been reported as containing toxic compounds. On the other hand, toxic dinoflagellate (*Alexandrium minutum*) red tides have occurred annually during spring since 1986 near metropolitan Adelaide. Other paralytic shellfish poisoning (PSP) dinoflagellates, such as *Alexandrium catenella* and *Gymnodinium catenatum*, have also become a problem during the 1980s in the southern states. For example, blooms of the latter have resulted in shellfish farm closures in Tasmania for up to 6 months. The sudden occurrence of toxic species and their detection in ship ballast water has led to speculation that they have recently been introduced to Australia rather than indicating a rise in eutrophication.

There are however, toxic dinoflagellates endemic to tropical Australian waters: they include *Gambierdiscus toxicus*, the cause of ciguatera poisoning from fish (Hallegraeff 1992). There is also recent fossil record of the tropical PSP species *Pyrodinium bahamense* as far south as Newcastle, but unless ocean warming eventuates, Hallegraeff suggests the latter is unlikely to return to Australian waters.

Epidemiology and standards

Epidemiological studies

A range of prospective epidemiological studies have been undertaken worldwide over the last decade (Corbett et al. 1993; Harrington et al. 1993; Cheung et al. 1991). Most, however, rely on insufficient water samples (of variable volumes) and poor recovery of stressed cells resulting in an error of at least 50% in estimating the likely concentration of bacteria to which an individual swimmer is exposed (Fleisher 1990). As a consequence, different index bacteria are proposed with varying correlations to illnesses.

Sewage release from surface ocean outfalls has definitively caused gastrointestinal illness in swimmers at nearby bathing beaches (Zagorski et al. 1984) and children are particularly vulnerable to such illnesses (Alexander et al. 1992). However, the majority of illness reported in prospective epidemiological studies, including two major studies undertaken in Sydney, have been respiratory and unrelated to standard faecal indicator bacteria presence (Corbett et al. 1993; Harrington et al. 1993; Cheung et al. 1991). Nevertheless, unidentified agents from sewage may cause some of these respiratory infections via aerosols, and several children apparently acquired a Norwalk-like gastrointestinal virus while swimming near a stormwater outlet in Sydney Harbour (Ferson et al. 1993). Except for the 1958 typhoid outbreak, no epidemiological study has been reported for Australian marine waters outside the Sydney region.

On the other hand, in Harrington et al.'s study, there was tenuous correlation between densities of the faecal bacterium *Clostridium perfringens* over 50 cfu/100 mL and increased respiratory illness (unpublished). Furthermore, staphylococci have strongly correlated with swimmer illness (respiratory and ear) at beaches with very high swimmer densities, suggesting staphylococci indicate person-to-person transmission rather than sewage effluent contamination (Cheung et al. 1991).

Microbiological Standards

The microbiological guidelines for recreational waters in Australia are expressed in terms of concentrations of colony forming units (cfu) of faecal coliforms and enterococci (NH&MRC 1990). These and the new draft standards (ANZECC 1992) are similar to those reported around the world for faecal indicator bacteria. However, what is clearly missing are standards for pathogens, including viruses, parasitic protozoa and the indigenous bacterial and algal pathogens. With the advent of molecular detection methods for such pathogens (Dorsch et al. 1994; Vesey et al. 1994; Enriquez et al. 1993; Tsai et al. 1993), it is timely to reassess microbiological standards.

The epidemiological evidence for accepting the enterococci standard proposed by Cabelli for marine waters (Cabelli et al. 1979) has been put in doubt (Fleisher 1991). Furthermore, the longer survival of enterococci in marine waters compared to faecal coliforms is of little consequence when in their absence, enteric viruses are present in either seawater or shellfish (Schaiberger et al. 1982).

On the other hand, the sewage indicator bacterium, *Clostridium perfringens* is significantly elevated at sewage/sludge disposal sites (Ashbolt et al. 1993; Hill et al. 1993). This bacterium may indicate the presence of long lived faecal micro-organisms as illustrated by elevated levels hundreds of kilometres down current of a sewage sludge dump site (Hill et al. 1993). Currently no standards exist for *Clostridium perfringens*, although a Hawaiian (Fujioka pers. comm.) and the Sydney (Harrington et al. 1993) epidemiological studies are evaluating its usefulness as a pathogen index organism.

National priorities

Monitoring programs

Coastal water microbiology has been monitored in the USA since 1955, mounting to a cost of $US130 million in 1989 (Ludwig et al. 1992). Most of this data had not been condensed into useful information (i.e. no computerised database). In addition, the data exist in pools of intensive samplings around discharge points, with very little overall study of the coastal waters. Hence, normal background levels are generally not known and monitoring programs are not coupled to the decision making and resource allocation processes.

A similar situation occurs in Australia. Government agencies require beach waters to be monitored, but streams, lagoons or stormwater drains nearby generally are not. For example, despite the use of submarine outfalls in cities like Sydney or Rio de Janeiro, unmonitored local stormwater discharges/leaky sewers still result in unacceptably high coliform counts at ocean bathing beaches during storm events (Kueh et al. 1991; Jordão & Leitão 1990).

New recreational and fisheries water standards

If indeed the current faecal indicator bacteria are poor predictors of health risk in temperate and/or tropical Australian waters, where should we be headed? Clearly the presence of fresh sewage material is a health risk; hence a coordinated national measure by applying ANZECC guidelines could be a move in the right direction. On the other hand, a health risk may be present in the absence of these short-lived indicators, or they may be unsatisfactory due to their growth in warmer waters or nonculturability.

Such problems can only be ironed out by a coordinated national program specifically designed to answer the various aspects mentioned above. Taking primary recreational waters as an example, a nationwide database of faecal indicator bacteria for bathing waters could lead onto a better understanding of their distribution (seasonal trend and the impacts of rainfall events) and relevance in protecting public health. However, additional data, such as time of sampling, state of tide during sampling and preceding rainfall is also required along with financial resources to reduce such data into useful information.

Furthermore, a simultaneous step should be to coordinated epidemiological data gathered by health departments and major pathology laboratories to identify likely water or fisheries exposure routes. A reasonable attempt to collect such data has been made for example, in the United States (Moore et al. 1994). Next, specific epidemiological studies and statistical evaluation of risk exposure from measured concentrations of key pathogens and indicators is required to assess what organisms to use and most importantly, what level of risk should be targeted by the Australian Standards. Additional focused monitoring and quantitative risk assessment studies would also be required to understand the role of the nonfaecal pathogens.

While the above example was for primary contact waters, secondary contact standards could rationally be deduced from the primary standards for each major water type (tropical, temperate, ocean, estuarine). Standards for fish/shellfish and fisheries waters should be generated by an analogous program focusing on these foods and source waters and applying quantitative risk assessment methods (Rose & Sobsey 1993) to aid in the establishment of Australian standards.

Until such a program is undertaken, little progress can be made on rationally setting national standards or using appropriate indicators/pathogens for environmental reporting. Key data are available from overseas on likely target micro-organisms, doses required to cause infection and models to apply to estimate risk exposure. However, no progress has been made on establishing health-based microbiological standards for primary or secondary contact waters since the US-EPA program in the late seventies (Cabelli et al. 1979). Therefore, to develop standards suitable for Australian waters we need to use the new methods available in Australia for monitoring target micro-organisms and collate the health data available from the larger pathology laboratories. In summary, it appears that a national priority and appropriately coordinated financing are all that are required to produce health-based standards to protect our diverse range of marine waters.

Conclusions

1. Traditional bacterial indicators do not reliably reflect the presence or absence of enteric pathogens in seawaters or sediments. Hence, despite little epidemiological evidence of health problems, the microbiological status of Australian marine waters is unclear. On the other hand, native microbial pathogens are of concern to Australian fisheries.
2. Enteric viruses should be regarded as indicators of faecal contamination, since they are probably more closely related to the causative agents of infections acquired by users of recreational waters rather than faecal coliforms or enterococci. Presence of bacteriophages may be a useful surrogate for enteric virus survival in seeding experiments.
3. Discharge of nondisinfected primary/secondary sewage effluent to bathing waters is expected to represent a local health risk without further dilution/die-off of at least 1000-fold, as can occur through deepwater ocean outfalls.
4. A national coordinated effort is required to collect and collate monitoring and epidemiological data, including the impact of toxic algae, for the setting of appropriate Australian Standards for primary contact, secondary contact and fisheries waters.
5. Research is required on the usefulness of new index organisms such as *Clostridium perfringens* and key enteric viruses, the significance of underestimation of faecal bacterial due to viable but nonculturable

forms, and the prevalence of the parasitic protozoa (*Giardia* and *Cryptosporidium*) and native pathogens in the Australian marine environment.

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The technical contribution by Dr N. Ashbolt was reviewed by Ms. Ann McNiell, State Water Laboratory, Melbourne.

Sewage: Sydney (NSW) - a case history

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Introduction

Many major coastal cities in the world discharge their wastewaters to the ocean. Wastewater may consist of a combination of domestic, industrial and agricultural effluent. Domestic sewage contains nutrients and bacteria, whilst wastewaters from industrial processes contain trace metals and chemicals, and agricultural effluent may contain chemicals including pesticides. Ocean conditions prevailing around the world's coastlines vary, as do effluent discharge quality and practices. Discharges can be from multi-port diffusers in deep water, or single pipe outfalls at the shoreline. Levels of treatment of the sewage also vary. Table 1 presents data on sewage discharges from a number of coastal cities.

Depending on local ocean conditions and the sewage discharge configuration, effluent may be rapidly dispersed and diluted with little visible trace, or may remain in a highly visible and relatively undiluted plume in the vicinity of a discharge point. Studies in the 1970s led to the conclusion that the coastal waters offshore from Sydney had a significant capacity to accept non-toxic organic and inorganic wastes. This factor has been particularly significant in the development of strategies to minimise impacts associated with the disposal of Sydney's sewage (Water Board 1989).

History

Sydney's first sewers, laid in the 1850s, drained raw sewage into the harbour. In 1888, two interceptor sewers were commenced, one to the ocean at Bondi, and one to a sewage farm beside Botany Bay, which was later extended to the ocean at Malabar. On the northern side of Sydney Harbour, a sewer to an ocean outlet at North Head was commenced in 1916 (see Figure 1). The major coastal treatment plants were completed at Bondi (1960s), Malabar (1974) and North Head (1984), and were designed to remove screenings, grit, grease, and settleable solids (Water Board 1989).

The ocean discharge points at North Head, Bondi and Malabar received 80% of Sydney's total sewage flow. The effluent was discharged less than 10 metres below the surface from rocky headlands, through single pipe outfalls. The remaining 20% continues to discharge from smaller coastal and inland treatment plants. In the early 1980s, after years of planning, the final decision was taken to divert the effluent of the three major sewage treatment plants from the shoreline to offshore deepwater outfalls, using multi-port diffusers in water depths of 60 to 80 metres.

Through the 1980s, public opinion had increasingly demanded action to overcome often severe pollution of bathing beaches near the

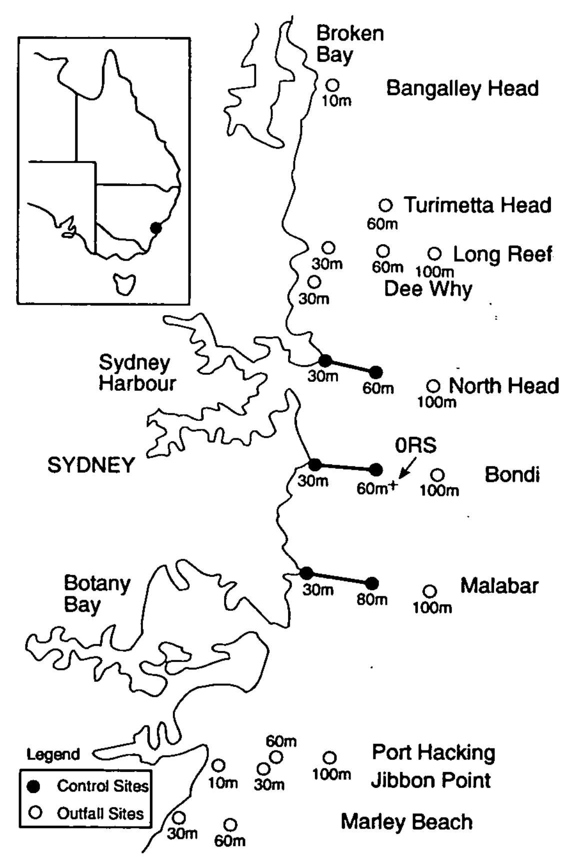

Figure 1: Sydney's urban coastline, showing the deepwater outfalls

outfalls. The first deepwater outfall was commissioned at Malabar in September 1990. In December 1990, the North Head deepwater outfall came on line, and the Bondi deepwater outfall commenced full operation in August 1991.

Framework for management of Sydney's sewage

The Water Board owns and operates Sydney's sewers and treatment plants, which collect and treat the effluent from domestic and industrial/commercial sources. The latter source accounts for approximately one third of the effluent quantity treated at the coastal treatment plants (Water Board 1989). The quality and quantity of the industrial/ commercial effluent entering the Water Board's sewers is controlled by the Board under trade waste agreements with each discharger. Construction of treatment plants, and the quality and quantity of discharges of treated effluent to the environment, are controlled by legislation administered by the NSW Environment Protection Authority (EPA).

Approvals and licences issued under the legislation may contain requirements for regular monitoring of discharged effluent and reporting to the EPA. Monitoring of the receiving environment may also be required. Approvals issued to the Water Board for construction of the Sydney deepwater outfalls had such a requirement.

Environmental monitoring

After conducting early pilot studies, the Sydney deepwater outfall monitoring program was begun by the Water Board in 1989. Its aim was to quantify the environmental impact of the change to deepwater discharge of most of Sydney's sewage effluent. Monitoring commenced prior to commissioning of the deepwater outfalls, and is to continue for two years beyond the commissioning of the last deepwater outfall. Responsibility for the implementation of this environmental monitoring program was transferred by Government decision to the EPA in 1990.

The monitoring program examines the complete sewage path, from treatment plant discharge, through dispersion in the ocean, and ending in accumulation in the biota and sediments. Oceanographic studies start with effluent discharge parameters, and examine the dispersive forces which act upon the discharged effluent using field-calibrated numerical models. Chemical and biological studies examine the fate of the effluent and its constituent pollutants, and measure the resulting environmental impacts. Particular studies examine levels of contaminants in fish and sentinel oysters, abundance and diversity of fish and macrobenthic organisms, and offshore and beach water quality. The monitoring program commenced in sufficient time to compile a set of baseline environmental data prior to commissioning of the first deepwater outfall. These data were to be compared with an equivalent set of post commissioning results.

State of Sydney's marine environment

The state of Sydney's marine environment is presented as it existed in 1991, during the period in which the deepwater outfalls were being commissioned. The major outfall at Bondi continued to discharge at the shoreline at this time. The results of the deepwater outfalls environmental monitoring program (EMP) are used to describe the situation.

Australian Water and Coastal Studies (1992) described the ocean off Sydney as having a

Table 1: Data on coastal sewage outfalls

Outfall	Service Population	Effluent Treatment	Water Depth (m)	Length (m)	Average Flow (ML/d)*	Risers x Ports
Sydney						
Malabar	1 500 000	Primary	80	2900	490	28 x 8
Bondi	600 000	Primary	60	1700	165	26 x 4
North Head	1 200 000	High Rate Primary	60	2900	385	36 x 6
The Hague	2 000 000	not known	15	2000	240	84
Los Angeles (Hyperion)	4 000 000	Adv. Primary	60	8000	1360	84 x 2
Rio de Janeiro	not known	Untreated	27	3300	not known	180
Montevideo	700 000	Screening	10	2150	310	25
San Diego	1 300 000	Adv. Primary	63	3450	720	29 x 2

* = megalitres per day
Source: Camp Dresser & McKee (1989)

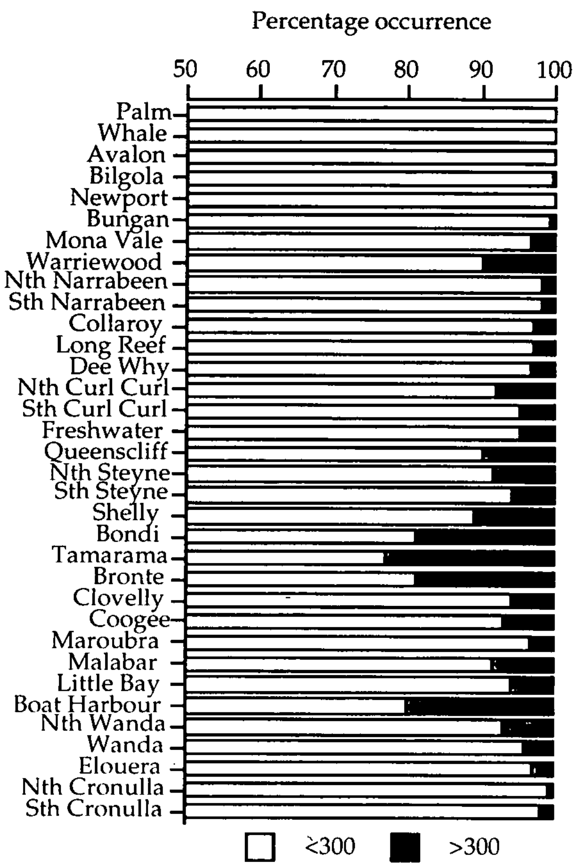

Figure 2: Faecal coliform densities at Sydney bathing beaches: October 1990 to August 1991 (categories are geometric means of colony numbers per 100 ml)

predominance of southerly flowing ocean currents, and temperature stratification of the water column. As a result, diluted sewage from the deepwater outfalls generally moved south, whilst remaining trapped below the water surface.

Figure 2 presents data on the quality of bathing waters at Sydney's beaches between late 1990 and mid 1991, as distributions of two categories of faecal coliform densities, measured from water samples taken four days per week. The lower faecal coliform category, less than 300 colony forming units (cfu) per 100 mL, corresponds to a level where waters pass NSW Department of Health guidelines for bathing waters.

Figure 2 showed that beaches at greater distances from the main sewage outfalls (north from Shelly beach and south from Malabar Beach) had much greater occurrences of the lower category of faecal coliform densities in the water than those closer to outfalls. There were discontinuities at Bondi, Warriewood and Boat Harbour, near continuing shoreline sewage outfalls.

In the ocean at the commissioned deepwater outfall sites, the sewage plumes were normally trapped in stratified waters. In the deeper half of the water column (greater than 30 m depth), mean faecal coliform densities were up to several hundreds of cfu/100 mL. In the upper half (top 30 m) of the water column, mean levels were less than 10 cfu/100 mL (Philip et al. 1993). At offshore sites away from the outfalls, mean faecal coliform levels in the upper 30 m of the water column were similarly low. Mean faecal coliform levels in the lower water column (greater than 30 m depth) at these sites were relatively higher, but of the same order of magnitude. This was attributed to the diluted deepwater plumes occasionally reaching these sites, whilst remaining at depth and being swept along by ocean currents.

Of a range of organochlorine compounds being investigated in sediments in 60 to 80 m water depth off Sydney, only hexachlorobenzene had been detected in the earliest samplings using relatively high detection limits. It was found in similar concentrations in locations across the study area, both close to and distant from the sewage outfalls (Philip et al. 1993). Trace metals concentrations varied both spatially and temporally across the same locations. All levels detected lay within the 'low contaminant status' range as defined by Thomas (1987).

Various studies overseas have shown that natural materials and contaminants can concentrate in the thin layer of sea water (the microlayer) extending from the air-sea interface to a depth of about 50 micrometres. The microlayer can have toxic or sublethal effects on microscopic organisms such as the eggs and larval stages of fish. Off Sydney, chemical and bacteriological studies of microlayers have been conducted near the main Sydney outfalls during their shoreline discharge phase. Overall, the chemical concentrations found by the EMP work at the three outfalls were considerably lower than those found in the overseas studies (Table 2) that used comparable sampling equipment (Rendell 1993).

Contaminant concentrations in marine biota were included in the deepwater outfalls environmental monitoring program. However, the statistical designs of these contaminant projects required that the data be compiled for the full term of the study, before detailed statistical analyses were carried out. Observations about contaminant concentrations in marine biota are presented in

Table 2: Comparisons of maximum concentrations of contaminants in microlayer samples from different studies

Study	Maximum microlayer concentration* ug/L			
	Total PAHs	Total metals	Total PCBs	Total pesticides
EMP (all sites)	0.28	50	0 (<0.004)	0.0069
Cross et al. (1987)	56	800	39	0.44
Hardy et al. (1987)	8000	4800	3.9	0.044
Sauer et al. (1989)	-	-	0.046	0
Hardy et al. (1990)	6.0#	73	-	-

* Except for metals the totals may not be based on the sum of exactly the same constituents. Results below the detection limit were assumed to be zero. Total metal values were based on the sum of concentrations for cadmium, lead, zinc, silver, and copper in the sample as was used in Hardy et al. (1987).
PAH - Poly Aromatic Hydrocarbon; PCB - Poly Chlorinated Biphenyl; # - Maximum mean over three sampling times.

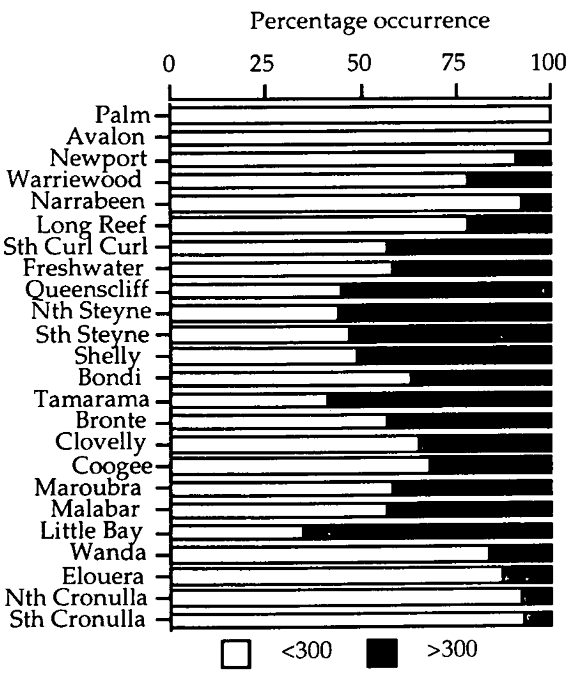

Figure 3: Sydney's beach water quality: December 1989 to September 1990 (categories are geometric means of colony numbers per 100 ml)

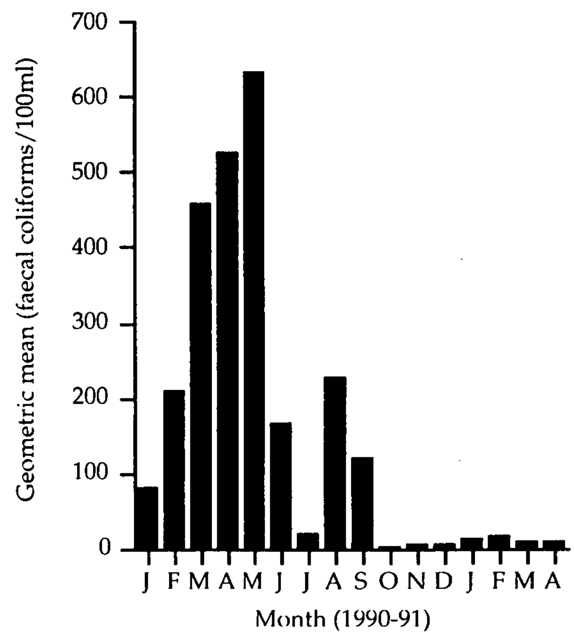

Figure 4: Trend in bacteria levels at Maroubra beach: 1990 to 1991

the Trends section of this report.

Trends in the state of Sydney's marine environment

The environmental state just described is predominantly the result of discharge of diluted sewage through the deepwater outfalls. Introducing results from 1988 to 1990, when all Sydney's sewage treatment plants were discharging at the shoreline, demonstrates the trends which have occurred in Sydney's marine environment over these years.

Prior to 1990, monitoring results showed that the marine waters and biota off Sydney had been significantly and adversely affected by the presence of sewage from treatment plants along some forty kilometres of Sydney's coastline. The seawater at many surfing beaches was often visibly discoloured, and contained particles of grease attributed in large part to the discharged sewage (CDM 1989). Water quality at beaches, in terms of levels of bacteria, did not meet Health Department guidelines for bathing waters on many occasions (Philip 1992). The flesh of some species of fish caught near the major shoreline discharge points contained contaminants at levels above residue limits set by the National Health and Medical Research Council (Mann & Ajani 1991).

Figure 3 presents the distribution of two categories of faecal coliform densities at Sydney's beaches between December 1989 and September 1990. It showed that famous Sydney beaches such as South Steyne (Manly) and Bondi failed the Health Department's bathing water quality guidelines more times than they passed. The trend from this poor environmental state to the present is reflected in Figure 4. This shows monthly coliform densities measured at Maroubra Beach near the Malabar sewage treatment plant, and spans the September 1990 commissioning date of the deepwater outfall. The reduction in bacterial densities was 100-fold. Most of Sydney's beaches now pass the NSW Health Department's bathing water quality guidelines 90% of the time (Beachwatch 1993).

The results from analyses of oysters deployed at the North Head and Malabar shoreline outfall sites, spanning the times of commissioning of deepwater outfalls at these sites, confirmed the pre- to post- commissioning trend observed for faecal coliforms.

Figure 5 (Philip et al 1993) shows that the concentrations of technical chlordane (*) in oysters dropped from significant levels to below the detection limit after commissioning in December 1990. Figure 6 shows a decrease in the concentration of technical chlordane (*) and dieldrin in one fish species, red morwong, known to be resident along the shoreline (P. Scanes, EPA, pers. comm.). This reduction in inshore contamination was a manifestation of the absence of the shoreline sewage plumes.

There has been wide variability in diversity and abundance of fish and seabed macro-invertebrates. A detailed interpretation of the

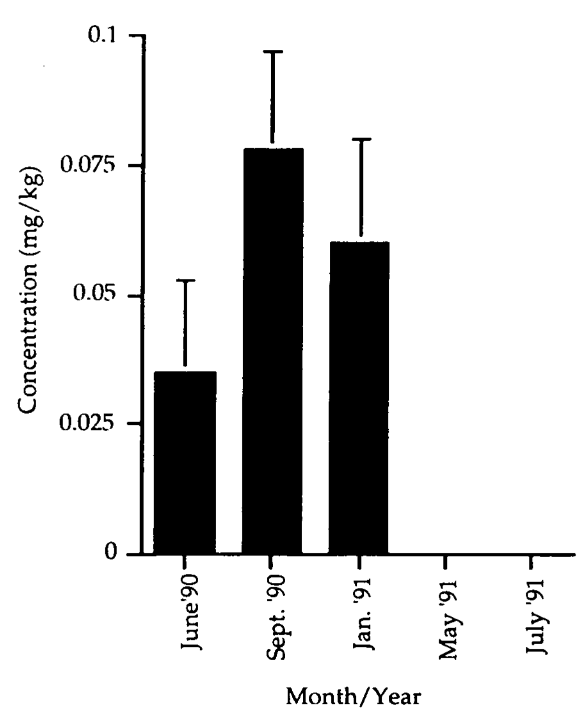

Figure 5: Trend in chlordane (*) levels in oysters from North Head shoreline outfall site - 1990 to 1991

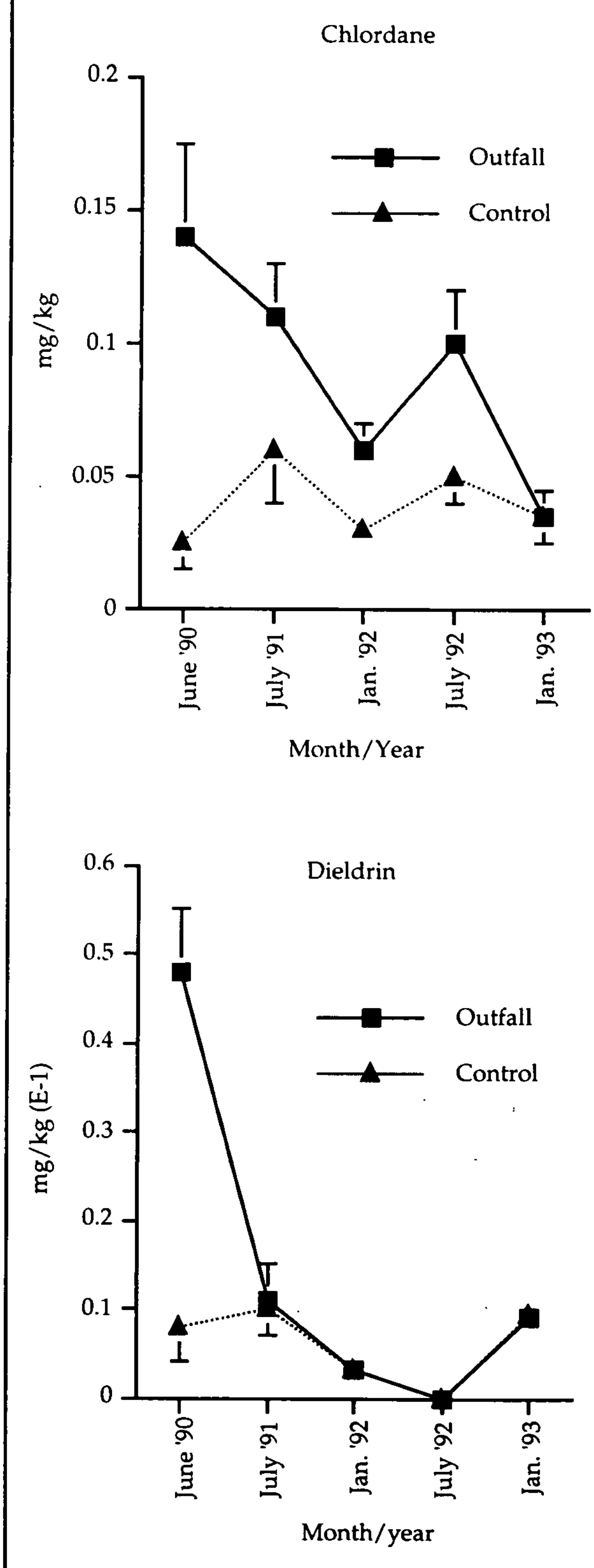

Figure 6: Trend in organochlorine levels in Red Morwong off Sydney - 1990 to 1992

state of Sydney's marine environment up to 1993, as determined by the EMP monitoring program is planned to be separately reported in late 1994.

Summary

The adverse impacts of the discharge of Sydney's urban sewage into the marine environment have been reduced by the commissioning of the deepwater outfalls. Improvements have occurred to the quality of the bathing waters at beaches close to the outfalls, and no increased adverse impacts have resulted at other Sydney beaches further away. Levels of contaminants measured in fish and deployed oysters along the shoreline have reduced.

Monitoring of the marine environment off Sydney is continuing, to determine whether the discharge of sewage in deep water, at greater dilutions than was previously achieved at the shoreline, will have any reduced impacts in the longer term. The continuing monitoring will also provide data to assist decision making on the need for improvements to the treatment processes, which would reduce the load of contaminants still being discharged into Sydney's marine environment.

Acknowledgments: The permission of the Environment Protection Authority (EPA) to present the material in this review is acknowledged. The work of EPA project teams and contractors, in carrying through the projects represented here, is gratefully acknowledged.

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This technical paper by N. Philip was reviewed by Dr D. W. Connell & Professor J. Middleton.

Occurrence and effects of petroleum hydrocarbons on Australia's marine environment

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Sources and composition of petroleum and associated products

Both crude and refined petroleum usually consist of hundreds of chemical substances. Common petroleum products together with their basic characteristics are shown in Table 1. Chemically the components of crude petroleum can be divided into two classes: alkanes and aromatic hydrocarbons. The aromatic hydrocarbons include the environmentally suspect polycyclic aromatic hydrocarbons (PAHs) which occur in relatively low concentrations in most petroleum substances. Refined petroleum products can contain the alkenes which are prepared synthetically and are somewhat similar to the alkanes in environmental properties.

All of these products are transported in Australia or manufactured in Australian petrochemical plants. In addition crude petroleum is shipped into Australia and around the Australian coast. Marine oil fields operate in Bass Strait and on the North West shelf at present with others being developed in other marine areas. Most of these activities have generated discharges during production processes as well as accidental spills. Petroleum substances also occur in relatively low concentrations in sewage (Connell 1974) and urban run-off, but the total amount discharged is relatively high due to the large volumes involved.

Environmental properties of petroleum and associated products

Properties which influence the behaviour of petroleum compounds in the environment are listed in Tables 1 and 2. The natural gases and petrol have relatively low boiling points and evaporate readily from the surface of water. As a general rule they evaporate within 24 hours. The other petroleum products exhibit increasing persistence in the environment with increasing number of carbon atoms and boiling point. At the extreme end of the range the asphalts and residual oils exhibit long-term persistence, usually over several decades.

Some physicochemical and biological properties of typical representatives of the alkane and aromatic hydrocarbon groups are shown in Table 2. All members exhibit relatively low solubility in water with benzene (a common petrol component) being the most soluble at 1780 mg/L. The octanol/water partition coefficient (K_{OW}) is an important property reflecting the lipid solubility of a substance. The most biologically active substances, in terms of toxicity and bio-accumulation, have log KOW values between 2 and 6, and are referred to as lipophilic compounds. Many hydrocarbons fall into this group as illustrated by the data in Table 2.

As a group the alkanes have very low toxicity which is reflected by the low toxicity of n-hexane and n-decane recorded in Table 2. On the other hand the aromatic compounds (benzene, naphthalene and benzo(a)pyrene) are described as 'relatively toxic' to aquatic organisms. Persistence in biota is low for the n-alkanes but higher for the aromatic hydrocarbons,

Table 1: Characteristics of some petroleum products

Petroleum Product	Hydrocarbon Types Present	Boiling Point Range (°C)	Number of Carbon Atoms
Natural gas	Alkanes	<20	1-6
Petrol	Alkanes and Aromatics	20-200	4-12
Kerosene, Jet fuel and Diesel	Alkanes and Aromatics	185-345	10-20
Lubricating oil	Alkanes	345-540	18-45
Asphalt and Residual oils	Complex Aromatic	>540	>40

Table 2: Physicochemical and biological properties of some typical petroleum hydrocarbons[1]

Compound	Aqueous Solubility (mg/L)	Octanol-Water Partition Coefficient (log K_{OW})	Toxicity (mg/L)	Persistences (half-life in days)
n-hexane	9.5	813 (2.91)	> 100 (fish)[2]	ca 4 (mussels)
n-decane	0.004	3.8×10^{5} (5.58)	> 10 (mussel larvae)[3]	ca 4 (mussels)
n-tetradecane	2.82×10^{-4}	1.6×10^{7} (7.20)	Not known	ca 4 (mussels)
benzene[4]	1780	135 (2.13)	5.8-46 (several fish species)[2]	Not known
naphthalene[4]	32	2290 (3.36)	1.24-150 (fish)[2]	1.5 (mussels)
benzo(a)pyrene[4]	0.0037	1.1×10^{6} (6.04)	> 1 (marine worms)[2]	16 (mussels)

1 Data from Verschueren (1983)
2 LC_{50} (96 hrs)
3 EC_{50}
4 Aromatic substances. All others are alkanes.

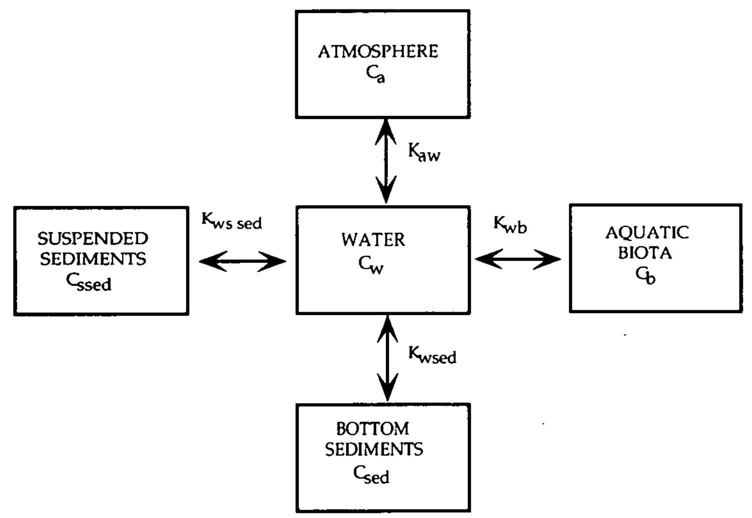

Figure 1: Distribution patterns of a chemical in different phases in the environment. The equilibrium constants (K values) are shown between phases and concentrations in a phase at equilibrium (C values) indicated.

particularly the PAHs (e.g. benzo(a)pyrene, Table 2). It is important to note that many of the aromatic hydrocarbons, particularly the PAHs, are carcinogens which have been implicated in many a wide range of human health problems and also disease problems with aquatic organisms (Grimmer 1983).

Another important environmental property of the PAHs is that they have strong bio-concentration capacities in aquatic organisms. For example naphthalene, the simplest PAH, has a bio-concentration factor (concentration in fish:concentration in water) of 426, while for benzo(a)anthracene it is 10 000 and for pyrene, 2690 (Connell 1990; Connell & Schüürmann 1988). Also, the PAHs and other hydrocarbons have strong affinities with bottom sediments, dependent on the amount of organic matter present. Gravels and coarse sands have little capacity to take up hydrocarbons whereas silts and muds with a high organic matter content have a strong capacity to take up lipophilic hydrocarbons. This is reflected in the high K_{OC} values (concentration in organic carbon containing matter in sediment:concentration in water) which range up to about 10^{6}.

In the discussion above the environmental properties of petroleum are discussed in terms of the individual substances. Petroleum products occur as complex mixtures of individual substances. The environmental properties of petroleum products can be inferred from the properties of the individual substances but these may be modified to some extent by the presence of other substances.

Levels of petroleum hydrocarbons in the marine environment

The marine environment can be considered to consist of several phases, as illustrated in Figure 1, and when a lipophilic chemical is discharged to the marine environment it distributes into these phases. In general terms the sediments and biota have a relatively strong affinity for lipophilic hydrocarbons, and concentrations in these phases would be expected to be relatively high. On the other hand concentrations in water would be expected to be low, due principally to the very low water solubility of lipophilic hydrocarbons (Kayal & Connell 1990; Connell & Hawker 1986).

An outline of the concentrations of petroleum hydrocarbons reported in Australian waters and sediments is shown in Table 3. It is noteworthy that, as expected, the concentrations in water are

Table 3: Occurrence of petroleum hydrocarbons in Australian waters and sediments

WATERS Location	**Concentration [µg/L] and (Hydrocarbon Type)**	**Reference**
Brisbane River	0.10-0.13 (PAHs)	Kayal and Connell (1989)
	0.13-0.28 (PAHs)	Smith et al. (1991)
Western Port	<0.1-7.1 (petroleum)	Burns and Smith (1980)
Parramatta River	0.17-0.41 (PAHs)	Smith et al. (1991)
Port Phillip Bay	0.2-22.6 (petroleum)	Burns and Smith (1981)
	0.25-0.70 (total hydrocarbons)	Murray et al. (1988)
Yarra River	0.05-0.41 (PAHs)	Smith et al. (1991)
Great Barrier Reef	0.29 (petroleum)	Coates et al. (1986)
SEDIMENTS Location	**Concentration [mg/kg, unless noted] and (Hydrocarbon Type)**	
Brisbane River	3.9-16.1 (dry wt.) (PAHs)	Kayal (1991); Kayal and Connell (1989)
Western Port	2.3-5,271 (dry wt.) (total hydrocarbons)	Burns and Smith (1977)
Parramatta River	0.1-13.6% (grease)	Furzer (1975)
Yarra River estuary	0.12-10.9 (PAHs)	Bagg et al. (1981)
	10.81 (PAHs)	Maher et al. (1979)
Yarra River/Hudson's Bay - Port Phillip Bay	43-955 (petroleum hydrocarbons)	Burns and Smith (1982)
Corio/Geelong-Port Phillip Bay/Corio Bay	6-1516 (petroleum hydrocarbons)	Burns and Smith (1982)
	0.49-3.0 (PAHs)	Bagg et al. (1981)
Mallacouta Inlet	0.80-0.11 (PAHs)	Bagg et al. (1981)
	0.80 (PAHs)	Maher et al. (1979)
Rowley Shelf, WA	0.015-0.05 (dry wt.) (alkanes)	Pendoley (1992)
Great Barrier Reef	0.2-0.8 (dry wt.) (hydrocarbons)	Coates et al. (1986)

Table 4: Occurrence of petroleum hydrocarbons in Australian marine biota.

Biota	Location	Hydrocarbon Type	Concentration (mg/kg)	Reference
Seabirds	Brisbane River	petroleum hydrocarbons (unresolved complex mixture)	up to 1038	Miller and Connell (1980)
Fish	South Qld	kerosene	up to 270	Connell (1974)
Fish	Great Barrier Reef	hydrocarbons	up to 0.3	Coates et al. (1986)
Corals	Great Barrier Reef	hydrocarbons	0.06-3.1 (lipid wt.)	Coates et al. (1986)
Clams	Great Barrier Reef	hydrocarbons	0.06-0.1 (lipid wt.)	Coates et al. (1986)
Mussels	Western Port and Port Phillip Bay	petroleum	up to 4.4 (lipid wt.)	Burns and Smith (1982)
Oysters	Rowley Shelf (WA)	petroleum	up to 4.9 (lipid wt.)	Pendoley (1992)

very low compared with those in sediments. As a general rule less than about 1 mg L^{-1} total hydrocarbons in water, as are observed in the Great Barrier Reef (GBR), represent background levels. Port Phillip Bay and Western Port exhibit background levels but with some zones of contamination. PAHs are natural components of the environment and occur in trace concentrations in rivers. The concentrations in rivers reported in Table 3 represent levels which occur in these waters due to urban and sewage discharges and also there are sometimes petrochemical industry activities in the vicinity. Similar general observations to those on waters can be made regarding the sediments. However Rowley Shelf in the north-west of WA exhibits a low level of petroleum contamination in a region where urban and sewage discharges are very low: such contamination as occurs is believed to be associated with the various activities associated with the extraction of petrochemicals in the area.

The observed occurrence of petroleum hydrocarbons in biota in Australian waters is shown in Table 4. Background concentrations are difficult to define since many animals and plants produce hydrocarbons which may be difficult to distinguish from petroleum hydrocarbons. The levels in GBR biota probably represent background levels, with corals exhibiting the highest concentrations of up to 3.1 mg/kg (lipid weight). Clearly the fish and birds from south Qld were heavily contaminated, with levels up to 270 and 1038 mg/kg, respectively. Western Port and Port Phillip Bay contain zones where biota are effectively free of contamination as well as zones where biota are contaminated . Rowley Shelf (WA) biota, as with the sediments, seems to exhibit a low level of petroleum contamination.

Environmental impacts

Oil spills of various sizes occur periodically in the Australian marine environment. Fortunately most of these have been on a small scale or have occurred in circumstances resulting in limited damage to the marine environment. For example, incidents of two relatively large spills from vessels, the *Ocean Grandeur* (Torres Strait) and *Kirki* (WA), occurred under conditions which resulted in natural dispersal of the spilled oil to the open sea. Nevertheless these larger incidents have the potential to cause immense damage, particularly to intertidal and subtidal ecosystems such as coral reefs, mangroves, seagrass communities and so on. Additionally, major spills at sea may have less obvious but serious long-term consequences for marine communities, such as detrimental effects on planktonic phases of marine organisms.

There are several reported investigations of the effects on mangrove communities in Botany Bay of smaller scale oil spills originating from oil handling facilities (Allaway et al. 1985; Anink et al. 1985; Allaway 1982). Seedling mortalities, defoliation of lower zones of trees and shrubs, mortalities of invertebrates and other adverse effects are described over areas of up to 73 hectares. These spill investigations are in accord with several controlled experiments demonstrating the toxicity of crude oils to Australian mangroves (McGuinness 1990; Wardrop et al. 1987).

Although oil spills of a relatively minor size occur frequently in GBR waters, there is a lack of any systematic investigation of their effects on coral reef ecosystems (Craik 1991). Similar oil spills occur generally throughout the Australian marine environment but there are few scientific reports on chronic or acute ecological impacts. Birds and other animals utilising surface waters, such as seals, are vulnerable to spilled oil and reports of oiled birds occasionally appear in the media (Anon 1992). In addition the toxicity of various oil spill dispersants to Australian marine biota has been demonstrated (Wardrop et al. 1987; McManus & Connell 1972). Thorhaug (1992) has recently reviewed the impact of oil spills and clean-up procedures on selected marine communities in international waters, and Miller (1982) has reviewed the lethal and sublethal effects of petroleum hydrocarbons in the marine environment.

The occurrence of different concentrations of petroleum in the marine environment was reviewed in the previous section. Adverse effects can result from the occurrence of petroleum substances in seafood. In many instances both in Australia and overseas, contamination has resulted in 'tainting', rendering the seafood unacceptable to consumers. A well known Australian example of tainting of fish, with resultant economic damage to the fishery comes, from southern Qld (Connell 1979, 1978, 1975, 1974; Shipton et al. 1970). In the 1960s sea mullet became contaminated in the Brisbane River and subsequently moved along the coast on their spawning 'run', causing tainted fish to occur along a considerable length of coast. The source of hydrocarbon contamination was an untreated sewage discharge, but the introduction of sewage treatment has essentially eliminated this problem. Any seafood listed in Table 4 as containing petroleum hydrocarbons has the potential to become tainted if concentrations of the causative substances reach high enough levels (Connell & Miller 1981a).

Petroleum hydrocarbons causing tainting usually contain a reasonable proportion of PAHs and other aromatic hydrocarbons. Many of those substances are believed to be human carcinogens (Grimmer 1983). However the dose received by consumers is usually low since contaminated seafood comprise only a small proportion of the diet of most populations (Connell & Miller 1981b).

Several responses can be expected from organisms exposed to sublethal levels of petroleum hydrocarbons. For example Chapman et al. (1988) found that gastropods exhibited reduced activity on exposure to sublethal concentrations of diesel. A range of detrimental physiological responses are possible, resulting in histopathological effects such as abnormal growth, occurrence of tumours and so on. These have not had extensive evaluation in Australian waters. Effects on larval stages of marine species may be significant: however the necessary systematic studies have not been conducted.

Indicators for monitoring environmental impacts

A range of chemical, histopathological and ecological indicators can be used to evaluate the effects of petroleum in Australian waters. The most valuable chemical indicator is the occurrence of petroleum in sediments. Water is not a satisfactory medium for monitoring, since concentrations of petroleum are very low and occurrence can be changed by weather conditions and seasonal changes. On the other hand sediments exhibit relatively high concentrations and are not affected by the factors mentioned previously.

Current results suggest that in Australian waters petroleum contamination comes from urban areas and areas where petrochemical industries operate. A national program of monitoring sediments in selected areas would be appropriate on a geographic and periodic time scale which relates to potential changes in petroleum status. In areas where elevated concentrations were detected histopathological investigations should be instituted. Should this indicate significant effects, an ecological program monitoring population, community and ecosystem effects would be appropriate.

In this work it would also be valuable to establish background information against which changes in the parameters mentioned could be evaluated. Levels of petroleum hydrocarbons and related histopathological effects which occur in a representative locations relatively free from petroleum-related activities would provide a suitable background baseline.

A publicly available national marine environment database to record this information would enhance the value of the data. It could then be used to determine management strategies in affected areas as well for the development of discharge standards and criteria.

Existing and desirable management strategies

A continued effort is needed to manage petroleum spillages and reduce them to the minimum level practicable. Specific attention is required to ensure discharge levels of petroleum from sewerage plants and industrial operations are specified in licence conditions and kept to a minimum. Little attention has been paid to urban run-off as a source of petroleum and other pollutants in Australian cities. There should be encouragement for local governments to introduce programs to improve the quality of urban run-off. While conducting these programs there is a need for the development of more soundly based standards and criteria to protect the marine environment from damage.

Summary review

Petroleum spillages occur periodically in the Australian marine environment. The adverse effects of these are difficult to evaluate, but available observations indicate limited damage to mangrove ecosystems and seabirds has occurred. However spills, sewage and urban run-off contribute sublethal levels of petroleum substances to the marine environment. Elevated concentrations of petroleum occur in water, sediment and biota adjacent to urban locations or in the vicinity of petrochemical industries but are at very low levels elsewhere. These sublethal levels have had a number of adverse effects, including tainting of seafood, and have the potential to cause detrimental histopathological changes to organisms. There is a need to establish the background occurrence and effects of petroleum in some representative locations and to monitor vulnerable areas. More control and management of petroleum in sewage, industrial discharges and urban run-off is needed.

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The problem of chlorinated compounds in Australia's marine environment

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Summary

A range of organochlorine compounds, including herbicides, insecticides, fungicides and polychlorinated biphenyls (PCBs) have been used in Australia. In addition, other compounds, including the dioxins and dibenzofurans, have been produced as a result of chlorination processes, or by combustion. Despite the fact that such organochlorines are recognised internationally as important contaminants in marine environments, few well designed studies have been implemented in Australia to elucidate local occurrence and distribution. Those which have been performed suffer from several problems, including the lack of adequate definition of monitoring objectives. At present, it is impossible to determine whether organochlorines are significant contaminants in Australian coastal waters, but the best evidence suggests that, as is the situation in Northern Hemisphere countries, organochlorines occur in highest concentrations close to urban and industrialised centres, or where sewage discharge and run-off from rural areas has a major influence. In order to develop and maintain an adequate level of environmental protection, there remains a need in Australia to embrace nationwide monitoring programs for the continued surveillance of organochlorine compounds, and to develop an analytical capability sufficient to allow the quantification of dioxins and dibenzofurans. Techniques for assessing the chronic toxicity of organochlorines in local waters also need to be developed.

Organochlorine contaminants: a brief overview

During the past two decades, the presence of organochlorine contaminants in the oceans of the world has caused considerable concern. The term 'organochlorines' is the name applied to a group of organic compounds which contain chlorine. Most of these compounds are synthetic, and during the last 50 years some 60 000 different organochlorines of industrial significance have been manufactured (ANZEC 1991). Organochlorines have achieved wide usage in Australia, especially as insecticides (e.g. DDT, lindane, chlordane, dieldrin, aldrin and heptachlor), fungicides (e.g. hexachlorobenzene and the chlorinated phenols such as pentachlorophenol), and herbicides (e.g. 2,4-D and 2,4,5-T). In addition to the pesticides, polychlorinated biphenyls (PCBs) were used extensively as dielectric, or insulating fluids in large transformers and capacitors, and as additives in hydraulic fluids, surface coating materials, plastics and lubricants. Organochlorine compounds such as the dioxins and dibenzofurans have also been produced unwittingly as by-products of chemical or combustion processes.

As well as the contaminants mentioned above, it is estimated that between 500 and 1000 new organic substances have been introduced worldwide each year, but their fate in the environment is largely unknown. As an example of this, concentrations of so-called 'bound chlorine' in fish fat (a representation of total organochlorine contamination) range from 30 to 200 parts per million (ppm), of which only 5-10 ppm can be attributed to the well recognised and researched contaminants such as DDT, PCBs, dioxins and chlorophenols (GESAMP 1990). The possible sources of the remainder are thought to be the result of industry-related activities, including chlorination processes (e.g. as a part of sewage treatment or pulp mill bleaching operations), metal smelting, and the burning of chlorine-containing compounds.

The key properties of organochlorines which cause concern are persistence and toxicity. Many organochlorines, such as the pesticides, were deliberately produced because of their toxicity; the fact that they were also persistent had advantages in that they remained effective against their targets for prolonged periods. Other compounds (such as the PCBs) were used in industry because of their stability, and were later found to produce toxic responses in many organisms.

Persistence combined with toxicity implies that organochlorines are a long-term problem when they enter marine ecosystems, and this concern is compounded by the ability of organochlorines to accumulate in the tissues of living organisms (Phillips 1993). This phenomenon, known as 'bio-accumulation', may seem anomalous given that organochlorines (as a general rule) are poorly soluble in water. However, organochlorines are also very soluble in fats, including those which are found in the tissues of living organisms. Thus, relatively small amounts of organochlorines present in water may be preferentially transferred and accumulated in the fats of aquatic plants or animals, and the resulting concentrations may be as much as 500 000 times or more than in surrounding waters.

Living organisms may also accumulate organochlorines through their food via a process termed 'biomagnification'. This is a stepwise process, in which persistent contaminants such as organochlorines are transferred through food chains or food webs as successive organisms feed upon each other. As a result, concentrations of organochlorines may increase with trophic levels, the highest concentrations being observed in higher consumers including certain fish, marine mammals, birds, or humans. There are some doubts that this process contributes significantly to the accumulation of organochlorines by all living organisms, but it may be a significant factor at the highest trophic levels (Phillips 1993).

Sources of organochlorines

The sources of organochlorines in the Australian marine environment have recently been reviewed by ANZEC (1991). Organochlorines enter marine systems from a variety of sources, which can be broadly defined as 'point sources' (arising from a single source or location) and 'diffuse sources', which are more widespread and hence more difficult to accurately define. Examples of point sources include sewage or factory discharges via pipelines, or rivers and streams discharging directly to the ocean. Point sources are relatively easy to recognise and manage, but measuring organochlorines in them is often difficult as the substances are poorly water soluble and hence difficult to detect using routine chemical techniques. Nonetheless, these low concentrations are often of toxicological significance, due to bio-accumulation. Diffuse sources include atmospheric fallout, run-off from land, and ground water leaching. Because of their widespread nature, diffuse sources of organochlorines are also difficult to measure, and are much more difficult to control than point sources.

In recent years, the use of many organochlorines in Australia has been limited, due to concerns over their persistence and toxicity. Table 1 summarises the situation of the commonest organochlorines which have been or are still being used in Australia. It can be seen from this table that the use and/or disposal of these substances has directly contributed to their widespread presence in the Australian environment.

Unlike the pesticides and PCBs, dioxins and their close chemical relatives the dibenzofurans are produced unintentionally during the manufacture of other organochlorines, during the chlorination of waste materials, or by combustion processes. Although there are many different dioxins and dibenzofurans, most concern has been directed towards 2,3,7,8-tetrachlorodibenzodioxin (2,3,7,8-TCDD) and 2,3,7,8-tetrachlorodibenzofurans (2,3,7,8-TCDF), which are considered to be highly toxic. It should be noted, however, that dioxins and dibenzofurans are usually present in environmental samples at extremely low concentrations (in terms of parts per trillion or less), and that 2,3,7,8-TCDD and 2,3,7,8-TCDF usually make up only a small fraction of the total dioxins present.

The principle sources of dioxins and dibenzofurans in Australia have been reviewed by ANZEC (1991). Chemical processes producing the substances include pulp and paper mills where chlorine is used as a bleaching agent, and the manufacture of such substances as pentachlorophenols and 2,4,5-T (the latter has now ceased in Australia, although considerable waste remains to be disposed). Other industrial processes are also possible sources, and these include copper smelting, magnesium and nickel production, vinyl chloride production (or the use of recycled polyvinyl chloride in steel mills), and the regeneration of used petrochemical plant catalysts.

Combustion processes are also known to produce significant amounts of dioxins and dibenzofurans. For example, in Canada and Sweden incineration of wastes (especially

Table 1: Summary of past and present uses of organochlorines (after ANZEC 1991)

ORGANOCHLORINE	USES
Insecticides	
DDT, Endrin, HCB	No permitted uses
BHCs other than Lindane	No permitted uses
Lindane (-BHC)	Banned in WA. Withdrawn in all states for use in control of insect pests in stored seeds. Used to control white grubs & symphylids in pineapples (Qld), ectoparasites on food (all states except NSW, Tas. and WA), and head lice on humans (except in Vic. and WA).
Chlordane	Quarantine purposes (Vic.). Control of termites in all states except Tas.
Heptachlor	Control of funnel ant in some cane growing areas of Qld, subject to permit. Control of termites in all states except Tas.
Aldrin	Control of termites in all states except Tas. Currently being phased out in SA and WA.
Dieldrin	Soldier Fly control in cane growing areas of Qld, subject to permit. Control of termites in all states except Vic., WA and Tas. Generally no longer available commercially for termite control.
Herbicides	
2,4-D	Used to control broad-leaved weeds in crops, water weeds and vegetation near drains, and in domestic gardens.
2,4,5-T	Used in the past as a herbicide against broad-leaved woody plants, and as a defoliant. Dioxins found present as a contaminant in some commercial preparations. Use and manufacture in Australia has ceased.
Fungicides	
Hexachlorobenzene (HCB)	Use well controlled in the past. Does not present problems experienced with widescale use in Northern Hemisphere. Now banned in Australia.
Chlorinated Phenols	Widely used in Australia. Readily degrade in aerobic environments. Formulations of pentachlorophenol contaminated by dioxins and bibenzofurans.
Polychlorinated Biphenyls (PCBs)	Never manufactured in Australia; imported for use. Use restricted from 1975 to 'closed system' uses where contact with the environment is unlikely.

municipal and industrial wastes) and the subsequent disposal of ash and sludge are the major sources of dioxins and dibenzofurans. In Sweden, the next most important sources are metals reclamation and pulp and paper mills. In the former West Germany, the major dioxin and dibenzofuran contribution to the environment arose from pentachlorophenol use, followed by municipal incinerators and emissions from motor vehicles running on leaded petrol (ANZEC 1991). In Australia, the various sources of dioxins and dibenzofurans have not been quantified, although it is unlikely that municipal incineration plays a key role as this process does not occur in Australia to the same extent as in Europe and Scandinavia.

Sewage discharge has been recognised as a potential source of organochlorine compounds to the Australian marine environment (Portmann 1974). Organochlorine compounds in sewage may arise from industrial or domestic sources, or new compounds may be formed in the sewage as the result of disinfection processes utilising chlorine. The latter processes give rise to many different (and largely unknown) organochlorine compounds, which are poorly understood both chemically and toxicologically (Koppermann et al. 1975). Similarly, chlorination processes in the pulp and paper industry give rise to a suite of organochlorine compounds (apart from dioxins and dibenzofurans) which are poorly understood. Such compounds are often measured in total as AOX, or 'adsorbable organic halide'. About 300 of the AOX components have been chemically characterised, but the higher molecular weight compounds are still not well understood. In Australia, pulp mill effluents are subject to primary and secondary treatment prior to discharge, but the fate of many of the chlorinated components upon passage through such processes has not been established, and it is possible that these compounds are being discharged to the marine environment. Process improvements, however, have lead to considerably less AOXs being formed during pulp and paper production.

Organochlorine concentrations in the marine environment

The concentrations of organochlorine residues in Australian marine waters, sediments and biota have been extensively reviewed by a number of authors (Connell 1981; Richardson & Waid 1982; Richardson et al. 1987; Farrugia 1986; Olsen 1988; Brodie 1989; Lincoln Smith & Mann 1989a, 1989b; ANZEC 1991; Thompson et al. 1992; Phillips et al. 1992). What is clear from these reviews is the lack of an integrated perspective on organochlorines in Australian marine waters, as most investigations have been undertaken in localised areas surrounding potential 'hotspots'. Indeed, most work on organochlorines has been conducted near Australia's major eastern seaboard cities, and only few data exist for what can be considered as 'baseline' or relatively pristine environments (Martin & Richardson 1991).

Pesticides

It is surprising that, given Australia's past (and present) usage of organochlorine pesticides and the propensity for run-off from rural and urban usage, there have been few definitive monitoring programs conducted in coastal waters. Thus, in comparison with the northern hemisphere, little is known about the extent of local contamination.

Because of the low concentrations of organochlorine pesticides found in waters, most analyses have been conducted on sediments and biota. Such surveys have been done for two purposes:

1. to establish spatial and (to a far lesser extent) temporal differences in organochlorine distribution; and
2. to measure organochlorines in organisms consumed as food to establish if acceptable health limits have been exceeded.

These two objectives are frequently confused in monitoring programs, and interpreting data accumulated for the latter objective in terms of spatial and temporal distribution is often fraught with difficulties. Many of the food species tested are mobile, and may not necessarily indicate that high concentrations are the result of contamination in the capture locality. In addition, meaningful comparisons between different species of organisms, and between the different analytical techniques utilised by various workers, are difficult to draw. Nonetheless some inferences are possible, but these highlight the need for monitoring programs with firmly established aims and methodologies if data are to be obtained which accurately indicate the extent of organochlorine contamination in Australian coastal waters.

Work to date indicates that although organochlorine pesticides can be found in Australia's marine organisms, concentrations are relatively low, except where discharges arise from urbanised areas or as a result of run-off from intensively-farmed rural areas. Thus, Olafson (1978) and Smillie and Waid (1984) were able to report low concentrations of organochlorine pesticides (DDT, DDE and lindane) in corals, fish and molluscs obtained from the Great Barrier Reef. However, studies undertaken in Moreton Bay, which receives discharge from the Brisbane River, indicate considerably higher concentrations of organochlorines, including DDT and its metabolites (ANZEC 1991).

In South Australia, HCB, lindane, dieldrin and DDTs were measured in 79 fish samples during the 1970s (ANZEC 1991). Only six specimens were pesticide free, and analyses indicated that maximum residue limits (MRLs) for dieldrin and total DDTs (the sum of DDT and its metabolites) were exceeded in some samples. A survey of organochlorine contamination in waters and sediments in Western Australia (ANZEC 1991; Thompson et al. 1992) indicated that river flushing following rainfall contributed relatively high loadings of chlordane, total DDTs, dieldrin, heptachlor and heptachlor epoxide, and that Environment Protection Authority (EPA) criteria for the maintenance and preservation of marine aquatic ecosystems were exceeded in 63% of waters sampled.

In Victoria, few studies on organochlorine pesticide contamination of coastal waters have been undertaken (Thompson et al. 1992). Measurement of DDTs in sediments from Port Phillip Bay and the Yarra River showed relatively low concentrations in comparison with international data (Murray 1987), although concentrations of these substances in Australian fur seals were at least as high as those reported in Northern Hemisphere species (Smillie & Waid 1987).

Most work on the organochlorine pesticides in Australian waters has been conducted around sewage outfalls in the Sydney area (see Thompson et al. 1992; Lincoln Smith & Mann 1989a, 1989b; ANZEC 1991). Fish caught off Sydney's sewage outfalls (in particular the Malabar outfall which contains the highest proportion of industrial waste) contained high concentrations of organochlorine pesticides. For example, red morwong (*Cheilodactylus fuscus*) flesh exceeded the MRLs for BHC and heptachlor epoxide by 122 times and 52 times, respectively. Chlordane and hexachlorobenzene concentrations were also found to exceed MRLs in some samples taken off the Sydney coast, and other organochlorine pesticides including dieldrin, oxychlordane and the DDTs were also detected. In 1989, fishing was prohibited within 500 m of the main outfalls. Further studies around the new deep water ocean outfalls (Mann & Ajani 1991) have indicated the presence of organochlorine pesticides in the flesh of rubberlip morwong (*Nemadactylus douglasii*), and 65% of the samples taken from 30m depth contained organochlorines higher than the National Health & Medical Research Council's MRLs [see NH&MRC (1988) for details of MRL standards applying to pesticides, agricultural chemicals and noxious substances in food].

Polychlorinated Biphenyls (PCBs)

The presence of PCBs in Australia's coastal environment has been reviewed by Richardson et al. (1987). The major regional study of PCBs in Australia was undertaken in Port Phillip Bay, Victoria during the mid to late 1970s (Richardson & Waid 1982; Richardson 1983). This survey utilised the U.S. 'Mussel Watch' approach, which uses mussels (e.g. *Mytilus edulis*) as 'sentinel' (or nonmotile, indicator) organisms. Bivalve shellfish (such as mussels and oysters) have been used as indicators of contamination in many studies worldwide. The successful use of bivalves is based upon the sedentary nature of the organisms and their ability to accumulate contaminants from the water column in proportion to that in surrounding waters.

In Port Phillip Bay, PCBs were found in all 87 samples of shellfish and in 27 sediment samples. Highest concentrations occurred near the densely populated and industrialised areas of Melbourne and Geelong, whilst lowest concentrations occurred offshore or near localities distant from urbanised areas. Little follow-up work has been done on PCBs in Victoria, but limited sampling (see Richardson et al. 1987; Phillips et al. 1992) has shown that PCB concentrations in Corio Bay, near the city of Geelong, have fallen markedly whilst those in Hobsons Bay near Melbourne have remained similar to those recorded in the late 1970s. The explanation for the fall in Corio Bay concentrations is believed to be related to the closure of an industrial tip in the locality (Richardson et al. 1987).

In other areas of Australia, few surveys of PCBs have been conducted. In Queensland, PCBs have been detected in fish samples from the Brisbane River (Shaw & Connell 1980), and in low concentrations in corals, fish and dugong from the Great Barrier Reef and the Gulf of Carpentaria (Smillie & Waid 1984). These latter data are representative of the low levels of contamination in remote areas. However, in the Sydney area PCBs have been detected in mullet (*Mugil cephalus*) caught in Port Jackson and Botany Bay (Woollard & Settle 1978; State Pollution Control Commission 1979), and in red morwong off Malabar. It is notable that more recent surveys of the Sydney sewage outfalls (Lincoln Smith & Mann 1989a) have recorded a continued decrease in PCB concentrations, presumable due to the restrictions placed on the use of PCBs in the mid 1970s and efforts to more appropriately dispose of the substances in recent years.

PCBs have been measured in Australian surface waters by Tanabe et al. (1982). Concentrations were highest in near-shore waters, but overall the levels measured were less than those in the Atlantic.

Dioxins and dibenzofurans

Although dioxins and dibenzofurans are recognised as significant waste problems in the Australian environment (see Thompson et al. 1992), few studies of these substances have been made in the marine environment. This is largely due to the lack of local analytical capability and the expensive nature of analyses, which require high resolution gas chromatography/mass spectrometry to detect extremely low concentrations. At present, the only published studies have been in urbanised areas, or associated with pulp and paper mills.

In Sydney, dioxins have been measured in Homebush Bay, adjacent to contaminated landfill sites. Dioxins (including 2,3,7,8-TCDD) were measured in fish and sediments, and as a result fishing has been banned in the area (Rubinstein & Wicklund 1991). In Melbourne, dioxins and dibenzofurans have been measured in association with the sewage system and in the sediments of Port Phillip Bay (Thompson et al. 1992; Phillips et al. 1992). In general, the more toxic forms of dioxins and dibenzofurans were present in low concentrations.

Low concentrations of 2,3,7,8-TCDD and its related dibenzofuran (2,3,7,8-TCDF) have been found in effluents to marine receiving waters from 3 of 4 pulp mills in Australia using chlorine bleaching processes (Thompson et al. 1992). Due to the risk of contamination associated with the pulp and paper industry, Australia is currently increasing its research capability with regard to dioxins and dibenzofurans through the National Pulp Mills Research Program, and a more definitive picture of the influence of this source of dioxins and dibenzofurans is expected in the future.

Other chlorinated contaminants

Little work has been undertaken to date on other, unknown chlorinated compounds in marine waters. Total AOXs are measured in pulp mill effluents and in certain sewage discharges, but at the moment few data are available for review, and no statement can be made about their status in the Australian marine environment. This is an area in which further research is urgently required in this country.

Environmental effects of organochlorine contaminants

Connell (1981) and ANZEC (1991) have summarised the effects of organochlorines in the marine environment. These include, for example, fish kills associated with waters near agricultural areas receiving organochlorine pesticide treatment; peregrine falcon eggshell thinning associated with DDT concentrations; sublethal effects on crabs as a result of DDT in surrounding waters; and possible synergistic effects of DDTs and PCBs on algae. It is notable from these reviews that there have been very few studies of the effects of organochlorines on Australian species, and it is fair to assume that this is more a reflection of a lack of research effort rather than a lack of contamination in local waters (ANZEC 1991).

Consideration of the effects of organochlorines in the Australian marine environment is mainly based on the results of overseas investigations, and local evidence of specific effects relies more on anecdotal evidence than hard, scientific fact. Studies in Australia have mostly defined the occurrence and distribution of organochlorines in marine biota rather than investigating possible effects. Furthermore, in contrast to the routine chronic toxicity testing (using local species) of effluents discharged to the marine environment in such countries as the United States, there remains a lack of specific tests which can be applied to the Australian marine environment. This is a situation which urgently requires attention, and should be the subject of an intensive research effort in this country (Martin & Richardson 1991).

With regard to the possible effects of persistent organochlorines on human health, ANZEC (1991) notes that the majority of fish eaten by Australians is taken from waters which contain relatively low concentrations of organochlorine contaminants. This statement is based on the results of regular market basket surveys by the National Health and Medical Research Council (NH&MRC), which examine the concentrations of such organochlorine compounds as aldrin, BHC, DDT and its metabolites, dieldrin, heptachlor, HCB, lindane and PCBs in foods including fish. Of these contaminants, only DDE and PCBs were detected in locally caught or imported fish samples, and dietary intakes have been estimated to be below the Food and Agriculture Organisation and World Health Organisation acceptable daily intakes.

Monitoring programs related to organochlorines

The relative paucity of data relating to organochlorines in the Australian marine environment is a direct result of the lack of monitoring activities which have been undertaken to date. Of particular note is the lack of consistent nationwide data upon which a definitive statement may be made regarding the status of organochlorine contamination. Such data are unlikely to be based on analyses of waters alone, given the analytical problems associated with the very low concentrations usually encountered in waters. Rather, a network of sites at which bio-accumulator organisms (e.g. bivalve shellfish) are analysed seems a more reasonable alternative, despite the fact that different organisms may have to be used to cover the range of environments found around the Australian coastline. A monitoring network of this nature has been suggested to the Federal Government by Bremner and Richardson (1986).

Martin and Richardson (1991) have recently reviewed marine contaminant monitoring practices in Australia, relating them to those in California. They note that there are many similarities between the two localities, and that monitoring activities in both California and Australia have suffered (to a greater or lesser extent) from the same deficiencies. These include:

insufficient numbers of study sites, and thus a poor data base with which to understand spatial differences in the distribution of contaminants. This is particularly true for pristine sites, which tend to be neglected in the rush to characterise pollutant 'hotspots';

insufficient repeated monitoring at fixed sites, thus degrading the overall data set and resulting in an inability to detect changes over time;

no analyses of certain contaminants, especially dioxins and dibenzofurans and those compounds resulting from chlorination processes;

inadequate commitment to coordination and funding of monitoring programs , and changes in political directions which have lead to a lack of ability to study nationwide status and trends;

lack of firm and consistent regulations relating to organochlorine contamination, developed on a nationwide basis;

a reluctance to rationalise existing monitoring programs so that consistency can be obtained within individual States or throughout the country;

inadequate opportunities for related research, which has degraded efforts to develop chronic toxicity tests specific to local environments.

Martin and Richardson (1991) note that current practices of marine contaminant monitoring in many parts of Australia are in need of a thorough review, and that new programs with well defined aims and objectives need to be developed. Perhaps most importantly, a commitment needs to be made to these programs such that they deliver long-term, reliable information upon which future policy in Australia can be established or refined.

Management of organochlorines in Australia

Thompson et al. (1992) have reviewed the current management practices and legislative requirements relating to hazardous wastes (including organochlorines) in Australia. They note that the Australian regulation of chemicals and hazardous wastes is extremely complex due mainly to:

the States and Territories taking primary responsibility for these matters, resulting in a lack of uniformity nationwide;

the introduction of legislation on a 'piecemeal' basis in reaction to specific State problems;

regulation of organochlorines on a 'sectoral' basis, with different legislation (and different government departments) relating to the protection of public, worker or farmers' health, consumers and the environment.

As an example of the multitude of legislation which confounds the regulation of organochlorines in Australia, Thompson et al. (1992) cite New South Wales, where 72 pieces of legislation administered by 19 departments relate to the control of toxic and hazardous chemicals. They discuss aspects of the Federal regulation of organochlorines in the areas of control of agricultural and veterinary chemicals, the control of industrial chemicals, the management of intractable wastes, the regulations surrounding sea dumping, and highlight areas where improvements are being made, including:

the *Agricultural and Veterinary Chemicals Act*, 1988, by which the Australian Agricultural and Veterinary Chemicals Council evaluates chemical products. Deficiencies in this Act (relating to public information and right-to-know provisions) are being addressed through legislation that transfers the registration of rural chemicals from the States to the Commonwealth (the National Registration Scheme);

the *National Industrial Chemicals (Notification and Assessment) Act*, 1989, which established the National Industrial Chemicals Notification and Assessment Scheme (NICNAS), aimed at protecting workers, the public and the environment from the harmful effects of industrial chemicals;

the Independent Panel on Intractable Waste, established in 1991, which examined alternative technologies for the disposal of intractable wastes; and the *Hazardous Waste (Regulation of Exports and Imports) Act*, 1989, which allows Australia to fulfil international agreements made under the Basel Convention on the Control of Transboundary Movements of Hazardous Wastes and their Disposal;

the *Environment Protection (Sea Dumping) Act, 1981*, which allows Australia to fulfil its international obligations under the London Dumping Convention.

Summary

Despite the fact that organochlorines are recognised internationally as important contaminants in marine environments, few well designed studies have been implemented in Australia to elucidate local occurrence and distribution. Those which have been performed suffer from several problems, including the lack of adequate definition of monitoring objectives. At present, it is impossible to determine whether organochlorines are significant contaminants in Australian coastal waters, but the best evidence suggests that, as is the situation in Northern Hemisphere countries (see GESAMP 1990), organochlorines occur in highest concentrations close to urban and industrialised centres, or where run-off from rural areas has a major influence. In order to develop and maintain an adequate level of environmental protection, there remains a need in Australia to embrace nationwide monitoring programs for the continued surveillance of organochlorine compounds, and to develop an analytical capability sufficient to allow the quantification of dioxins and dibenzofurans. Techniques for assessing the chronic toxicity of organochlorines in local waters also need to be developed.

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Heavy metals and tributyltin in Australian coastal and estuarine waters

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Introduction

Heavy metals are a major anthropogenic contaminant of estuarine and coastal waters. Their inputs include urban run-off, industrial effluents, mining operations and atmospheric depositions, and may be in particulate or dissolved forms. Although many are essential biological elements, all have the potential to be toxic to organisms above certain threshold concentrations, and for the protection of aquatic biota it is important that these limits not be exceeded in aquatic environments.

The oceans are often mistakenly seen as a boundless sink for heavy metals and other contaminants. In shallow near-shore waters where dispersion and dilution processes are less effective, the measured concentrations of heavy metals are generally noticeably higher than in open ocean waters. Australian coastal waters are generally characterised by currents which follow the coastline and are less readily able to transport contaminants to deeper waters. In deeper waters, metal concentrations are extremely low, and there are noticeable differences between metal concentrations in surface and bottom waters, the profiles often correlating with nutrient concentrations and organic particulate matter. In near-shore waters, however, such differences are less pronounced.

Although some metals such as molybdenum and uranium are highly conserved in marine waters, most are not, attaching to suspended particulates, and ultimately accumulating in bottom sediments. The sediment load of coastal waters is noticeably higher than open ocean waters. This is particularly so in the estuarine zone where, with increasing salinity the precipitation of iron hydrous oxides will scavenge and co-precipitate soluble metal species.

The bio-availability of both dissolved and particulate heavy metals is critically dependent on chemical form, and a great deal of research, much of it pioneered in Australia, has been devoted to methods which can detect specific species classes whose biological impacts can be predicted (Florence & Batley 1980). There are limited measurements of the speciation of metals in waters, but for the most part data are available only for total metal concentrations. These usually refer to the dissolved metal concentrations defined by filtration through a 0.45 µm membrane filter.

Sources

Heavy metals

A number of texts have discussed the sources of heavy metals to estuaries and coastal waters (Furness & Rainbow 1990; Salomons & Forstner 1984; Forstner & Wittman 1979). In the northern hemisphere, the higher population densities and greater urbanisation and industrialisation have led to quite high contamination of coastal waters by metals. In Australia, potential problems are restricted to less than a dozen major cities located on the coastal fringe. These include the state capitals, and a number of other highly industrialised cities, such as Wollongong and Newcastle.

Because most heavy metals tend to accumulate in sediments, their presence in the water column is usually the result of recent inputs. Metal concentrations can vary significantly over short distances and as a function of tide. Single measurements at a given site may indicate contamination, but to fully assess its magnitude a more complex sampling program which considers spatial and temporal variability is required. For this, integrating samplers, which continuously preconcentrate metals from solution over periods from hours to days, have been used.

Major estuarine contamination is generally directly traceable to an industrial (or mining) source, although urban run-off can be responsible for an increase in metal concentrations above background open ocean values. In Australia, industrial and mining sources could include smelters, power stations, paper mills, port facilities including ore or coal loaders, sewage treatment works, oil refineries and other chemical and industrial manufacturing plants.

In the late 1970s, following international disasters involving mercury and cadmium, heavy metals were identified as the major pollution threat facing mankind. At that time the Derwent River in Tasmania was severely affected by discharges of metallurgical waste, partially treated sewage, and effluent from a pulp mill operation and a chlor-alkali plant (Bloom & Ayling 1977), and was described as 'one of the most polluted areas of the world' (Forstner & Wittman 1979). The past twenty years have seen a worldwide response to this threat. In Australia too, stricter controls on aqueous discharges have been progressively imposed by state and federal environmental agencies (ANZECC 1992). These have succeeded in dramatically reducing the dissolved concentrations of heavy metals in estuarine waters. Evidence for this change is available from routine monitoring data held by these agencies. More often these relate to concentrations at the point of discharge rather than the dispersed concentrations in the estuary, because firstly this is the point of regulation, and secondly the concentrations are demonstrably higher and therefore analyses are easier and more reliable. Sediments equally reflect the improvement in discharge controls, with the concentrations of heavy metals in uppermost layers being lower than those in deeper sediments. Examples of this have been found for Lake Illawarra and Lake Macquarie (NSW), as well as in offshore sediments.

Tributyltin

Tributyltin (TBT) has been used in Australia as an active ingredient in marine antifouling paints since the early 1970s. Modern copolymer paint formulations have an initial high leach rate of TBT which, within days, reaches a standard and constant value of around 4 µg TBT cm^{-2} day^{-1}. The half-life of TBT in seawater is around six hours, but it rapidly partitions either to suspended sediments or to the surface microlayer. In sediments its half-life has been estimated at around 3.5 years (Batley et al. 1992). Dissolved TBT has adverse impacts on oysters (and to a lesser extent other bivalves), with tissue growth diminished at the expense of shell, leading in some species to shell thickening or curling. These impacts led to what is effectively a worldwide ban on the use of TBT in paints on ships below 25 m in length. In Australia, the ban took effect in Australia around 1988. TBT is however still used in paints for larger ocean-going vessels, and can therefore be detected in the waters of most major harbours, and especially in sediments in the vicinity of dockyards.

Butyltins from sources other than marine paints can also enter the water column. Dibutyltin is used as a catalyst in the plastics industry. TBT is used as an algaecide in boiler water cooling circuits. Dibutyltin is the primary degradation product of TBT, but has a comparatively minor impact on the environment. Other alkyltins have been used in pesticide formulations, but none have been detected in coastal waters.

Analysis

Heavy metals

There have been many studies of heavy metals in Australia's marine waters over recent years and in reviewing these data it is important to recognise the problems associated with the sampling and analysis of waters for metals at natural concentrations (Batley 1989). In many instances the significance of the data is questionable, usually because inadequate precautions were taken to avoid contamination during sampling.

Recent studies (Ahlers et al. 1990; Hunter & Tyler 1987) have reinforced the need for extreme care in the sampling of marine waters for trace metal analysis. The earliest concerns were raised by Patterson & Settle (1976) and Schaule & Patterson (1981). More recent analyses of open ocean water samples in which special precautions were taken in preparation of sample containers, sampling operation, and sample handling and analysis, have shown metal concentrations to be lower than originally anticipated. For example, open ocean concentrations of zinc, lead, copper, cadmium and chromium are now estimated to be in the ranges 0.003-0.6, 0.001-0.04, 0.03-0.4, 0.0001-0.12 and 0.1-0.3 µg L^{-1} respectively (Bruland et al. 1991; Bruland & Franks 1983; Bruland 1980).

Because of these low concentrations and the problems of contamination, sample handling and analysis must be carried out in a clean laboratory, especially designed so that airborne particles are absent. Unless these facilities have been used, the results of analyses can be considered of questionable validity.

Few analytical techniques are capable of detecting heavy metals at sub microgram per litre concentrations, principally because of interference due to matrix components. For a limited number of metals, anodic stripping voltammetry (ASV) permits direct analysis by in situ electrochemical pre concentration at a mercury electrode, followed by a voltage scan which sequentially dissolves or strips the reduced metals from the electrode, producing a current peak for each which is concentration dependent. Graphite furnace atomic absorption spectrometry (AAS) is the other commonly used technique. For most metals, preliminary extraction with a complexing agent is required to obtain sufficient pre concentration to enable detection of ambient concentrations. Rosman et al. (1982) and Rosman & De Laeter (1980) have successfully applied isotope dilution mass spectrometry to the detection of cadmium and other heavy metals at nanogram per litre concentrations in waters off Western Australia.

More recently, integrating samplers have been used to preconcentrate heavy metals. These pump large volumes of water (typically >10 L) through a column containing a chelating resin. The metals are eluted with acid and detected by a spectrometric technique such as graphite furnace AAS or inductively coupled plasma mass spectrometry (ICPMS). The method does not necessarily recover total metals, because not all species are dissociated on the resin, and incomplete recovery from the columns is a problem.

Tributyltin

Butyltins can be extracted from sediments, waters and biota using non-aqueous solvents preferably in the presence of a complexing agent, troplone. Reaction with sodium borohydride forms volatile hydrides which can be trapped on a chromatographic column, and separated by thermal desorption, with subsequent detection by quartz furnace atomic absorption spectrometry (Batley et al. 1989b). Alternatively, the hydrides can be solvent extracted and determined directly by gas chromatography using electron capture or flame photometric detection (Batley et al. 1989a). CSIRO's Centre for Advanced Analytical Chemistry in Sydney is the only laboratory in Australia currently experienced in the analysis of TBT in waters at baseline concentrations.

Heavy metals in estuarine and coastal waters

There are few Australian laboratories with the demonstrated ability to determine ultratrace metal concentrations in seawater. Consequently there are few reliable data available for heavy metals in Australian coastal waters. Sampling and analysis of open ocean waters is particularly demanding and is the special expertise of the CSIRO Division of Oceanography in Hobart.

Table 1: Dissolved heavy metal concentrations ($\mu g\ L^{-1}$) in some Australian estuarine waters

Estuary	Cu	Pb	Cd	Zn	Hg	As	Co	Ni	Cr	Reference
ANZECC Guidelines	5	5	2	50	0.1	50	-	15	50	ANZECC (1992)
Central Port Phillip Bay, Vic.	0.6	<0.8	<0.05	<2	<0.002	2.8	-	-	-	EPA (1991)
Corio Bay, Vic.	1.1	<0.8	0.2	<2	<0.002	3.2	-	-	-	EPA (1991)
Port Hacking Estuary, NSW	0.5	0.4	0.2	-	-	-	-	-	-	Batley and Gardner (1978)
North Lake Macquarie, NSW	1.5	1.6	1.9	5.2	-	-	-	-	-	Batley (1987)
South Lake Macquarie, NSW	1.2	0.1	0.2	1.0	-	-	-	-	-	Batley (1987)
Lake Munmorah, NSW	1.5	0.2	0.1	2.7	0.02	1.7	-	-	2.1	Batley and Brockbank, unpubl.
Port Augusta, S.A.	0.45	0.54	0.37	<10	-	-	0	-	-	Ferguson (1983)
Port Pirie, S.A. (offshore)	0.25	5.1	0.32	47	-	-	-	-	-	Ferguson (1983)
Macquarie Harbour, Tas.[a]	7	-	0.03	2.0	-	-	-	0.5	-	Carpenter et al. (1991)
Derwent River, Tas.	1.2 (7.3)	0.23 (4.6)	0.05 (0.51)	3.4 (40)	0.034 (3.83)	<6.0	0.030 (0.10)	0.27 (1.1)	-	Mackey, unpubl.
Ross River, Qld	0.24 (1.21)	0.11 (0.59)	0.07 (0.20)					0.3		Jones and Thomas (1988); Jones (1981)

a Data are the highest found. Numbers in parentheses are total metals

Metal concentrations are expected to be higher in estuarine waters, and these have naturally been the focus of local and state agencies. Many of the data for heavy metals in estuarine waters have been documented in internal reports of the monitoring agencies, and the resources of this investigation did not enable a detailed survey to be undertaken. Instead, a compilation of results for a number of key estuaries has been reported (Table 1) and evaluated in terms of recently formulated water quality standards (ANZECC 1992). Organisations who use the ANZECC water quality guidelines should also be familiar with the national and site-specific guidelines produced by the USEPA (Stephan et al. 1985). These guidelines provide insights into data base requirements for criteria derivation and relevant laboratory and field data, as well as the toxicological and ecological significance of a substance. In addition USEPA discuss strengths and limitations of criteria and provide guidance on their implication.

The data for estuaries in New South Wales, Victoria, South Australia, Tasmania and Queensland (Table 1) show that in most instances heavy metal concentrations in waters were below the recommended ANZECC guideline concentrations. In some cases measured concentrations are an order of magnitude above seawater concentrations. Notable examples of the latter are Port Pirie (SA), where the impact of effluents from the nearby lead-zinc smelter is clearly seen (Ferguson 1983), and the Lake Macquarie (NSW) sites.

When making comparisons it is important to note that these data are for dissolved metals only, i.e. filterable through a 0.45 µm membrane filter. Total metals, especially for estuarine waters having a high particulate load, can be appreciably higher, as shown in the examples from the Derwent River in Tasmania and the Ross Estuary in Queensland. Considerable emphasis has been given to the determination of the chemical forms (speciation) of metals in waters (Batley 1989; Florence & Batley 1980), since in assessing biological impact it is important to recognise that not all forms of dissolved metals are bio-available. In coastal waters this is generally of little consequence since the concentrations are well below recommended water quality standards.

The trace metal data reported in Tables 1 and 2, are, in many instances, obtained from single samplings. However, concentrations can be quite variable in estuarine and coastal waters, both as a result of changing inputs, or from the temporal effects of biological, chemical and physical interactions. For example, the presence of the blue-green alga *Trichodesmium* in the Ross River estuary in late winter and spring when the estuary was hypersaline, was accompanied by an increase in copper, lead and cadmium (Jones 1992; Jones & Thomas 1988). At close inshore locations, available copper concentrations were found to increase some 600% to 0.9 µg L^{-1} during senescence of this alga (Jones et al. 1982), and in later investigations at an offshore reef, similar effects were noted for other trace elements (Jones et al. 1986). The combination of a local source of

Table 2: Dissolved heavy metal concentrations (µg L^{-1}) in some Australian coastal waters

Site	Cu	Pb	Cd	Zn	Hg	As	Se	Co	Ni	Cr	References
NE Pacific Ocean	0.03	-	-	0.006	-	-	-	-	0.12	-	Bruland et al. (1991)
Bate Bay, NSW	0.3	0.2	0.06	-	-	-	-	-	-	-	Batley and Gardner (1978)
Pacific Ocean, 8 km off Port Jackson, NSW	<0.2	0.04	0.01	0.1	0.02	1.4	0.08	-	-	0.1	Batley and Brockbank, unpubl.
Pacific Ocean off Maroubra, NSW	0.09	0.03	0.01	0.2	0.01	1.0	<0.01	0.04	0.2	0.3	Batley and Brockbank, unpubl.
Pacific Ocean off Eden, NSW	0.03	<0.01	<0.005	<0.04	<0.0015	1.5	<0.073	-	0.18	0.1	Apte et al. (1994)
Lizard Island, GBR, Qld	0.13	<0.06	<0.01	0.1	<0.002	-	-	-	-	-	Denton and Burdon-Jones (1986)
Heron Island, GBR, Qld	0.15	<0.06	<0.01	0.12	<0.001	-	-	-	-	-	Denton and Burdon-Jones (1986)
Tasman Sea	0.06	-	0.02	0.08	-	-	-	-	-	0.12	Mackey, unpubl.
Indian Ocean, 20 km west of Fremantle	-	0.018	0.002	0.030	-	-	-	-	-	-	Rosman et al. (1982)
Cleveland Bay, GBR, Qld	0.1	0.15	0.07	1.02	-	-	-	-	0.13	-	Jones (1981)

a Total metal concentrations; typical values for surface waters
GBR-Great Barrier Reef

Table 3: Trace metals ($\mu g\ g^{-1}$) in some Australian estuarine sediments

Site	Depth cm	Zn	Pb	Cu	Cd	Reference
Corio Bay offshore	Surface	4-400	2-210	2-50	0.1-13	Smith (1978)
Corio Bay	Surface	14-166	14-100	4-35	0.2-9	Fabris (1983)
Port Phillip Bay, near shore	Surface	21	8	1.5	0.8	Talbot et al. (1976)
Port Phillip Bay, offshore	Surface	40	22	8	2	EPA (1976)
Port Phillip Bay, near Werribee Treatment Complex	Surface	9-300	<20-140	<5-75	<5	MBW (1991)
Lake Munmorah	5	150	40	70	-	Batley et al. (1990)
	60	50	20	50		
Tuggerah Lakes	5	110	40	20	-	Batley et al. (1990)
	70	80	30	30		
Lake Macquarie North	55	2400	1200	170	160	Batley (1987)
Lake Macquarie South		150	68	20	4	
Blackwattle Bay, Sydney Harbour	10	1150	520	180	3	Batley (1986)
Port Kembla Harbour	10	380	113	113	2	Batley and Low (1985)
Quibray Bay, Botany Bay	10	25	10	3	0.5	Batley, unpubl.
Sydney coast (100m water depth), typical high values for clay-silt	Surface	60	15	14	-	Batley and Brockbank, unpubl.
Central GBR (John Brewer Reef)	Surface	5	0.6	0.2	-	Jones (1992)
Cleveland Bay, Qld	10	24-460	<5-53	15-70	-	Reichelt and Jones (1992)
Saunders Bay	Surface	6-18	<0.5-5	1-3		
Bowling Green Bay	Surface	10-26	<0.5-4	1-4	-	Burdon-Jones et al. (1977)

copper and copper-binding organics released by the algal ligands were believed to be responsible for metal concentrations approaching guideline values.

Metal concentrations in coastal waters are usually much lower than estuarine waters (Table 2), with values for both well below those reported for northern hemisphere coastal waters where industrialisation has had a greater impact, and where discharge standards have been more relaxed (Forstner & Wittman 1979). Values for dissolved metals did not differ greatly from waters of the Great Barrier Reef to those in the Pacific Ocean off New South Wales or the Indian Ocean off Western Australia. The detection limits for these data were lowest in the latter samples where isotope dilution mass spectrometry was used as the method of analysis. Concentrations were higher closer to shore, for example copper in Bate Bay (NSW) or dissolved zinc in Cleveland Bay (Qld), suggesting local contamination. A recent detailed study by Apte et al. (1994) of waters from sites off the NSW coast, using ultraclean techniques for both sampling and analysis, has yielded data which are amongst the lowest reported for Australian waters.

Heavy metals in sediments

In estuarine and inshore waters, as a result of both particulate and dissolved inputs, heavy metals will be enriched in suspended sediments as they are in the dissolved phase, and there is now substantial evidence of heavy metal enrichment in the bottom sediments of coastal waters. This is greatest in sediments close to the mouth of estuaries, reflecting their role as a contaminant source. Enrichment will be a function of grain size, being greatest with the smaller area, higher surface area clay and silt particles than with sandy sediments.

Data for some Australian sediments are shown in Table 3. The lowest numbers reflect a high sand content in the samples. Industrial and port activities, stormwater run-off and sewage discharges have been identified, in the studies quoted, as the major sources of the high concentrations observed for metals such as zinc, lead and copper.

Speciation of metals in sediments is sometimes necessary to delineate pollution sources, especially to distinguish between mineralised or lattice-held metals and the more bio-available fractions (Kersten & Forstner 1989). For this purpose, the use of dilute acid or complexing agents such as ethylenediamine tetraacetic acid (EDTA) as extractants has been found to yield concentrations which most closely relate to the bio-available fraction, and this fraction has been measured in sediments from a number of Australian estuaries (Reichelt and Jones 1992; Batley, 1987).

The definition of sediment quality guidelines is still the subject of extensive research both in Australia and abroad. While no definitive guidelines have yet been published, typically

Table 4: Zinc and copper ($\mu g\ g^{-1}$) in some Australian oysters

Site	Zn	Cu	Reference
Hook Island, close to resort, Great Barrier Reef	1225-9648 (2864)	121-459 (256)	Stump (1988)
Hook Island, distant from resort, Great Barrier Reef	131-1314 (500)	16-397 (127)	Stump (1988)
Rattlesnake Island, Great Barrier Reef [b]	1441-2095 (1723)	263-439 (339)	Burdon-Jones et al. (1977)
Dampier Archipelago, WA	55-1800[a]	31-200[a]	Talbot (1985)
Shark Bay, Vic.	(180)[a]	-	McConchie et al. (1988)
Darwin Harbour, NT	109-611[a]	18-58[a]	Peerzada and Dickinson (1988)
Townsville Harbour, Qld[b]	673-20 906 (8253)	508-3246 (1635)	Jones (1981), Jones (1992)
Georges River, NSW	80-665[a]	3-48[a]	Mackay et al. (1975)
Georges River, NSW	440-760[a] (592)	19-89[a] (53)	Batley et al. (1992)
Tasman River, Tas.	1700-14 000	200-1700	Ayling (1974)

a Wet weight.
b Seasonal study.
Numbers in parentheses are total metals

concentrations of 100 µg Zn g^{-1}, 50 µg Pb g^{-1}, 25 µg Cu g^{-1} and 8 µg As g^{-1} have been considered as guidelines for dredged sediment disposal in Canada, and these are not too dissimilar to suggested EPA values. On this basis, most of clay-silt sediments in urban and industrialised estuaries around Australia would exceed the guideline values (Table 3).

Heavy metals in biota

There is a plethora of data for metals in biota because of the importance of food chain bio-accumulation and ultimate human consumption. The National Health and Medical Research Council (NH&MRC) set guideline concentrations for heavy metals in fish, crustaceans and molluscs. There have been isolated instances of certain organisms exceeding these guideline concentrations and this has generally been attributable to point source pollution from urban or industrial sources. Such contamination has often been detected in waters and sediments from where the biota was collected, indicating the route of metal uptake. More often, because organisms bio-accumulate and effectively integrate metal loads, biota provide a more reliable measure of the presence of metal pollution in waters. It should be noted that bio-accumulation will be dependent on the chemical form of heavy metals, with ionic species being generally more bio-available than bound or complexed forms. The analysis of biota is also a less exacting task than that of waters. There are a number of examples where uptake and biomagnification of metal concentrations occurs via the food chain, for example through algae or aquatic plants, but the original source is always water or sediment.

Mercury is a metal that has received considerable attention since the early 1970s. Recent measurements of fish species, at the top of the food chain, still show tissue concentrations exceeding the NH&MRC guideline value of 0.5 µg g^{-1} (wet wt), at sites such as Port Phillip Bay (Walker 1982, 1981) and Townsville's coastal waters (Denton & Breck 1981). Elevated concentrations of other metals have in the past been detected in fish, oysters and seagrasses near smelters, refineries and other heavy industry. The number of such instances has now been reduced in New South Wales and Victoria because of more stringent discharge controls, and this is likely to occur in other States.

In tropical Queensland waters an extensive investigation of metal levels in a wide range of organisms collected from urbanised and industrial locations has been undertaken (Burdon-Jones et al. 1977; Burdon-Jones et al. 1975). These studies highlighted significant metal contamination in various marine organisms collected from sites close to port activities, urbanisation and the discharge of sewage. One of the objectives of this study was to identify useful monitoring organisms. The oysters, *Saccostrea amassa* and *Saccostrea echinata* fulfilled many of the requirements of a useful monitoring organism in that they are abundant in the intertidal zone, sessile, euryhaline, long-lived, and exhibit high concentration factors for a number of metals. Mean concentrations of zinc exceeded the NH&MRC guideline values in *S. amassa* from Cleveland Bay (Table 4) close to port activities and sewage discharges (Jones 1992) and were comparable to values for oysters from the

Table 5: Tributyltin in Australian coastal waters

Site	Date	TBT (ng Sn L^{-1})	Reference
Georges River, NSW	Pre-ban	8-40	Batley et al. (1989a)
Georges River, NSW	Post-ban	1-11	Batley and Scammell, unpubl.
Kogarah Bay, NSW	Pre-ban	100	Batley et al. (1989a)
Port Phillip Bay, Vic.	Pre-ban	3-23	Batley and Scammell (1991)
Southport, Qld	Pre-ban	45	Batley and Scammell (1991)

polluted Tamar Estuary in Tasmania (Ayling 1974). The upper range of zinc concentrations (38 700 µg g^{-1} dry weight) was close to the highest recorded concentration of zinc in oysters (Bloom & Ayling 1977). Copper concentrations were even higher than reported for the Tamar Estuary (Jones 1992). Comparison of zinc concentrations in oysters from a range of locations throughout Australia clearly indicates their usefulness as a monitoring organism (Table 4).

In the Georges River (NSW), an urban estuary, oysters (*Saccostrea commercialis*) in 1988 had copper concentrations exceeding the guideline concentration of 70 µg g^{-1}. This was attributed to antifouling paints (Batley et al. 1992). Copper has long been a component of antifouling paints but mixtures of copper and TBT have been found to bee more effective against the range of biofouling organisms. Copper was elevated in oysters because of a synergism involving TBT, but following the banning of the latter in paints, the copper concentrations returned to below the limit.

Tributyltin in estuarine and coastal waters

The guideline concentration for TBT in marine waters is 2 ng L^{-1}. Prior to banning, measurements had been obtained for water samples from a number of sites in NSW and other states (Table 5). Particularly near dockyards or other areas of high shipping densities, concentrations of TBT in excess of 100 ng Sn L^{-1} were found. In waterways inaccessible to large vessels, concentrations ranged from 10-100 ng Sn L^{-1}. Following banning, concentrations were in most instances near or below the guideline value.

Tributyltin in sediments

As with heavy metals, the greatest concentrations of TBT are found in sediments. Data have been obtained for sites in Vic., Qld, SA, WA and NSW for their respective environmental authorities, and the results have been similar for each location (Batley & Scammell 1991; Witney 1991; EPAWA 1990).

Typically concentrations in sediments are low and are of minor environmental concern. In the vicinity of marinas, however, sediment concentrations can exceed 1 µg Sn g^{-1} or higher, the latter usually the result of paint flakes hydroblasted from boat hulls (Batley & Scammell 1991). Surveys of estuaries around Sydney have shown that, following banning, TBT concentrations are now lower in the upper 1-3 cm of sediment, depending on sedimentation rate. A maximum concentration appears below this depth, diminishing in deeper sediments consistent with its successive degradation to dibutyltin, monobutyltin and inorganic tin.

Tributyltin in biota

The range of data on TBT in Australian aquatic biota has been summarised in several publications (Batley & Scammell 1991; Maher & Batley 1990; Scammell et al. 1990). The most significant impacts have been on intertidal oysters but with an apparent lack of impact on subtidal oysters. The reason for this is postulated to be related to enrichment of TBT in the surface microlayer. Reductions in populations of scallops in Victoria might also be attributable to the impact of TBT in the microlayer on larval scallops.

Both problems have now been reversed with the banning of TBT (Batley et al. 1992). Oyster growth is now normal, with TBT barely detectable, and larger than ever scallop populations are being reported in Port Phillip Bay.

The observations in gastropods of imposex (the induction of male reproductive organs in female animals) caused by TBT has been examined in a number of sites in NSW (Ahsanullah and Wilson, unpublished results). The impact of banning on this phenomenon has not been reported. There appears to have been no significant impacts or accumulation of TBT by other aquatic biota.

Summary

While this review has concentrated principally on the eastern seaboard of Australia, at sites known to the author, the results can be considered typical of the continent's coastal waters. In general heavy metal concentrations in coastal waters are low and in most cases approach open ocean values, although the data base is small. In estuaries, concentrations are higher and in limited cases where point source inputs are responsible, values are found which exceed water quality guidelines. This represents an improvement in conditions of a decade or two earlier, and reflects improved discharge controls for some states. TBT concentrations in estuarine waters have also decreased to close to or below ANZECC guideline values since its banning in antifouling paints for small boats.

The impacts from lowering dissolved metals in waste discharges is seen in recent data for sediments and biota. Surface sediments generally show lower metal concentrations than at depth, but the top 50 cm of most urbanised and industrialised estuaries are contaminated with heavy metals, especially lead and zinc. Biota accumulating metals from either waters or sediments, can reflect significant contamination, and in some cases this leads to values in excess of the NH&MRC guidelines.

In conclusion, however, it is felt that comparable baseline studies are still needed for other coastal zones and estuaries throughout Australia. Furthermore, knowledge of the physical and chemical processes in coastal waters is essential in order to understand the mechanisms involved in the distribution of pollutants from their points of entry. Care needs to be taken when extrapolating water quality guidelines to tropical environments where considerable variation in temperature and salinity can occur at different times of the year.

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Ocean litter stranded on Australian coasts

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Introduction

Oceans are the ultimate sink for many by-products of human activities, but present data are inadequate to manage this large component of the global commons (Brundtland 1987). Research on ocean pollution has concentrated on the local impacts of diffuse marine pollutants such as oils and other toxins, radionucleides, sewage and other organic wastes, mostly near inhabited coasts in the enclosed seas of the Northern Hemisphere. Conspicuous marine pollutants, including floating marine debris, or ocean litter, have been the subject of conferences in 1985 (Shomura & Yoshida 1985), 1989 (Shomura & Godfrey 1991), and in 1994 (Miami 1994a, 1994b). These have emphasised the detrimental effects of ocean litter on marine wildlife, especially through entanglement (Laist 1994, 1987), ingestion of plastics (Robards et al. 1994), and economic impacts on fisheries and tourism (Miami 1994b). 'Ghost fishing' by abandoned fishing gear results in the entanglement of marine wildlife (Carr & Harris 1994). It is the passive equivalent of drift netting, a wasteful commercial fishing technique which also involves indiscriminate entanglement, and is opposed by Australia.

In Australia, occurrences of ingestion of plastic by seabirds and turtles, and entanglement of cetaceans, seals, seabirds and fish are known, but research has been opportunistic and the overall extent of damage to marine wildlife has not been estimated. The animals mainly affected appear to be the Australian fur seal in Tasmanian waters, and other pinnipeds and cetaceans off Western Australia.

A daily input of more than 600 000 plastic containers into the oceans was attributed to shipping in 1982 and in 1975 the U.S. National Academy of Science estimated that 6.4 million tonnes of litter were jettisoned from ships at sea each year (Laist 1987). Marine pollution from land-based sources (LBS) has been seen mainly in regional terms, but is now a source of global concern (Nollkaemper 1992). Liffmann (1994) stated that 'not unlike other forms of marine pollution, land based sources [of marine debris] are a much more significant factor than are vessels', and quoted other authors saying that vessels account for only about 10% of all pollutants entering the oceans. A new international convention for the protection of the oceans from all sources of pollution has been mooted (Davis 1990).

International concern about ocean litter led to the Marine Pollution Convention ('MARPOL') regulating the dumping of rubbish at sea, including a total ban on jettisoning plastic. Australia ratified MARPOL in November 1990, but there are presently no data being collected in Australian waters or coasts to discover whether its provisions are having any effect. Recent research in the South Atlantic suggests that they are not (Ryan & Moloney 1993).

There is a general lack of quantitative information about ocean litter, but informed opinion suggests that it is a growing environmental problem worldwide (Miami 1994a, 1994b). This paper discusses some methods of sampling and quantifying ocean litter, the sources of human litter and natural flotsam stranded ashore, the suitability of Australian beaches for monitoring ocean litter, and the movement of floating litter by wind and sea. A summary of what is known about ocean litter on Australian beaches is followed by recommendations for future research and monitoring of stranded litter, as a basis for the enforcement of MARPOL regulations in the seas surrounding Australia.

Sampling ocean litter

Ocean litter might be detected and monitored by:

a) remote sensing from satellites or aircraft;
b) sampling at sea from ships using neuston nets, or;

c) collection and measurement when washed ashore.

The Marine Debris Survey Manual (Ribic et al. 1992) summarised different techniques, based mainly on experience of such surveys in the North Pacific and North Atlantic.

Remote sensing of floating litter from aircraft or satellites is beyond the capacity of present methods of detection, except for large items in very calm seas, and would not distinguish between natural and synthetic objects. Matsumura et al. (1994) described line transect sightings of 136 000 items of marine debris over 926 000 nautical miles from Japanese fishing vessels in 1987-91, with densities varying from 20-40 items per mile^2 in coastal waters and within the Pacific Gyre north of Hawaii, to 0.2 items per mile^2 in the North Equatorial Current. Earlier findings from these surveys were mapped by IMO (1991). No such quantitative data are known from sightings of debris in Australian seas. Ship-borne sampling using neuston trawls is a more reliable method of measuring ocean litter at sea, but is slow and costly. Neuston trawling in the surface waters of the New Zealand sector of the Southern Ocean suggested that there were less that 20 items of litter per km^2 in seas to the south of the subtropical convergence, and up to some 2000 per km^2 in the Tasman Sea (Gregory 1985). Trawl surveys showed that granules from plastic fabrication were locally abundant in seas near manufacturing ports, and may be an underrated contaminant of the oceans (Gregory 1994, 1990). No such trawl surveys have been undertaken in Australian waters.

Systematic beach litter surveys have been used (mostly in North America and Europe) to estimate litter pollution levels in nearby seas. Such beach surveys demand frequent repetition, employing standardised techniques, and a rigorous systematic approach (Ribic 1994; Ribic et al. 1992). Most data on the types and origins of beach litter on Australian mainland coasts are concentrated near cities where land-based litter is most abundant. Greenpeace's *Adopt-a-beach litter* campaign used survey forms which

> 'were designed for recreational urban beaches, as opposed to those located near industrial sites. This was because Greenpeace envisaged that people would adopt beaches they visited frequently and, most likely, lived near' (Greenpeace Australia 1992).

Notwithstanding this bias towards popular beaches, Greenpeace paid special attention to 'Debris resulting from fishing practices' and found a total of 34 158 fishing related articles on just 0.8% of the Australian coastline. They tried to distinguish 'hook, line and bait' (HLB) fishing gear (from shore fishing) and maritime 'net fishing' (N) gear, but found it difficult to identify the source of the articles collected. Comparison of *Adopt-a-beach* yields, with those from remote coasts not visited by beachgoers, could be valuable in sourcing fishing litter, provided the data are collected in comparable ways. Other popular beach-cleaning events such as *Clean Up Australia* distinguish land-based and marine items in their yields but do not separate the maritime component from litter originating as a result of shore-based fishing. Although valuable in keeping beaches clear of rubbish, *Adopt-a-Beach* and *Clean-up Australia* are of limited use in measuring pollution of nearby seas by floating litter. Gregory and Ryan (1994) advocated 'long term monitoring of isolated and remote beaches as a cheap and efficient way of measuring at-sea loads' [of marine debris] and stressed the importance of time-series surveys 'from which accumulation rates can be determined'.

Origins of human litter on beaches

Artefacts stranded on beaches have four possible sources:

a) *Items left behind by beachgoers ('tourist trash')*. This typically consists of picnic items (rugs/sheeting, plates, cups, carrybags, bottles, suntans, food packaging), clothing (hats, garments, thongs), recreational fishing and swimming gear (snorkel, masks, flippers, surfboards, fishing line). Some tourist trash betrays its land-based origin because the items do not float in surf (unstoppered glass bottles, paper, cardboard).

b) *Land litter washed into estuaries or along coasts from creeks draining catchments with industrialised and urban areas.* This may include items of domestic garbage, sewage, industrial wastes and packaging, and pellets from plastic fabrication plants. Agriculture and mariculture litter (e.g. fertiliser bags, oyster boxes) may also be locally important. While much agriculture, mariculture and industrial beach litter is easily sourced, commonly-used items such as plastic

Table 1: Composition of the Australian coastline

State	Sand (km)	% Aust coast	Rock (km)	Dune-rock (km)	Mud (km)
Queensland	3574	11.8	568	-	1714
N.S.W.	1168	3.9	507	-	14
Victoria	1112	3.7	329	68	177
S.Australia	2058	6.8	484	424	281
W.Australia	4766	15.7	2535	472	1985
N.Territory	2195	7.3	345	22	2308
Tasmania	1061	3.5	1084	-	32
Totals	**15921**	**52.7**	**5852**	**986**	**6511**

containers may come from land or maritime sources.

c) Maritime litter from coastal and inshore shipping and fisheries on the continental shelf, including litter from drilling and extraction platforms, and inshore recreational craft. This consists of domestic rubbish similar to that from shore sources, but typically includes discarded glass bottles, light globes and fluorescent tubes, and fishing gear such as crates, bait boxes, nets, lines, ropes and floats.

d) Oceanic litter carried from sources beyond the continental shelf. This includes fishing gear, and many of the same items as litter from closer inshore. Long distance ocean litter may carry encrusting organisms or other evidence of periods afloat. Items with foreign markings may have been jettisoned a long way from the Australian coast, but many foreign vessels fish near Australia, making oceanic litter difficult to distinguish from (c) above.

Identifying sources of beach litter is difficult or impossible for items used both ashore and afloat. Features such as buoyancy, purpose and manufacturer; and circumstantial evidence from seasonal timing in relation to shore-bound tourist activity and location, may enable the source of much beach litter to be determined.

Natural flotsam on beaches

Some stranded beach litter comes from natural sources:

geological:	pumice, bitumen and tarballs from seabed oil seeps;
plant:	seaweed, drift seeds (e.g. coconuts, nickar beans), resins;
animal:	cuttlebones, dead fish, seals, cetaceans and seabirds.

Bitumen and tar may have natural or artificial origins (oil spills, tanker washouts). Natural flotsam often carries chemical or taxonomic indications of its source, and thus provides information on the direction from which artifacts entrained in the same water bodies have come. Natural flotsam should therefore be recorded in surveys undertaken to determine the origins of human beach litter.

Beach types for ocean litter monitoring

The Australian coastline has been categorised according to the material at the water's edge, based on a length of about 30 000 km for the continent (Galloway & Bahr 1979). Numbers in Table 1 (Galloway et al. 1984) exclude about 1000 km of water (behind chains of inshore islands), alluvial fans and artificial coasts associated with ports, harbours and sea walls.

Stranded litter is smashed or concealed on rocky coasts, and lost amongst salt marsh plants or mangroves on muddy coasts. Over half the Australian coastline is sandy: it provides the most suitable beaches for observation and monitoring stranded ocean litter.

The preferred characteristics of any beach for ocean litter monitoring can be summarised as:

Geographical :-

1. Facing the major wind systems and ocean currents which operate across the more-or-less defined area of sea whose litter is to be monitored.

Geomorphological:-

2. Sand, gravel or shingle beach, without reefs causing heavy surf to break offshore (which

may smash glass before it reaches the beach).
3. Nourished by offshore sands, rather than by nearby rivers contributing land-based sediments (which may be associated with terrestrial litter).
4. Backed by a dune system with an understood relationship to the beach sands.
5. Having a uniform sediment compartment, at least 5 km long, with minimal longshore drift of sand.
6. Moderate beach gradients with a small tidal range, so that high and low water strandlines are close enough to be sampled simultaneously, and litter is not shunted across wide strandflats.

Ecological:-
7. No dense subtidal seagrass or algal growth offshore, whose storm debris can smother stranded ocean litter.
8. No dense land vegetation, in dunes, mangroves or back-dune swamps behind the beach, in which windblown litter can be lost.

Social and economic:-
9. Remote from human settlements, seldom visited by tourists, and without easy access to motor vehicles.
10. Without human settlements or industry in catchments contributing litter directly or indirectly to the beach.
11. Without nearby inshore fisheries, mariculture, or anchorages which are used by fishing boats or recreational craft.
12. Within reach of a rubbish tip, to which beach litter can be removed from the beach/dune system.

Collecting and sorting beach litter requires an active, observant and flexible but disciplined labour force. An important practical consideration in selecting a beach for ocean litter monitoring is the accessibility of sources of beachcombing labour, such as local schools. The periodic deployment of such labour can also be used to educate, and promote community involvement in environmental concerns on a global, as well as a local scale.

All beaches absorb continuously varying amounts of energy from the atmosphere and the ocean, including the incessant effects of rhythmic tides. They are therefore in a state of constant flux, within which ocean litter is only a tiny component, caught up in the large scale interactions of wind, sand and sea along the coast. Without some understanding of this dynamic system within which stranded litter is moved and broken down (and eventually becomes incorporated into the coastal sediments), it is impossible to interpret beachcombing yields, and their relation to inputs of litter from the ocean.

Movement of litter and other drifting objects at sea

Ocean and inshore currents are generally used to indicate the directions from which litter approaches the coastline, and therefore the seas in which it is likely to have been jettisoned. But experiments with drifting objects, and observations of natural flotsam, suggest that wind is often a more important vector than current.

Smith (1991) stated

> 'Many studies (e.g. Brown 1991, Jokiel 1990) have shown that wind direction provides the best prediction of floating objects' drift speed and direction, at least where currents are not strong. Reported 'surface' currents in generalised maps actually represent integrated transport of all water above the thermocline (Jokiel 1990); those published for mariners and based on ship reports reflect more accurately true surface flows, but they tend to have low levels of constancy (e.g. Federal Dept of Transport, 1987). ... Results of short-term observations of surface water movement (Pickard 1986) and longer-term drift card experiments (Collins & Walker, quoted by Parnell, 1988) in the Great Barrier Reef region are consistent with these generalisations. Floating objects presenting a large freeboard are even more strongly propelled by the wind (Coombs & Landis 1966)'

Figure 1 shows both the major offshore currents and the prevailing winds affecting Australasia, but inshore currents, tidal streams and local winds (e.g. land and sea breezes) fluctuate greatly in strength and direction. The form of coastal dunes, and shape of coastal shrubs can be useful indicators of the dominant inshore wind directions. Wind-driven movements of floating litter may therefore be difficult to predict near the shore, and rafts of litter probably strand episodically, contrary to any predictions based on the major currents and prevailing winds.

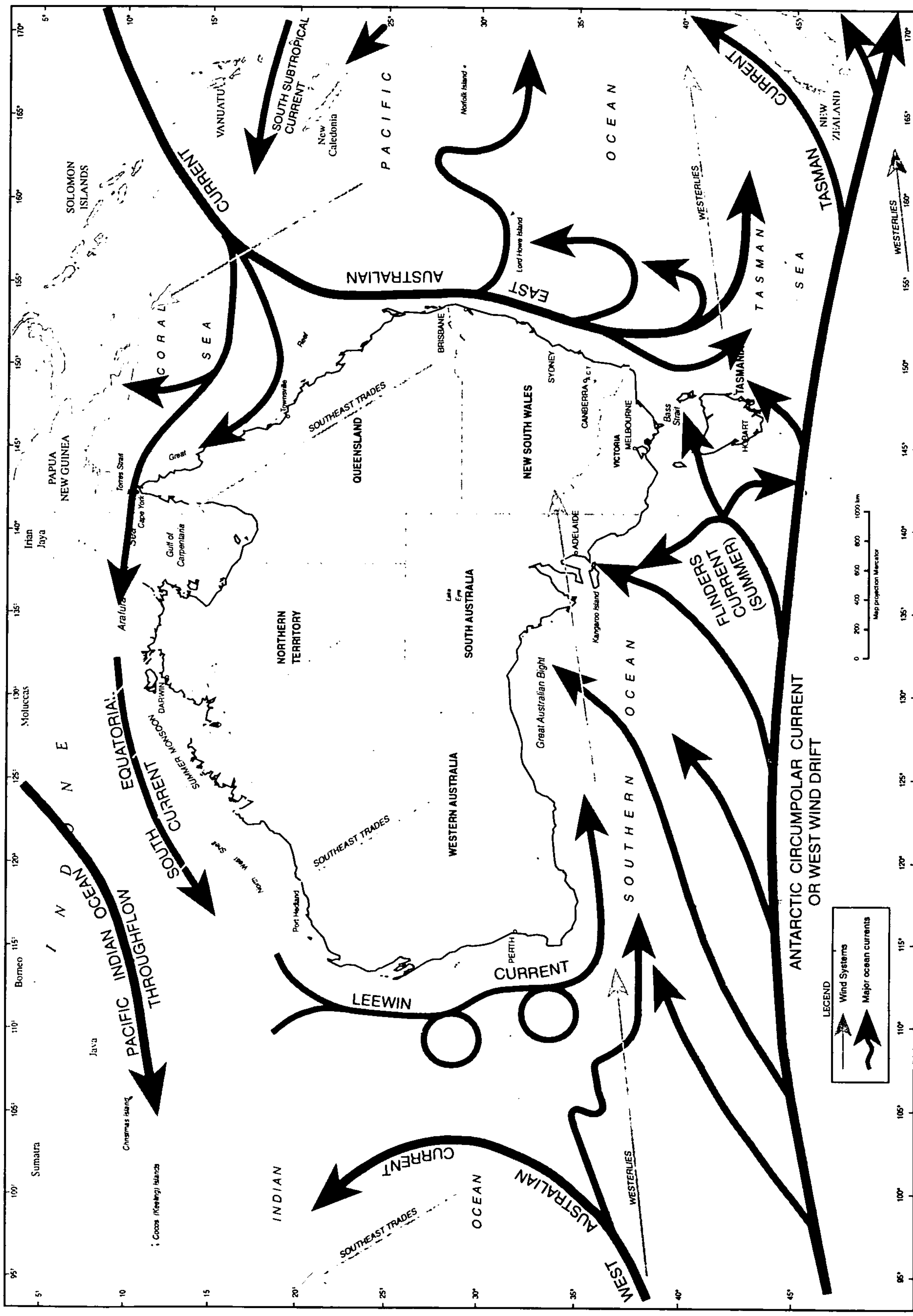

Figure 1: Major ocean currents and wind systems influencing Australia

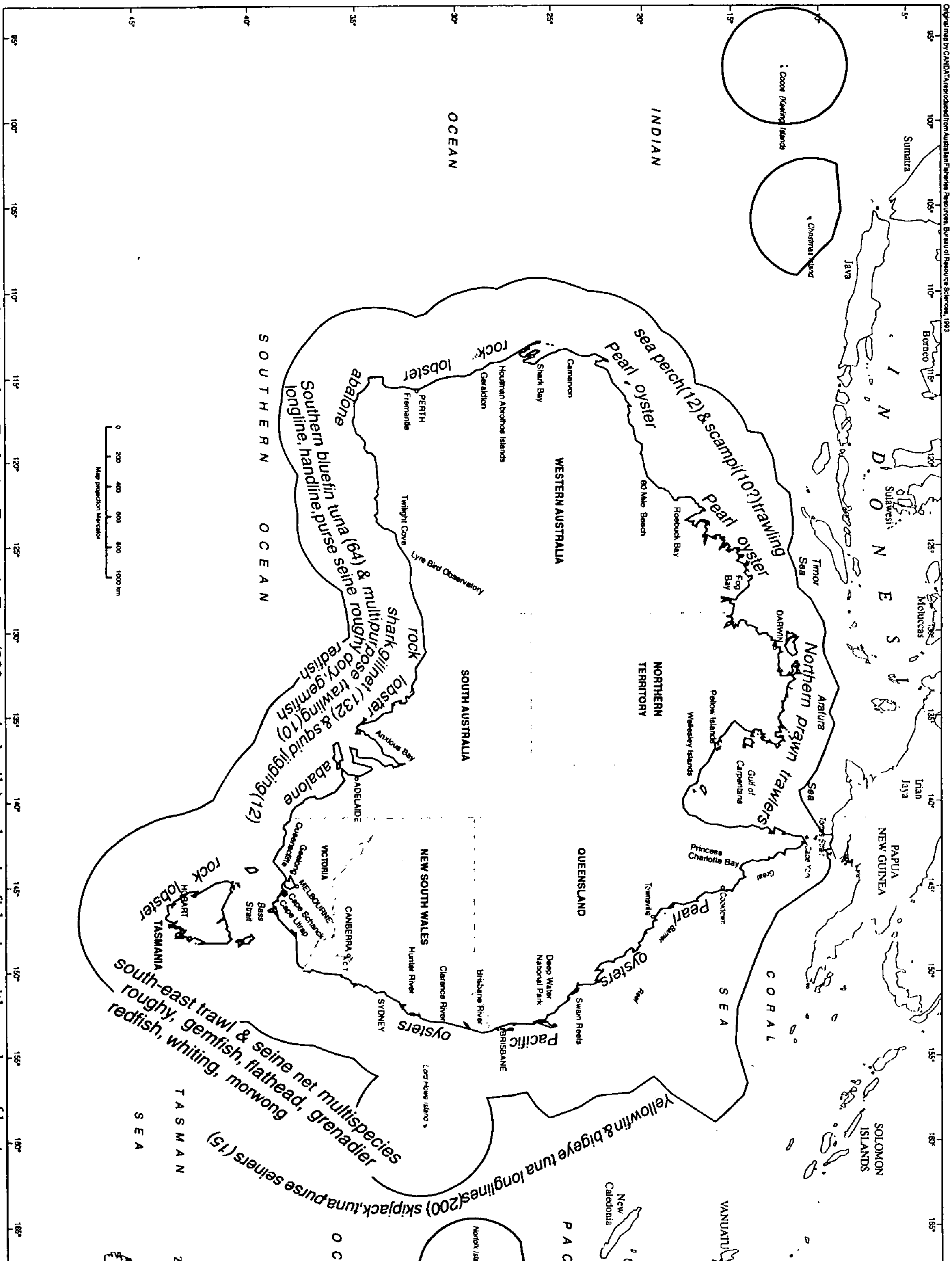

Figure 2: The Australian Exclusive Economic Zone (200 nautical mile) and major fisheries, with numbers of boats involved according to Background Fisheries Statistics No.6 (Fisheries Policy Branch, Canberra, August 1992).

Sources, quantities and nature of ocean litter stranded on Australian beaches

Very little systematic or quantitative data are available. Anecdotal accounts suggest that most Australian beach litter (see Table 2) of marine origin comes from recreational craft and inshore shipping, and from fisheries on the nearby continental shelf (Figure 2).

East Coast, including the Great Barrier Reef (South East Cape, Tasmania to Cape York, Queensland)

Beach litter on the Great Barrier Reef comes from the Coral Sea, Melanesia and the equatorial South West Pacific in the East Australian Current (Creswell 1987). Natural flotsam from Fiji, New Caledonia and Vanuatu drifts westwards across the southern Coral Sea, and is blown towards the coast in the prevailing south-east trade winds. There is an increase in human-derived flotsam, expressed as bottle:coconut ratios, west of the Chesterfield (outermost) Reefs towards Swain Reefs and the central Queensland coast. Pumice on Great Barrier Reef islands and the Australian east coast has been attributed to volcanic eruptions near Tonga and the Kermadec Islands. Plastic containers, rope and thongs are not abundant on cays in the southern Coral Sea, but glass bottles (often Japanese) are common, presumably from shipping and longline and purse seine tuna fisheries (Smith 1992). Cays in East Torres Strait had some glass and plastic bottles in December 1992, with bottle:coconut ratios of 8:55 on three cays (J.M.B. Smith, University of New England, pers. comm.).

Polystyrene floats, fishing litter and bottles were seen at Princess Charlotte Bay in the mid 1980s, but beaches there were generally clear of litter (J. Grindrod, Geography Dept., Monash University, Melbourne, pers. comm.). Nearby Davie and Tydeman Reef Cays, which are probably washed over by storms every few years, had 127 glass bottles, some light globes, thongs and assorted timber on their beaches, together with plastic bottles, pipe, rope, netting and foam in December 1992 (J.M.B. Smith, University of New England, pers. comm.). Daintree beaches were 'remarkably clear' of any litter in September 1992, but some visitors recall much tourist trash there in previous summers. In November 1992, '500 items, plastics predominating' and weighing 8.4 kg, were seen in a litter survey on a 1.1 km length of beach at Deepwater National Park, north of Bundaberg, but this included much tourist trash (Woodall 1993). In south-east Tasmania, 68% of beach litter came from fishing and boating sources in 1990-91. Rope comprised 30% of all plastic beach litter, with large inputs from trawl fisheries and fish farms (Slater l991a, 1991b).

Drift seeds and pumice are carried south in the East Australian Current to beaches in south-east Australia, where they are presumably joined by land litter from the Brisbane, Clarence and Hunter rivers, whose sediment plumes are visible in satellite imagery. Anecdotal evidence from observations along NSW coasts, suggests that plastic and glass bottles are ubiquitous, and that tourist trash is usually abundant and persistent, mixed with estuarine industrial contributions and ocean litter on east coast beaches.

Such fragmentary data give little idea of the quantity of litter on east coast beaches. Data from the *Adopt-a-beach* clearances on the east coast (where most of this activity has taken place) may yield some information on ocean litter. However the data have not been recorded in a way that litter can be quantified, or the land and marine litter distinguished. In a survey of shoreline litter near Brisbane, Sydney and Melbourne, O'Callaghan (1993) compared beach litter with that in gutters, drains and creeks, and concluded that most stranded litter came from land-based sources. Observers note that littering of beaches in southern NSW varies seasonally and with preceding weather: clear after winter storms, but heavily littered after calm spells in the summer holidays.

North Coast (Cape York, Queensland, to North West Cape, WA)

South-east trade winds drive surface waters westwards through Torres Strait and between Australia and Indonesia. The South Equatorial Current also sets westward through Indonesia into the Indian Ocean. These flows in the tropics to the north of Australia, contribute to the build-up of waters off north-west WA, forming the headwaters of the Leeuwin Current.

No systematic studies of natural flotsam or synthetic litter are known from beach on these coasts, but anecdotal evidence suggests that away from settlements, fishing litter is abundant. Prawn fisheries in the Gulf of Carpentaria contribute litter which is lost among mangroves, but sandy beaches between the Pellew and Wellesley Islands were badly

Table 2: Ocean litter on Australian beaches*

		EAST COAST		NORTH COAST	WEST COAST		SOUTH COAST	
	East Tasmania	**Queensland**	**Barrier Reef**	**Arnhem Land**	**Abrolhos Coast**	**East Bight**	**Bass Strait**	**West Tasmania**
Offshore currents								
Dominant current		East Australian Current		Equatorial Counter	Leeuwin		Antarctic Circumpolar Current	
Direction	N to S	N to S	NE to SW	East-West	North-South		West-East (strongest in winter)	
Natural flotsam								
Pumice	+	+	abundant	?	?	common	present	present
Bitumen (seabed seeps)			none	?	?	occasional	?	+
Dammar resins					?	occasional	?	?
Drift seeds & fruits	?	+	coconuts +	coconuts +	Indomalesian	rare	?	rare
Human beach litter	(items)	(weight)				(weight)	(items)	(items)
Hard plastic (moulded)	} 65%	33% fB	scarce Sf	locally abund.	?	30% Fbsd	38%	}55%
Soft plastic (flexible)	} Fb	3% b	scarce f	?	?	} 32%	36%	} Fb
Foamed plastic	9%	?5%	rare f	?	?	} Fb	4%	3%
Virgin plastic		?local				none	local	?
Glass	9% B	40% Br	common S	+	?	31% Fbrd	4%	17%
Metal	5% B	2% b	very rare f	?	?	5% sb	4%	23% B
Wood	(woodchips)	6% s	scarce s	?	?	+ S	+	forest waste
Fabrics & footwear	+	8% B	common s	?	?	+ b	+	+
Ocean litter surveys	1990-91	1992	1986-1990			10/91,92,93	4-5/93	1990-91
Site	south-east	Deepwater NP	Swains Reef			Anxious Bay	Gippsland	SW Heritage
Beach length (km)	composite	1.1 km	reefs & cays			26 km	179 km	?
Investigator	Slater 1991b	Woodall 1993	Smith 1992			Wace 1994a	Coast Trek	Slater '91b/2
Items per km beach	c.300	455					84	350
Weight per km beach		7.6 kg				13 - 15kg	7 kg	
Anecdotal data	Hughes	Nicholson, Grindrod	Smith	Bowman	Wallensky, Minton	Eyre Bird Observatory	Heislers	Wace
Litter-generating fisheries	SE trawl, squid	I/I tuna, E trawl	recreational, E tuna	prawn	rock lobster	GAB trawl, SB tuna, squid scallop, shark rocklobster	SE fishery	squid, rocklobster

* Stranded ocean litter collected from some Australian beaches, based on published and unpublished written accounts of systematic analyses of beachcombings organized by investigators listed, supplemented by anecdotal evidence of beach litter from marine and land sources. Data from *Clean-Up-Australia*, and Greenpeace's *Adopt a Beach* activities are not included. The information in this table is not definitive: it categorises and quantifies ocean litter stranded on Australian beaches, to invite the incorporation of more data

Key: Beach litter generators - B=beachgoers; I=urban/industrial; R=recreational craft; F=shelf/inshore fisheries; S=shipping/ferries (upper case for major inputs, lower case for important but lesser contributions to stranded ocean litter); GAB Great Australian Bight; SB Southern Bluefin polluted with fishing gear during the late 1980s, and lesser pollution of beaches in Arnhem Land (west side of Elcho Island, Croker Island, Port Essington, Melville Island and Fog Bay) was noted in the late 1980s to early 1990s. Beaches on the west coast of Bathurst Island remained 'pristine' in 1991, but proposed mineral sand mining there and on Melville Island will lead to future inputs of land-based litter (D. Bowman Northern Territory Conservation Commission, pers. comm.). Land litter on beaches near Darwin is apparently localised, and beach cleanup activities organised. Researchers studying turtle nesting mention Indonesian debris on some northern beaches (D. Curl, Turtle-watcher on Northern Territory coasts, Jabiru, NT0886, pers. comm.).

Litter on sandy beaches in Roebuck Bay comes from Broome, but Cable Beach nearby is 'very clean'. The northern end of the 80 Mile Beach has 'negligible litter', with some fishing floats, old rope and netting. Plastic bottles and other plastic litter, ropes and nets, are also stranded in small quantities on 80 Mile Beach. Quantities of thick rubber or plastic sheeting, out of which the soles of thongs have been cut, are abundant on Cocos-Keeling Island beaches (E. Wallensky, Australian National University, Canberra, pers. comm.). Such footwear is hand made, and the surplus sheeting jettisoned from Indonesian fishing boats.

West Coast (North West Cape to Cape Leeuwin, WA)

Kenneally (1972) recorded seeds and fruits of nine genera of tropical species, probably carried south from Indonesia in the Leeuwin Current and stranded on beaches in south-west Western Australia. Such naturally occurring long distance drift items are now outnumbered by human debris from the offshore rock lobster fishery, and from sources near Carnarvon, Geraldton and Fremantle. The recent discovery of the egg of an extinct elephant bird from Madagascar on an old shoreline near Cervantes, indicates that long distance drifts across the Indian Ocean to Western Australia have taken place in prehistoric times.

The semiarid to arid sandy/aeolianite coast from Geraldton to North West Cape is poorly known. Its ocean litter has not been systematically surveyed, although concern has been expressed over plastic waste and other marine pollutants on this coast (Young 1989). Oceanic litter from the Antarctic Circumpolar Current in the Indian Ocean joins litter from the tropics in the southward-flowing Leeuwin Current, and is deposited on beaches from North West Cape to Shark Bay, and south to Cape Leeuwin and into the Bight (Creswell 1991). Beaches on the Houtman Abrolhos Islands have craypots, ropes and other debris from the rock lobster fishery, in addition to litter taken ashore by fishers (E. Wallensky, Australian National University, Canberra, pers. comm.).

South Coast (Cape Leeuwin, WA, to South East Point, Tasmania)

Strandings of pumice, bottles and drift cards show that a huge area of the Southern Ocean from Australia westwards through the southern Indian Ocean to the Atlantic coast of South America contributes some litter to the south coast of Australia (Wace 1991, 1990). Similar strandings of bitumen and dammar resins indicate Indonesian origins of this natural flotsam (McKirdy et al. 1994; Padley et al. 1993). This oceanic litter and natural flotsam is shepherded southwards along the Indian Ocean coast of Western Australia and into the Great Australian Bight by the Leeuwin Current (Figure 1).

Systematic attempts to identify the types and quantities of ocean litter on a beach remote from human settlements in southern Australia have been made at Eyre Bird Observatory in the western Bight coast, south of Cocklebiddy. Seven clearances of a 1 km open beach in 1990-91 yielded 494 items, mostly fishing gear: 5% glass, 47% moulded plastic, 41% flexible plastic (including 12% rope), 4% metal and 3% wooden items. Here as elsewhere, plastic forms the major component of ocean litter (Wace 1991; G. Goodried, Curator, Eyre Bird Observatory, pers. comm.). Longshore drift of tourist trash from Twilight Cove to the west may account for some items.

Three clearances of litter along the 26 km beach at Anxious Bay in north-west Eyre Peninsula in October 1991, 1992 and 1993 yielded 343, 399 and 228 kg respectively; averages of 13.2, 15.3 and 8.8 kg of litter per kilometre. About two thirds (by weight) of the Anxious Bay litter consisted of plastic ropes, nets, floats, strapping and containers, mostly abandoned gear from fisheries in the Great Australian Bight including cod-ends and netting from orange roughy and other trawl fisheries, longline floats from tuna boats, rock lobster pots, and bait baskets from inshore fisheries (Wace 1994a; IMO 1992). Yields of ocean litter for these successive

October clearances at Anxious Bay should not be interpreted as annual inputs (see Table 3), because an unknown quantity of litter is banked in the beach sands and dunes. Nevertheless, preliminary work for the establishment of a baseline for ocean litter inputs to one large beach in southern Australia has been established. Comparable studies are needed from other coasts. If the Anxious Bay yields typify the contribution of fisheries and shipping to littering Australian beaches, the quantities of marine debris stranding along the whole Australian coastline must be some hundreds of tonnes a year from these sources alone.

In Victoria, beaches west of the entrance to Port Phillip Bay, are heavily polluted by plastic food and drink containers, and plastic sheeting and rope. It was estimated that 66% of the plastic rubbish was tourist trash. Some had been washed down the Barwon River from industrial sites in Geelong, but much was foreign, and could be sourced to the visits of particular ships to Melbourne. Plastic pellets from local fabrication plants, plastic fragments in the dunes and beach sands, and the presence of surfactants and foam all form part of a study of beach pollution carried out by Queenscliffe High School students. It illustrates the value of using beach pollution studies in the school curriculum, and the practical difficulty of sourcing beach litter on a densely populated coast near a major port, with tourists, industry, shipping and fisheries all contributing to the mess on these beaches (Hunt & Gray 1990). Many plastic bottles had been chewed by sharks, or nibbled by smaller fish.

Beach litter surveys in November 1991 of 7.2 km of residential and surfing coast east of Port Phillip Heads yielded 163 items per km, 69% of which were plastic. Much of the litter was tourist trash. Further east in September 1992, 8 km of beaches not used by surfers yielded 130 litter items per km, averaging 8 kg/km, (75% plastic by number of items, 30% by weight) . A detailed survey of the Gippsland coast in April-May 1993 yielded 84 litter items per km over 179 km of beach (7 kg/km over 67 km where the items were weighed). This comprised about 50% plastic (items and weight) but included land litter left by casual beach users and shore fishers on 90 Mile Beach (D. Heislers, Leader, beach litter surveys on Victorian Bass Strait coasts, pers. comm.).

In Tasmania, 150 surveys of marine debris from 88 beaches all over the State, yielded 50 211 items (January 1990-June 1991), or about 300 items/km. (IMO 1992; Slater 1992, 1991b). West coast beaches yielded 300-350 items per km, 61% of which were plastic, and 80% of which originated from offshore fisheries. Seventy-nine *Adopt-a-beach* litter clearances from March to May 1992 yielded 45 515 items of marine debris (about 600 items/km), but included some heavily populated coasts with litter inputs from land sources and inshore recreational craft (Dick 1992). Plastic items, including rope, strapping bands and fishing nets, buoys and lines, made up 65% of the yield. Inaccessible northern beaches on King Island (Bass Strait) had 10-15 glass bottles per kmin April 1994, but very little plastic (Wace 1994c). Tasmanian, Victorian and South Australian data are hard to compare, because urban beaches were included, and litter items were counted (rather than weighed) in all Tasmanian and most Victorian clearances.

The impacts of litter on marine wildlife seems to have been more intensively studied in Victorian and Tasmanian waters (Pembertonet al. 1992; Slater 1991a; Brown et al. 1986; Skira 1986) than elsewhere on Australian coasts.

Australian oceanic islands

Repeated surveys of beach litter on Heard and Macquarie Islands, (where there is no local production of human litter) suggest minimum accumulation rates of 13 items at Heard Island, and 9 per kilometre of coast per year at Macquarie Island (Slip & Burton 1992, 1990). These annual clearances continue, but the beachcombers are having difficulty in removing heavy items which have floated ashore, so that they will not be re-counted in future clearances (N. Perrin, Australian Antarctic Division, Tasmania, pers. comm.). No systematic surveys of beach litter are known from Australian islands in the Indian Ocean or the Tasman Sea. Christmas Island lacks suitable beaches. Local tourist trash may confuse ocean litter signatures on Norfolk and Lord Howe beaches. Casual observations from Cocos-Keeling (E. Wallensky, Australian National University, Canberra, pers. comm.) suggest that they may be good monitoring sites for ocean litter in the north-east sector of the southern Indian Ocean.

Conclusions

From these meagre data, some tentative conclusions are possible concerning the origins,

Table 3: Example of litter survey form format and data

SOUTHERN OCEAN MEGA-LITTER ON AUSTRALIAN STRANDLINES

Anxious Bay, Eyre Peninsula, South Australia

Kms: 26 **Beachcombers**: Australian National University, South Australian Fisheries, Elliston & Streaky Bay schools **Date**: 9-15 Oct 1992

LAND LITTER:(= NEVER-BUOYANT AREIFACTS)
few newspapers, 70 unstoppered glass bottles, dirty nappy.
few plasticised & waxed cardboard milk & softdrink cartons

NATURAL BEACH LITTER:
seaweed/seagrass: very little seaweed, scattered
nickars, coconuts, other seeds: none
derelict birds, insects: 4 prions (none banded), many moths
marine animal remains: many cuttlebones, few fish & seahorses
pumice: intermittent, mostly small (collected/Diane Padley)
bitumen: sporadic, small pieces (collected/Diane Padley)
damma (Dipterocarp) resin: occasional (collected/Diane Padley)

AREIFACTS: i.e. MANUFACTURED & SYNTHETIC BUOYANT MATERIALS ON BEACH

HARD (MOULDED) PLASTIC = 121kg (4.7kg/km)
liquid containers: >100, few toothmarked, many fragmented
drums, buckets: numerous, many toothmarked and scored (? squidbeaks)
crates: c.10, (incl. S.African milk crate from 1991), many fragments
craypots: few complete, many pieces (red)
bait baskets: >50, (mostly black)
fishbuoys, floats: numerous
surfboards: none
syringes: one, no needle
fragments: very large numbers, (uncountable)

FOAMED AND SOFT (FLEXIBLE) PLASTIC = 126.7kg (4.9kg/km)
fishbouys, floats: numerous
surfboards: none
trays, cups, plates: none, some styrofoam fragments
beercan yokes: 7 only
bags, sheeting: few, mostly fragments
rope: very abundant, mostly short yellow & orange pieces
strapping tape: abundant, mostly blue fragments (43 uncut loops),
fishlines: few tangled
thicknets: numerous, mostly orange pieces
monfilament nets: none

VIRGIN PLASTIC PELLETS: none (but plastic fragments in beach sands)

GLASS = 123kg (4.7kg/km)
bottles: 238 (70 unstoppered incl. 65 from 0-5 & 25-26km, few echoes)
jars: 31 (most with lids)
light globes: 21 (incl. spotlight & large floodlight globe)
flourescent tubes: 20
car light assembly: 1 (from 0-1km marks)

ALUMINIUM (weight included with iron & steel)
drink cans: 30 (incl. 5 @ 0-5km & 21 @ 25-26km) [=$1.30 !]
aerosol cans: 31

RON & STEEL = 10.5 + 9kg
drums: 1, plus rusty bits of at least 2 more
drink/aerosol cans: 8, plus rusty fragments
buoy: 1 (@ 15-20km, wt = 9kg - taken to Elliston school)

RUBBER (weights included with soft plastic)
rawsheet: none
gloves: 12
thongs: c.20 (no pairs), 4 shoes, 1 boot
car radiator hose: 1 (@ 25-26km)

WOOD (not collected)
worked timber - spars & planks: sporadic driftwood
unworked trunks & branches: none

CELLULOSE (not weighed)
paper: some newspapers @ 25-26km
cardboard: beer and other cartons @ 25-26km
plasticised & waxed cartons: sporadic, most @ 0-5 & 25-26km

CLOTH, HESSIAN (not weighed)
sacking: very little
garments: one old shirt, remains of hat

TOTALS: 398.7 KG in 26KM = 15.3 kg litter collected per km of beach

sources, nature, and quantities of ocean litter stranded on Australian beaches:

Geographical origins and sources

Much ocean litter on Australian beaches is jettisoned from fishing vessels near the coast and on the continental shelves, and from shipping using Australian ports. Inshore shipping (and ferries?) probably make the biggest contribution at the approaches to ports, while offshore shipping litter dominates on the Barrier Reef. Most beach litter near cities, and on tourist-frequented coasts has local land-based origins.

Most natural flotsam from outside Australian waters stranding on our coasts, originates in the tropics and is carried south along our west and east coasts by the Leeuwin and East Australian Currents respectively. On tropical coasts, the south-east trade winds force flotsam ashore from the Tasman Sea through the Great Barrier Reef. Some flotsam from middle to high latitudes reaches southern Australia from the South Atlantic and Indian Oceans in the Antarctic Circumpolar Current and the Westerlies. Ocean litter which reaches Australia from distant sources is driven by these surface forces. Although ocean litter from distant sources follows similar courses to natural flotsam, it makes only a tiny contribution to beach litter as compared to that from local sources, especially fishing and coastal shipping. In general, Australians are littering their own beaches with debris jettisoned from vessels operating out of Australian ports, although foreign shipping and fishing vessels may contribute much litter regionally.

The sources of beach litter in Australia are influenced by proximity to cities (as in Israel: Golik & Gertner 1992). On remote coasts, fisheries are the most important beach litter source (as in Alaska: Merrell 1985). Coastal and offshore shipping are important sources at approaches to ports, and along coasts and islands in heavily trafficked seas (as in Western Europe: Dixon & Dixon 1983).

Materials

As elsewhere in the world, more than 50% of stranded ocean litter (whether measured by weight or number of items) now consists of plastics, especially rope, netting and fishing gear, and containers (complete or fragmented). Glass objects (bottles, jars, light globes, fluorescent tubes) are ubiquitous but less abundant. Wood, metal, rubber and fabric are usually present in smaller quantities. Driftwood may have maritime or terrestrial origins, but wood may be abundant on beaches near logging operations (Wace 1994b).

Quantities

The rates and quantities of ocean litter stranding ashore on Australian coasts at any one time, obviously varies widely with situation, season, and preceding weather conditions. Total beach litter yields range up to 500 items or more per km, accumulated over unknown times, but this figure includes some tourist trash. Remote beaches free of tourist trash and other land-based sources, may yield up to 15 kg/km of litter from offshore. Such figures from annual beach clearances can not be accepted as reliable estimates of annual inputs until more studies are made of the rates of natural breakdown and banking of stranded litter. There are no data on which to base reliable estimates of the quantities of litter on beaches, because there is no agreement on the best way to measure it or to distinguish oceanic litter from litter derived locally (i.e. tourist trash, urban, domestic, agricultural, maricultural, industrial wastes).

Recommendations

1. *Standardised accounting methods should be developed to classify and measure beach litter for monitoring purposes.* Such beach litter accounting should be developed recognising overseas beach litter survey techniques (e.g. Ribic et al. 1992). An important step in developing uniform accounting would be the adoption of a concise standard data sheet from which beachcombing yields could be entered directly into portable computers using simple software, written for the purpose. The data sheet used at Anxious Bay could form the basis for standard recording of Australian beach litter, but would probably need amendment for the tropics. Because of the importance of plastics in ocean pollution, the Plastics Institute of Australia should be involved in development of a beach litter taxonomy.

2. *A program of systematic monitoring of ocean litter stranded on Australian beaches should be established* in the States and the Northern Territory, in cooperation with the Federal Department of Transport (Shipping and MARPOL responsibilities), and State and Federal fisheries authorities. Such monitoring should be undertaken annually or more often, on a few remote beaches selected as representative of major coastal provinces and fishing grounds. The advice of coastal geomorphologists, oceanographers and State fisheries authorities should be sought in beach selection, taking into account the criteria set out in this report. In the tropics, beach litter monitoring by coastwatch teams could be linked to existing quarantine and customs surveillance, especially by providing periodic access to otherwise inaccessible coasts. Although data from ongoing Australian beach litter clearances may yield some useful information about maritime litter, ocean litter monitoring should not be seen merely as an addition to activities such as *Clean-Up Australia* and *Adopt-a-beach*, because these concentrate on urban and recreational beaches, and do not distinguish land-based and marine litter.

3. *Offshore neuston trawls should be carried out* to establish a relationship between beachcombing yields, and floating litter in nearby seas.

4. *Biochemical analyses of encrusting biota* growing on floating and stranded litter should be undertaken, to test for pesticide residues, and other toxic substances which may be

accumulating in marine food chains in Australian waters .

5. *Yields from these beachcombing and neuston net trawlings should be sourced so far as possible* to manufacturers and distributors of the offending materials, and to the ocean litterers - notably to seaborne activities such as fisheries, ferries, recreational craft, and shipping. Audited results of the beachcombings should be used as the basis of an education campaign to reduce the amount of litter deposited at sea, and aimed at the ocean litterers themselves, as well as the beachgoing public and other land-based sources of beach litter. Yields should be considered as a basis for levying insurance premiums, or imposing fines on the littering industries or individuals, in order to cover the costs of monitoring and disposing of beach litter. Such arrangements could become a potent means of enforcing the 'polluter pays' principle in the MARPOL 73/78 convention.

6. *Beachcombing yields should also be used as a basis of research* to develop buoyant plastic fishing gear and containers which are biodegradable in the sea. Contacts should be sought with overseas workers studying aquatic and photo-degradation of plastics in the marine environment (Gauldie 1992; Andrady 1988).

Australia's geographical position, receiving flotsam and jetsam from the tropical south-west Pacific, and from enormous areas of the Southern Ocean, provides an opportunity to monitor the condition of these seas, as it is registered in the ocean litter cast ashore along our sparsely inhabited coasts. No other southern continent has such a longitudinal extent of ice-free coast. No other continent has poleward flowing currents on both east and west coasts. The major ocean currents approaching Australia deliver flotsam and jetsam from both tropical and high latitude oceans to sandy beaches here where it can be systematically monitored. Anecdotal evidence suggests that most Australian beaches are less strewn with litter than those in corresponding latitudes of the Northern Hemisphere, but that they are more littered than beaches in Fiji and New Zealand.

An Australian beach litter monitoring program could establish important baselines for pollution of the seas around our continent, and in the Southern Hemisphere more generally, and thus contribute to management of the world oceans.

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Entanglement of Australian fur seals in human debris

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The entanglement of Australian fur seals in human debris in Tasmanian waters has been documented over four breeding seasons (1989-1993) both at Australian fur seal breeding colonies and on an opportunistic basis at haul-out sites in Bass Strait and southern Tasmania. The results to these surveys are summarised here and the methods of data collection and analysis are outlined in Pemberton et al. (1992).

A total of 136 Australian fur seals (*Arctocephalus pusillus*) and one New Zealand fur seal (*A. forsteri*) with neck collars were observed over the four year study period.

Polyethylene trawl nets accounted for 42% of neck collars, polypropylene straps (packaging straps) 29%, monofilament gill nets 15% and nylon rope 11%. Other incidental items included steel rings (n=2) and a rubber loop.

The high rate of entanglement of seals in packaging straps has continued despite the successful development by SAFCOL and the Department of Parks Wildlife and Heritage (Tasmania), of a bait box which does not require straps. Observations of the bait boxes being loaded onto fishing boats at Tasmanian ports suggests that whilst the strapless bait box was accepted by some sectors of the industry, they are not being used to a significant extent. It is also noteworthy that some of the straps (white and green) observed on the seals are rarely used by the Tasmanian fishing industry and probably originate from other nations fishing either in Australian waters or on the high seas west of Tasmania. The West Wind Drift will continue to bring flotsam and jetsam to Tasmanian waters as long as there is a source for this material west of Tasmania and therefore attempts to reduce the amount of debris have to be directed at both international and national levels.

Trawl nets continue to be the major source of entanglement materials for seals in Tasmanian waters. This form of entanglement is probably the most lethal as the typically large pieces of entangled netting is both resilient to wear and buoyant. The frequency of occurrence of trawl net entanglements has increased in southern waters and is now similar to that in Bass Strait. This increase was anticipated as the orange roughy fishery shifted from operating over soft bottom to fishing over rocky sea mounts off southern Tasmania. This resulted in an increase in the number of nets lost due to 'hook-ups' (Pemberton et al. 1992). This trend may change as a reduction in the quota of orange roughy to the south-east trawl fishery has been offset by an increase in the quota of blue grenadier. A major fishery occurs off western Tasmania and west and east of Bass Strait over the continental shelf.

The majority of entangled seals were juvenile and subadult animals (75%). Amongst adult animals, females (23%) were entangled more often than males (2%). As suggested by Pemberton et al. (1992), this probably reflects the tendency for smaller animals to become entangled and the consequent mortality of these animals due to physical injury imposed by the increased restriction of neck collars as the animals grow. Some seals however do survive entanglement. A total of 15% of seals classified as 'entangled' had circular scars around their necks. The only neck collars which were observed to be fraying whilst on the seals were packaging straps. It is probable, therefore, that the majority of those seals that survive entanglement had this form of entanglement, although monofilament and rope neck collars have been found on breeding colonies suggesting that these do occasionally fall off.

The entanglement of fur seals is a management issue for both the conservation of the species and the alleviation of suffering by individual seals. The mean incidence of entanglement of Australian fur seals was 1.6+/-1% (n=22, range 0-3.4). This rate must be considered as an underestimate because of the probable mortality of seals entangled in debris which prevent the animals travelling to haul-out sites, and hence being observed. This incidence of entanglement is higher than that recorded for northern fur seals (*Callorhinus ursinus*) where entanglement was implicated in the decline of this species (Fowler 1987). The entanglement of Australian fur seals is therefore considered to be a threat to the status of the population and consequently a conservation problem. Also, because entanglement results in gross wounding to the seals and prolonged suffering over years, there is a moral responsibility by wildlife organisations to alleviate the suffering of individual seals.

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Acronyms

AAS	Atomic Absorption Spectrometry
AMD	Acid Mine Drainage
ANCA	Australian Nature Conservation Agency
ANZEC	Australian and New Zealand Environment Council
ANZECC	Australian and New Zealand Environment and Conservation Council
AOX	Adsorbable Organic Halogen (Edyvane)
AOX	Adsorbable Organic Halide (Richardson)
APM	Australian Paper Manufacturers
APM	Australian Paper Mills (Rees)
APPM	Australian Pulp and Paper Manufacturers
ASV	Anode Stripping Voltammetry
BHAS	Broken Hill Associated Smelters
BHP	Broken Hill Proprietry
BOD	Biochemical Oxygen Demand
CCNT	Conservation Commission of the Northern Territory
CEPA	Commonwealth Environmental Protection Agency
CSIRO	Commonwealth Scientific and Industrial Research Organisation
DASET	Department of Arts, Sport, Environment and Territories
DBIRD	Department of Business, Industry and Regional Development
DELM	Department of Environment and Land Management
DENR	Department of Environment and Natural Resources
DEP	Department of Environment and Planning
DEST	Department of Environment, Sport and Territories
DOE	Department of Environment
DPI	Department of Primary Industries
DSP	Diarrhetic Shellfish Poisoning
EAC	East Australian Current
EDTA	Ethylenediamine tetraacetic acid
EMP	Environmental Monitoring Program
EOX	Extractable Organic Halogen
EPA	Environment Protection Authority
EPCSA	Environment Protection Council of South Australia
ESD	Ecologically Sustainable Development
GBRMPA	Great Barrier Reef Marine Park Authority
GIS	Geographical Information Service
HCB	Hexaclorobenzene
ICPMS	Inductively Coupled Plasma Mass Spectrometry
ILZRO	International Lead Zinc Research Organisation
IUCN	International Union for the Conservation of Nature
MARPOL	International Convention for the Prevention of Pollution from Ships
MERP	Marine Environment Protection Agency
MRL	Maximum Residual Limits
NFR	Nonfilterable Residues
NH&MRC	National Health and Medical Research Council
NSW	New South Wales
NT	Northern Territory
OPUD	Office of Planning and Urban Development
PAH	Polyaromatic Hydrocarbon
PCB	Polychlorinated Biphenyl
PSP	Paralytic Shellfish Poisoning
QCFO	Queensland Commercial Fishing Organisation
QDEH	Queensland Department of Environment and Heritage
QDPI	Queensland Department of Primary Industries
Qld	Queensland
QTTC	Queensland Tourism and Travel Corporation
RAC	Resource Assessment Commission
SA	South Australia
SAAC	South Australian Aquaculture Committee
SAFIC	South Australian Fishing Industry Council
SARDI	South Australian Research and Development Institute
SCUBA	Self Contained Underwater Breathing Apparatus
SEMP	Shellfish Environmental Monitoring Program
SEPP	State Environment Protection Policy
SPCC	State Pollution Control Commission
STP	Sewage Treatment Plant
TAG	Technical Advisory Group
Tas.	Tasmania
TBT	Tributyltin
TCM	Total Catchment Management
TEMP	Tuna Environmental Monitoring Program
UNEP	United Nations Environment Program
USEPA	United States Environment Protection Agency
Vic.	Victoria
WA	Western Australia