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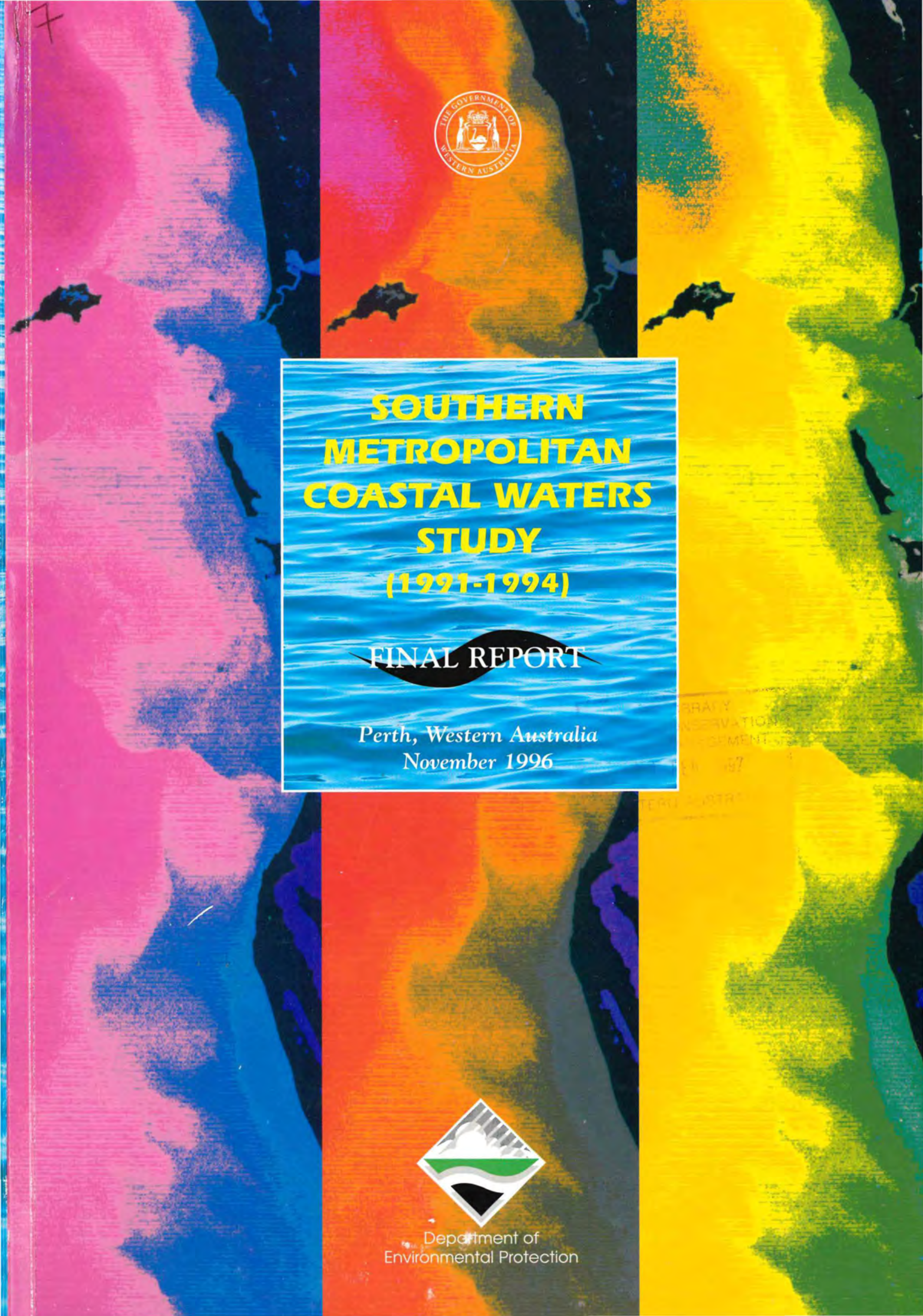


SOUTHERN METROPOLITAN COASTAL WATERS STUDY (1991-1994)

FINAL REPORT

Perth, Western Australia
November 1996

Department of
Environmental Protection

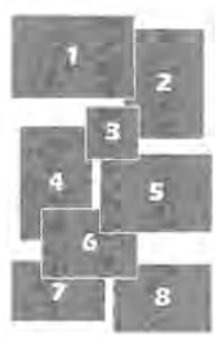
Front cover

Based on Landsat Thematic Mapper pseudo colour image of the waters off Perth recorded from an altitude of 720 km at 0930, 14 August 1991. (see Plate 5.1-1, c and d)

Back cover

A compilation of photographs taken in the southern metropolitan coastal waters of Perth as part of the Department of Environmental Protection's 'Snap a Snapper' photographic competition.

1. Sea lion by Pip O'dell
2. Urchin on seagrass by Pip O'dell
3. Stinging anemone on soft sediment by Peter Nicholas
4. Little Scorpionfish in seagrass by Pip O'dell
5. Tube anemone on soft sediment by Peter Nicholas
6. Starfish on reef by Margaret Nicholas
7. Starfish in sand by Anne Storrie
8. Breaksea Cod by Peter Nicholas



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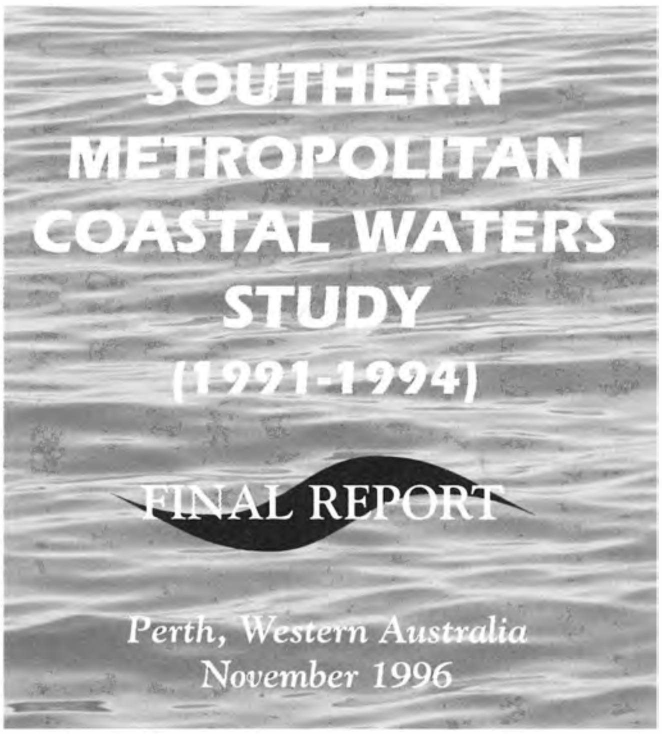

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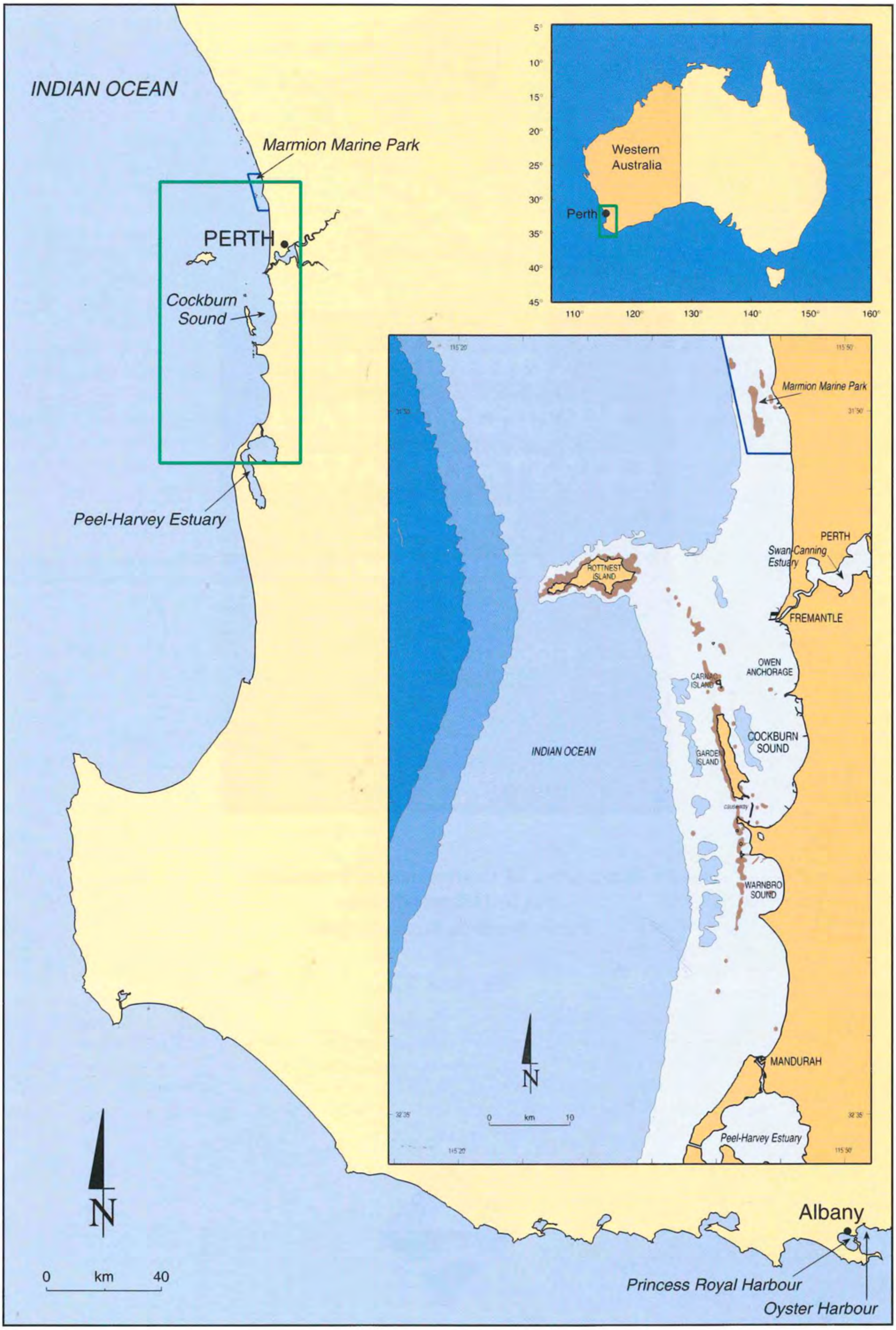
Report 17

Department of
Environmental Protection


INDIAN OCEAN
Marmion Marine Park
PERTH
Cockburn Sound
Peel-Harvey Estuary
Western Australia
Perth
5°
10°
15°
20°
25°
30°
35°
40°
45°
110°
120°
130°
140°
150°
160°
Marmion Marine Park
PERTH
Swan-Canning Estuary
FREMANTLE
ROTTNEST ISLAND
OWEN ANCHORAGE
CARNAC ISLAND
COCKBURN SOUND
GARDEN ISLAND
INDIAN OCEAN
WARNBRO SOUND
MANDURAH
Peel-Harvey Estuary
N
0 km 10
N
0 km 40
Albany
Princess Royal Harbour
Oyster Harbour

PREFACE

In the late 1960s to early 1970s most of the seagrass meadows in Cockburn Sound were severely degraded as a result of nutrient enrichment of the waters of the Sound. Sediments, water and biota were also contaminated with toxicants and pathogens. These impacts were attributed to waste discharges to these waters since the mid-1950s. At the same time environmental problems were emerging in the Peel-Harvey Estuary as a result of contaminants entering the estuary in runoff from the surrounding agricultural catchments. These problems gradually worsened over the next decade. In 1986 the Department of Conservation and Environment released a report detailing a major loss of seagrass meadows and a general decline in the environmental quality of Princess Royal Harbour and Oyster Harbour in Albany. Once again, chronic waste discharges and agricultural inputs were considered to be responsible for these problems. In all cases the quality of these waterbodies was significantly degraded by the time major environmental studies were initiated. Since then several hundred million dollars have been invested in remedial measures and, although the environmental quality of all three areas has improved significantly, many of the natural attributes that were lost are unlikely to ever return completely.

The major lesson to come out of these case studies was that an improved understanding of the ecological response time-frames and long-term environmental implications of chronic waste inputs to aquatic environments was urgently required if a repetition of these problems was to be avoided in other parts of Western Australia. The need to take a more pro-active approach to the issue of waste inputs to the coastal waters of Perth was acknowledged in 1985 when the Environmental Protection Authority funded a three year post-graduate study into the nutrient dynamics of Perth's coastal waters. This study focussed on nitrogen dynamics as significant quantities of this substance enter the coastal waters off Perth from land-based sources and it is generally accepted to be the macronutrient that limits algal productivity in these waters. In 1987, the Environmental Protection Authority initiated a proposal to undertake a study into the long-term environmental implications of waste inputs to the southern metropolitan coastal waters. This initiative recognised the need to improve the knowledge base required for a more strategic approach to waste management issues in Western Australia's marine environment as well as the need to address continuing concerns about the impact of waste discharges to Cockburn Sound. The Southern Metropolitan Coastal Waters Study began in mid-1991 and this report is an outcome of this initiative.

In 1990 the Environmental Protection Authority assessed a proposal from the Water Authority of Western Australia (WAWA) to construct and operate a second domestic wastewater ocean outlet into the waters of Marmion Marine Park. The environmental impact assessment concluded that the proposal was environmentally acceptable provided the nutrient loadings from both outfalls did not exceed the maximum loads set for the original single outfall. This decision was based on the conclusion that the short- and long-term impacts of increased contaminant loadings to these waters could not be predicted with adequate certainty. The Environmental Protection Authority also indicated that in the event of the WAWA (or now the newly formed Water Corporation) submitting further proposals to increase loads to the coastal waters of Perth, it would be necessary for the environmental implications of domestic wastewater discharges to be better understood. The Environmental Protection Authority therefore recommended that the WAWA undertake further studies. As a result, the WAWA initiated the Wastewater 2040 studies which included the Perth Coastal Waters Study (PCWS). This study focussed on the current and potential environmental effects of domestic wastewater discharges to the marine environment off Perth to the year 2040. Companion studies examined alternatives to ocean discharge and community attitudes to the issue of domestic waste disposal.

The Southern Metropolitan Coastal Waters Study (SMCWS) and the Wastewater 2040 studies were complementary studies initiated to provide the information necessary to address current problems and facilitate strategic environmental planning and management of waste inputs to Perth's coastal waters.

This report presents a summary and recommendations of the Southern Metropolitan Coastal Waters Study which was conducted from 1991 to 1994. A brief report of the Study is also outlined in a short companion document to facilitate wider distribution of the findings, actions and recommendations (SMCWS Summary Report, Department of Environmental Protection, 1996). These two reports have been prepared not only for those responsible for making management decisions but also for private organisations and individuals with an interest in the marine environment off Perth.

PRECIS

The coastal waters and beaches of Perth are major community assets to the people of Western Australia and significant attractions to overseas visitors. The biogeographic location of these waters results in an interesting mixture of tropical and temperate marine species and the local geomorphology provides a wide variety of coastal features ranging from protected embayments and lagoons to more exposed beaches, offshore reefs and islands. These features combined with the high water clarity and Mediterranean climate are the reasons why many of the recreational pursuits of the people of Perth are focussed on the coastal environment. In addition to recreation, these waters are used for commercial activities such as fishing and shipping and also as receiving waters for the disposal of domestic and industrial wastes.

The Southern Metropolitan Coastal Waters Study (SMCWS) was initiated to address the increasing threats to the environmental quality of these waters as a result of waste inputs from the rapidly developing areas to the south of Perth. It was considered that, despite the application of waste minimisation, recycling and reuse procedures, certain waste streams (e.g. domestic wastewater) would continue to increase in proportion with population growth, at least in the short term. Furthermore, the option of diverting these waste streams away from ocean discharge would have large infrastructure implications and hence would require considerable lead times. Certain other diffuse source waste inputs (e.g. from agricultural runoff and groundwater discharge) are known to be more difficult to manage and can be reduced only over long time scales through sustained catchment and groundwater management programmes. By contrast, the remaining major industrial waste dischargers to Cockburn Sound are planning to cease direct effluent discharges to these waters by early next century.

As outlined in the Progress Report of the SMCWS (Simpson *et al.* 1993), the study sought to provide a clearer understanding of the current and projected environmental quality status of these coastal waters in relation to the effects of waste discharge and to identify, as far as possible, the long-term ecological consequences of these discharges. The information base provided by the study has given new impetus to the development of management strategies to alleviate existing environmental problems due to waste discharges in areas such as Cockburn Sound and, perhaps even more importantly, to prevent the onset of such problems in other areas which still enjoy a high level of environmental quality.

The studies have significantly improved the knowledge of background levels and natural variability in Perth's marine environment and of key processes in relation to waste discharge and its chronic effects. The characterisation studies, process-related studies and modelling studies carried out as part of the SMCWS provide an improved basis for iterative, adaptive environmental management of these waters, based on monitoring, compliance testing and prediction. There is, however, an ongoing need to improve on the information base and prediction techniques to reduce the inherent uncertainties associated with managing natural systems; this reduction in uncertainty is to the benefit of environmental users and regulators alike.

The formulation of an environmental management strategy for Perth's coastal waters requires a clearly enunciated set of environmental quality objectives, based on the ecological and cultural values of these waters that are to be protected, both for present and future generations. The Environmental Protection Authority is undertaking a consultative process to finalise the draft Environmental Quality Objectives (EQOs), outlined in Chapter 3, and apply them to these waters. This process is an integral part of a key recommendation of the study to develop a formal Environmental Protection Policy for Perth's coastal waters.

An important part of the SMCWS, particularly in relation to the EQO process, has been to develop and assemble a set of draft quantitative environmental quality criteria and indicators for the study area which, if satisfied, will help ensure the long-term sustainability of the ecological and community values of these waters. The indicators, used in conjunction with well-designed monitoring programmes, will also provide early warning signs of the onset of environmental impacts, and hence facilitate prompt management action.

Apart from Cockburn Sound, eastern Owen Anchorage and in the vicinity of the Cape Peron wastewater outfall and the estuarine entrances, the southern metropolitan coastal waters of Perth are in a 'close to pristine' condition. The water quality is high, apart from when these waters are influenced by estuarine plumes, and the seagrass communities are generally healthy, other than in southern Cockburn Sound and eastern Owen Anchorage. Toxic contamination of waters, sediments and biota by heavy metals, polychlorinated biphenyls, pesticides and hydrocarbons is generally low by world standards and, apart from some areas in Cockburn Sound and Owen Anchorage, poses little threat to marine life and public health. The microbiological quality of the waters along the coastline is well within health criteria and these waters are generally safe for swimming and related activities.

The outflows from the Peel-Harvey and Swan-Canning estuaries have a widespread impact on the water quality of the southern metropolitan coastal waters, particularly during winter. Nutrient concentrations and phytoplankton abundance are elevated significantly above background during periods of high river flow which usually occur between July and September.

The problems associated with the heavy metal, hydrocarbon and pathogen contamination of areas of Cockburn Sound and Owen Anchorage in the 1970s are greatly diminished due to reduced inputs of these contaminants as a result of better waste treatment practices and cessation of discharges following relocation of some industries. Benthic invertebrate faunal communities in the northern half of the Cockburn Sound basin appear to be recovering.

A major environmental issue in Cockburn Sound is the poor water quality caused by continuing nutrient enrichment of these waters. Under these conditions seagrass meadows, particularly in southern Cockburn Sound, are continuing to decline due to light starvation, albeit at a much slower rate than in the 1960s and 1970s, and there is no evidence of *Posidonia* seagrass recovery in the sound. This study has determined the light requirements for *Posidonia* seagrass meadows to survive and has specified the improvements in water clarity and quality conditions which, if achieved, would enable these light requirements to again be met on the eastern margin of the sound, where broad expanses of these meadows once thrived. Significant reductions in nutrient inputs are required to achieve these improved water quality conditions, and this would be a pre-requisite to successful restoration of this area, should the appropriate technologies become available.

There have also been significant losses of seagrass meadow in areas of Owen Anchorage since the early 1980s, apart from the direct losses due to limesands mining. These losses are possibly due to a combination of natural and anthropogenic influences.

Nutrient inputs from the Cape Peron wastewater outfall into Sepia Depression are causing significant increases in nutrient concentrations and phytoplankton abundance up to about five kilometres from the outfall. The long-term ecological implications of these changes are unknown but the very presence and extent of the changes indicate that proposals to further increase nutrient loads to these waters from this source should be treated with extreme caution.

The use of organotin-based antifouling paints on the hulls of commercial ships and recreational vessels operating in Perth's coastal waters has led to extensive tributyltin (TBT) contamination of the sediments and biota of the study area. Levels of TBT in mussels in some areas exceed health criteria and may pose a public health risk if the mussels are consumed. A reproductive disorder in a bioindicator species, known to be caused by TBT contamination, was widespread throughout the study area and experience from many parts of the world suggests that other marine species in Perth's coastal waters may be seriously affected by this substance. Large ships and associated maintenance facilities are the likely current major source of TBT contamination to these waters.

The introduction of foreign organisms into Australian waters has had widespread ecological and economic impacts. At least eighteen species are thought to have been introduced to Perth's coastal waters via the disposal of ships' ballast water and sediment or from dislodgement off the hulls of vessels. The disposal of ballast waters and continuing TBT contamination represent serious threats to the ecological and cultural attributes of these waters. A coordinated effort by international, national and state agencies will be required to address these problems.

Many of the commercial and recreational activities in the metropolitan coastal waters rely on the maintenance of the ecological integrity of these ecosystems and this can only be achieved by understanding the cumulative long-term environmental implications of human activities on these waters. In addition this information must be integrated into a strategic environmental planning process and into appropriate management structures. The current sectoral approach to environmental management of these waters is unlikely to achieve the required level of coordination and integration. As such there is a need to develop more appropriate management frameworks for Perth's coastal waters if a repetition of the problems that have occurred in the coastal environments of many cities around the world are to be avoided.

EXECUTIVE SUMMARY

This report presents the findings and recommendations of the Southern Metropolitan Coastal Waters Study 1991-1994 (SMCWS) and synthesizes the results of many inter-related technical and scientific investigations by a wide cross-section of the marine scientific community, drawn from State, Commonwealth and local government agencies, Universities and private consultants. The study arose from a need, identified by the Western Australian Environmental Protection Authority, for a better technical basis from which to manage existing and projected waste inputs to the marine environment of Perth.

The primary aims of the SMCWS were to develop an understanding of the cumulative impacts and long-term environmental consequences of contaminant inputs to these waters and to facilitate the development of a comprehensive environmental management strategy for the southern metropolitan coastal waters of Perth. The study area encompassed the waters from Fremantle to Mandurah and west from the mainland coast to the mid continental shelf in recognition of the spatial scales of the projected urban development of Perth over the coming decades and of the physical and biological processes in these waters. Particular emphasis was placed on the partially-enclosed 'core' areas of Cockburn Sound, Owen Anchorage and the Shoalwater Islands Marine Park, including Warnbro Sound.

The following major outcomes were achieved:

- quantification of past, current and projected contaminant inputs;
- determination of the current status of the environment;
- establishment of a baseline for the detection of future environmental changes;
- a greater understanding of the processes controlling the movement of water-borne contaminants;
- a greater understanding of the key ecological processes underlying the relationships between human activities and environmental quality;
- development of draft Environmental Quality Criteria for these waters, and;
- formulation of actions and recommendations to address existing and potential environmental problems.

The formal establishment of Environmental Quality Objectives (EQOs) for the entire metropolitan coastal waters of Perth will provide the basis for long-term planning and environmental management of existing and proposed waste inputs to these waters. The conclusions and recommendations in this report are based on draft EQOs, proposed by the EPA and DEP, in regard to regulating waste inputs to these waters.

Nutrient inputs, water quality and seagrass health

An extensive programme of physical, chemical and biological water quality monitoring was conducted in the 'core' areas and throughout the wider SMCWS area. These data were used to characterise 'natural' or background water quality conditions and to assess the effects of anthropogenic point and diffuse source inputs of nutrients on ambient water quality.

Wider metropolitan coastal waters

During high river flow periods the Peel-Harvey and Swan-Canning estuary outflows were spread widely and caused significant elevations of nutrient concentrations in seawater over much of the nearshore study area. Phytoplankton levels were also elevated above background during these periods and phytoplankton populations have been dominated by a single species of silicoflagellate in recent years. This dominance is of concern since 'blooms' of a closely related silicoflagellate species have been linked to broadscale eutrophication of coastal waters in Europe. Water quality at other times of the year is generally high throughout the southern metropolitan coastal waters, apart from Cockburn Sound and near domestic wastewater outfalls and estuary mouths.

Total annual nitrogen loading into Perth's wider coastal waters is currently about 6000 tonnes, mostly from ocean disposal of domestic wastewater, from estuarine outflows, and to a lesser extent, from groundwater and industrial inputs. Current projections indicate that annual nitrogen loadings will increase to 10 000 tonnes by 2021, mainly from domestic wastewater disposal. The study findings suggest that widespread, nutrient-related changes are already occurring in these waters during winter. The long-term ecological impacts of the projected nutrient inputs are unknown and extreme caution should be exercised in considering further proposals involving nutrient discharge to these waters.

Cockburn Sound

The current water quality, as measured by chlorophyll *a* concentrations and light attenuation, of Cockburn Sound is only slightly better than during the late 1970s, when the water quality of the Sound was in its poorest recorded state. The presence of a high biomass of epiphytes on the leaves of seagrass in southwest Cockburn Sound, and the chronic high phytoplankton biomass clearly illustrate the eutrophic condition of the sound. Inorganic nitrogen is the macronutrient limiting algal growth in these waters and inputs need to be significantly reduced if the water quality of the sound is to improve. The Contaminants Input Inventory indicates that direct industrial discharges of nutrients to Cockburn Sound have been significantly reduced in recent years. Furthermore groundwater sources now represent 70% of the total external loading of nitrogen to the sound and about 80% of this load originates from contaminated groundwater under industrial estates.

Seagrass dieback in Cockburn Sound has slowed considerably since the early 1970s, although some losses are still occurring, particularly in the southern end of the sound. There is no evidence of *Posidonia* seagrass recovery in the sound.

Owen Anchorage

There has been significant improvement in the nutrient-related water quality of Owen Anchorage since the 1970s, when the area was in its poorest recorded state. This is largely due to a relatively recent reduction of direct nitrogen inputs to these waters from industrial sources. The water quality of Owen Anchorage declines from about July to September and this appears to be caused mainly by nutrient-enriched outflows from the Swan-Canning Estuary.

There have been significant losses of seagrasses on parts of Parmelia and Success banks since the early 1970s due to shellsand mining operations, smothering by mobile sand sheets and nutrient pollution, particularly on the eastern side of Owen Anchorage.

Shoalwater Islands Marine Park

The water quality of Warnbro Sound is high and appears to be largely unchanged since the late 1970s. However, nutrient inputs from remote sources such as the Peel-Harvey Estuary have been identified and this may have important implications for long-term management of the water quality within the Shoalwater Islands Marine Park.

Seagrass meadows are generally in good 'health' although small losses have occurred in some areas, probably due to the natural process of sediment smothering. Seagrasses in the northern part of Warnbro Sound appear to be under moderate nutrient stress.

Sepia Depression (> 5 km south of the Cape Peron wastewater outfall)

The water quality of Sepia Depression, further than 5 km south of the Cape Peron wastewater outfall, during the non-winter months is high and appears to have remained largely unchanged since the late 1970s. During July to September, however, the water quality in this area declines due predominantly to the influence of the Peel-Harvey Estuary outflow.

Sepia Depression (within 5 km of the Cape Peron wastewater outfall)

In recent years, significantly elevated (above background) nutrient and phytoplankton concentrations have been regularly recorded during the non-winter months within about 5 km of the Cape Peron wastewater outfall. Nutrient loadings from this outfall site are projected to increase rapidly over the coming decades (i.e. in proportion with the population growth of the southern metropolitan area) if current treatment and disposal practices remain. When the recent changes in water quality are considered in the context of these projected loading increases, there is cause for considerable concern.

Dawesville Channel

Preliminary water quality studies by the Department of Transport, near the Dawesville Channel, indicate that nutrient and chlorophyll *a* concentrations are significantly elevated above background levels for the southern metropolitan coastal waters, particularly in the area between the Channel and the southern part of Comet Bay. These elevations are a direct result of the outflow from the Peel-Harvey Estuary and indicate that the ouflows are having a measurable effect on the surrounding coastal waters. The ecological impact of these outflows on adjacent benthic communities is currently unknown.

Toxic contamination

Extensive baseline surveys of organochlorine and organophosphate pesticides, polychlorinated biphenyls (PCBs), hydrocarbons, organotin compounds and heavy metals in sediments and the tissue of the blue mussel *Mytilus edulis* were undertaken during the study. The concentrations of pesticides, PCBs, aliphatic and polycyclic aromatic hydrocarbons in sediments and mussels throughout the study area were generally very low. Areas where these substances could be detected were generally within or immediately adjacent to harbours, marinas, wharves and ship maintenance facilities. The concentrations of all these substances throughout the study area were well below ecological and human health criteria, apart from PCBs and the pesticide DDT in sediments which exceeded the criteria at less than 1% and about 15% (most of which were just above the DDT criterion) of the sites respectively. These findings indicate that the southern coastal waters of Perth are not significantly contaminated with these substances.

Tributyltin

Organotin-based paints were developed to prevent marine fouling on the hulls of vessels. The use of these paints has led to serious environmental problems at many locations around the world as a result of contamination by tributyltin (TBT), the most active ingredient in these paints. TBT contamination in marine sediments and mussels (as an indicator of water contamination) of the study area is widespread with the highest contamination occurring in harbours, marinas and near industrial and naval wharves. Some values are amongst the highest recorded in Australia. The high frequency of occurrence of the reproductive disorder, *imposex*, in the mollusc *Thais orbita* indicates that this substance is having a significant biological impact in the coastal waters of Perth, including Rottnest Island. Whether other marine species are being affected is unknown but experience in other parts of the world indicate that this is highly likely. The concentrations of TBT in mussels also exceeded health criteria, primarily at sites in harbours and near the wharves in Cockburn Sound, indicating that the consumption of mussels from these sites poses a risk to human health.

The use of TBT-based antifouling paints was restricted in Western Australia by regulations, effective from 1 November 1991, which prohibit the use of this substance on boats under 25 m and restrict its usage to low leaching forms on boats over 25 m. Surveys conducted in 1991 and 1994 suggest that the regulations have been effective in minimising further TBT contamination from small boats. However, sediment TBT concentrations have increased significantly, since 1991, in areas predominantly associated with the use and maintenance of vessels over 25 m, suggesting that vessels over 25 m are the major current source of TBT to Perth's coastal waters.

The widespread contamination of TBT in the waters and sediments of the study area and the widespread deleterious effect on *Thais orbita,* combined with the extreme toxicity of this substance to a wide range of marine plants and animals, indicates that TBT contamination of the coastal waters is an issue of extreme concern.

Heavy metals

Heavy metal concentrations in the sediments and mussels of Perth's southern coastal waters are generally low and the concentrations of most metals have decreased significantly in Cockburn Sound and Owen Anchorage since the late 1970s, mainly as a result of substantial reductions in the discharge of heavy metals from industrial sources. Highest concentrations were found within harbours/marinas, along the eastern shoreline and the southern half of Cockburn Sound and, to a lesser extent, along the shoreline of Owen Anchorage.

The sediment concentrations of total arsenic and total mercury at some sites in Cockburn Sound and eastern Owen Anchorage exceeded the draft ecological criteria. Concentrations of heavy metals in mussels were generally below the values indicative of contamination, apart from zinc, which exceeded the indicative value at most sites in Cockburn Sound and eastern Owen Anchorage. The concentrations of all heavy metals included in the 1994 mussel survey were below the health criteria for edible seafood at every site within the study area.

The species richness of benthic invertebrate fauna is lowest in the deep basin sediments of southern Cockburn Sound where, as a result of present and past discharges, heavy metal accumulation in the sediments is highest. Species richness of the benthic fauna in Cockburn Sound is inversely correlated to a suite of contaminant concentrations in the basin sediments.

The concentrations of most heavy metals in the water, sediments and biota of Sepia Depression, near the Cape Peron outfall, are below ecological and human health criteria. Concentrations of mercury in water exceeded ecological and health criteria on occasions at sites up to one kilometre from the outfall. The concentrations of cadmium and zinc in mussels at sites up to four kilometres from the outfall exceeded the values indicative of contamination.

Microbiological quality of coastal waters and beaches

Regular monitoring carried out by the Department of Health and the City of Rockingham indicate that the waters of Perth's metropolitan beaches do not pose a risk to human health via direct contact recreation. Neither does there appear to be a health risk in relation to the consumption of seafood, with the exception of waters off Palm Beach and Safety Bay Jetty.

Results from microbiological surveys by the Water Corporation (formerly the WAWA) in Sepia Depression indicate that faecal coliform concentrations in water are occasionally above the human health criteria for direct contact recreation and for the maintenance of aquatic organisms for human consumption for up to about two and four kilometres, respectively, from the end of the Cape Peron outfall. These results suggest a potential public health risk exists in these waters over spatial scales approximately prescribed by these distances.

Introduction of foreign organisms

The introduction of foreign marine organisms has caused widespread biological and commercial impacts in many parts of the world. At least 18 foreign species have established in the coastal waters of Perth and are considered to have been introduced via discharge of ships' ballast water and associated sediments, or dislodged from the hulls of ships. In 1994, ships visiting the Port of Fremantle discharged an estimated 3.5 million tonnes of ballast water into port waters. Shipping activity in these waters is projected to increase significantly over the next decade; therefore the potential for the introduction of foreign organisms into the coastal waters of Perth from this source will increase proportionally if current ballast water disposal practices remain. Given the high conservation, recreational and commercial value of these waters, the introduction of foreign organisms via ballast water disposal represents a significant threat to the ecological and cultural values of the metropolitan coastal waters of Perth.

Direct Impacts

Shellsand mining

Cockburn Cement Ltd is currently mining shellsands from Success Bank for the manufacture of lime and cement. This operation removes seagrasses along with the shellsand and increases the water depth to seabed from about 5 to 15 m. Since operations began in 1972, approximately 100 ha of seabed have been modified and 90 ha of seagrass meadows have been removed. The environmental implications of Cockburn Cement's long-term mining operation on the Owen Anchorage/Cockburn Sound ecosystem area are currently being determined through a separate EPA/DEP process. Consequently, no recommendations are made on the long-term impacts of shellsand mining in Owen Anchorage in this report.

Management

With more than 70% of the population of Western Australia living within 20 km of the metropolitan coastline, the coastal waters of Perth are one of the most used and, therefore, 'valuable' parts of Western Australia's marine environment. Perth's population is expected to increase by about 50% over the next 30 years resulting in significant industrial and urban expansion along the coastal strip and increasing recreational activity in the coastal waters. This growth will inevitably lead to greater demands being placed on Perth's marine environment. As these demands increase, so too does the need to provide sound environmental management.

The primary objective of the SMCWS and PCWS was to provide the information required for improved management of current impacts and better strategic planning to prevent a repetition of the problems that have occurred in the marine waters of many coastal cities around the world. Currently there is no formal integrating management framework for these waters. Instead, environmental management is presently the responsibility of numerous individual agencies, operating across four jurisdictions, ranging from local government by-laws to international treaties. The current situation does not provide the level of integration necessary to ensure the multiple use of these waters is both socially equitable and ecologically sustainable. Such integration would give due recognition to all the major linkages between land-based activities and coastal water quality and would seek, through agreement between the local, state and the commonwealth government, to encompass the important, natural spatial and temporal scales of contaminant transport and ecological effect throughout Perth's coastal waters.

PROPOSED ACTIONS AND RECOMMENDATIONS

Findings from the Southern Metropolitan Coastal Waters Study have given rise to a number of proposed Department of Environmental Protection (DEP) actions and recommendations to the Environmental Protection Authority (EPA). The background and rationale for the actions and recommendations are presented in Chapter 7.

The proposed DEP action items list those actions the DEP would take, subject to State Government endorsement, to address existing environmental problems or concerns. The DEP recommendations to the EPA are primarily concerned with the adoption of policy positions aimed at addressing specific environmental issues or problems identified in the Study, as well as the broader need for long-term protection of the marine environment through coordinated environmental management of Perth's coastal waters.

Problems or issues are considered to arise where the draft Environmental Quality Objectives (EQOs) for Perth's southern metropolitan coastal waters, as expressed by the associated draft Environmental Quality Criteria, are not being met, or appear unlikely to be met in the future if current trends continue. The conclusions and subsequent proposed actions and recommendations in this report are based on the draft EQOs, proposed by the DEP and EPA, in regard to regulating waste inputs to these waters.

The draft EQOs are discussed in detail in Chapter 3 and will be finalised after a separate, consultative process being undertaken by the EPA. The desired outcome of this consultative process, which will involve the public and all major users of Perth's marine environment, is an agreed set of EQOs, associated criteria and zones where the EQOs will apply. As a result of this process, some of the following DEP proposed actions and recommendations to the EPA may change.

PROPOSED DEP ACTIONS

Subject to State Government endorsement, the DEP would implement the following actions. Several of these actions refer to Part V of the E P Act (ie. Part V of the Environmental Protection Act, 1986-1993) which provides for the DEP to grant or refuse a works approval for premises with a known potential to pollute. Part V also provides for the DEP to grant (subject to conditions, eg. monitoring requirements) or refuse a waste discharge licence for such premises.

Action 1

Under Part V of the E P Act, the Department of Environmental Protection will require that major contributors of nutrients to Cockburn Sound implement a nutrient management strategy to ensure that the draft Environmental Quality Objective 2 (i.e. the maintenance of ecosystem integrity) for these waters (excluding designated exclusion zones) is achieved by 31 March 2002, with appropriate annual environmental performance indicators.

Action 2

Under Part V of the E P Act, the Department of Environmental Protection will require that major contributors of nutrients to Cockburn Sound jointly undertake annual water quality monitoring programmes in Cockburn Sound until the draft Environmental Quality Objective 2 (i.e. the maintenance of ecosystem integrity) for these waters (excluding designated exclusion zones) is finalised, and the finalised EQO 2 is achieved and maintained for at least two years. Future monitoring requirements will be reviewed at this time.

Action 3

Under Part V of the E P Act, the Department of Environmental Protection will not issue works approvals or licenses to increase nutrient loads, particularly nitrogen, to Cockburn Sound until the draft Environmental Quality Objective 2 (i.e. the maintenance of ecosystem integrity) for these waters (excluding designated exclusion zones) is finalised, and the finalised EQO 2 is achieved and maintained for at least two years at levels that would permit consideration of further loadings.

Action 4

The Department of Environmental Protection will request the Water and Rivers Commission to extend its water quality monitoring programmes of the Swan-Canning Estuary to include the waters of Owen Anchorage/Gage Roads.

Action 5

Under Part V of the E P Act, the Department of Environmental Protection will require that the Water Corporation implements a nutrient management strategy to ensure that draft Environmental Quality Objective 2 (i.e. the maintenance of ecosystem integrity) for waters of Sepia Depression influenced by the Cape Peron outfall (excluding designated exclusion zones), is achieved by 31 March 2002, with appropriate annual environmental performance indicators.

Action 6

Under Part V of the E P Act, the Department of Environmental Protection will require that the Water Corporation revises the environmental monitoring programmes for the Cape Peron outfall, by 31 December 1997, consistent with Action 5.

Action 7

Under Part V of the E P Act, the Department of Environmental Protection will not issue works approvals or licenses to increase nutrient loads, particularly nitrogen, to Sepia Depression until the draft Environmental Quality Objective 2 (i.e. the maintenance of ecosystem integrity) for these waters (excluding designated exclusion zones) is finalised, and the finalised EQO 2 is achieved and maintained for at least two years at levels that would permit consideration of further loadings.

Action 8

The Department of Environmental Protection will request the Water and Rivers Commission and the Department of Transport to implement monitoring programmes in the nearshore coastal waters influenced by the Dawesville and Mandurah Channel outflows and to provide annual performance indicators in relation to the catchment management objective outlined in Recommendation 6.

Action 9

The Department of Environmental Protection will recommend to the Minister for the Environment that the Western Australian Government request the Australian Government to initiate further action with international agencies to prohibit the use of tributyltin-based antifouling paints on all vessels, or reduce allowable tributyltin release rates to levels that would achieve the criteria for draft Environmental Quality Objective 2 (i.e. maintenance of ecosystem integrity) and draft Environmental Quality Objective 3 (i.e. maintenance of aquatic life for human consumption) in Perth's coastal waters.

Action 10

The Department of Environmental Protection will recommend that the Western Australian Minister for the Environment request ANZECC to review its existing, recommended tributyltin release rate for antifouling paints used on Australian registered vessels (including naval vessels) greater than 25 m in length, with a view to prohibiting the use of this substance or reducing the allowable tributyltin release rate to levels that would achieve the criteria for draft Environmental Quality Objective 2 (i.e. maintenance of ecosystem integrity) and draft Environmental Quality Objective 3 (i.e. maintenance of aquatic life for human consumption) in Perth's coastal waters.

Action 11

The Department of Environmental Protection will recommend to the Minister for the Environment that the Western Australian Government coordinate the implementation of incentives to encourage 'TBT-free' ships to Western Australian ports.

Action 12

The Department of Environmental Protection will identify and license ship-maintenance and ship-building facilities that use or remove organotin compounds so that waste materials containing organotin are disposed of in an environmentally acceptable way.

Action 13

The Department of Environmental Protection will request the Department of Health to investigate the potential health implications of the exceedances of the tributyltin criterion for draft Environmental Quality Objective 3 (i.e. maintenance of aquatic life for human consumption) and, if necessary, implement a public health risk minimisation strategy.

Action 14

Under Part V of the E P Act, the Department of Environmental Protection will require that the major contributors of arsenic to Perth's southern metropolitan coastal waters investigate the ecological implications of the current levels of arsenic in sediments with a view to the development of arsenic criteria, and implement (if necessary) an arsenic management strategy, with appropriate annual environmental performance indicators, to ensure draft Environmental Quality Objective 2 (i.e. maintenance of ecosystem integrity), for areas influenced by their discharges (excluding designated exclusion zones), is achieved by 31 December 1999.

Action 15

Under Part V of the E P Act, the Department of Environmental Protection will require that major contributors of zinc to Perth's southern metropolitan coastal waters investigate the ecological implications of the current levels of zinc in mussels with a view to the development of zinc criteria, and implement (if necessary) a zinc management strategy, with appropriate annual environmental performance indicators, to ensure draft Environmental Quality Objective 2 (i.e. maintenance of ecosystem integrity), for areas influenced by their discharges (excluding designated exclusion zones), is achieved by 31 December 1999.

Action 16

That, in relation to possible synergistic effects of heavy metal and polycyclic aromatic hydrocarbon contamination on benthic faunal communities, the Department of Environmental Protection will require that major dischargers of these substances to Cockburn Sound conduct investigations (e.g. ecotoxicological) to evaluate this possibility, develop criteria as appropriate and implement a management strategy as required by 31 December 2001.

Action 17

Under Part V of the E P Act, the Department of Environmental Protection will require that current major contributors of heavy metals and polycyclic aromatic hydrocarbons to Cockburn Sound jointly undertake triennial monitoring programmes of basin sediment contamination and benthic community structure in Cockburn Sound from 1998 until the relevant criteria are met or until the input of these contaminants to Cockburn Sound from these contributors ceases.

Action 18

The Department of Environmental Protection will request the Water Corporation to implement a contaminant management strategy, with appropriate annual environmental performance indicators, to minimise contaminant inputs to the Shoalwater Islands Marine Park from the Waikiki and Forrester Road main drains.

Action 19

Under Part V of the E P Act, the Department of Environmental Protection will require that, with respect to the operation of the Cape Peron ocean outfall, the Water Corporation investigates the ecological implications of the current levels of cadmium in mussels with a view to developing cadmium criteria and implement (if necessary) a cadmium management strategy, with annual environmental performance indicators, to ensure that draft Environmental Quality Objective 2 (i.e. maintenance of ecosystem integrity) and draft Environmental Quality Objective 3 (i.e. maintenance of aquatic life for human consumption) for areas of Sepia Depression influenced by the Cape Peron outfall (excluding designated exclusion zones) are achieved, by 31 March 2002.

Action 20

Under Part V of the E P Act, the Department of Environmental Protection will require that the Water Corporation implements a faecal coliform monitoring and management strategy, with annual environmental performance indicators, to ensure that criteria for draft Environmental Quality Objective 3 (i.e. maintenance of aquatic life for human consumption) and draft Environmental Quality Objective 4 (i.e. maintenance of recreation values) for areas of Sepia Depression influenced by the Cape Peron outfall (excluding designated exclusion zones) are achieved by 31 March 2002.

Action 21

The Department of Environmental Protection will request the Department of Health to investigate the cause/s of the chronic exceedance of the faecal coliform criterion for draft Environmental Quality Objective 3 (i.e. maintenance of aquatic life for human consumption) at Palm Beach and Safety Bay and, if necessary, implement a management strategy to reduce faecal contamination to levels consistent with this criterion.

Action 22

The Department of Environmental Protection will request the Department of Health to assess the microbiological water quality at areas commonly used both for boat mooring and direct contact recreation, at Rottnest Island, in association with the Rottnest Island Authority, and at Garden Island, in association with the Commonwealth Department of Defence and, if necessary, to implement a management strategy to reduce faecal contamination to levels consistent with draft Environmental Quality Objective 3 (i.e. maintenance of aquatic life for human consumption) and draft Environmental Quality Objective 4 (i.e. maintenance of recreation values).

Action 23

The Department of Environmental Protection will recommend to the Minister for the Environment that the Western Australian Government request the Australian Government to encourage the International Maritime Organisation to expedite, as a matter of high priority, finalisation of an Annex to the International Convention for the Prevention of Pollution from Ships (MARPOL 1973/78) requiring new vessels, especially bulk carriers and tankers, to have upgraded ballast water management systems.

Action 24

The Department of Environmental Protection will recommend to the Minister for the Environment that the Western Australian Government request the Australian Government to encourage the International Maritime Organisation to research strategies, such as in-transit sterilisation of ballast waters, to minimise risk of introduced organisms from ballast water discharge to Australian waters.

Action 25

The Department of Environmental Protection will recommend to the Minister for the Environment that the Western Australian Government request the Australian Government to implement incentives to encourage ships with appropriate ballast water management systems.

Action 26

The Department of Environmental Protection will request the Department of Transport and Fremantle Port Authority to further encourage ship operators to adopt the guidelines recommended in the Australian *Draft Ballast Water Management Strategy*.

Action 27

The Department of Environmental Protection will request the Department of Transport and Fremantle Port Authority to jointly examine the Australian Draft Ballast Water Management Strategy and implement practical measures as soon as possible.

Action 28

The Department of Environmental Protection will request the CSIRO Centre for Research on Introduced Marine Pests to give high priority to research activities related to the formulation of ballast water risk minimisation strategies.

Action 29

The Department of Environmental Protection will request the Inter-Departmental Committee on Aquaculture to develop guidelines to ensure that the cumulative impacts of existing aquaculture ventures are taken into account when evaluating new aquaculture proposals, and to assist in achieving/maintaining ecosystem integrity, recreation values and aesthetic values in the southern coastal waters of Perth.

Action 30

Under Part V of the E P Act, the Department of Environmental Protection will require that the Water Corporation implements a management strategy, with appropriate annual environmental performance indicators, to ensure that draft Environmental Quality Objective 5 (i.e. maintenance of aesthetic values), for areas of Sepia Depression influenced by the Cape Peron outfall (excluding designated exclusion zones), is achieved by 31 March 2002.

Action 31

The Department of Environmental Protection will request the Fremantle Port Authority, in collaboration with major port users, to implement codes of practice for vessel wash down and cargo handling operations to further reduce and minimise impacts on the aesthetic quality of Perth's coastal waters from these operations in port waters.

Action 32

The Department of Environmental Protection will recommend to the Minister for the Environment that the Western Australian Government establish a formal framework to coordinate environmental management within Perth's metropolitan coastal waters (nominally Dawesville Channel to Yanchep) and between these waters and their land catchments.

Action 33

The Department of Environmental Protection will recommend to the Minister for the Environment that the Western Australian Government develop a Memorandum of Understanding with the Australian Government to facilitate coordinated environmental management of State and Commonwealth waters off Perth.

Action 34

The Department of Environmental Protection will recommend to the Minister for the Environment that the Western Australian Government consider including the Rottnest Island Aquatic Reserve within the statewide marine conservation reserve system under the CALM Act 1984.

Action 35

The Department of Environmental Protection will support the proposed extension of the Shoalwater Islands Marine Park, northward off the western shores of Garden Island and Carnac Island, and seaward to the limit of the State Territorial Sea, consistent with the report and recommendations of the Marine Parks and Reserves Selection Working Group (1994).

Action 36

The Department of Environmental Protection will ensure further studies are undertaken to determine whether the winter silicoflagellate blooms observed in Perth's coastal waters over recent years are occurring in comparable temperate waters of Western Australia that are not significantly influenced by anthropogenic waste inputs.

Action 37

The Department of Environmental Protection will ensure that monitoring of sediments and biota for tributyltin in Perth's coastal waters is conducted every three years or until draft Environmental Quality Objective 2 (i.e. maintenance of ecosystem integrity) and draft Environmental Quality Objective 3 (i.e. maintenance of aquatic life for human consumption) are achieved.

Action 38

The Department of Environmental Protection will ensure that the elevated mercury levels in the sediments of Cockburn Sound and Owen Anchorage are further investigated, with a view to identifying the sources, the ecological implications and the necessity for remedial action.

Action 39

The Department of Environmental Protection will ensure that the health and distribution of seagrass meadow communities in Cockburn Sound and Owen Anchorage are monitored annually and triennially, respectively.

Action 40

The Department of Environmental Protection will request the CSIRO to continue and expand its long-term water quality monitoring programme in Perth's coastal waters.

Action 41

The Department of Environmental Protection will ensure that an inventory of contaminant inputs to Perth's coastal waters is updated annually and made publicly available.

RECOMMENDATIONS TO THE EPA

The DEP recommends that the EPA adopt the following policy positions with respect to managing specific environmental issues and problems and ensuring the long-term protection and strategic environmental management of the marine environment:

Recommendation 1

That environmental protection policies and integrated catchment management strategies for the catchments of the Swan-Canning and Peel-Harvey estuaries incorporate the objective of minimising nutrient inputs to marine waters to assist in achieving draft Environmental Quality Objective 2 (i.e. the maintenance of ecosystem integrity) for these waters (excluding designated exclusion zones).

Recommendation 2

That the Environmental Protection Authority adopt a presumption against any proposals to increase nutrient loads, particularly nitrogen, to Cockburn Sound until the draft Environmental Quality Objective 2 (i.e. the maintenance of ecosystem integrity) for these waters (excluding designated exclusion zones) is finalised, and the finalised EQO 2 is achieved and maintained for at least two years at levels that would permit consideration of further loadings.

Recommendation 3

That environmental protection policies and integrated catchment management strategies for the catchment of the Swan-Canning Estuary incorporate the objective of minimising nutrient inputs to Owen Anchorage to assist in maintaining draft Environmental Quality Objective 2 (i.e. the maintenance of ecosystem integrity) for these waters (excluding designated exclusion zones).

Recommendation 4

That environmental protection policies and integrated catchment management strategies for the catchment of the Peel-Harvey Estuary incorporate the objective of minimising nutrient inputs to the coastal waters to assist in maintaining draft Environmental Quality Objective 2 (i.e. the maintenance of ecosystem integrity) for the waters of the Shoalwater Islands Marine Park.

Recommendation 5

That the Environmental Protection Authority adopt a presumption against any proposals to increase nutrient loads, particularly nitrogen, to Sepia Depression until the draft Environmental Quality Objective 2 (i.e. the maintenance of ecosystem integrity) for these waters (excluding designated exclusion zones) is finalised, and the finalised EQO 2 is achieved and maintained for at least two years at levels that would permit consideration of further loadings.

Recommendation 6

That environmental protection policies and integrated catchment management strategies for the catchment of the Peel-Harvey Estuary incorporate the objective of minimising nutrient inputs to coastal waters to assist in achieving draft Environmental Quality Objective 2 (i.e. the maintenance of ecosystem integrity) for the coastal waters influenced by the Dawesville and Mandurah Channel outflows.

Recommendation 7

That the Environmental Protection Authority adopt a presumption against any proposals that would increase nutrient loads, particularly nitrogen, to the nearshore coastal waters influenced by the Dawesville and Mandurah Channel outflows until the draft Environmental Quality Objective 2 (i.e. the maintenance of ecosystem integrity) for these waters (excluding designated exclusion zones) is finalised, and the finalised EQO 2 is achieved and maintained for at least two years at levels that would permit consideration of further loadings.

Recommendation 8

That the Environmental Protection Authority recommend to the Minister for the Environment that the Western Australian Government establish a formal framework to coordinate environmental management within Perth's metropolitan coastal waters (nominally Dawesville Channel to Yanchep) and between these waters and their land catchments.

Recommendation 9

That the Environmental Protection Authority recommend to the Minister for the Environment that the Western Australian Government develop a Memorandum of Understanding with the Australian Government to facilitate coordinated environmental management of State and Commonwealth waters off Perth.

Recommendation 10

That the Environmental Protection Authority develop an Environmental Protection Policy for Perth's coastal waters.

Recommendation 11

That the Environmental Protection Authority formally review the Environmental Quality Objectives for Perth's coastal waters every seven years.

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- Swan River Trust
- The University of Canterbury (Department of Civil Engineering)
- The University of New South Wales (Departments of Mathematics, Water Engineering, Geography and Oceanography)
- The University of Western Australia (Departments of Botany, Zoology, Geography and the Centre for Water Research)
- Water Authority of Western Australia
- Waterways Commission
- Western Australian Museum (Division of Natural and Social Sciences)
- Western Australian Police Department (Water Police Branch)

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Department of Fisheries	T M Burden, R J Kirk and crews of the vessels *Phillip King* and *Flinders*
Royal Australian Navy (HMAS Stirling)	A Daish, P J Fowler, D Jenkins, D R Thomas, L J Webster and crews of the vessels *AWB4006* and YTM Tug *Tammar*
Royal Australian Navy (Diving)	Clearance Diving Team 4: C A Davies-Graham, R L Gallop, P W Koerber, N W Mader, G A Payne and crews of the vessels *NWB* and *D L Stark*
Western Australian Police Department	K J Doy and crew of the vessel *Cygnet IV*
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PART I

BACKGROUND

This part of the report consists of three chapters and presents background information about the study and the regional setting of the metropolitan coastal waters of Perth. Chapter 1 briefly outlines the environmental history of these waters and the factors that prompted the Study. The scope, objectives and organisational arrangements are also outlined. Chapter 2 provides the regional physical, biological and cultural setting of the Study and Chapter 3 outlines the draft environmental quality objectives and criteria for these waters. Locality names, bathymetry and major geomorphological features of the southern metropolitan coastal waters are found in Figures 1.1-1 and 1.1-2, and in Plate 4.1-1.

1. THE SOUTHERN METROPOLITAN COASTAL WATERS STUDY

1.1 Introduction

The need to provide a better ecological basis for regulating industrial and domestic waste emissions into Perth's metropolitan marine environment was highlighted by the continuing impacts of waste discharges into Cockburn Sound throughout the 1980s (Cary *et al.* 1991) and also by the environmental degradation that occurred in the Albany harbours over the same period (Simpson and Masini, 1990). Both cases clearly demonstrated the deleterious effects of excessive waste inputs (particularly nutrients) into temperate marine ecosystems of Western Australia and the need for environmental managers to better understand the relationship between cumulative, long-term contaminant loadings and environmental quality. It was clear that, without estimates and scenarios of the environmental impacts of a range of contaminant loadings, coupled with a clear statement of environmental quality objectives, pollution prevention regulations by the Department of Environmental Protection (DEP) and environmental impact assessments by the Environmental Protection Authority (EPA) would be less effective in preventing further environmental degradation resulting from waste inputs to Perth's southern coastal waters.

The Southern Metropolitan Coastal Waters Study (SMCWS) was initiated to address this issue and, specifically, to provide a better technical basis for the long-term management of Cockburn Sound and the surrounding southern metropolitan coastal waters of Perth.

1.1.1 Perth's southern metropolitan coastal waters

The protected coastal waters off the southern metropolitan coastline of Perth are utilised intensively for industrial, commercial and recreational purposes. Over the past 40 years, wastes have been routinely discharged into Cockburn Sound and, to a lesser extent, Owen Anchorage. Extensive phytoplankton and epiphyte blooms occurred during the 1960s and 1970s as a result of these discharges. The blooms reduced light availability to the benthic plant communities, resulting in a catastrophic loss of seagrasses in Cockburn Sound (Department of Conservation and Environment, 1979). Nutrients contained in these discharges, particularly nitrogen, were identified as the primary cause of the algal blooms. In addition, a range of contaminants such as heavy metals in these discharges contaminated sediments and biota in some areas of Cockburn Sound and Owen Anchorage. Following recommendations of the Cockburn Sound Environmental Study (Department of Conservation and Environment, 1979), the discharge of contaminants into these waters decreased substantially during the early to mid-1980s. A long-term monitoring programme was established to assess the ecosystem response to these management measures and included a 14-week survey of the nutrient-related water quality of Cockburn Sound and Owen Anchorage during summer every two to three years.

The first survey, in the summer of 1982/83, showed a significant improvement in water quality (measured as water clarity and phytoplankton chlorophyll *a* concentration) and a reduction in the rate of seagrass loss (Chiffings and McComb, 1983). The results of the next two surveys, in 1984/85 and 1986/87, indicated that this improvement had been maintained (Hillman, 1986; Hillman and Bastyan, 1988). However by the summer of 1989/90 the water quality in Cockburn Sound had deteriorated significantly and was approaching a state comparable to that of the late 1970s (Cary *et al.* 1991). This decline coincided with a significant increase (over the previous survey) in nitrogen loading from the CSBP/KNC outfall, the major point source of nitrogen into these waters at the time.

A positive relationship was derived between mean nitrogen loading from this source and mean water quality (measured as chlorophyll *a*) of Cockburn Sound for these four surveys (Cary *et al.* 1991). The relationship suggested that, for mean chlorophyll *a* concentrations in summer to fall to the 1982/83 levels (i.e. 0.8 $\mu g\ l^{-1}$), total nitrogen loadings from this source should be less than about 600 kg d^{-1}, approximately the mean loading during the 1982/83 survey. Nitrogen loading into Cockburn Sound from the CSBP/KNC outfall is now significantly lower than in 1989/90 (Martinick *et al.* 1993). Despite the reduction in nutrient load from this source the water quality of Cockburn Sound has remained at approximately the level of the summer of 1989/90 and this suggests that other factors are becoming more important as nutrient loads from industrial outfalls decrease (Bastyan and Paling, 1992; Bastyan *et al.* 1994; Cary and D'Adamo, 1995).

1.2 Need for study

The 'sudden' loss of seagrass meadows on east Parmelia Bank in the early 1980s (Le Provost, Semeniuk and Chalmer, 1986) and the nutrient-related water quality results of the summer monitoring programmes in Cockburn Sound during the 1980s (summarised in Cary *et al.* 1995a) again demonstrate the reduced resilience of marine ecosystems that have a history of environmental disturbance. These data underlined the importance of gaining a better understanding of the cumulative environmental impacts of waste inputs, particularly nutrients, into the shallow, temperate, coastal waters of Western Australia.

In recognition of this situation, the Department of Conservation and Environment (now the DEP) provided funding to the University of Western Australia, between 1986 and 1989, for a post-graduate study of the nutrient dynamics of Perth's coastal waters (Paling, 1991). Since nitrogen is generally considered to be the macronutrient limiting algal growth in Perth's coastal marine waters (Department of Conservation and Environment, 1979; Chiffings and McComb, 1983), this project provided a good basis for more comprehensive studies of Perth's metropolitan coastal waters by DEP and the Water Authority of Western Australia (WAWA; now the Water Corporation).

The population of Perth is projected to increase by about 50% over the next 30 years, from about 1.3 to 1.9 million (Western Australian Planning Commission, 1995). Associated with this increase there is likely to be a corresponding increase in commercial and industrial activity. The metropolitan coastal waters are currently used as receiving waters for some industrial waste and for more than 95% of Perth's reticulated domestic wastewater (i.e. sewage effluent). The sheltered waters of Cockburn Sound, Owen Anchorage and Warnbro Sound are the focus of recreational activities in the southern coastal waters of Perth and with the rapid spread of urban development in the southern metropolitan area, these activities are predicted to increase dramatically over the coming decade. Commercial activities such as aquaculture are also increasing. As these demands increase, so too does the need for sound environmental management. Thus the need for a better understanding of the cumulative long-term impacts of waste inputs to these waters has never been more urgent. The Southern Metropolitan Coastal Waters Study (1991-1994) was initiated to address this general need.

1.3 Scope, objectives and study area

1.3.1 Scope

Marine ecosystems are affected by development (e.g. ports and marinas), exploitation of natural resources, (e.g. fishing and mining), and waste inputs (e.g. industrial and domestic waste disposal). In Western Australia, development proposals are subject to the environmental impacts assessment process under the Environmental Protection Act 1986; waste discharges to the marine environment of Western Australia are also regulated under this Act. The impact of waste inputs, in relation to the loss of key habitats and the effects of toxic substances on the marine environment, is a central focus of the SMCWS. Public health issues, commercial and recreational fishing and marine wildlife are managed under other statutes and therefore are not considered directly in the SMCWS. The emphasis on the impacts of waste inputs on environmental quality, therefore, complements existing sectoral marine management frameworks in the metropolitan coastal waters of Perth.

1.3.2 Objectives

The overall goal of the *The Southern Metropolitan Coastal Waters Study* was to develop an understanding of the environmental consequences of the cumulative impacts of waste discharges to these waters to help develop a comprehensive environmental management strategy for the southern metropolitan coastal waters of Perth.

The broad objectives of the SMCWS were to —

- quantify past, present and projected contaminant inputs;
- determine the current status of the environment;
- establish a baseline for the detection of future environmental changes;
- predict the movement of water-borne contaminants;
- develop ecological simulation models to predict the environmental consequences of future nutrient input scenarios; and
- develop draft ecological environmental quality objectives and criteria for these waters.

From a regulatory point of view the primary objective of the SMCWS was to determine the cumulative environmental impacts of contaminant inputs to the southern metropolitan coastal waters of Perth, with particular emphasis on the semi-enclosed areas of Cockburn Sound and Owen Anchorage. This has been achieved and will provide a better technical basis, firstly, for environmental impact assessment by the EPA of new proposals involving significant waste inputs to the southern metropolitan coastal waters of Perth, and secondly for regulation by the DEP of existing waste discharges through improved licence setting. A secondary objective, also achieved, was to provide quantitative descriptions and baseline information necessary for current and future environmental management of these waters.

1.3.3 Study area

Locality names, bathymetry and major geomorphological features of the southern metropolitan coastal waters are found in Figures 1.1-1 and 1.1-2. The study area included the 'core' areas of Cockburn Sound, Owen Anchorage, Shoalwater Islands Marine Park (including Warnbro Sound) and, to a lesser extent, Sepia Depression. The wider southern metropolitan coastal waters, south to Mandurah and west across the continental shelf, were also included in recognition of the spatial scales of key processes affecting the 'core' areas (e.g. the Leeuwin Current and the Peel-Harvey Estuary outflow). Areas that will inevitably become part of the wider metropolitan coastal waters over the coming decades, such as Comet Bay, were also included for future reference.

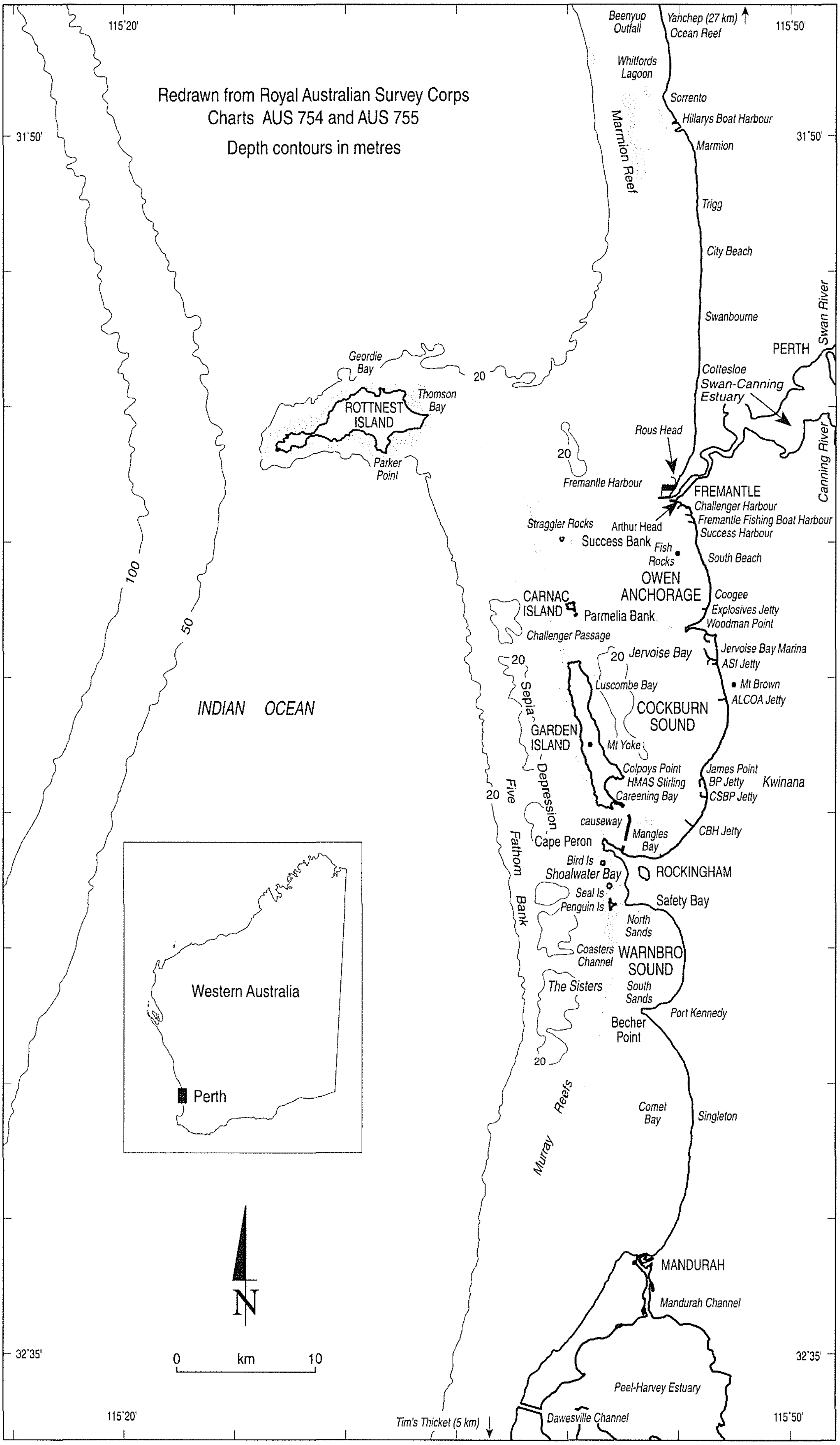

Figure 1.1-1. Location map of the study area.

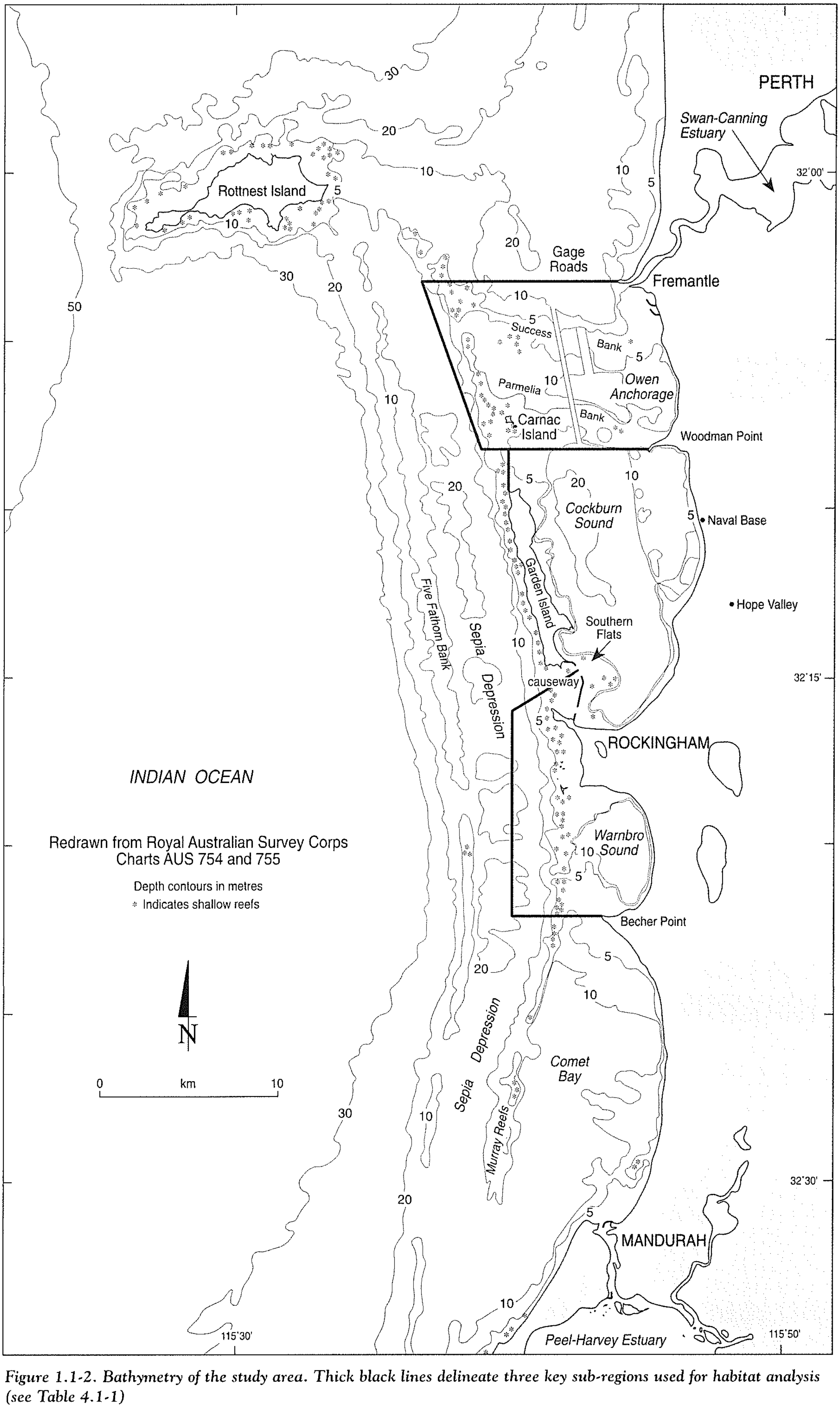

Figure 1.1-2. Bathymetry of the study area. Thick black lines delineate three key sub-regions used for habitat analysis (see Table 4.1-1)

1.4 Study approach

1.4.1 Philosophical approach

The philosophical approach of the EPA and the DEP to waste management centres around several key points: that society produces waste; that even with the best recycling and re-use programmes, some of this waste will inevitably require disposal to the natural environment; that any amount of waste entering the environment causes change, even if this alteration cannot be measured. Furthermore, between the state of no change (i.e. from natural variation) and the state just before the ecological integrity of the natural system in question begins to break down, there is a level of change in the natural environment, induced by human activity, that society may be prepared to accept for economic, social or other perceived benefits. The implicit assumption here is that the maintenance of biodiversity and ecosystem integrity are fundamental environmental attributes that should never be compromised (IUCN/UNEP/WWF, 1991). With this proviso, the acceptable level of change is determined, therefore, from a balance between existing and perceived future uses on the condition that the principle of inter-generational equity (i.e. avoidance of significant irreversible change) is maintained (Boelens, 1992; Nip and Udo de Haes, 1995). In this context unacceptable change is defined in terms of measurable departures from clearly stated environmental quality objectives (GESAMP,1991a). In practice, measurable environmental parameters would be compared with established environmental quality criteria (surrogates for the environmental quality objectives) to help assess whether these objectives are being achieved.

The long-term management of the metropolitan coastal waters of Perth therefore requires that a clearly stated set of environmental quality objectives have been designated; that links between human activity and environmental quality are sufficiently well understood (i.e. with an acceptable level of confidence); that appropriate environmental quality criteria are identified, and that effective management frameworks, including regulatory (e.g. licence setting, environmental impact assessment), feed-back (e.g. monitoring) and auditing mechanisms, exist. This combination of prediction, acknowledgment of uncertainty, monitoring and review allows management strategies to have the flexibility to be adapted to an improved knowledge base provided by well-planned monitoring programs (ANZECC, 1992). Therefore, a collective long-term vision of what society wants for these waters and a scientific understanding of the impact of human activities on these natural systems are required before management strategies can be put in place which ensure that long-term environmental sustainability is achieved.

1.4.2 Technical approach

The technical programme of the SMCWS involved the following major components. Readers are referred to the SMCWS Progress Report (Simpson *et al.* 1993) for further details.

Review of existing information

A bibliography of existing environmental information relevant to the SMCWS was compiled in 1991 (Cary and Ryall, 1992). The physical oceanography of the region was reviewed (Hearn, 1991b; D'Adamo, 1992) and a preliminary study of the nutrient dynamics of Perth's coastal waters (Paling, 1991) was funded by the DEP.

Inventory and baseline studies

Inventory and baseline studies were undertaken to describe the current status of aspects of the physical, chemical and biological environment of the study area and to provide improved quantitative baseline data sets from which future changes can be measured. Selected examples are described briefly below:

Information on the major benthic habitat types (e.g. seagrass meadows, reefs) and other important marine resources in the study area were mapped. Commercial and recreational uses of the study area were also mapped.

An inventory of past (from 1950), recent (1991) and projected (to 2021) contaminant inputs to the southern metropolitan coastal waters from point (e.g. industrial outfalls) and diffuse (e.g. groundwater inflow) sources was compiled and produced as a report (Martinick *et al.* 1993) with an accompanying electronic database. The database is maintained and updated by the Pollution Prevention Division of the DEP. The inventory was updated to provide 1994 estimates of contaminant loadings to these waters (Muriale and Cary, 1995).

The contaminant inputs inventory was used to plan comprehensive surveys of the main chemical contaminants (e.g. heavy metals, hydrocarbons, pesticides, organotins) likely to occur in the study area. Sediments and mussel tissue were analysed to assess the current state of contamination of the study area and to provide a quantitative baseline from which to measure future changes. Surveys of the chemical (e.g. nutrients), physical (e.g. light attenuation, temperature, salinity, dissolved oxygen) and biological (e.g. chlorophyll *a*, phytoplankton, zooplankton) characteristics of the water column throughout the study area were also undertaken for the same reasons. Faunal studies examined the benthic invertebrate communities of the deep basins of Cockburn and Warnbro Sounds and the effect of tributyltin (TBT) on *Thais orbita*, a molluscan bioindicator of TBT contamination.

Environmental quality objectives

Environmental quality objectives (EQOs) represent the long-term goals of an environmental management programme and relate to both ecological (i.e. maintenance of biodiversity and ecosystem integrity) and cultural values (i.e. maintenance of community uses and aspirations) of natural systems. Ecological EQOs are fundamental management goals whereas cultural EQOs are, by definition, negotiable and generally derived from a balance between existing and future uses after due consideration of economic, social and political factors. The process of establishing EQOs for the southern metropolitan coastal waters of Perth was formally separated from the SMCWS scientific studies by the EPA in 1992. Many of the proposed actions and recommendations of this report are, however, based on draft environmental quality objectives, proposed by the EPA and the DEP, and outlined in Chapter 3.

Process studies

A number of studies were undertaken to provide an understanding of the key physical and biological processes, particularly in relation to nutrient enrichment of Perth's southern coastal waters. These studies, and a complementary suite of studies undertaken within the Perth Coastal Waters Study (PCWS; see Section 1.7), provided basic information requirements for the joint development of a nutrient-effects ecological model.

Quantifying links between contaminant inputs and environmental quality

The approaches used to establish quantitative links between the contaminant inputs and the EQOs reflect the current and predicted nature of the two major classes of pollutants entering Perth's coastal waters: biostimulants, particularly nitrogen and phosphorus, and toxic substances such as heavy metals.

The productivity of temperate coastal ecosystems off Western Australia is limited by the availability of total inorganic nitrogen (TIN). TIN is a constituent of domestic sewage and estuarine outflows. Significant loads of TIN enter Cockburn Sound from point (e.g. industrial outfalls and surface drains) and diffuse (e.g. groundwater) sources. Excessive inputs of this nutrient stimulate nuisance growths of algae (e.g. phytoplankton and epiphytic algae) which can lead to destruction of benthic plant communities through light starvation via shading and smothering. Seagrasses grow in the more protected inshore areas of Perth's coastal waters, where both water residence times are longer and nutrient inputs are highest, making these communities more vulnerable to nutrient enrichment than offshore algal communities. This differential vulnerability, coupled with the ecological importance of seagrasses in Perth's coastal waters and the apparent inability of the *Posidonia* species to regenerate naturally, at least within several decades, provides the rationale for developing a model to link dissolved inorganic nitrogen inputs to *Posidonia* survival. This conceptual model provided the basis for designing ecological process studies in the field to refine the understanding of the key cause-effect pathways.

A modelling approach was used to determine, in conjunction with long-term monitoring data and SMCWS field studies, the relationships between nutrient loadings to Cockburn Sound and environmental quality. A nutrient-effects ecological model was developed in collaboration with the WAWA (see section 1.7) and has hydrodynamic transport and biological components. The model simulates the effect of inorganic nitrogen inputs on light availability to benthic plant communities, particularly seagrasses, a critically important biological component of the coastal ecosystems off Perth.

Contaminant concentrations in sediments and mussel tissue were compared to draft environmental quality criteria for the relevant EQOs to determine if pollution problems exist and if further regulation of contaminant inputs is required.

1.4.3 Identifying environmental quality criteria

Environmental quality criteria (EQC) are benchmarks which can be used to assess whether the EQOs are being achieved or maintained. Once the EQOs are determined and expressed spatially over the study area, the results of water, sediment and biota surveys can be compared with the appropriate criteria to see whether a management response is required. In areas where several EQOs overlap, the most sensitive EQC for each contaminant can be utilised to protect the other prescribed EQOs.

In the SMCWS, the draft water quality criteria for toxicants were drawn from the Australian Water Quality Guidelines (ANZECC, 1992) and the Draft Western Australian Water Quality Guidelines (EPA, 1993). Human health related criteria for maintenance of edible biota were drawn from the Australian Food Standards (NH&MRC, 1990). Sediment quality criteria were based, in the first instance, on the published reviews of Long *et al.* (1995) and Waite *et al.* (1991). If not available from these sources, the approach adopted by the Commonwealth Environmental Protection Authority (Pollution Research Pty Ltd. 1994), which uses values related to 'natural background' levels in sediments, was used.

As part of the scientific component of the SMCWS, draft nutrient-related EQC were developed by identifying key parameters of the critical biological pathways in the ecological model. These parameters relate to either direct or indirect biological responses to nutrient inputs. In the case of nutrient pollution, changes in epiphyte species composition and biomass on the leaves of seagrasses, and changes in phytoplankton abundance and water column light attenuation, are more useful indicators of eutrophication than are the concentrations of nutrients in the water column, because these concentrations can vary markedly due to the confounding processes of biological uptake and release.

The formal designation of environmental quality objectives and criteria for Perth's coastal waters will be undertaken through a consultative process, as outlined in section 3.6.

1.4.4 Community consultation

Community groups and relevant local and state government authorities were regularly briefed on the objectives, progress and findings of the SMCWS. Talks were also delivered to schools and user groups and an educational booklet and pamplets were distributed widely at the beginning of the Study.

1.5 Quality assurance

1.5.1 Data collection

Physical data

The quality of data collected by field instrumentation (e.g. current meters, CTD profilers) was assured by pre-deployment and post-deployment calibrations being undertaken at suitable laboratories. Wave, sea level, solar radiation, wind data were obtained directly from local organisations that routinely collect this information for their own requirements and maintain internal quality assurance programmes.

Chemical data

Sample collection employed standard methodology. Chemical analyses were undertaken independently at either laboratories registered by the National Association of Testing Authorities (NATA) or university research laboratories. For example, all nutrient and chlorophyll *a* analyses were conducted by the Marine and Freshwater Research Laboratory at Murdoch University, organotin analyses were conducted by the CSIRO Centre for Advanced Analytical Chemistry at Lucas Heights (NSW) and organic pollutants and heavy metals by the Chemistry Centre of Western Australia.

Biological data

Sample collection employed standard methodology and taxonomy was undertaken by specialists from local and interstate museums and universities.

1.5.2 Modelling

Physical

Time-series of current speed and direction predicted by the hydrodynamical model under real forcing conditions were statistically compared to time-series of currents measured under the same forcing conditions to assess the reliability of the hydrodynamical model predictions. Similarly the evolution of the physical structure of the water column as predicted by the model under specified forcing conditions was compared to the evolution of the measured structure of the water under the same conditions. Direct indicators of water movement such as satellite imagery were also used to help assess the validity of the hydrodynamic simulations as was the distribution and spread of phytoplankton emanating from the estuaries.

Ecological

Validation experiments for the seagrass, epiphyte and light attenuation sub-models were carried out to assess the accuracy of the predictions of the main biological components of the ecological model.

1.5.3 Data interpretation

The quality assurance of data interpretation was carried out by the standard process of peer review. All SMCWS technical reports published by the DEP were reviewed by recognised experts before publication. Scientific papers were subject to the usual process of anonymous review. Quality assurance of the process information used in the development of the ecological modelling component was carried out within the PCWS quality assurance programme. This involved providing documentation to the PCWS Modelling Group for ratification before inclusion into the model.

1.6 Reporting and data archiving

The reporting schedule of the SMCWS had four principal objectives:

- *to provide a statement on the progress of the SMCWS at about the halfway mark.*
  This was achieved by publishing the SMCWS Progress Report (Simpson *et al.* 1993).

- *to provide a clear statement of the summary, conclusions and recommendations at the completion of the SMCWS.*
  This was achieved by publishing the SMCWS Final Report - this report.

- *to provide an accessible, defensible technical basis for the recommendations in the final report.*
  This was achieved by publishing independently reviewed technical reports and scientific papers and;

- *to ensure the data collected during the SMCWS is archived and retrievable.*
  This was achieved by appending a SMCWS Publications Directory to the Final Report and lodging all technical publications and data reports, with copies of supporting digital information, in the DEP library.

1.7 Relationship to the Wastewater 2040 studies

In 1990, the EPA assessed a proposal from the Water Authority of Western Australia (WAWA) to construct and operate a second domestic wastewater ocean outlet into the waters of Marmion Marine Park (EPA, 1990). The environmental impact assessment concluded that the proposal was environmentally acceptable provided the nutrient loadings from both outfalls did not exceed the maximum loads set for the original single outfall. This decision was based on the conclusion that the short- and long-term impacts of increased nutrient loading to these waters could not be predicted with adequate certainty. The EPA also indicated that in the event of the WAWA (now the Water Corporation) submitting further proposals to increase nutrient loads to the coastal waters of Perth in the future, it would be necessary for the environmental impacts to be better understood.

As a result, the WAWA undertook the Perth Coastal Waters Study (PCWS) which focussed on the environmental effects of domestic wastewater discharges to the marine environment off the metropolitan coastline of Perth. This study provides information to help assess the short- and long-term cumulative environmental impacts of existing and proposed wastewater outfalls into the coastal waters off Perth to the year 2040. Companion studies examined alternatives to ocean discharge and community attitudes to domestic waste disposal. Together, these studies will help determine the environmental and economic 'costs' of different disposal options, thereby contributing to the information base necessary for planning the disposal of Perth's domestic wastewater into the 21st century.

Specifically, the PCWS examined the 'local' effects of the existing Beenyup outfalls at Ocean Reef on the biological communities of the Marmion Marine Park and adjacent waters, and provided information on the potential short- and long-term implications of chronic nutrient and toxic contaminant loadings to these waters. Other aspects of the study examined the potential short- and long-term effects of wastewater discharges from all existing and proposed (to the year 2040) outfalls on the wider metropolitan coastal waters off Perth (nominally Mandurah to Yanchep).

The PCWS overlapped with the SMCWS, both technically and geographically and, as a result, a significant level of collaboration occurred between the two studies. The biological communities of the northern and southern metropolitan coastal waters are similar and, consequently, the biological components of the ecological model were developed jointly. Although the geographical separation and different geomorphology of the two local study areas required that their hydrodynamics be investigated and modelled independently, a significant level of collaboration existed in the shelf-scale oceanography programmes.

The combined technical output of the SMCWS and the PCWS provides a comprehensive information base on which to plan the long-term management of Perth's metropolitan coastal waters (see section 3.6).

1.8 Current management frameworks

The environmental and resource management of Perth's coastal waters is undertaken by numerous agencies with responsibilities operating across four jurisdictions from local council by-laws, state and federal government legislation to international treaties. Separate legislative regimes cover State waters, Commonwealth territorial seas, the Australian Fishing Zone and Australia's Exclusive Economic Zone (Figure 1.1-3). Further complexity is added by special management arrangements that exist between the federal and state governments. This section summarises the principal legislation and agreements that are currently relevant to resource and environmental management of Perth's coastal waters. Further details of Commonwealth and State environmental legislation can be found in Fabricius (1993).

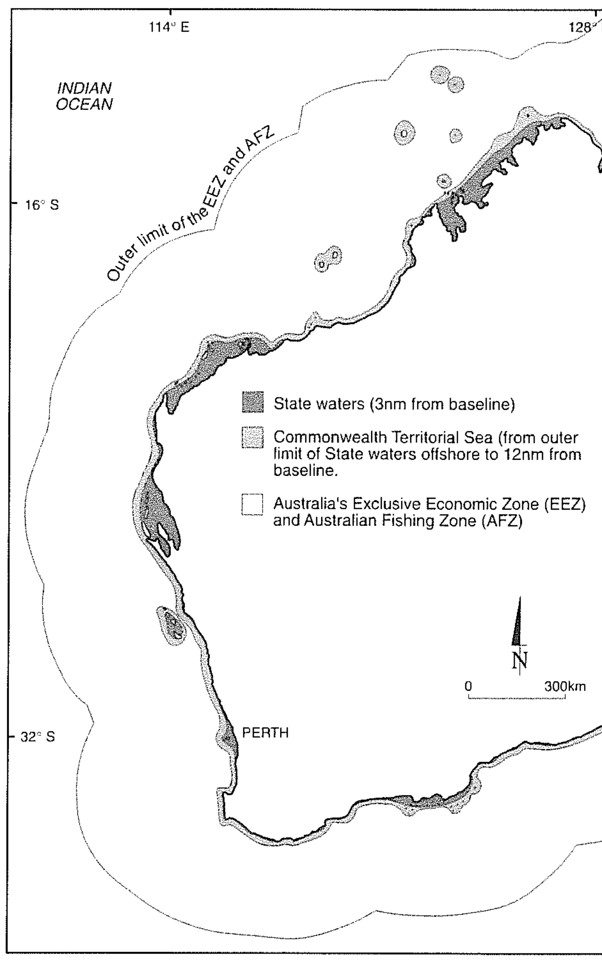

Figure 1.1-3. Zonation of legislative regimes applying to the marine waters off Western Australia.

1.8.1 International agreements

Australia is a signatory to several international agreements that are relevant to the management of Perth's coastal waters.

Convention on Wetlands of International Importance Especially as Waterfowl Habitat (Ramsar Convention): The Ramsar Convention encourages signatories to establish reserves on wetlands and to work towards increasing waterfowl populations. The Peel-Harvey Estuary is one of only nine Western Australian wetlands included on the Ramsar List of Wetlands of International Importance.

Agreement between the Government of the People's Republic of China and the Government of Australia for the Protection of Migratory Birds and their Environment (CAMBA).

Agreement between the Government of Japan and the Government of Australia for the Protection of Migratory Birds and Birds in Danger of Extinction and their Habitat (JAMBA).

The CAMBA and JAMBA agreements aim to promote international cooperation to protect birds that migrate between Australia and these two countries. As a result of the JAMBA agreement, several Australian migratory bird species that visit Japan have been declared as threatened fauna under the Western Australian Wildlife Conservation Act (see below).

1.8.2 Commonwealth legislation

Australian Heritage Commission Act 1975: This Act established the Australian Heritage Commission whose function it is to maintain a list of properties of outstanding cultural and natural value, known as the register of the National Estate. In Western Australia 88 wetlands are included on the National Estate register including the Swan-Canning and Peel-Harvey estuaries.

National Parks and Wildlife Conservation Act 1975: This Act, in conjunction with the Whale Protection Act 1980, is administered by the Australian Nature Conservation Agency (ANCA) and gives the Commonwealth Government powers to protect wildlife and to establish and manage marine parks on Commonwealth lands and waters.

Protection of the Sea (Prevention of Pollution from Ships) Act 1983: This Act implements the International Convention for the Prevention of Pollution from Ships 1973 (Marpol) which deals with most forms of marine pollution from ships except the disposal of land-generated waste into the sea by dumping (see below). The Marpol convention includes five technical Annexes and the Commonwealth legislation has ratified Annexes I, II, III and V, dealing respectively with oil, noxious liquid substances, packaged harmful substances and garbage. The Act is administered by the Australian Maritime Safety Authority.

Environment Protection (Sea Dumping) Act 1981: This Act implements the Convention on Marine Pollution by Dumping of Wastes and Other Matter 1972 and regulates the dumping of wastes and other materials into State and Commonwealth waters. Provisions of the Act prohibit the dumping of listed wastes and license the dumping of all other wastes. The Act also requires an environmental assessment of all proposals to dump excavated material and dredge spoil at sea. The Act is administered by the Commonwealth Environment Protection Agency.

Environment Protection (Impact of Proposals) Act 1974: This Act provides for the assessment of the environmental implications of proposals and decisions involving the Commonwealth Government and its agencies, throughout Commonwealth waters and where Federal approval is required. The Act is administered by the Commonwealth Environment Protection Agency.

Fisheries Act 1991: Fishing in Commonwealth waters and foreign fishing throughout the Australian Fishing Zone is managed under this Act. Jurisdictional structures provide for the joint management of certain fisheries by the State and Commonwealth (Fisheries Department of Western Australia, 1988).

Offshore Minerals Act 1994: This Acts enables the Commonwealth to regulate offshore mineral exploration and development in Commonwealth waters.

Petroleum (Submerged Lands) Act 1967: This Act provides for the exploitation of petroleum resources in Commonwealth waters by granting to licensees rights to explore for and develop offshore deposits. The Act is administered by both the State and the Commonwealth. Under this Act, Western Australian laws are extended offshore beyond State waters for the purposes of petroleum development.

Control of Naval Waters Act 1918: Provides for the control of naval waters (e.g. the eastern and western sides of Garden Island), the removal of hazards and for the implementation and observance of measures designed to ensure safe navigation which reduces the chance of environmental damage caused by collision at sea.

1.8.3 State legislation

State Agreement Acts: State Agreement Acts give Parliamentary endorsement to resource development agreements between the State and other parties. Examples in Perth's southern coastal area include the Cement Works (Cockburn Cement Limited) Agreement Act 1971 and the Industrial Lands (CSBP and Farmers Limited) Agreement

Act 1976. Since the assent of the Environmental Protection Act (1986), most Agreement Acts have included environmental clauses. The consent of both Houses of Parliament is required to overturn an Agreement Act.

The Environmental Protection Act 1986-1993 (or as amended): This Act provides for the establishment of the Environmental Protection Authority (EPA) and Department for the prevention, control and abatement of environmental pollution and for the conservation, preservation, protection, enhancement and management of the environment. Three different regulatory tools available under the Act are relevant to the management of coastal waters, namely environmental protection policies, environmental impact assessment and licensing of waste discharge. The Environmental Protection Act binds the Crown. The Act prevails over other State legislation with the exception of State Agreement Acts which received Royal Assent before 1 January 1972.

Wildlife Conservation Act 1950: The Wildlife Conservation Act provides a legislative basis for the conservation and protection of flora and fauna in Western Australia, including State waters, and is administered by the Department of Conservation and Land Management (CALM).

Conservation and Land Management Act 1984: The Conservation and Land Management Act provides for the protection of flora and fauna through a Statewide system of reserves. There are two marine conservation reserves in Perth's coastal waters: Marmion Marine Park and the Shoalwater Islands Marine Park. In addition, all the offshore islands and most of the islets in Perth's coastal waters are A Class Reserves under this Act.

Fish Resources Management Act 1994: This Act provides for the management of the fish resources of the State. Recreational fishing and over fifty commercial fisheries are managed by the Fisheries Department of Western Australia under this Act. In addition, under arrangements of the Offshore Constitutional Settlement, two other commercial fisheries are managed under the Act, by the State and Commonwealth, as joint authority fisheries. The objects of the Act are to conserve fish stocks and their habitat, develop fishing and aquaculture industries, achieve optimum economic, social and other benefits from the use of fish resources and to ensure that the exploitation of those resources is carried out in a sustainable manner.

Petroleum (Submerged Lands) Act 1982: This Act may be exercised with respect to any areas of land under the sea (within State jurisdiction), whether Crown or private and provides a right of access for the purpose of searching and operations of obtaining petroleum.

Mining Act 1978: This Act is principally concerned with land-based operations and is particularly relevant to mining activities occurring in the coastal zone. The Mining Act extends offshore into State waters, but does not mirror the Commonwealth Offshore Mineral Act (see above).

The Pollution of Waters by Oil and Noxious Substances Act 1987: This Act complements Commonwealth legislation (see above) however it only refers to the polluting of the sea with oil and noxious substances, as defined in Annex I and II of the Marpol Convention (1973). The Act is administered by the Western Australian Department of Transport.

Fremantle Port Authority Act 1902 and Ports (Functions) Act 1993: These Acts define all the functions relating to the operations of the Fremantle Port Authority. The Authority is the strategic manager of the Fremantle Port and the present boundaries include a land and sea area of nearly 900 square kilometres that covers most of the coastal waters, from Cockburn Sound to approximately Hillarys Boat Harbour, and offshore for about 15 km (FPA, 1994).

Waterways Conservation Act 1976: This Act established the Waterways Commission to manage and maintain the quality of the State's waterways including the Peel-Harvey estuary.

Swan River Trust Act 1988: This Act established the Swan River Trust with responsibilities for planning, protecting and managing Perth's river system.

Rottnest Island Authority Act 1987: This Act establishes the Rottnest Island Authority to control and manage the island for a variety of recreational and cultural purposes, for the protection of the flora and fauna of the island, and to maintain, protect and repair its natural environment.

Water Authority Act 1984: Rights in Water and Irrigation Act 1914; Metropolitan Water Supply, Sewage and Drainage Act 1909: These Acts provide legislative tools to regulate water supply, drainage and wastewater disposal.

Water and Rivers Commission Act 1995: This Act establishes a Commission with functions relating to water resources, including functions under various written laws.

Water Corporation Act 1995: This Act establishes a corporation with the function of providing water services, including functions under various written laws.

Metropolitan Region Town Planning Scheme Act 1956; State Planning Commission Act 1985; Local Government Act 1960: The Ministry for Planning and the Western Australian Planning Commission coordinate planning under the provisions of these Acts and through amendments to the Metropolitan Region Scheme. These statutes also create zoning controls and development approval processes which local authorities administer.

2. REGIONAL OVERVIEW OF PERTH'S SOUTHERN COASTAL WATERS

2.1 Introduction

Chapter 2 provides the general, regional information required by the reader to consider the more detailed findings of the Southern Metropolitan Coastal Waters Study, presented in Chapters 4, 5 and 6. This Chapter deals with the physical and biological setting and historical usage of Perth's southern coastal waters. The individual sections are brief outlines only and the interested reader is referred to key references for further details.

2.2 Climate and oceanography

2.2.1 Climate

The southwest of Australia has a 'Mediterranean' climate characterised by hot, dry summers and cool, wet winters (Australian Bureau of Statistics, 1989). The synoptic-scale weather patterns of the region are controlled by the migration of the anticyclonic belt from about 40 ^{o}S in January to about 30 ^{o}S in July. Thus, in summer, southwest Australia lies in the tropical low pressure region and enters the high pressure belt in winter. This belt rotates eastward around the globe and results in synoptic variations in the barometric pressure field at periods of about 7-10 days, with synoptic weather patterns broadly reflecting this periodicity (Breckling, 1989; Steedman and Craig, 1979).

The climate over southwest Australia is related to the size and intensity of the pressure systems within the anticyclonic belt and also to the latitudinal position of the belt. From about October to April, the belt is displaced just south of the Australian continent and stable anticyclonic high pressure cells produce a predominantly easterly airflow (the southeast trades). From May to September the anticyclonic belt is located across the continent, and associated high pressure cells produce predominantly westerly winds (the roaring forties). During winter in particular, these high pressure cells are periodically displaced by low pressure systems that approach from the southwest bringing strong winds (~ 15-20 m s^{-1}) and rain from the northwest, west and southwest (Breckling, 1989). Wind speeds over the Perth region generally range between about 5 and 15 m s^{-1} throughout the year. Differential heating and cooling across the coastline results in a land-sea breeze diurnal cycle which is superimposed on the regional pattern and is most apparent in spring and summer. South-southwesterly onshore winds, associated with sea-breezes, occur on over 250 days each year although the strongest sea-breezes (10-15 m s^{-1}) occur mainly during the mid-spring to summer period. Apart from storms that accompany low pressure systems, the wind field becomes weaker and more variable throughout autumn and winter. Further details of the seasonal characteristics of wind patterns in Perth's coastal waters can be found in Steedman and Craig (1979), Breckling (1989) and Hearn (1991).

The annual average rainfall for the Perth region is about 900 mm with over 80% occurring from May to September. Annual average evaporation is about 1700 mm with maximum and minimum rates of about 9 mm d^{-1} and 2 mm d^{-1} occuring in January and June, respectively. Long-term records indicate that the mean daily maximum air temperature is 29 ^{o}C in summer and 18 ^{o}C in winter; the mean daily minimum air temperature is 17 ^{o}C in summer and 9 ^{o}C in winter. Cloud cover, based on the daily average of readings at 0900, 1500 and 2100 hrs, varies from about 30% during January to 60% during June. Maximum daily short-wave radiation is typically about 750-1000 W m^{-2} in summer and 250-500 W m^{-2} in winter. Details of the climate of southwest Australia are available in Western Australian Year Books (e.g. Australian Bureau of Statistics, 1989) and seasonal measurements of heat fluxes, such as short-wave radiation, for Perth's coastal waters are presented in Pattiaratchi *et al.* (1995).

2.2.2 Waves

The wave climate of Perth's metropolitan coastal waters consists of oceanic swell and wind-generated sea waves. The swell generally develops in the Southern and south Indian oceans and approaches the Western Australian coastline from the southwest. The wave trains are refracted as they approach the coast arriving predominantly from the west-southwest at the metropolitan coastline. Waverider buoy records, collected over a 12-month period from south-southwest of Rottnest Island, indicate that the wave climate off Perth has the following characteristics (Department of Transport, unpublished data): swell waves have a mean significant period of about 12 seconds and significant wave heights of between about 0.5 and 5 m with a mean annual value of approximately 1.8 m. Swell is highest in winter and spring and lowest during summer and early autumn. Wind waves have a mean significant period of less than about 8 seconds and significant wave heights of between about 0.3 and 3.3 m, with a mean annual value of approximately 1.3 m which varies according to seasonal wind patterns.

Swell waves have their heights significantly attenuated as they approach the coast and cross the reefs, banks and sills between Fremantle and Warnbro Sound. For example, numerical modelling studies indicate that oceanic swell height is reduced to 10% at the shorelines of Owen Anchorage and Warnbro Sound and to about 5% in southern Cockburn Sound (Cockburn Cement Ltd. 1995; Department of Transport, unpublished data).

Currents generated by wave pumping are believed to be significant only in regions of constricted or shoaling bathymetry such as within reef areas and over banks, sills and swash zones (Pattiaratchi *et al.* 1995). The influence of wave set-up and radiation stress (Longuet-Higgins and Stewart, 1964) on mean circulation and flushing off Perth is considered to be relatively minor compared to wind-driven processes (Pattiaratchi *et al.* 1995).

2.2.3 Tides, wind and barometrically forced water level variations and low frequency oscillations

The metropolitan coastal waters off Perth are located in the southwest Australian micro-tidal zone where tides are mainly diurnal with spring tidal amplitudes of less than 1 m (Easton, 1970). The *National Tide Tables*, which are published annually by the Department of Defence, show that the predicted astronomical tidal range at Fremantle varies from about 0.1 to 0.9 m. Tidal current speeds for the continental shelf waters off Perth are typically less than about 0.02 m s^{-1} (Steedman and Craig, 1983; Hearn *et al.* 1985; van Senden, 1991; Pattiaratchi *et al.* 1995). Variations from predicted astronomical tide heights are due mainly to wind-stress and barometric pressure variations which, in combination, can alter the water level by up to 0.9 m, but more typically by 0.3 m. During the occasional passage (usually less than once per year) of tropical cyclonic depressions down the southwest coast these meteorological effects can alter coastal water levels off Perth by up to 1 m (Hodgkin and Di Lollo, 1958; Fandry *et al.* 1984).

Low frequency oscillations of water level off the west coast of Australia have characteristic periods of 5-10 days and ranges of 0.1-0.3 m (Webster, 1983). These may be associated with propagating coastally-trapped waves, such as continental shelf waves and Kelvin waves, or may be directly forced by the passage of synoptic meteorological systems (Fandry *et al.* 1984). Low frequency oscillations have been estimated to cause currents of order 0.1 m s^{-1} in the shelf zone off Perth (Hearn, 1991; Smith *et al.* 1991; van Senden, 1991), however the present understanding of these processes off the coast of southwest Australia is limited.

2.2.4 Large scale wind-driven circulation and the influence of the Leeuwin Current

Traditional atlases show a mean northward oceanic current off Western Australia which is the eastern portion of the large Indian Ocean anti-clockwise gyre. However, closer to the coast (within about 500 km) of Western Australia the Leeuwin Current flows southward for most of the year as a warm, low salinity, tropical water mass, driven by a north to south steric height gradient (Cresswell and Golding, 1980). This poleward flow of the Leeuwin Current is believed to be the primary reason for the absence of significant upwelling along the west coast, despite the prominence of equatorward winds during much of the year that would otherwise favour upwelling, as is the case along the west coasts of South America and South Africa (Pearce, 1991).

The Leeuwin Current typically flows over the continental shelf and slope with maximum core speeds of order 0.5-1.5 m s^{-1} just beyond the shelf edge (Pearce and Griffiths, 1991; Cresswell, 1991; Pearce, 1991). The current is of order 50-100 km wide and about 200 m deep off the southwest coast in winter. It is significantly weaker in summer due to the strength of opposing south-southwesterly winds. Beneath the Leeuwin Current is a colder undercurrent that carries South Indian Central Water northward (Cresswell, 1991). The Leeuwin Current frequently meanders and breaks out to sea forming warm anticyclonic eddies, of order 200 km in width, which are separated by cold core cyclonic cells. Between the eastern edge of the current and the coast, billows, of order 20 km, can approach the nearshore zone and influence the hydrodynamics of the inner-shelf waters (Mills *et al.* 1994, 1996).

During El Nino-Southern Oscillation (ENSO) events (Bureau of Meteorology, 1994; Philander, 1990) annual mean coastal sea levels along the west coast of Australia are lower and the waters along the outer shelf are cooler and more saline (Pearce and Phillips, 1988). These features are considered to be indicative of a weaker Leeuwin Current that is perhaps associated with a weaker flow through the Indonesian Archipelago during ENSO events (Pearce, 1991).

2.2.5 Wind-driven currents

In summer, wind stress has a predominant northward component and is the principal agent driving flow over the continental shelf off Perth (Cresswell and Golding, 1980; Steedman and Associates, in Binnie and Partners, 1981; Steedman and Craig, 1983; Hearn, 1991; Pattiaratchi *et al.* 1995). In autumn and winter, the mean wind stress is less than in summer. Although water flow in the nearshore zone is still largely dominated by wind in autumn and winter, flows over the mid- and outer-continental shelf are frequently southward, dominated by the steric height gradient (see section 2.2.4). This allows the eastern edge of the Leeuwin Current to occupy the mid-shelf zone resulting in a strong temperature front at its interface with cooler nearshore water. It is believed that regional currents, for example the Leeuwin Current, do not contribute significantly to the mean flow field in the semi-enclosed embayments, such as Cockburn Sound, when compared to wind-driven flow (Steedman and Craig, 1983).

The mean wind-driven transport within the nearshore zone is predominantly shore-parallel with speeds typically of order 0.1 m s^{-1} within a range of about 0-0.25 m s^{-1} (Hearn, 1991; D'Adamo, 1992; Steedman and Associates, in Binnie and

Partners, 1981; Pattiaratchi *et al.* 1995). However, the effect of the complex bathymetry, such as the reefs, islands and semi-enclosed embayments, results in bathymetric steering of wind-driven flows. Wind-induced sea-surface slopes within semi-enclosed embayments lead to local pressure gradient forces which drive sub-surface return flows, significantly influencing the basin-scale circulation patterns. The presence of sills further restricts the flushing of these embayments.

2.2.6 Density effects

Atmospheric heating and cooling, inputs of freshwater from estuaries, evaporation, and the transport of buoyant Leeuwin Current waters give rise to horizontal and vertical differences in water temperature, salinity and, therefore, density throughout the study region. The salinity of the nearshore zone varies annually from about 34 to 37 pss, the temperature from about 15-24 ^{o}C, and the density from about 1024.5 to 1026.4 kg m^{-3}. The effects of heating, cooling, evaporation and freshwater discharges are most significant in the relatively shallow nearshore zone shoreward of the major reef lines, while the influence of the Leeuwin Current is most prevalent over the mid- to outer-shelf waters. Evaporation throughout summer and autumn leads to a hypersaline nearshore zone with the embayments in particular displaying salinities significantly higher than the offshore shelf waters. From about June to September buoyant discharges from the Swan-Canning and Peel-Harvey estuaries are relatively strong, with rates from the Swan-Canning Estuary, for example, of up to 10^7 m^3 per day. As a result of these influences cross-shelf density differences of up to about 1 kg m^{-3} occur between the nearshore embayments and the mid-shelf region and can drive horizontal flows with speeds of order 0.1 m s^{-1}. Vertical density differences, typically up to about 0.5 kg m^{-3}, are also present throughout the year except under conditions of strong vertical mixing. Further details on the influence that density gradients can have on the hydrodynamics of Perth's coastal zone can be found in Hearn (1991), D'Adamo (1992), Pattiaratchi *et al.* (1995), D'Adamo and Mills (1995b) and Pearce and Church (submitted).

2.2.7 Effects of the earth's rotation

The dynamical influence of the earth's rotation is equivalent to an additional force acting perpendicularly to the direction of water movement. If this 'force' is not balanced, the flow direction will be significantly deviated in the anti-clockwise sense (southern hemisphere) after a time equivalent to the inertial period. At the latitude of Perth this is approximately one day. For water current speeds of order 0.1 m s^{-1}, typical of the study area, rotational effects become significant in water bodies which have horizontal dimensions of several kilometres or more. In such cases the Rossby number (Csanady, 1982) is less than one.

The combined presence of vertical density stratification and rotation introduces another dynamical length scale, the baroclinic radius of deformation (Csanady, 1982). If the width of a density-stratified water body exceeds this length scale, rotational effects can lead to upwelling and downwelling of the density structure transverse to the principal current direction. These dynamical responses are contained within boundary (e.g. coastal) zones having widths that scale with the baroclinic deformation radius.

The embayments, channels and inner-shelf areas of Perth's southern coastal waters have typical vertical density differences of 0.1 to 0.5 kg m^{-3} in the upper 20 m of the water column (D'Adamo and Mills, 1995b), and this corresponds to a baroclinic radius of deformation in the range 1-3 km. Cockburn Sound is about 15 km long and up to 10 km wide. The basins of Warnbro Sound, Mangles Bay and Owen Anchorage, although smaller than Cockburn Sound, nonetheless have horizontal dimensions which scale with the baroclinic radius of deformation.

2.3 Geomorphology and sedimentology

The geomorphology of the Perth region is dominated by a series of ancient dune systems, their positions reflecting migrations of the shoreline in response to a falling trend in the global sea level during the last 300 000 years, overlayed with marked sea level fluctuations due to glacial and interglacial periods. With the end of a glacial period about 100 000 years BP the sea level rose and remained, fluctuating around a high, as shelly sediments were moved onshore. As a result the Tamala dune system formed, consisting of a series of shore-parallel limestone dune ridges, some lying to the east of today's coast and others, submerged or partially submerged, lying to the west (Figures 1.1-1, 1.1-2 and Plate 4.1-1). From west to east these include the Five Fathom Bank Ridge, the Garden Island Ridge and the Spearwood Ridge. During the most recent glaciation, occurring between 60 000-25 000 years BP, the shoreline advanced beyond Rottnest Island and the coastal plain was again subject to exposure and weathering.

As a result of the Holocene sea level rise (between about 20 000 and 6000 years BP) several of the Tamala Limestone ridges were drowned, although a number of prominent islands remained along the Garden Island Ridge. The nearshore embayments of Owen Anchorage, Cockburn Sound, Warnbro Sound and Comet Bay were formed by inundation of the depression between the Spearwood Ridge and the Garden Island Ridge. Along the eastern side of Cockburn Sound and Owen Anchorage the Spearwood Ridge was eroded by wave action leaving a shelf about 3 km wide and submerged to a depth of up to 10 m. The gradual erosion of the Five Fathom Bank Ridge and the Garden Island Ridge led to the formation of banks and sills such as the South Sands and North Sands of Warnbro Sound, Parmelia Bank and Success Bank, north of Cockburn Sound, and the tombolo between Rottnest Island and Fremantle. The areal extent of the embayments enclosed by these deposits has since decreased during the Recent (beginning about 6000 years ago) sea-level fall of about 5 m resulting in greater physical isolation from the adjacent ocean due to the shallowing of profiles along the reef lines, and the ongoing development of the banks and sills.

The sedimentology of the nearshore embayments is characterised by two main types of sediments which built up during the Holocene sea-level rise. The fringing banks and sills, mentioned above, are comprised mainly of carbonate-quartz sands derived primarily from material eroded from the adjacent Pleistocene eolianite ridges (mainly the Garden Island Ridge and Spearwood Ridge) by intense wave action and, secondarily, from shell material within the seagrass meadows which formed. Wave patterns east of the Garden Island Ridge were significantly changed as these fringing banks developed. As a result fine carbonate sands and organic material, transported from the banks by littoral drift, were trapped in the deeper basins of Cockburn and Warnbro sounds leading to an accumulation of organically rich carbonate muds. In the vicinity of the estuarine outflows there is a greater proportion of fluvial sediments primarily derived from overland runoff of siliceous material and, as a consequence, the carbonate content of the sediments within these regions is lower than the surrounding nearshore zone (see section 4.5.1). West of the Five Fathom Bank Ridge, the sediment composition is dominated by mobile carbonate sands with the relative proportion of organic material significantly less than within the embayments.

Littoral sand drift along the coastline south of Perth is predominantly northward throughout the year due to the prevailing southwesterly swell (Department of Conservation and Environment, 1980a; Public Works Department of Western Australia, 1984). However, the direction of littoral drift is intermittently reversed to be southward due to shorter period waves associated with storms from the northwest, particularly in winter and spring.

Further details on the geomorphology and sedimentology of Perth's southern coastal zone can be found in France (1978), Department of Conservation and Environment (1980b), Searle *et al.* (1988) and Semeniuk *et al.* (1989).

2.4 Biology

The west coast of Western Australia between Cape Leeuwin and Northwest Cape is a zone of biogeographic overlap where tropical species of the 'Northern Australian Region' and the warm-temperate species of the 'Southern Australian Region' intermingle (Wilson and Allen, 1987). Perth is near the southern extremity of this broad overlap zone and, although the marine flora and fauna of the Perth region are dominated by temperate species, many tropical fish, corals and other animals occur in these waters (Hutchins, 1991; Veron and Marsh, 1988).

The tropical species that occur in these waters owe their existence to the Leeuwin Current, a southward flow of warm, tropical water off the Western Australian coastline (Cresswell and Golding, 1980). In winter the current maintains relatively high temperatures in Perth's offshore waters compared with inshore. The current is strongest during autumn and winter and although it travels predominantly along the continental shelf break, large 'billows' or incursions come within five kilometres of the metropolitan coastline on occasions (Mills *et al.* 1994). The stronger offshore influence of the current is reflected in the more tropical composition of reef communities further offshore. At Rottnest Island for instance, the molluscan fauna at the western end of the island is comprised of almost 50% tropical species whereas 12 km away, at the eastern end of the island, the molluscan fauna consists of about 33% tropical species and is almost identical to the mainland assemblage (Morgan and Wells, 1991). Similarly, tropical scleractinian corals are common at Rottnest Island but only occur as isolated colonies near Fremantle and in Cockburn Sound and Owen Anchorage. Tropical pelagic fishes are also common during late summer and autumn in the offshore coastal waters of Perth.

Apart from commercially important species and visually dominant plants and animals, little is known about much of the marine flora and fauna of the Perth coastal region. This lack of knowledge was highlighted recently when 83 species, new to science, were described from samples collected from Rottnest Island during a two week marine biological workshop held in 1991 (Wells *et al.* 1993). Some obviously diverse groups, such as sponges, are almost completely undescribed. The zooplankton, phytoplankton and benthic fauna of Cockburn and Warnbro sounds were poorly understood until the present study.

The surface waters off the southwestern coast of Australia are generally nutrient-poor and have low pelagic productivity compared to the western coasts of South America and Africa where upwelling of nutrient-rich subsurface waters promote rapid phytoplankton growth and high secondary production of zooplankton and fish (Morgan and Wells, 1991). The low rates of pelagic productivity in the coastal waters off southwest Australia result in high light penetration to benthic primary producers which provide much of the primary production needed to 'drive' these ecosystems (Paling, 1991). Seagrasses and macroalgae are the dominant marine macrophytes found along the Perth metropolitan coastline, occupying substantial areas and producing large quantities of organic matter (Table 2.4-1). In addition these communities, particularly the meadow-forming seagrasses, provide substrata for diverse assemblages of small plants and animals, habitat and nursery areas for fish and invertebrates, a means of trapping and binding sediments and a medium for storing and recycling nutrients (Larkum *et al.* 1989).

Table 2.4-1. Comparative area, dry weight, production and nitrogen content of seagrass and macroalgae in the area bounded by Woodman Point, Five Fathom Bank and Becher Point in waters less than 20 m deep (after Paling, 1991).

Community type	Area (km^2)	Total dry weight (tonnes)	Production (tonnes yr^{-1})	Nitrogen content (tonnes)
Seagrass	21.14	21 851	333.7	294.7
Macroalgae	35.85	60 290	1862.7	1066.0

The diversity of seagrass genera (10) and species (25) in Western Australia is unequalled anywhere in the world (Kirkman and Walker, 1989) and although 18 species have been recorded in the Perth region, 90% of the total seagrass cover occurs in monospecific or mixed-species meadows comprised of only four species: *Posidonia sinuosa*, *Posidonia australis*, *Amphibolis antarctica* and *Amphibolis griffithii*. The most robust and conspicuous genus is *Posidonia* and eight of the nine species occurring world-wide occur in Western Australia and are either endemic to this State or to the southern coast of Australia (Kirkman and Walker, 1989). The high degree of endemism in this genus is presumably related to the oligotrophic, clear waters off the southwest coast of Australia and the ability of this genus, with its robust rhizomes, to survive in and stabilise unconsolidated sandy substrata.

The dominant macrophyte community in different parts of the coastal waters of Perth is largely determined by substratum, light availability and degree of exposure to swell waves. Seagrass meadows are only found on relatively stable sediments whereas attached macroalgae are found on solid substrata such as limestone reefs and pavement. The macroalgal community is dominated by the kelp, *Ecklonia radiata*, which forms 95% of the algal biomass on reefs and by *Sargassum* spp. which are seasonally important on the shallower reefs in the summer months (Paling, 1991). Rhodophytes are more common on the deeper limestone pavements (Ottaway and Simpson, 1986). Areas of mobile sediment are inhabited by microscopic algae attached to sand grains or inhabiting the spaces between them, and are colonised, from time to time, by non-meadow forming seagrass species (Masini, 1990).

With respect to major benthic habitats, two distinct zones occur within the coastal waters of the study area, largely as a result of differential exposure to waves: the protected *nearshore coastal* zone, east of the Garden Island Ridge (Zone A), and the exposed *offshore coastal* zone, west of this ridge system (Zone B), as shown in Figure 2.4-1. The relatively sheltered nearshore zone contains extensive seagrass meadows and only a small amount of sub-tidal reef. Extensive areas of unvegetated substrate occur on the sand banks and in the sedimentary basins of Cockburn Sound, Warnbro Sound and Owen Anchorage. The offshore zone is characterised by algal-dominated reefs and large areas of unvegetated, mobile sands (see also Figure 4.1-1).

The ability of the plant communities in the coastal waters of Perth to maintain high rates of productivity under conditions of low nutrient availability (Paling, 1991) also makes them vulnerable to nutrient pollution.

Habitat type	Zone A		Zone B	
	(km^2)	(%)	(km^2)	(%)
Sand (including sparse seagrass)	197.5	58.4	40.2	14.1
Seagrass meadow	41.7	12.3	1.5	0.5
Subtidal reef (algal dominated)	4.0	1.2	125.1	44.0
Intertidal reef (algal dominated)	< 0.1	< 0.1	0.5	0.2
Coarse sand	0.0	0.0	117.0	41.2
Fine sand and silt	10.2	3.0	0.0	0.0
Silt	84.9	25.1	0.0	0.0
Total	338.3	100.0	284.3	100.0

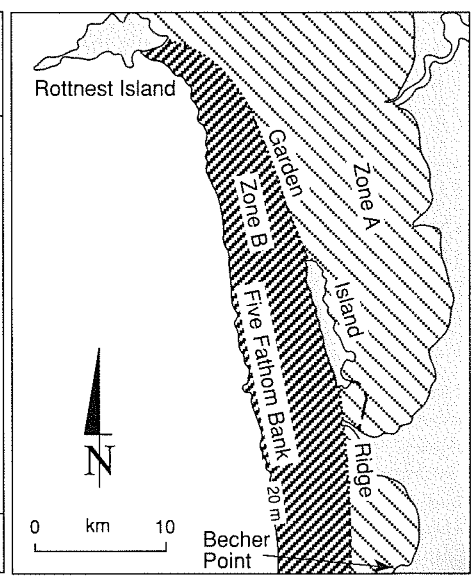

Figure 2.4-1. Area of major benthic habitats in Zone A and Zone B.

Seagrasses, in particular, are easily outcompeted and displaced by algae under nutrient-enriched conditions (e.g. Department of Conservation and Environment, 1979; Simpson and Masini, 1990). The low ambient nutrient status of the coastal waters off southern Western Australia, the high degree of endemism and the key ecological role of seagrass communities in these nearshore ecosystems reduces the usefulness of information collected at similar latitudes in other parts of the world in relation to the management of nutrient pollution.

2.5 Inter-annual variation

Few long-term (>10 y) physical and biological data sets exist for the metropolitan coastal waters of Perth. As a result, some of the conditions (e.g. the phytoplankton community characteristics) which occur during relatively short-term studies, such as the SMCWS, cannot be put into an appropriate temporal perspective. Furthermore, without a clear understanding of the range of natural variation, the interpretation of the results of such studies, in regard to separating anthropogenic and natural influences on the natural environment, is more problematical.

Water quality parameters have been monitored at a site about 5 km west of the western end of Rottnest Island by the CSIRO since the early 1970s and in Cockburn Sound, by the DEP, since the mid-1970s. Although somewhat limited, these data provide valuable long-term information on aspects of Perth's coastal waters. Another data set that provides a further indication of the inter-annual variability of selected physical and biological parameters in these waters is shown in Figure 2.5-1. These data indicate that global meteorological processes, such as those represented by the Southern Oscillation Index, have a significant influence on the seasonal and inter-annual mean sea level variations off Western Australia, and therefore on the strength of the Leeuwin Current. The strength of the Leeuwin Current, in turn, influences the annual temperature and salinity cycle of these waters and recruitment patterns of the western rock lobster.

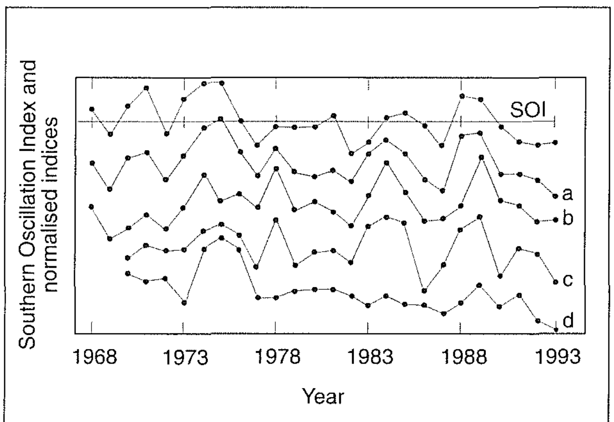

Figure 2.5-1. Inter-annual variation of the Southern Oscillation Index (SOI) and normalised indices of (a) mean sea level at Fremantle, (b) puerulus (lobster larvae) settlement index from reefs off Geraldton, (c) mean sea temperature (10 m) near Rottnest Island and (d) mean salinity (10 m) near Rottnest Island. Figure updated from Pearce and Phillips (1988) with data from CSIRO, Department of Fisheries of Western Australia and Flinders University of South Australia.

These examples serve to illustrate the critical importance of establishing long-term data sets in order to develop an understanding of the nature and causes of long-term variability in these ecosystems.

2.6 Usage

Large areas of the coastal waters off Perth are protected from the full force of oceanic swell by the chains of reefs, banks and islands that run approximately parallel to the coast. These features and the mediterranean climate provide ideal conditions for marine-based recreational activities such as swimming, boating and fishing. Perth's coastal waters are also used for conservation, education, commercial fishing, shipping, defence purposes and as receiving waters for industrial and domestic wastes.

In 1993 the population of Western Australia was 1.7 million with 1.2 million people living in the Perth metropolitan area within about 20 km of the coastline (Australian Bureau of Statistics, 1995). Overall the population of the Perth metropolitan region is predicted to increase by about 50% by the year 2021, whereas the population of the coastal municipalities is predicted to more than double (Department of Planning and Urban Development, 1990), indicating a strong bias towards urban expansion along the coast. Major recreational activities in the coastal zone include swimming, board-riding, yachting, sightseeing, boating and fishing, and many of these activities are focussed on the more protected sections of the coast such as Cockburn and Warnbro sounds.

Three marine conservation reserves have been declared in Perth's marine waters and several sections of the coastal strip have been declared as terrestrial conservation and recreation reserves. These reserves have been created to better manage the intense recreational usage of the coastal zone.

Over 95% of Perth's reticulated domestic wastewater is currently discharged to the nearshore coastal zone via ocean outfalls off Ocean Reef, Swanbourne and Cape Peron. The wastewater from the Ocean Reef and Swanbourne outfalls is treated to secondary level while that from the Cape Peron outfall undergoes primary treatment (Water Authority of Western Australia, 1995). About 200 million litres of domestic wastewater is discharged to the coastal waters of Perth each day and this volume is predicted to increase to about 600 million litres per day by the year 2040 if current ocean disposal practices are maintained.

Commercial and industrial use of Perth's coastal zone is primarily focussed along the southern metropolitan coastline and adjacent marine waters between Fremantle and Rockingham. These uses include manufacturing and processing industries, harbours and marinas, ports, shipping, ship-building and maintenance facilities, subtidal sand mining, oil exploration, aquaculture, fishing and tourism (Dames and Moore Pty. Ltd. 1993). The major centre of industrial development is along the Kwinana industrial strip, along the eastern shoreline of Cockburn Sound, and the adjacent marine environment is used as a mooring and loading/unloading area for shipping, and also as a receiving environment for industrial wastes. Heavy industries in the Kwinana area include mineral and metal processing plants, a fertiliser company, a petroleum refinery, grain storage and handling facilities, gas and chemical factories and a power station. Many of these industries discharge wastes to Cockburn Sound.

In the recent past there was a limited number of commercial and industrial sites along the eastern shoreline of Owen Anchorage. Industries included meatworks, fellmongers, fish processors and a tannery, with several of these companies discharging wastes to Owen Anchorage (Muriale and Cary, 1995). Most of these industries have closed down, relocated, or are soon to be relocated to the Coogee Redevelopment Project (Biotechnology Park) which will incorporate a centralised waste treatment facility.

The principal port for the region is at the mouth of the Swan River at Fremantle Harbour, with smaller shipping facilities servicing individual industries in Cockburn Sound. In 1993/1994 1700 ships used Fremantle Harbour, with a total tonnage of 29 million tonnes (Fremantle Port Authority, 1994). A major ship building and maintenance area is situated at Jervoise Bay, and this services both commercial and naval vessels.

Shellsands are mined on Success and Parmelia banks and a washing plant is located at Woodman Point (Cockburn Cement Ltd. 1995). The mining is carried out in accordance with the Cement Works (Cockburn Cement Limited) Agreement Act 1971 which provides the company with a mining lease area within an 8 km radius from Coogee Beach until the year 2011, with rights of extension until 2021. In 1994, 1.6 million tonnes of shellsand was mined and this is forecast to increase to about 2 million tonnes per year by the year 2001.

Seismic and exploratory drilling for petroleum resources has been carried out in recent years in the coastal waters north and south of Rottnest Island. There are no production wells operating off the Perth metropolitan coastline at present.

Aquaculture consists mainly of mussel farming in northern Cockburn Sound within a lease area of about 1 km^2, and within a smaller lease area in southeast Warnbro Sound. Mussel spat are collected from near the Cooperative Bulk Handling jetty in southeastern Cockburn Sound. A number of important commercial fisheries operate off the metropolitan coast, with the largest being the western rock lobster (*Panulirus cygnus*) fishery. In addition, abalone, scallops, prawns (by trawling), baitfish, finfish, scale fish and crabs are fished in the region. Trawling is open to most of the area between Fremantle and Cape Naturaliste however trawling usually occurs on about five fishing grounds in the southern metropolitan coastal waters off Cottesloe and Comet Bay (Laurenson *et al.* 1993).

Marine-based tourism in Perth's coastal waters is based mainly on sightseeing, swimming, ecotourism, diving, boating and fishing and is developing as an important and expanding component of the local economy.

A major naval facility is situated at HMAS Stirling on the southern end of Garden Island and in the adjacent waters of Careening Bay. A significant percentage of Garden Island is zoned for defence purposes and Commonwealth jurisdiction extends seaward by up to 1 km from the shoreline of Garden Island. A solid-fill causeway, with two bridge openings, connects HMAS Stirling and the mainland near Cape Peron.

2.7 Historical environmental impacts

The major historical environmental impacts in Perth's coastal waters have been from industrial and domestic waste inputs, dredging of shipping channels, subtidal sand-mining, recreational use and establishment of defence facilities. Many of the impacts are similar to those which have occurred in the marine waters of many coastal cities of the world and have involved impacts on sensitive marine habitats and contamination of sediments and biota with toxic materials.

Industrial development along the eastern shoreline of Cockburn Sound and Owen Anchorage in the 1950s and 1960s resulted in direct and diffuse inputs of contaminants into these waterbodies (Department of Conservation and Environment, 1979; Martinick *et al.* 1993). The Woodman Point wastewater outfall into northern Cockburn Sound was a major additional source of nutrients, bacteria and heavy metals until 1984 when the waste stream was diverted to Sepia Depression. These waste discharges contributed to widespread eutrophication in Cockburn Sound resulting in extensive dieback of seagrasses and localised toxic contamination of sediments and biota (Department of Conservation and Environment, 1979).

The dredging of the Fremantle Port Authority shipping channel and the creation of a second 'shipping channel' through Success and Parmelia banks has resulted in the loss of over 150 ha of seagrass meadow, the major benthic primary producer in these sheltered waters. If mining of shellsand continues on Success Bank until 2021, using current practices, almost 600 ha of seagrass meadow could be lost.

The southern end of Garden Island and the adjacent waters of Careening Bay are utilised for defence purposes by the Royal Australian Navy. Berthing facilities, jetties and coastal structures have significantly changed the nature of the shoreline. The causeway (with two bridge openings) across the southern entrance of Cockburn Sound was completed in 1973 and reduced the flushing of the sound (Maritime Works Branch, 1977a).

2.8 Past studies

Since the early 1970s there have been several environmental studies by Commonwealth and State agencies and industry that have made a significant contribution to the understanding of the marine environment of Cockburn Sound and surrounding waters. In many cases these studies have provided valuable data for the present study. Cary and Ryall (1992) completed a comprehensive bibliography of past environmental studies of the southern metropolitan coastal waters of Perth.

Following the decision to build a causeway linking Garden Island to the mainland, just west of Rockingham, the Department of Defence and the Fremantle Port Authority undertook studies between 1970 and 1975 to assess the effect that the causeway would have on the hydrodynamics of Cockburn Sound (Maritime Works Branch, 1977a). In the latter stages of this study, the focus was re-directed towards ecological processes in recognition of the observed deterioration in water quality and the widespread loss of seagrass meadows in the sound (Maritime Works Branch, 1977b).

The Western Australian Government approved funding for a major three-year environmental study of these waters in 1976. The objective of the Cockburn Sound Environmental Study (1976-1979) was to provide the necessary technical information to arrest the ecological deterioration of the sound and, through ancillary studies of recreational usage and public attitudes, provide the basis for developing a comprehensive environmental strategy for these waters (Department of Conservation and Environment, 1979).

Numerous localised environmental impact studies have been conducted in Perth's southern waters by industry and government agencies and these are detailed in Cary and Ryall (1992). Many of these studies arise from the formal requirements of the EPA environmental impact assessment process, and from monitoring requirements attached as conditions to the granting of point source discharge licences by the DEP. The following examples briefly describe some of the major on-going studies. The Water Authority of Western Australia conducted extensive environmental studies in Sepia Depression between 1981 and 1983, before commissioning the Cape Peron Outfall in 1984, and monitoring has continued, at least annually, since then (Halpern Glick Maunsell, 1992). In Owen Anchorage, Cockburn Cement Ltd. has dredged shellsands from Parmelia and Success banks since 1972 (Cockburn Cement Ltd. 1995). Over this period Cockburn Cement Ltd. has undertaken numerous ecological and oceanographic studies to assess the direct and indirect environmental impacts of the mining operation on the seagrass meadows on these banks, on bank stability and on the adjacent shoreline. Currently Cockburn Cement Ltd. is undertaking an extensive suite of studies, including seagrass meadow restoration studies, to provide further information on the environmental acceptability of its proposed long-term mining operations in Owen Anchorage (Cockburn Cement Ltd. 1995).

Further details of past studies can be found in the reviews of Department of Defence (1988) and the bibliography of Cary and Ryall (1991).

2.9 Community attitudes

An assessment of past and present community attitudes was based on the results of surveys conducted by Feilman and Associates (1978), Fenton and Syme (1987), Syme and Nancarrow (1988), Dames and Moore Pty. Ltd. (1993), Dielesen (1994), WAWA (1994) and CSIRO (1994). The main findings relevant to the SMCWS are outlined briefly below.

The community believed that in 1994 there was a better balance between the level of industrial development and recreational usage in the Cockburn Sound region than existed during the 1970s (Table 2.9-1). This change probably reflects the absence of new, major industrial developments in the area since this time (Dames and Moore Pty. Ltd. 1993), the increase in recreation facilities and the more recent establishment of several conservation and recreation reserves in the coastal zone (see section 4.1.4).

Table 2.9-1. Community response to the question "Cockburn Sound is currently used for both recreation and industry - what should its future use be?".

Response	1978 (%)	1994 (%)
Maintain present balance	28	50
More recreation	66	44
More industry	0	2
More industry and recreation	6	2
Do not care	0	2

In relation to the environmental quality of Cockburn Sound the majority of respondents, in both 1978 and 1994, believed that the environmental problems in the sound were serious but could be controlled by stringent pollution prevention measures (Table 2.9-2). In 1994, the majority (57%) of the people interviewed believed that if further industrial expansion was to occur in the Cockburn Sound region it

Table 2.9-2. Community response to the question "What do you believe is the extent of environmental problems in Cockburn Sound?".

Response	1978 (%)	1994 (%)
Are serious but can be controlled	77	63
Are not serious	9	20
Are necessary because of need for industry and employment	12	7
Are beyond control	2	7
Do not exist	0	3

should be located away from the coast. A further 27% indicated that there should be no further industrial expansion (Table 2.9-3). These results indicate that there has been a significant change since the 1970s in community attitudes to the use of the coastal strip.

Table 2.9-3. Community response to the question "If more industry needs to be developed - where should it go?".

Response	1978 (%)	1994 (%)
Not on coast - inland	41	57
No further expansion	22	27
Infill near existing industries	13	9
Woodman Point to Naval Base	10	7
Other	14	0

The beaches of Rockingham, Safety Bay, Waikiki and, to a lesser extent, Kwinana are still the most popular beaches in the study area, as they were in the 1970s. These beaches are relatively sheltered, have a range of facilities nearby and are generally considered to be relatively unpolluted.

Employment opportunities were seen as the major benefit of industry in the region, both in the past and in the future, though less so in the future (Syme and Nancarrow, 1988). Pollution, on the other hand, was seen as the major negative impact associated with industrial development. The community believes that the biggest issue to be dealt with in the Kwinana Industrial Area was industrial pollution and that this issue was more important than additional employment and transport opportunities (Dames and Moore Pty. Ltd. 1993).

Community attitudes concerning the collection and disposal of domestic wastewater were surveyed by WAWA (1994) and CSIRO (1995). Environmental and human health concerns were dominant. There was significant community concern over the long-term environmental implications of wastewater disposal as the population of Perth increases. Furthermore, the community believes that domestic wastewater should be considered as a resource which, in our relatively dry region with its limited water resources, should be recycled/reused as efficiently as possible.

Recent surveys of community attitudes to the southern metropolitan coastal waters off Perth, particularly Cockburn Sound support earlier findings that environmental issues are a high priority to the community. The community also now expects to be far more involved in determining appropriate uses of these waters than in previous years. Chapter 3 outlines the draft environmental quality objectives, proposed by the EPA and the DEP, for these waters and outlines a process to integrate environmental planning and management for the entire metropolitan coastal waters. A key part of this process will be to formally designate environmental quality objectives and criteria for Perth's coastal waters and this will be achieved through consultation with the general community and user groups.

3. DRAFT ENVIRONMENTAL QUALITY OBJECTIVES

This chapter describes the draft Environmental Quality Objectives (EQOs) proposed by the EPA and the DEP for Perth's southern metropolitan coastal waters. Draft Environmental Quality Criteria (EQC) that apply to these EQOs are also provided. Key definitions, philosophy and current practice of the EPA and DEP to waste management issues in Western Australia are also outlined. The process by which the Southern Metropolitan Coastal Waters Study and the Perth Coastal Waters Study will be integrated and, with stakeholder and community consultation, used to set Environmental Quality Objectives and Criteria for all of Perth's coastal waters and the subsequent development of an integrated management response in relation to waste inputs, is also described.

3.1 Introduction

In a report on the state of the global marine environment, the Group of Experts on the Scientific Aspects of Marine Pollution (GESAMP) concluded that if waste inputs to the marine environment were allowed to increase unchecked, then, especially in view of the continuing growth of human populations, the marine environment could deteriorate significantly in the next decade (GESAMP, 1990a). In Australia, declining marine and coastal water/sediment quality and losses of sensitive marine and coastal habitat were identified in a State of the Marine Environment Report as the most serious issues affecting Australia's marine environment (Zann, 1995). There is little doubt that marine ecosystems and their resources are under increasing strain from human population growth in most parts of the world and much of this deterioration is caused by excessive loads of contaminants from industrial, domestic and agricultural sources (GESAMP, 1990a; Zann, 1995). Clearly, long-term ecological sustainability of the marine environment is unlikely in the face of continued development unless limits are put on human activities that affect the environment, and in particular on the amounts and types of anthropogenic wastes discharged to coastal waters. These limits will generally be ecosystem and contaminant specific and relate directly to maintaining ecological integrity and current and future uses.

The coastal marine environments off Perth support a high diversity of animal and plant life consisting of both tropical and temperate species (Hutchins, 1991; see section 2.4). Sheltered embayments, such as Cockburn and Warnbro sounds, are particularly valuable natural assets; the community derives pleasure from their aesthetic qualities and uses their natural attributes for a variety of purposes. The Cockburn Sound region is also of particular economic importance as a major site for heavy industry in Western Australia. Uses of the southern coastal waters of Perth include education, conservation, recreation, commercial activities and transport; these waters also receive industrial and municipal wastes (see sections 2.6 and 4.1). Some of these uses can be clearly incompatible with each other. For example, discharged waste can reduce the cleanliness and quality of waters near the discharge point making these areas unsuitable for activities such as swimming. In principle, one use must not be allowed to degrade the waterbody to the extent that other attributes or uses are impaired. For this reason, and for the sustainable use of the environment, uses of marine waters in Western Australia are designated by the EPA. These uses were formerly termed *beneficial uses* (Department of Conservation and Environment, 1981). The term *environmental value* has replaced *beneficial use* in the national and draft Western Australian water quality guidelines (ANZECC, 1992; EPA, 1993) and is used here for consistency.

3.2 Definitions

3.2.1 Biodiversity

Biodiversity refers to the variety of all life forms: the different plants, animals and micro-organisms, the genes they contain and the ecosystems they form (Department of Environment, Sport and Territories, 1993). In this study the term biodiversity is used in the context of genetic diversity at the species level as opposed to assemblages of species. The intent of protecting biodiversity is to ensure that no 'species' becomes extinct.

3.2.2 Ecosystem integrity

Ecosystem integrity is defined as "*... the ability to support and maintain a balanced, integrative, adaptive community of organisms having a species composition, diversity and functional organisation comparable to that of natural habitat of the region*" (Karr, 1991; Karr and Dudley, 1981). Ecosystem integrity can be expressed in terms of structure (e.g. species richness) and function (e.g. primary production).

3.2.3 Environmental values

An environmental value is any natural (i.e. ecological) attribute or societal use of the environment that is conducive to public benefit, welfare, safety or health. Environmental values require protection from the detrimental effects of any direct or indirect alteration of the environment. The National (ANZECC, 1992) and draft Western Australian (EPA, 1993) water quality guidelines identify two key environmental values that are relevant to the southern metropolitan coastal waters of Perth:

- *Ecosystem Protection, and*
- *Recreation and Aesthetics.*

A third environmental value *'Industrial Water Supply'* is also relevant. From an industrial water supply perspective, water from Cockburn Sound is used primarily for cooling purposes and, as the quality requirements for this purpose are much lower than for *Ecosystem Protection* and for *Recreation and Aesthetics*, the environmental value *'Industrial Water Supply'* is protected by default and will not be considered further in this report.

3.2.4 Environmental quality objectives

Environmental quality objectives represent the goals of an environmental management programme and relate to both ecological (i.e. maintenance of biodiversity and ecosystem integrity) and cultural values (i.e. maintenance of community uses and aspirations) of natural systems. Ecological EQOs are fundamental management goals whereas cultural EQOs are, by definition, negotiable and generally derived from a balance between existing and future uses after due consideration of economic, social and political factors.

3.2.5 Environmental quality criteria

Environmental quality criteria (EQC) are the benchmarks upon which a decision or judgement may be made concerning the ability of the environment of a given quality to meet a designated environmental quality objective. The criteria for ecological EQOs and some cultural EQOs (e.g. maintenance of aquatic life for human consumption) are determined on the basis of technical information. Criteria for other cultural EQOs, such as the maintenance of aesthetic values, are determined in a more subjective manner.

3.2.6 Outfalls, mixing zones and exclusion zones

Outfalls are points of effluent discharge to waterbodies and they are surrounded by an area where the concentration of pollutants rapidly decreases through dilution. This area is termed an 'initial dilution zone' or 'mixing zone'. Well-designed and properly sited outfalls promote rapid mixing of pollutants in small mixing zones and minimal drift of undiluted effluent. However, it is not always possible to maintain water quality within the mixing zone at levels below the relevant EQC, leading to the situation where some or all of the EQOs are not met and the corresponding Environmental Values are not protected within a zone around an outfall.

It is important that the total area where Environmental Values are not protected is minimised. Hence the individual sizes and total number of these areas should be restricted to ensure the overall value of the environment is not significantly depreciated. To this end such areas may be declared 'exclusion zones' where some or all of the Environmental Values are not protected. In some instances designated exclusion zones may vary in size according to which Environmental Value is being protected.

This formal designation serves a dual purpose; it provides clear guidance for regulation and management and also informs the community of where, and to what extent, designated Environmental Values are not protected. Other areas where exclusion zones may be designated are within some types of harbours and around industrial and naval wharves.

3.3 Approaches to waste management

3.3.1 Philosophical approach

Although the philosophical approach of the EPA and the DEP to waste management is outlined in section 1.4.1, it is repeated here to provide the background for the following sections. This approach centres around several key points: that society produces waste; that even with the best recycling and re-use programmes, some of this waste will inevitably require disposal to the natural environment, and that any amount of waste entering the environment causes change, even if this alteration cannot be measured. Furthermore, between the state of no change (i.e. from natural variation) and the state just before the ecological integrity of the natural system in question begins to break down, there is a level of change in the natural environment, induced by human activity, that society may be prepared to accept for economic, social or other perceived benefits. The implicit assumption here is that the maintenance of biodiversity and ecosystem integrity are fundamental environmental attributes that should never be compromised (IUCN/UNEP/WWF, 1991). With this proviso, the acceptable level of change is determined, therefore, from a balance between existing and perceived future uses on the condition that the principle of inter-generational equity (i.e. avoidance of significant irreversible change) is maintained (Boelens, 1992; Nip and Udo de Haes, 1995). In this context unacceptable change is defined in terms of measurable departures from clearly stated environmental quality objectives (GESAMP, 1991a).

The approach outlined above is consistent with the State Conservation Strategy for Western Australia (Department of Conservation and Environment, 1987) and with the National and World Conservation Strategies.

The long-term management of the metropolitan coastal waters of Perth therefore requires that a clearly stated set of environmental quality objectives has been designated; that links between human activity and environmental quality are sufficiently well-understood (i.e. with an acceptable level of confidence); that appropriate environmental quality criteria are identified, and that effective management frameworks, including regulatory (e.g. licence setting, environmental impact assessment), feed-back (e.g. monitoring) and auditing mechanisms, exist. This combination of prediction, acknowledgment of uncertainty, monitoring and review allows management strategies to have the flexibility to be adapted to an improved knowledge base provided by well-planned monitoring programs (ANZECC, 1992). Therefore, a collective long-term vision of what society wants for these waters and a good scientific understanding of the impact of human activities on these natural systems are required before management strategies can be put in place which ensure that long-term environmental sustainability is achieved.

3.3.2 Different pollutant types

All chemical pollutants can be broadly classified as either biostimulants (e.g. nutrients) or toxicants, including bio-accumulatory substances. Examples of this latter group include naturally-occurring substances, such as heavy metals and hydrocarbons, and synthetic substances, such as polychlorinated biphenyls and organotin compounds. The environmental impact of these two pollutant types is different and therefore different waste management strategies are needed to prevent undesirable impacts resulting from their input to the environment.

The extent to which most toxic substances affect aquatic biota is primarily related to their concentration. Adverse effects occur when concentrations exceed critical thresholds and therefore the best approach for managing the environmental impacts of naturally-occurring toxic substances is through the application of concentration-related environmental quality criteria, based on the results of toxicity studies. In contrast, the introduction of synthetic toxic materials such as DDT and TBT into the marine environment has caused widespread deleterious effects at extremely low concentrations (Bryan and Gibbs, 1990; GESAMP, 1990b; GESAMP, 1991b; Long *et al.* 1995). Given the paucity of scientific knowledge of the ecological effects of many synthetic substances, the only safe control approach for this class of pollutant is to completely restrict their entry into natural ecosystems.

Nutrients are rarely toxic, even at concentrations found in the most polluted marine waters. Rather, excessive inputs of nutrients stimulate the growth of planktonic, epiphytic and benthic algae, leading to the decline of benthic plant communities through light starvation induced by shading and smothering (Department of Conservation and Environment, 1979; Shepherd *et al.* 1989; Simpson and Masini, 1990). A concentration-based approach (e.g. water quality criteria) to nutrient management is therefore inappropriate because it provides insufficient protection of environmental (ecological) values, due to the confounding effects of nutrient uptake by marine plants (Nixon *et al.* 1986; Costa *et al.* 1992). An alternative approach is to adopt the concept that a natural system has the capacity to receive some level of anthropogenic nutrient input without unacceptable changes occurring.

This concept has been defined using a variety of terms including: *assimilative capacity* or *environmental capacity* (GESAMP, 1986; EPA, 1990c; Masini *et al.* 1992); *receiving capacity* or *absorptive capacity* (UNESCO, 1988; WAWA, 1994) and *carrying capacity* (French, 1991; Jenkins, 1991). Regardless of what name is used this concept is now recognised as central to the principle of ecological sustainability (IUCN, 1991; Jenkins, 1991; Folke *et al.* 1993). An ecosystem-based approach linking nutrient loadings to environmental response is outlined in Pearce (1993) and this approach has been adopted in the SMCWS.

3.3.3 Marine pollution prevention practices

The EPA can prevent pollution under the Environmental Protection Act 1986 through environmental impact assessment procedures and through the development of Environmental Protection Policies.

A primary role of the DEP is to prevent pollution and this is achieved by controlling the discharge of wastes to the environment. 'Pollution' in this sense refers to any direct or indirect alteration of the environment to its detriment or degradation. A licence is required by any industry prescribed in regulations under the Environmental Protection Act 1986. 'Prescribed premises' include those that are likely to cause pollution of the environment if their waste discharges are not controlled. The DEP can revoke, suspend or change the conditions of licences at any time.

Licences set limits for effluent discharges and vary according to discharge type, quality and quantity and also according to the characteristics of the receiving waterbody, including consideration of cumulative impacts. Effluent discharge limits are expressed as either the maximum allowable concentration of each contaminant in the effluent stream, and/or as a total load of each contaminant per unit time, and are generally accompanied by monitoring requirements, both of the effluent and of the receiving environment. Monitoring requirements are an integral component of the licences and are set to ensure that all waste treatment equipment is operating efficiently, discharge limits are not exceeded, and the environment is protected. Sampling frequency varies according to the parameter being measured and the timescale and severity of potential impacts. The DEP can independently take samples of the effluent stream to check the reliability of the licensee's self-monitoring programmes.

Breaches of a licence condition constitute an offence under the Environmental Protection Act 1986. Additional powers under this Act include the provision to serve a Pollution Abatement Notice which may require the discharger to prevent or abate pollution within a specified time. If the Notice is not complied with and serious pollution continues, the Minister for the Environment may close the offending operation.

3.4 Environmental quality objectives

Environmental quality objectives (EQOs) represent the long-term aims or goals of an environmental management programme. The intent of this EQO-based management approach is:

To maintain the environmental quality of the water body for the widest possible range of environmental values, while recognising the current and projected future uses (eg. recreation, aquaculture, industry, etc.)

This EQO-based framework for environmental management was originally developed for essentially "non-degraded" ecosystems. To ensure that the framework applies consistently, but appropriately, to all coastal ecosystems in the State it also needs to accommodate areas that have a history of environmental degradation. Management responses in these areas can be complicated by a number of factors. For example, the original cause of the problems may have ceased or may be on-going, contaminant inputs may originate from multiple points or from diffuse sources, cumulative impacts are likely to be present and the cause-effect relationships complex. Additionally, some changes that can occur (such as loss of *Posidonia* seagrass meadows) are essentially irreversible. Given these considerations the intent of the EQO-based management approach in these historically degraded areas is:

To improve the environmental quality of the water body so that the widest possible range of environmental values is maintained or restored, while recognising the current and projected future uses (eg. recreation, aquaculture, industry, etc.)

The process of establishing EQOs and associated criteria for the southern metropolitan coastal waters of Perth was separated from the scientific studies of the SMCWS by the EPA in 1992. This process was revised by the current EPA in 1995, to establish EQOs/EQC for the entire metropolitan coastal waters of Perth. This is seen as a way to integrate the technical output of the SMCWS and the PCWS to provide the basis for strategic planning and long-term environmental management of existing and proposed waste inputs to Perth's coastal waters (see section 3.6).

The proposed actions and recommendations in this report are based on draft EQOs and their associated criteria, proposed by the EPA and DEP for the southern metropolitan coastal waters of Perth. Although the general nature of the draft EQOs outlined below is unlikely to change when the EQO process is finalised, the spatial boundaries of the EQOs may be modified as a result of consultation with user groups and the community. Designated exclusion zones, where some or all of the EQOs do not apply, are also likely to be formally established. Draft environmental quality criteria for the EQOs are outlined in section 3.5.

DRAFT ENVIRONMENTAL QUALITY OBJECTIVE 1:

Maintenance of Biodiversity

Biodiversity values (see above for definition) will be protected in the entire southern metropolitan coastal waters of Perth.

This objective means that if biodiversity values are found to exist in the southern metropolitan coastal waters of Perth (e.g. the discovery of local endemic species), these values will be automatically protected.

DRAFT ENVIRONMENTAL QUALITY OBJECTIVE 2:

Maintenance of Ecosystem Integrity

The EPA considers that the maintenance of the ecological integrity of the southern metropolitan coastal waters of Perth is a fundamental value requiring a high level of protection and should be considered in terms of ecosystem structure and function. In 1981, the EPA recognised three levels or classes in relation to the protection of aquatic ecosystems (Department of Conservation and Environment, 1981). An eight-level system has been proposed for the Marmion Marine Park (Hillman *et al.* 1995) which encompasses all of the ecological environmental quality objectives outlined in this section. The system adopted here for draft EQO 2 is largely based on the three-class system developed by the EPA in 1981, with cross-referencing to the system proposed by Hillman *et al.* (1995).

EQO 2 (Class I) - Conservation Zone

> *This represents maximum protection for ecosystems and corresponds to the environmental quality of a natural or pristine state. Coastal waters subject to such a level of protection should not receive any waste discharges whatsoever, nor be affected by human-made changes within the catchment or surrounds of the waterbody, or the waterbody itself.*

Class I protection allows no detectable* change† in water or sediment quality and no change in the abundance/biomass or diversity of biota. Existing or proposed Marine Nature Reserves and sanctuary zones in Marine Parks are examples where this level of protection would apply. Class I is equivalent to Level 2 of Hillman *et al.* (1995).

EQO 2 (Class II) - Multiple Use Zone

> *This represents a high level of protection which requires that any contaminant discharges or human-made changes which do occur may be readily absorbed or withstood by the waterbody without any detectable effects on the biota or the functioning of the ecosystem.*

* detectable is defined in terms of sampling and analytical methodologies approved by the DEP

† change is defined in relation to departures from natural background conditions

Class II protection allows detectable changes in water and sediment quality outside designated exclusion zones, but not exceeding specified environmental quality criteria, and no detectable change in the abundance/biomass or diversity of biota. This class of protection would apply to the general coastal waters of Western Australia and to areas including *General Use* zones in Marine Parks, Marine Management Areas and Fish Habitat Protection Areas. Class II is equivalent to Level 3 of Hillman *et al.* (1995).

EQO 2 (Class III) - Industrial Buffer Zone

> *This represents a moderate level of protection which requires that contaminant discharges and human-made changes cause neither a detectable change in the diversity of the biota, nor loss of ecological function. However, resultant changes in the abundance/biomass of biota within this zone may occur.*

Class III protection allows detectable changes in water and sediment quality outside designated exclusion zones, but not exceeding specified environmental quality criteria for toxicants, and allows changes in the abundance/biomass but not the diversity of the biota, providing there is no loss of ecological integrity of the ecosystem of which this zone is part. Coastal waters immediately next to major development sites would be an example where this level of protection could apply. Class III is approximately equivalent to Level 5 of Hillman *et al.* (1995).

Spatial boundaries of Draft EQO 2

EQO 2 (Class I): Not applicable to the southern metropolitan coastal waters of Perth

EQO 2 (Class II): The entire southern coastal waters of Perth (State waters between Fremantle and Dawesville), apart from waters designated EQO 2 (Class III) or designated exclusion zones.

EQO 2 (Class III): The spatial boundaries of waters designated EQO 2 (Class III) in the southern metropolitan coastal waters of Perth will be determined during the EQO/EQC consulative process (see section 3.6).

DRAFT ENVIRONMENTAL QUALITY OBJECTIVE 3:

Maintenance of Aquatic Life (including molluscs) for Human Consumption

Aquatic life for human consumption will be maintained in the entire southern metropolitan coastal waters of Perth, apart from designated exclusion zones.

DRAFT ENVIRONMENTAL QUALITY OBJECTIVE 4:

Maintenance of Recreational Values

Recreational values, particularly direct contact recreation, will be protected in the entire southern metropolitan coastal waters of Perth, apart from designated exclusion zones.

DRAFT ENVIRONMENTAL QUALITY OBJECTIVE 5:

Maintenance of Aesthetic Values

The aesthetic values of the marine environment will be protected in the entire southern metropolitan coastal waters of Perth, apart from designated exclusion zones.

3.5 Environmental quality criteria

Environmental quality criteria (EQC) are established to provide a benchmark to help ensure that designated environmental quality objectives are met and environmental values are protected. The goal of environmental management should be that the relevant EQC is never exceeded. However, a single isolated exceedence of a particular parameter value would not necessarily indicate that an EQO had not been met. In practice, exceedences that were widespread, or commonly occurring in a localised area, would mean that the corresponding EQO had not been met. When an agreed EQO has not been met, the discharger(s) of the contaminant(s) would be required to develop and implement a remedial management program, with appropriate performance indicators, to ensure the EQO is met within a specified time frame. Regulators would be expected to recommend against any further increases in cumulative loading of that contaminant until EQOs were met and maintained at levels that would permit the consideration of further increases in loads. Reductions in allowable cumulative loadings through the amendment of licence conditions may also become necessary.

Criteria can be expressed either in narrative or quantitative form. The Western Australian draft water quality guidelines (EPA, 1993) form the basis of many of the draft EQC described below. These guidelines are largely drawn from the Australian Water Quality Guidelines for Fresh and Marine Waters (ANZECC, 1992) which were also adopted in the draft 'criteria for conservation of flora and fauna' proposed in the PCWS (Hillman *et al.* 1995).

The results of the SMCWS and the PCWS demonstrate the inherent spatial and temporal variability in the physical and chemical characteristics within Perth's coastal waters, and highlight the difficulty in establishing criteria that can be generally applied throughout Perth's coastal waters.

The EPA (1993) and ANZECC (1992) guidelines acknowledge this inherent variability and suggest that 'ecosystem-specific' information be used to develop appropriate guideline values and indices of environmental quality. The draft quantitative and narrative criteria developed for the PCWS (Hillman *et al.* 1995) largely relate to the local waters of Marmion Marine Park, off Perth's northern coastline. The draft EQC described here relate to Perth's southern metropolitan coastal waters. A cross-reference between the draft EQOs and the appropriate EQC is provided in Table 3.5-1.

The following sections provide draft narrative and quantitative criteria relating to each draft EQO. The schedules of criteria cover a broad spectrum of parameters and substances, but clearly some substances or parameters will be more relevant than others in any given situation and these would provide the focus for monitoring or management programmes. The general process by which the EQOs and EQC for Perth's coastal waters will be formalised is outlined in Section 3.6.

3.5.1 Draft Environmental Quality Criteria for Maintenance of Biodiversity (EQO 1)

The draft EQC for EQO 1 are the natural background physical, chemical and biological characteristics of the prescribed area.

3.5.2 Draft Environmental Quality Criteria for Maintenance of Ecosystem Integrity (EQO 2)

3.5.2.1 EQO 2 (Class I) - Conservation Zone

The draft EQC for EQO 2 (Class I) are the natural physical, chemical and biological characteristics of the prescribed area.

3.5.2.2 EQO 2 (Class II) - Multiple Use Zone

Narrative EQC

Species richness, species composition, primary production and maintenance of ecological function are important attributes of ecosystems. Table 3.5-2 lists the draft narrative EQC in relation to these attributes for EQO 2 (Class II) protection.

Nutrient-related EQC

A concentration-based approach to nutrient management in isolation is inappropriate for local marine ecosystems because of the confounding influence of biological uptake (see section 3.3.2). Despite this limitation, nutrient concentrations that are above typical background concentrations can be indicative of an existing problem, or an emerging problem, and their spatial distribution can also help determine the source/s and the minimum 'zone of influence' of nutrient inputs.

Indicative background nutrient concentrations for Perth's southern coastal waters were determined from the results of water quality surveys in areas not considered to be influenced by anthropogenic inputs (see section 4.7). Expressed as a median and 90th percentile (in parentheses) value, the typical background nutrient concentrations were $< 6(9)$ µg l^{-1} for total inorganic nitrogen and $< 5(6)$ µg l^{-1} for total inorganic phosphorus.

Table 3.5-1. Key to tables linking draft environmental quality objectives to their criteria for the southern metropolitan coastal waters of Perth.

Draft environmental quality objective (EQO)	Table 3.5-2	3.5-3	3.5-4	3.5-5	3.5-6	3.5-7
EQO 1 (Maintenance of biodiversity)	Any change from the natural state is unacceptable					
EQO 2 (Maintenance of ecosystem integrity)						
Class I conservation zone	Any change from the natural state is unacceptable					
Class II multiple-use zone	*	*	*	*		*
Class III industrial buffer zone			*	*		
EQO 3 (Maintenance of aquatic life (including molluscs) for human consumption)			*		*	*
EQO 4 (Maintenance of recreational values)						*
EQO 5 (Maintenance of aesthetic values)						*

Table 3.5-2. Draft narrative environmental quality criteria for biological indicators for the maintenance of ecosystem integrity (EQO 2 Class II). Data from ANZECC (1992) and EPA (1993).

Biological indicator	Narrative environmental quality criteria
Species richness	*In any water body, the species richness of the predominant macrophyte, periphytic, phytoplanktonic, benthic and planktonic invertebrate or vertebrate assemblages, as measured by an appropriate standardised index, should not be altered.*
Species composition	*In any water body, impacts that result in significant changes in species composition compared with those in similar, local unimpacted systems should not be permitted.*
Primary production	*In any water body, net primary production should not vary from levels encountered in similar, local unimpacted habitats under similar light, temperature and nutrient loading regimes.*
Ecosystem function	*In any water body, changes that vary the relative importance of the detrital and grazing food chains should be minimised. Production to respiration ratios should not vary significantly from those of similar, local unimpacted systems.*

Light is essential for the survival of plants. The chlorophyll *a* component of water column light attenuation is a key factor regulating available light to benthic plant communities in Perth's coastal waters. Seagrasses are the dominant benthic primary producers in the shallower, more protected areas of these waters and epiphyte assemblages growing on the leaves of these plants further reduce light reaching the leaf surface (see section 5.3). Nutrient enrichment of coastal waters generally results in increased phytoplankton biomass (measured as chlorophyll *a* concentration) and can also induce changes in the species composition and increase the biomass of benthic and epiphytic algae. These changes reduce the amount of light reaching the original benthic plant communities; therefore, even small changes in these parameters will ultimately be ecologically significant and, for this reason, chronic departures from background conditions are considered to be environmentally unacceptable. The draft nutrient-related EQC for EQO 2 (Class II) reflect these considerations (see Table 3.5-7). Values for these criteria are provided for nearshore and offshore coastal waters of the study area during the non-winter period (Table 3.5-3). The ecological indicators (or parameters) used are consistent with those employed for the PCWS and detailed by Hillman *et al.* (1995), although some of the criteria values differ slightly due to the different characteristics of the two main areas in question.

Computer simulations linking environmental quality and seagrass health in Cockburn Sound (see section 6.3) indicate that under the conditions described by the draft nutrient-related EQC for nearshore coastal waters (see Table 3.5-3), seagrass meadows would have occupied 91% of the original area and would have contained 85% of the original biomass estimated to have been present before this waterbody was affected by waste inputs.

The draft nutrient-related EQC presented here will be finalised during the consultative process (outlined in section 3.6) which will occur following the release of this report.

Toxic and bioaccumulatory substance-related EQC

Water and sediment quality draft criteria for EQO 2 (Class II) are presented in Tables 3.5-4, 3.5-5 and 3.5-7.

Toxicant-related criteria for the maintenance of ecosystem integrity are reasonably well established for water (e.g. ANZECC, 1992) but there is no equivalent set of criteria for sediments. The draft EQC for toxicants in sediments proposed here are based on the comprehensive review of Long *et al.* (1995) who compiled sediment chemistry and biological effects data from over 80 independent studies and used these to determine probable-effects concentrations below which biological effects would rarely occur (effects range low, ERL) or would possibly occur (effects range median, ERM). Concentrations at or above the ERM represent a probable-effects range within which effects would be expected to frequently occur. The toxicity of a substance to biota depends on its bioavailability, and in sediments, bioavailability is dependent on a range of physical and chemical characteristics of the sediment and overlying water. The sediment-related EQC will be finalised, along with the other criteria, during the consultative process outlined in section 3.6.

Under the proposed scheme for applying these criteria to management in Class II zones, exceedence of the ERL value for a particular contaminant would require dischargers of that contaminant to undertake monitoring, support criteria refinement and implement remedial management action as required. Regulators would be expected to recommend against any further increases in loading of that contaminant until EQOs were met and maintained at levels that would permit the consideration of further increases in cumulative loads. Reductions in allowable cumulative loadings through the amendment of licence conditions may also become necessary.

Although criteria for metals and toxicants in biota have been developed to protect humans who consume seafood (see EQO 3) there are no equivalent and generally accepted criteria for the maintenance of ecosystem integrity. Chegwidden (1979) proposed a set of values for heavy metal concentrations in mussels for Cockburn Sound that he considered would be indicative of 'contamination' from an ecological perspective. These values, together with values for mercury (Neilson and Natham, 1975) and tributyltin

Table 3.5-3. Draft nutrient-related environmental quality criteria for the non-winter♦ period for the maintenance of ecosystem integrity (EQO 2 Class II).

Indicator	Units	EQO 2 (Class II) Nearshore coastal waters (zone A*) mean (upper 90% confidence interval)		EQO 2 (Class II) Offshore coastal waters (zone B*) mean (upper 90% confidence interval)	
Light attenuation coefficient**	(m^{-1})	0.08	(0.09)	0.04	(0.05)
Phytoplankton	(μg chl *a* l^{-1})	0.8	(1.1)	0.2	(0.4)
Epiphyte biomass	(mg dry weight cm^{-2} seagrass leaf)	0.5	(1.2)	0.5	(1.2)
Epiphyte carbonate content					
*** for waters ≤ 10m depth	(%)	≥ 20 - 40		≥ 20 - 40	
**** for waters > 10m depth	(%)	≥ 36 - 62		≥ 36 - 62	

♦ 'non-winter' refers here to an extended period ≥3 months when estuarine flows are weak, typically between November and May inclusive.

* see Figure 2.4-1

** $\log_{10}$ basis, multiply by 2.303 to obtain extinction coefficient of the exponential light decay curve

*** from Hillman *et al.* (1995), for waters ≤10m depth (K. Hillman, personal communication)

**** from the results of the SMCWS, for waters >10m depth.

(Page and Widdows, 1991) in wet mussel flesh are provided as inicators of levels that may be ecologically significant. The indicator concentrations (mg kg^{-1}) are: Cadmium, 0.5; Chromium, 0.5; Copper, 1.5; Iron, 40.0; Lead, 1.0; Manganese, 2.0; Mercury, 0.35; Nickel, 1.0; Tributyltin, <0.01; and Zinc, 30.0. The concentrations are for wet mussel flesh and are expressed as the metal except for tributyltin which is expressed as the compound.

The values for contaminants in biota listed above are not proposed for use as criteria but rather as general indicators of environmental quality. There is a recognised need for biota-quality criteria to be developed as benchmarks to assess ecosystem health, and it is intended that these be developed during the consultative EQO/EQC finalisation process that will occur after the release of this report (see section 3.6).

3.5.2.3 EQO 2 (Class III) - Industrial Buffer Zone

Toxic and bioaccumulatory substance-related EQC

Water and sediment quality draft criteria for EQO 2 (Class III) are presented in Tables 3.5-4 and 3.5-5. The management application of the sediment EQC in Class III zones differs to that described in section 3.5.2.2 for Class II zones. In the Class III zones, exceedence of the ERL value for a particular contaminant would require dischargers of that contaminant to initiate investigations which include environmental monitoring and criteria refinement. While levels of the contaminant in the environment are well below the ERM or refined criterion value, management response would be held to the investigation level. If levels in the environment exceed the ERM (ie. within the range where effects would be expected to frequently occur), or exceed the refined criterion, dischargers would be required to immediately develop and implement a remedial management plan. Under these circumstances regulators would be expected to recommend against any further increases in loading of that contaminant until EQOs were met and maintained at levels that would permit the consideration of further increases in loads. Reductions in allowable cumulative loadings through the amendment of licence conditions may also become necessary.

3.5.3 Draft Environmental Quality Criteria for Maintenance of Aquatic Life (including molluscs) for Human Consumption (EQO 3)

Water and biota quality draft criteria for EQO 3 are listed in Tables 3.5-4, 3.5-6 and 3.5-7.

3.5.4 Draft Environmental Quality Criteria for Maintenance of Recreational Values (EQO 4)

Water quality draft criteria for EQO 4 are presented in Table 3.5-7.

3.5.5 Draft Environmental Quality Criteria for Maintenance of Aesthetic Values (EQO 5)

Water quality draft criteria for EQO 5 are presented in Table 3.5-7.

Table 3.5-4. Draft environmental quality criteria for toxic and bio-accumulatory substances in water for the maintenance of ecosystem integrity (EQO 2 Class II, Class III) and the maintenance of aquatic life for human consumption (EQO 3). All units are $\mu g\ l^{-1}$ unless otherwise stated. Data from ANZECC (1992) and EPA (1993; as updated).

Indicator	EQO 2 (Class II, III) ($\mu g\ l^{-1}$)	EQO 3 ($\mu g\ l^{-1}$)	EQO 2 (Class II, III), EQO 3 ($\mu g\ l^{-1}$)
Inorganic toxicants			
Antimony	500.0		500.0
Arsenic	50.0	0.02*	0.02
Beryllium	NR	0.1*	0.1
Cadmium	2.0	0.2	0.2
Chromium	50.0	2.0	2.0
Copper	5.0	4.0	4.0
Cyanide	5.0		5.0
Fluoride	2000.0		2000.0
Lead	5.0	1.3	1.3
Manganese		100.0	100.0
Mercury	0.1		0.1
Nickel	15.0	100.0	15.0
Selenium	70.0		70.0
Silver	0.45		0.45
Sulfide	2.0		2.0
Thallium	20.0		20.0
Tin (tributyltin)	0.002		0.002
Zinc	20.0		20.0
Organic toxicants			
Acrylonitrile	NR	0.7	0.7
Benzidine	NR	0.0005	0.0005
Dichlorobenzidine	NR	0.02	0.02
Diphenylhydrazine	NR	0.6	0.6
Surfactants	*0.05 times the ninety-six hour LC_{50} determined in the receiving water, on the most sensitive important species in the region.*		
Halogenated aliphatic compounds			
Chlorinated ethanes:			
1,2-dichloroethane		240.0*	240.0
1,1,2-trichloroethane		40.0*	40.0
1,1,2,2-tetrachloroethane		11.0*	11.0
Hexachloroethane		9.0	9.0
Chlorinated ethylenes			
Chloroethylene (vinyl chloride)		530.0*	530.0
1,1-dichloroethylene		2.0*	2.0
Trichloroethylene		80.0*	80.0
Tetrachloroethylene		9.0*	9.0
Halogenated methanes			
Carbon tetrachloride		7.0*	7.0
Chloroform		16.0	16.0
Other halogenated methanes		16.0*	16.0
Other halogenated aliphatics			
Hexachlorobutadiene	0.3		0.3
Halogenated ethers			
bis (chloromethyl) ether		0.002	0.002
bis (2-chloroethyl) ether		1.0	1.0

(continued overleaf)

Table 3.5-4 continued.

Indicator	EQO 2 (Class II, III) (µg l^{-1})	EQO 3 (µg l^{-1})	EQO 2 (Class II, III), EQO 3 (µg l^{-1})
Hydrocarbons (total)	10.0		10.0
Monocyclic aromatic compounds			
Benzene	300.0		300.0
Chlorinated benzenes			
1,2,4,5-tetrachlorobenzene		50.0	50.0
Pentachlorobenzene		80.0	80.0
Hexachlorobenzene		0.0007	0.0007
Chlorinated phenols			
2,4,5-trichlorophenol	8.0		8.0
2,4,6-trinitrophenol		4.0	4.0
Pentachlorophenol	0.2		0.2
Dinitrotoluene		9.0	9.0
Nitrosamines			
N-nitrosodiethylamine		1.0	1.0
N-nitrosodimethylamine		16.0	16.0
N-nitrosodibuthylamine		0.6	0.6
N-nitropyrrolidine		90.0	90.0
N-nitrosodiphenylamine		16.0	16.0
Phenol	50.0		50.0
Pesticides			
Organochlorine			
Aldrin	0.002	0.00008	0.00008
Chlordane	0.004	0.0005	0.0005
DDE	0.014		0.014
DDT	0.0005	0.00003	0.00003
Dieldrin	0.002	0.00008	0.00008
Endosulfan	0.0007		0.0007
Endrin	0.003		0.003
Heptachlor	0.0003	0.0003	0.0003
Lindane	0.003		0.003
Methoxychlor	0.040		0.040
Mirex	0.001		0.001
Toxaphene	0.008		0.008
Organophosphate			
Chlorpyrifos	0.001		0.001
Demeton	0.100		0.100
Guthion (Azinphos-methyl)	0.010		0.010
Malathion	0.100		0.100
Parathion	0.004		0.004
Other pesticides			
Acrolein	0.2		0.2
Polycyclic aromatic compounds			
Polychlorinated biphenyls	0.001		0.001
Polycyclic aromatic hydrocarbons (PAHs)	3.0	0.03	0.03
2,3,7,8-tetrachlorodibenzodioxin		0.00000001	0.00000001
Radionuclides		0.4 Bq l^{-1}	0.4 Bq l^{-1}

NR: no recommendation made at this time

* Potential carcinogen, risk level 1:1 000 000

(continued from previous page)

Table 3.5-5. Draft environmental quality criteria for selected heavy metals and organic toxicants in sediments for the maintenance of ecosystem integrity (EQO 2 Class II, Class III). Values are expressed as dry weight. Data from Long et al. (1995), unless stated otherwise.

Indicator	EQO 2 (Class II, ERL) (expressed as the metal, in mg kg^{-1})	EQO 2 (Class II, ERL) (expressed as the compound, in µg kg^{-1})	EQO 2 (Class III, ERM) (expressed as the metal, in mg kg^{-1})	EQO 2 (Class III, ERM) (expressed as the compound, in µg kg^{-1})
Metal				
Aluminium	7140.0*		17850.0**	
Arsenic	8.2		70.0	
Cadmium	1.2		9.6	
Chromium	81.0		370.0	
Copper	34.0		270.0	
Nickel	20.9		51.6	
Lead	46.7		218.0	
Zinc	150.0		410.0	
Mercury	0.15		0.71	
Organic toxicant				
Polychlorinated biphenyls		22.7		180.0
DDT (total)		1.6		46.1
Polycyclic aromatic hydrocarbons (total)		4022.0		44792.0
Tributyltin***		10.0		50.0

Based on * twice and ** five times background concentration (Pollution Research Pty Ltd, 1994) in Warnbro Sound sediments from water depths >10 m
*** after Waite *et al.* (1991)

Table 3.5-6. Draft environmental quality criteria for toxic and bio-accumulatory substances in seafood for the maintenance of aquatic life for human consumption (EQO 3). Values are expressed as wet weight. Data from Australian Food Standard A12, unless stated otherwise.

Metal	Food	EQO 3 (mg kg^{-1} calculated as the metal)
Antimony	All foods not otherwise specified	1.5
Arsenic	Fish, crustaceans and molluscs, seaweed (edible kelp) (inorganic arsenic only)	1.0
	All foods not otherwise specified	1.0
Cadmium	Molluscs	2.0
	Fish, seaweed (edible kelp)	0.2
	All foods not otherwise specified	0.05
Copper	Molluscs	70.0
	All foods not otherwise specified	10.0
Lead	Molluscs	2.5
	All foods not otherwise specified	0.5
Mercury	Fish, crustaceans and molluscs	(mean level) 0.5*
	All foods not otherwise specified	0.03
Selenium	All foods not otherwise specified	1.0
Tin	All foods not otherwise specified	150.0
Tributyltin	All foods	0.146†^
Zinc	Oysters	1000.0
	All foods not otherwise specified	150.0

* Sampling provisions apply

^ calculated as mg tributyltin kg^{-1}

† World Health Organisation (1990)

Table 3.5-7. Draft environmental quality criteria for biological and physico-chemical indicators in water and biota for the maintenance of ecosystem integrity (EQO 2 Class II), aquatic life for human consumption (EQO 3), recreational (EQO 4) and aesthetic (EQO 5) values. Data from ANZECC (1992), EPA (1993), Hillman et al. (1995) and results from this study.

Indicator	EQO 2 (Class II)	EQO 3	EQO 4, 5	EQO 2-5
Biological Algae	Non-winter* average phytoplankton biomass (measured as chlorophyll *a*) and non-winter average epiphyte biomass (expressed as total dry weight/unit area of seagrass leaf), should not exceed the upper 90% confidence limit of the long-term average for these parameters in similar, local unimpacted systems. Mean of non-winter values for 3 years should not exceed the long-term non-winter averages of these parameters in similar, local unimpacted systems. Epiphyte $CaCO_3$ content should be $\geq$ 20% in water $\leq$ 10 m and $\geq$ 36% in water > 10 m.	No guideline. Toxins may be present in cyanobacteria and may be accumulated in other aquatic organisms	Macrophytes, phytoplankton scums, filamentous algal mats, etc. should not be present in excessive amounts. Direct contact should be discouraged if algal levels above 15 - 20 x 10^6 cells l^{-1}, depending on species	Non-winter* average phytoplankton biomass (measured as chlorophyll *a*) and non-winter average epiphyte biomass (expressed as total dry weight/unit area of seagrass leaf), should not exceed the upper 90% confidence limit of the long-term average for these parameters in similar, local unimpacted systems. Mean of non-winter values for 3 years should not exceed the long-term non-winter averages of these parameters in similar, local unimpacted systems. Epiphyte $CaCO_3$ content should be $\geq$ 20% in water $\leq$ 10 m and $\geq$ 36% in water > 10 m.
Molluscs	*Thais orbita.* > 5% imposex (incidence of penises in females) indicative of potential tributyltin contamination. *Mytilis edulis.* The occurrence of shell deformities may indicate chronic low level pollution.			*Thais orbita.* > 5% imposex (incidence of penises in females) indicative of potential tributyltin contamination. *Mytilis edulis.* The occurrence of shell deformities may indicate chronic low level pollution. Macrophytes, phytoplankton scums, filamentous algal mats, etc. should not be present in excessive amounts. Direct contact should be discouraged if algal levels exceed 15 - 20 x 10^6 cells l^{-1}, depending on species
Biotoxins *Gonyaulax* shellfish toxins		< 0.8 $\mu g\ g^{-1}$ shellfish		< 0.8 $\mu g\ g^{-1}$ shellfish.
Faecal coliforms		The median faecal coliform bacterial concentration should not exceed 14 MPN/100 ml, with no more than 10% of the samples exceeding 43 MPN/100 ml.	The median bacterial content over the bathing season should not exceed 150 faecal coliform /100 ml; or 35 enterococci organisms/100 ml.	The median faecal coliform bacterial concentration should not exceed 14 MPN/100 ml, with no more than 10% of the samples exceeding 43 MPN/100 ml.

* 'non-winter' refers here to an extended period $\geq$ 3 months when estuarine flows are weak, typically between November and May inclusive.

(continued overleaf)

Table 3.5-7 continued.

Indicator	EQO 2 (Class II)	EQO 3	EQO 4, 5	EQO 2-5
Physico-chemical				
Colour & clarity	< 10% change in euphotic depth. Non-winter* average light attenuation coefficent should not exceed the upper 90% confidence limit of the long-term average of similar, local unimpacted systems. Mean of non-winter values for 3 years should not exceed the long-term average of similar, local unimpacted systems.		≤ 20% reduction in clarity, < 10 point variation in hue based on Munsell scale, natural reflectance should not change by > 50%; no noticible hydrocarbon film or odour	< 10% change in euphotic depth. Non-winter* average light attenuation coefficent should not exceed the upper 90% confidence limit of the long-term average of similar, local unimpacted systems. Mean of non-winter values for 3 years should not exceed the long-term average of similar, local unimpacted systems. < 10 point variation in hue based on Munsell scale, natural reflectance should not change by > 50%; no noticible hydrocarbon film or odour
Dissolved oxygen	> 6 mg l^{-1}(> 80–90% saturation)			> 6 (> 80–90% saturation)
pH	< 0.2 pH unit change		5.0 - 9.0	< 0.2 pH unit change
Suspended particulate matter/turbidity	< 10% change seasonal mean concentration (see also colour & clarity)			< 10% change seasonal mean concentration (see also colour & clarity)
Temperature	< 2 °C increase		15 - 35 °C for prolonged exposure	< 2 °C increase

* 'non-winter' refers here to an extended period ≥ 3 months when estuarine flows are weak, typically between November and May inclusive.

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3.6 Management of Perth's coastal waters

In March 1995 the EPA decided to undertake a process to formally consider the issue of long-term environmental management of the entire metropolitan coastal waters of Perth, nominally from Dawesville to Yanchep. An integral part of this process is to establish Environmental Quality Objectives and Environmental Quality Criteria for these waters. Draft objectives and criteria, based on a common set of Environmental Values, have been established for the southern and northern sections of the metropolitan coastal waters during the Southern Metropolitan Coastal Waters Study (sections 3.4 and 3.5) and the Perth Coastal Waters Study (Hillman *et al.* 1995), respectively. These draft objectives and criteria will provide the basis for the consultative process between the EPA and community/user groups that is required before the objectives, their spatial application and the associated criteria are finalised. This integrative process is outlined in Figure 3.6-1 and will culminate with a report to Government regarding strategic environmental planning and management of Perth's coastal waters into the next century.

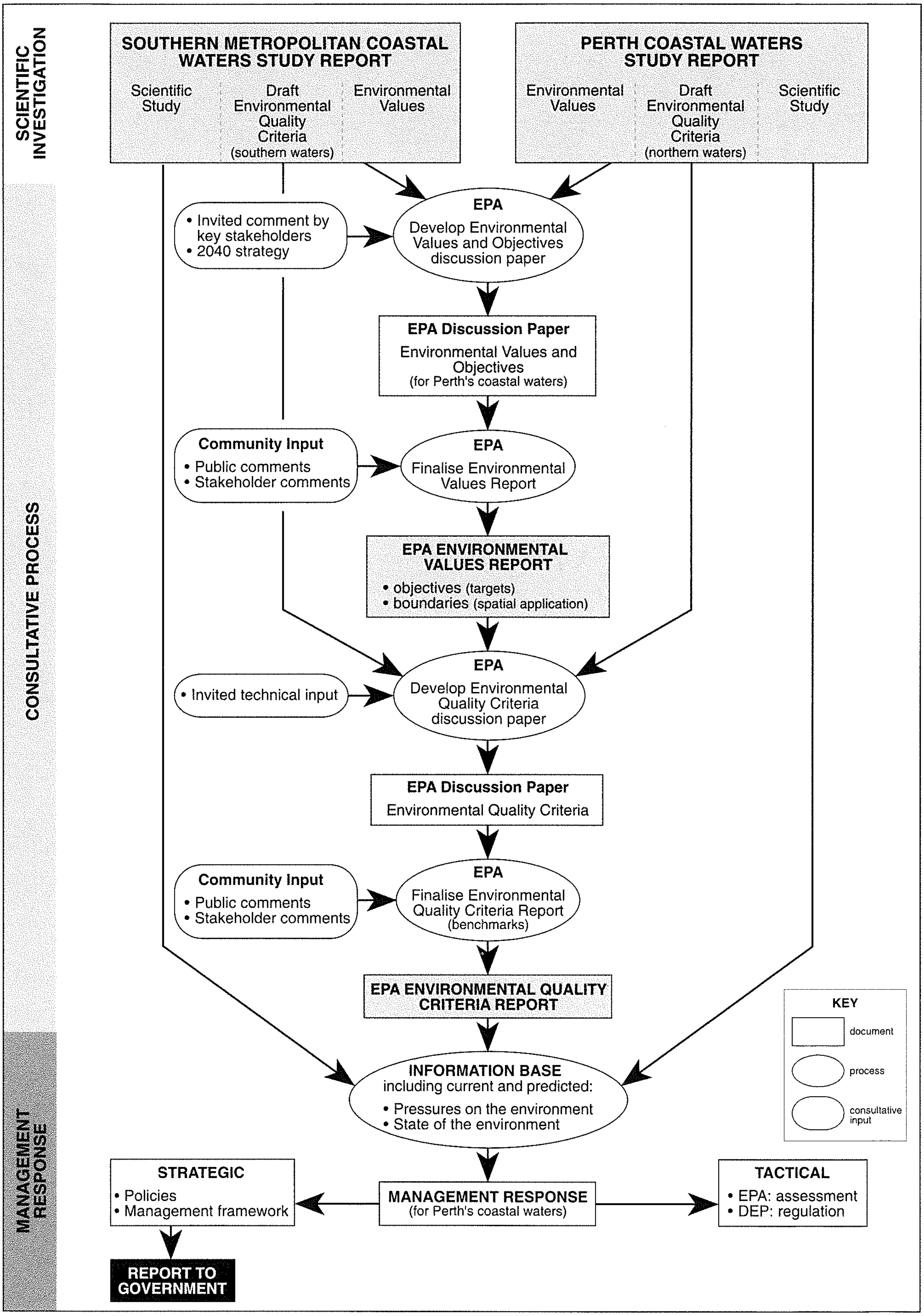


Figure 3.6-1. Flow diagram showing how the Southern Metropolitan Coastal Waters Study and the Perth Coastal Waters Study will be integrated and, with stakeholder and community consultation, used to set Environmental Quality Objectives and Criteria and provide the information needed to develop an integrated management response in relation to waste inputs to Perth's coastal waters.

PART II

SUMMARY OF STUDY FINDINGS

This part of the report consists of three chapters that outline the major findings of the scientific and technical investigations carried out during the Southern Metropolitan Coastal Waters Study. Chapter 4 discusses the findings of the inventory and baseline studies. These studies were undertaken to characterise the study area, provide a status report and to develop quantitative baseline data sets for future reference. Chapter 5 describes the results of process studies that were undertaken to identify the dominant factors that influence the physical, chemical and biological characteristics described in Chapter 4. Chapter 6 describes the integration of the physical and biological process information into simulation models of relevance to the environmental management of the study area.

4. INVENTORY AND BASELINE STUDIES

Inventory studies were undertaken to provide a current quantitative description of selected aspects of the biological and chemical environment of the study area. Where historical data do not exist these studies have the additional benefit of providing baseline data for future reference. Although most of the studies concentrated on the 'core' areas of Cockburn Sound, Owen Anchorage, Warnbro Sound and, to a lesser extent, Sepia Depression, other areas such as Comet Bay, Gage Roads and Rottnest Island were included in most of the surveys. This was done, firstly, in recognition of the need for a broader context within which to view the results from the 'core' study areas, secondly, in acknowledgment of the spatial scales of some of the influences on Perth's marine environment (e.g. the Peel-Harvey Estuary outflow), and thirdly, to take advantage of the minimal extra cost of obtaining additional baseline information in more distant areas that will inevitably become part of the metropolitan coastal waters in the near future. Many of the studies outlined in this chapter provide basic information requirements for the process studies and modelling, described in Chapters 5 and 6, respectively.

4.1 Coastal resource atlas

With approximately 70% of the State's population living within about 20 km of the metropolitan coastline (Western Australian Planning Commission, 1995) the coastal waters of Perth are the most used and, therefore, valuable part of Western Australia's marine environment. These waters are used for a wide variety of recreational activities including boating, swimming and fishing. They are also used for numerous commercial purposes such as shipping, aquaculture and large-scale shellsand mining. Furthermore, these waters receive considerable quantities of industrial and domestic wastewater, as well as urban and agricultural runoff. Effective environmental protection and management requires an understanding of the interaction of uses and equitable reconciliation of conflicts between these uses. To this end the current uses and major marine resources of the study area were identified and mapped in a Coastal Resource Atlas (CRA) as a series of computer-based maps.

Mapping of the bathymetry, major benthic habitats, marine biological resources and the current commercial and recreational usage of the study area was undertaken in collaboration with the Department of Transport (Maritime Division). This project served as a pilot project for the Coastal Resource Atlas of Western Australia (CRAWA), being developed collaboratively between the Coastal Information and Engineering Services Branch of the Department of Transport (DOT), the Department of Environmental Protection (DEP), the Remote Sensing Applications Centre of the Department of Land Administration (DOLA), the Department of Fisheries and the Australian Maritime Safety Authority.

The bathymetry of the southern metropolitan coastal waters was digitised from the original point soundings collected by the Royal Australian Navy and the DOT, to provide an accurate (± 0.1m depth), high resolution (< 50 m) digital database. The digital format provides the flexibility to use the data for a wide variety of purposes. In the present study the bathymetry was used for modelling, line contouring and production of three dimensional bathymetric relief maps (Plate 4.1-1). The digital information is owned by the DOT and archived at the Department's Marine Division.

The major benthic habitats of the southern metropolitan coastal waters were mapped to obtain an accurate representation of the current habitat distribution throughout the study area and as a reference for quantifying past and future habitat changes, such as seagrass cover (see section 4.2). The mapping was based on a high resolution (± 5 m), spatially rectified, digital image of the entire metropolitan coastal waters (Yanchep to Mandurah) that was recorded using a Geoscan airborne multi-spectral scanner during February 1993. The study area was classified according to the main benthic habitat types using an auto-classification technique described by Ong *et al.* (1995). As detailed in Ong *et al.* (1995), ground truthing at over 100 sites in the area indicated that this classification technique achieved a degree of accuracy of greater than 70% for the relatively shallow inshore habitats (< 10 m depth) and greater than 60% for the deeper offshore habitats (> 20 m depth). Benthic habitats were mapped in collaboration with the Remote Sensing Applications Centre of the Department of Land Administration, the Department of Agriculture and the Department of Transport. The digital information, jointly owned by the Department of Environmental Protection and the Water Corporation, is archived at the Remote Sensing Applications Centre.

4.1.1 Bathymetry

The major bathymetric features of the nearshore coastal waters of Perth are primarily the result of the Holocene sea level rise and the presence of three ridges of the Tamala Limestone dune ridge system (see section 2.3). The sea level rise and inundation of the depressions between these ridges led to the formation of Owen Anchorage, Cockburn Sound, Warnbro Sound and Sepia Depression. Onshore, the Spearwood Ridge is generally within 1-2 km of the present coastline. Offshore, the Garden Island Ridge now forms an extensive system of intertidal and subtidal limestone reefs and islands, extending from southern Comet Bay to Rottnest Island and including Penguin Island, Cape Peron, Garden Island, Carnac Island and the Straggler Rocks off Fremantle (Plate 4.1-1, Figure 1.1-1, Figure 1.1-2). Further offshore, Five Fathom Bank extends from Cape Bouvard, southwest of Mandurah, to the western end of Rottnest Island.

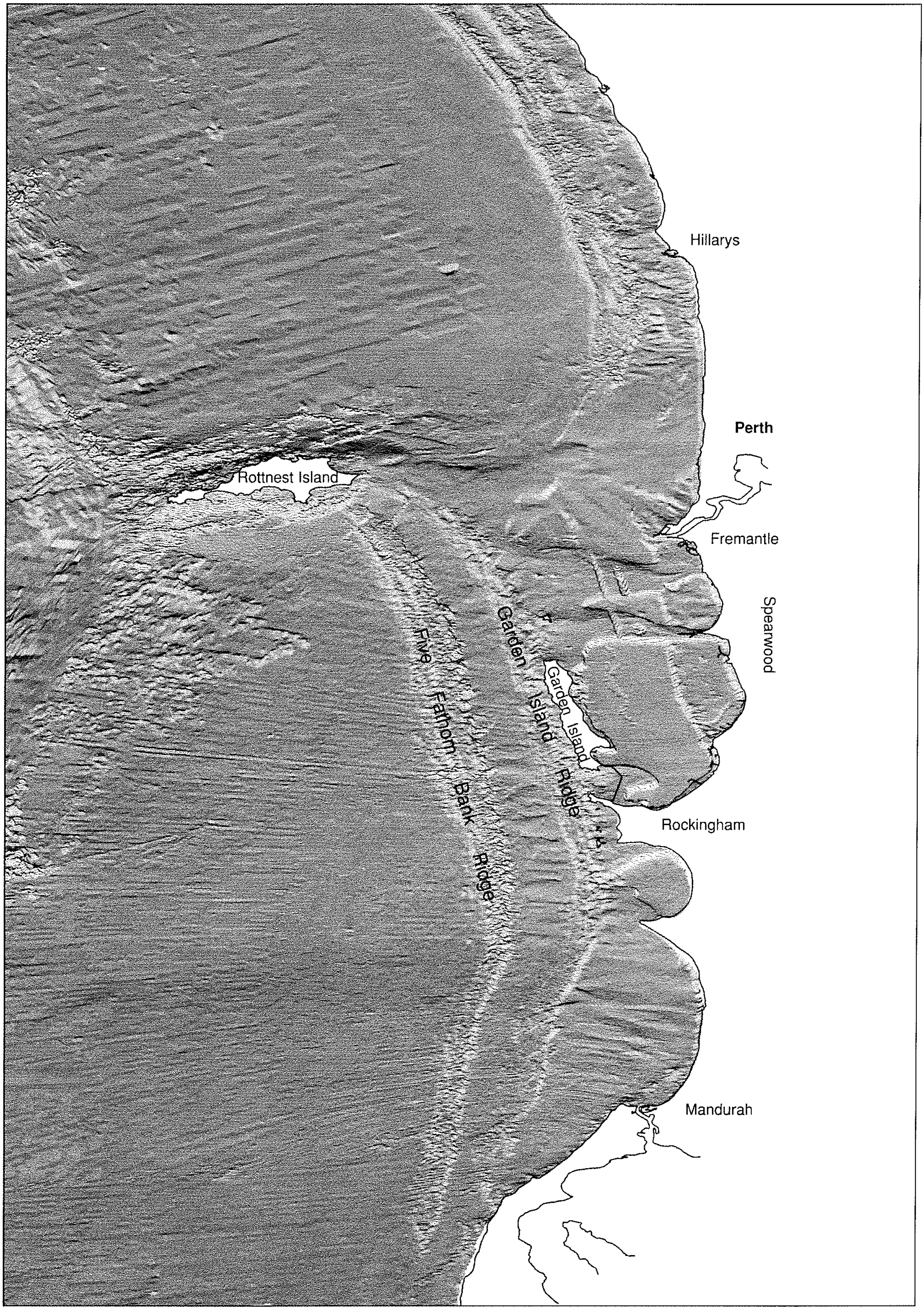


Plate 4.1-1. Three-dimensional bathymetry of the study area.

Between these offshore ridges is Sepia Depression with a maximum depth of approximately 20 m. West of the Five Fathom Bank the continental shelf gradually slopes to the shelf break about 50 km offshore. The offshore ridges, particularly the islands and reefs of the Garden Island Ridge, form a partial barrier that protects the coastline from offshore swell, creating relatively low energy coastal lagoons and embayments. In the coastal lagoons water depths are typically less than 10 m, whereas the basins of Cockburn and Warnbro sounds have maximum water depths exceeding 18 m. Parts of the offshore ridges have been eroded over this period and the material deposited as sediments to form shallow banks and sills inshore. Parmelia and Success Banks at the northern entrance to Cockburn Sound are two parallel sills (typically about 3-5 m deep) that were formed in this way. Similarly, at the southern entrance to the Sound is the Southern Flats, a shallow sill of 1-3 m depth. Along most of the eastern shoreline of Cockburn Sound and Owen Anchorage wave action has eroded the shoreline creating a submerged shelf about 3 km wide and 6-9 m deep.

4.1.2 Major benthic habitats

The mapped area was classified into seven benthic habitat types (Figure 4.1-1). The actual and percentage cover for each habitat type has been calculated for the entire mapped area, 'core' areas and other selected areas (Table 4.1-1).

Silt

The area classified as silt occurs in the protected deep basins (> 10 m depth) of Warnbro Sound and Cockburn Sound (Figure 4.1-1). Compared with sediments in more exposed areas, the sediments of these deep basins are relatively rich in organic silt (~ 20%) and fine carbonate sands (~ 25%). The floral and faunal assemblages that live on these soft sediments are found nowhere else on the central west coast of Western Australia (Wilson *et al.* 1978). The plant communities are impoverished, primarily due to light limitation, and are generally restricted to a few species of green and red algae, but benthic microalgae may be abundant. In contrast, there is a high diversity and abundance of fauna living on the seabed and in the upper layer of the sediments. Invertebrates are the most abundant and diverse component of the benthic fauna, particularly the polychaetes, crustaceans and molluscs. Species richness of the invertebrate fauna was generally high throughout the Warnbro Sound basin whereas, in Cockburn Sound, there was a distinct decrease in species richness from the northern to the southern parts of the basin (Cary *et al.* 1995c). Further details of the benthic fauna survey conducted in 1993 in the deep basins of Cockburn and Warnbro sounds are provided in section 4.10.

The area classified as silt occupies nearly 10% of the total area mapped. This habitat occupies about 60% of the area in Cockburn Sound and nearly 25% of the area in the Shoalwater Island Marine Park, containing Warnbro Sound (Table 4.1-1).

Fine sand and silt

The area classified as fine sand and silt only occurs in the deeper (~12 m) western part of Owen Anchorage (Figure 4.1-1), between the relatively shallow sills (3-5 m) of Parmelia and Success banks and inshore of the Fremantle Port Authority shipping channel (Figure 1.1-2). The sediments of this area consist of unconsolidated fine carbonate sands and silt with a relatively high organic content (~ 5%) compared to other parts of Owen Anchorage (~ 3%). The floral assemblage of this habitat is also generally impoverished, like the deep basins, due to light limitation (Le Provost, Dames and Moore, 1994). The benthic fauna, like the deep basins, is dominated by invertebrates living on the seabed (e.g. echinoderms) and in the surface sediments (e.g. polychaetes, crustaceans and molluscs).

Table 4.1-1. Area and relative cover of the major benthic habitats in three sub-regions of the study area (see Figure 1.1-2 for the sub-region boundaries) determined from a Geoscan image (see Figure 4.1-1).

Benthic habitat	Owen Anchorage		Cockburn Sound		Shoalwater Islands Marine Park		Owen Anchorage & Cockburn Sound		Total area mapped (Fig. 4.1-1)	
	(ha)	(%)	(ha)	(%)	(ha)	(%)	(ha)	(%)	(ha)	(%)
Silt	-	-	6940	60	1547	24	6940	31	8487	8
Fine sand and silt	1021	9	2	< 1	-	-	1024	5	1024	1
Sand (including sparse seagrass)	6537	61	3725	32	1927	30	10 262	46	38 262	35
Seagrass meadow	2099	19	750	7	849	13	2849	13	4611	4
Coarse sand	47	< 1	-	-	737	11	47	< 1	39 319	36
Subtidal reef	1110	10	68	< 1	1337	21	1177	5	15 927	15
Intertidal reef	-	-	-	-	18	< 1	-	-	347	< 1

The area classified as fine sand and silt represents only 1% of the total area mapped, but nearly 10% of the area of Owen Anchorage, where it mainly occurs (Table 4.1-1).

Sand (including sparse seagrass)

Bare sand with some areas of sparse seagrass occurs in the relatively shallow waters (< 10 m) inshore of the Garden Island Ridge (Figure 4.1-1). The sediments of this habitat are generally coarser, with a lower organic content compared to the deep sedimentary basins, reflecting a more energetic environment due to greater exposure to swell waves. Sparse seagrass generally occurs in areas that are sub-optimal for seagrass meadow development, such as areas that are subject to frequent remobilisation of sediments by wave action or areas off the banks, in deeper water, where wave action is reduced but light can be limiting. Macroalgal assemblages, growing attached to hard substrate, also occur in some parts of this habitat, especially along the eastern margin of Owen Anchorage where there are scattered outcrops of sub-tidal reef. Due to the dynamic nature of the sediment in this habitat, the fauna is generally impoverished and restricted to fish and burrowing invertebrates.

Bare sand covers approximately 35% of the total area mapped. Benthic microalgae are the dominant primary producers in these areas. These microphytobenthic communities can be quite productive and are comprised primarily of diatoms living on or between surficial sand grains (Masini, 1990). In Owen Anchorage bare sand is the dominant benthic habitat representing more than 60% of the sea-floor, whereas in Cockburn Sound and the Shoalwater Islands Marine Park slightly more than 30% of the benthic habitat is bare sand (Table 4.1-1).

Coarse sand

Coarse sand occurs in offshore areas in the relatively deep (> 15 m) and exposed waters to the west of the Garden Island Ridge (Figure 4.1-1). The sediments in this habitat consist of coarse carbonate sands and fragments of rock, shell and coral rubble, with a relatively low organic content. Although microphytobenthic communities may be present, macroflora do not generally occur in this habitat due to light limitation and sediment movement (Hillman *et al.* 1994). The benthic fauna is primarily restricted to fish and burrowing invertebrates, especially molluscs, with seasonally high abundances of echinoderms and crustaceans (Hillman *et al.* 1994).

Coarse sand is the dominant offshore benthic habitat and covers nearly 40% of the total area mapped (Table 4.1-1).

Seagrass meadow

Seagrass meadows are found on sandy substrate in the protected waters inshore of the Garden Island Ridge (Figure 4.1-1). Seagrasses occur to a depth of about 12 m in Warnbro Sound and Owen Anchorage. However, meadows of greatest extent and density occur in waters of less than 10 m, particularly on Parmelia and Success Banks. Ten species of seagrass have been recorded in the study area and the dominant species are *Posidonia sinuosa*, *P. australis*, *Amphibolis antarctica* and *A. griffithii*. Seagrass meadows usually occur as a mixture of *P. sinuosa* and *Amphibolis* spp. Monospecific stands of *P. sinuosa* occur on the leeward side of Penguin Island.

Seagrass meadows provide a source of food and shelter to many different animals including a number of important commercial and recreational species (Bell and Pollard, 1989). Studies in Cockburn Sound have also shown that seagrass meadows are important nursery grounds for a variety of juvenile fish and crustaceans (Vanderklift, 1994). The leaves and stems of seagrasses provide a substrate for a diversity of epiphytic plants (Burt *et al.* 1995a) and animals (D I Walker, personal communication) which form an important part of this community.

Seagrass meadows cover about 5% of the total area mapped (Table 4.1-1). In Owen Anchorage seagrass meadows represent about 20% of the benthic habitat compared to 15% in the Shoalwater Islands Marine Park and 5% in Cockburn Sound. Further details of the present health of seagrass in the study area and changes in seagrass distribution since the 1970s are provided in sections 4.2 and 4.3, respectively.

Subtidal reef

Subtidal reefs generally occur offshore as part of the reef chains that make up the Garden Island Ridge and Five Fathom Bank Ridge, respectively (Figure 4.1-1). There is a high diversity and abundance of plants and animals associated with these reefs (Le Provost *et al.* 1981; Gordon, 1986; Ottaway and Simpson, 1986). Large attached macroalgae such as the kelp, *Ecklonia radiata* and *Sargassum spp.*, dominate the flora of reefs in conjunction with an understorey of red coralline algae and non-calcareous foliose and filamentous algae. The diversity of the plant community combined with the reef structure provides habitat for a variety of immobile invertebrates such as ascidians, sponges and soft corals, and mobile invertebrates including crustaceans, echinoderms, molluscs and burrowing infauna.

Subtidal reefs represent 15% of the benthic habitat in the total area mapped. This habitat occupies more than 20% of the area in the Shoalwater Islands Marine Park compared with 10% for Owen Anchorage and less than 1% for Cockburn Sound (Table 4.1-1).

Intertidal reef

Intertidal reef platforms occur onshore at Cape Peron, however they are principally found offshore as fringing island reefs along the Garden Island Ridge (Figure 4.1-1). Species dominance, diversity and abundance on reef platforms are principally determined by the frequency of tidal submergence, giving rise to a zonation of community types (Benjamin, 1985). Seasonal factors such as summer desiccation and irregular sand-scouring result in significant seasonal changes in the biota, particularly the plant assemblage. The biota of the upper intertidal zone is relatively impoverished however some species of blue-green algae (e.g. *Calothrix confervicola*), rock crabs, littorinid snails and whelks are relatively common. Species of green (e.g. *Ulva lactuca*) and red (e.g. *Gelidium pusillum*) algae are abundant during winter. The lower levels of the reef platforms are less exposed and hence the algal assemblages are more stable and generally dominated by small red and green turf algae, sometimes with an overstorey of larger brown algae such as *Ecklonia radiata* and *Sargassum* spp., particularly around the raised rim on the seaward edge of these platforms. The dominant animals include the commercially important abalone, *Haliotis roei*, the common whelk, *Thais orbita*, marine snails such as the chiton, *Rhyssoplax torriana* and the large turban shell, *Turbo torquata*.

Intertidal reefs constitute less than 1% of the benthic habitat in the total area mapped, however they have a relatively high diversity of biota (Table 4.1-1).

4.1.3 Marine biological resources

Commercial fisheries

Six limited entry commercial fisheries, managed by the Fisheries Department of Western Australia, operate off the metropolitan coast of Perth (Figure 4.1-2). The largest and most important fishery is the Western Rock Lobster (*Panulirus cygnus*) and fishing grounds extend from the nearshore reefs to the shelf break. The coastal season presently extends from 15 November to 30 June and generally about 70 boats, or 10% of the total Western Australian rock lobster fishing fleet, is based at Fremantle (E Barker, personal communication). The average annual catch for the State's entire rock lobster industry is approximately 10 million tonnes and in the 1993/94 season total income to fishermen exceeded $300 million.

The abalone, *Haliotis roei*, fishery extends from Cape Bouvard to Moore River, about 80 km to the south and north of Perth, respectively, and currently involves 12 licences. *Haliotis roei* is commonly found on intertidal reef platforms and subtidal reef slopes. The main fishing grounds are on the nearshore reefs off the northern metropolitan area although abalone are found on intertidal reefs along the entire metropolitan coastline.

The Southwest Trawl Fishery targets scallops (*Amusium* spp.), western king prawn (*Penaeus latisulcatus*) and whiting (*Sillago* spp.). The prawn trawling grounds are confined principally to Comet Bay whereas the main scallop grounds are approximately 25 km due west of Comet Bay and about 10 km off Perth's northern coastline (Figure 4.1-2).

A wet-line and shark fishery involving only a few vessels operates in offshore waters along the metropolitan coast. There are presently nine vessels with access to the West Coast Purse Seine (WCPS) fishery and seven vessels with access to the Metropolitan Beach Bait (MBB) Fishery, operating in metropolitan waters. The target species of the WCPS fishery include pilchard (*Sardinops neopilchardus*), scaly mackerel (*Amblygaster postera*) and occasionally anchovy (*Engraulis australis*). The MBB fishery targets whitebait (*Hyperlophus vittatus*) and blue sprat (*Spratelloides robustus*).

The common mussel (*Mytilus edulis*) is farmed in six adjoining lease areas at the northeastern end of Garden Island and one lease in central Warnbro Sound (Figure 4.1-2). These farms supply the domestic market of Perth and export markets are currently being developed. Mussel spat (juveniles) are collected from near the Cooperative Bulk Handling jetty on the eastern side of Cockburn Sound. The spat are transferred to the lease areas, attached to drop lines and suspended in the upper part of the water column until they reach marketable size. The Cockburn Sound mussel fishery is managed by the Fisheries Department as a limited entry fishery.

Other Limited Entry Fisheries managed by the Fisheries Department include squid (*Sepioteuthis australis*) and octopus (*Octopus tetricus*) in Cockburn Sound and a crab fishery (*Portunus pelagicus*) in Cockburn and Warnbro sounds (Figure 4.1-2).

Marine mammals

Colonies of the Australian Sea Lion (*Neophoca cinerea*) occur on several islands along the metropolitan coast including Carnac Island and Seal Island within the study area (Figure 4.1-2). The New Zealand Fur Seal (*Actocephalus torsteri*), the Sub-antarctic Fur Seal (*A. tropicalis*) and the Leopard Seal (*Hydrurga leptonyx*) are occasional visitors to Perth's coastal waters (D Cockran, personal communication).

The Southern Right Whale (*Balaena glacialis*) and the Humpback Whale (*Megaptera novaeangliae*) are common in Perth's offshore waters during the migration period from July to November. Minke Whales (*Balaenoptera acutorostrata*) are also seen in these waters during this migration period, though in fewer numbers. The Pigmy Right Whale (*Caperea marginata*) is an occasional visitor to these offshore waters. These whales are rarely found in the nearshore waters of Perth.

The bottlenose dolphins (*Tursiops truncatus*) occur in inshore waters, sometimes forming 'resident' pods in particular areas. One such group occurs in Cockburn Sound and has become a popular tourist attraction. The striped dolphin (*Stenella caeruleoalba*) is an oceanic species generally occurring well offshore although this species has occasionally been found stranded on metropolitan beaches.

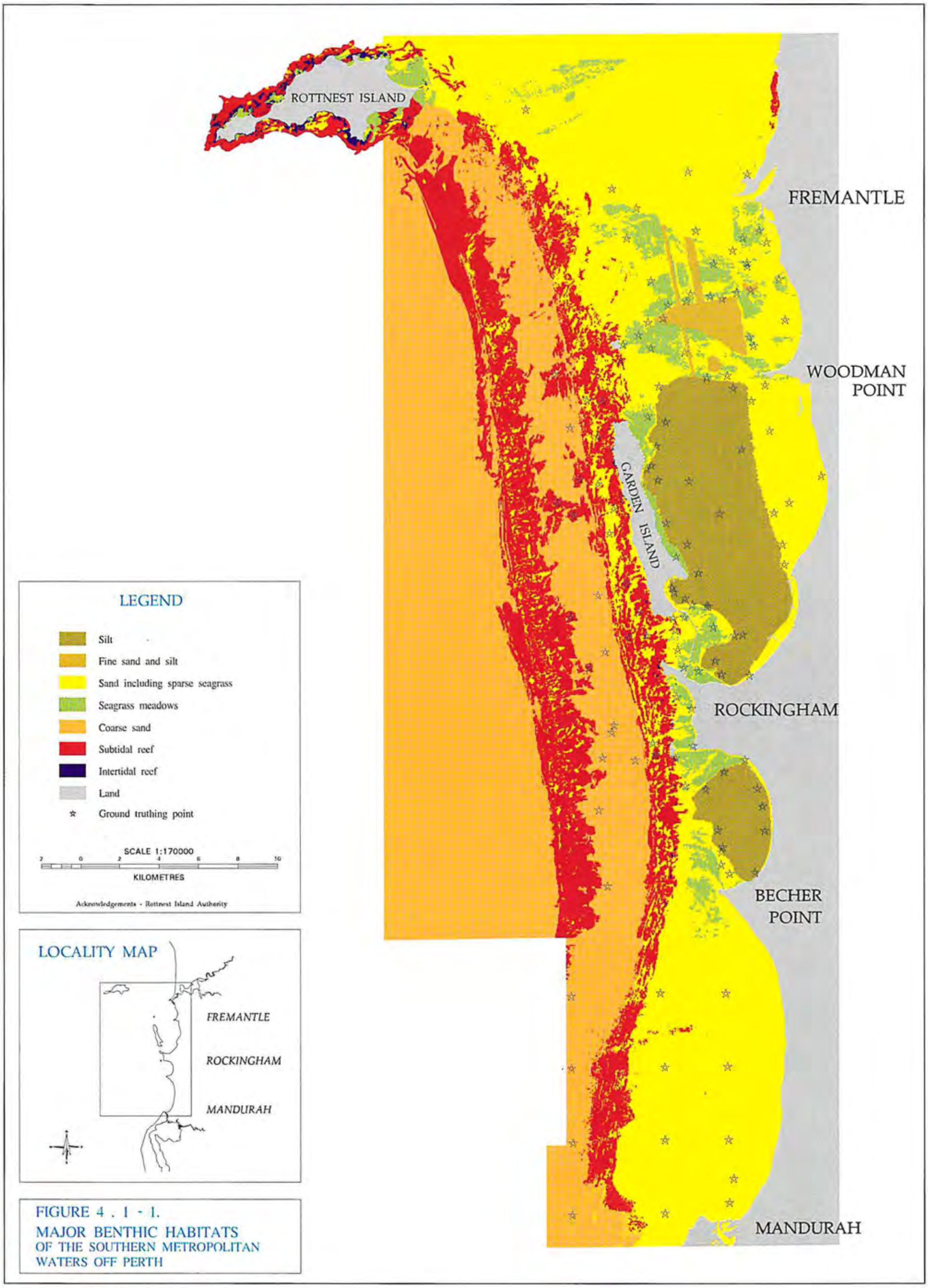


Figure 4.1-1. Major benthic habitats of the southern metropolitan waters off Perth.

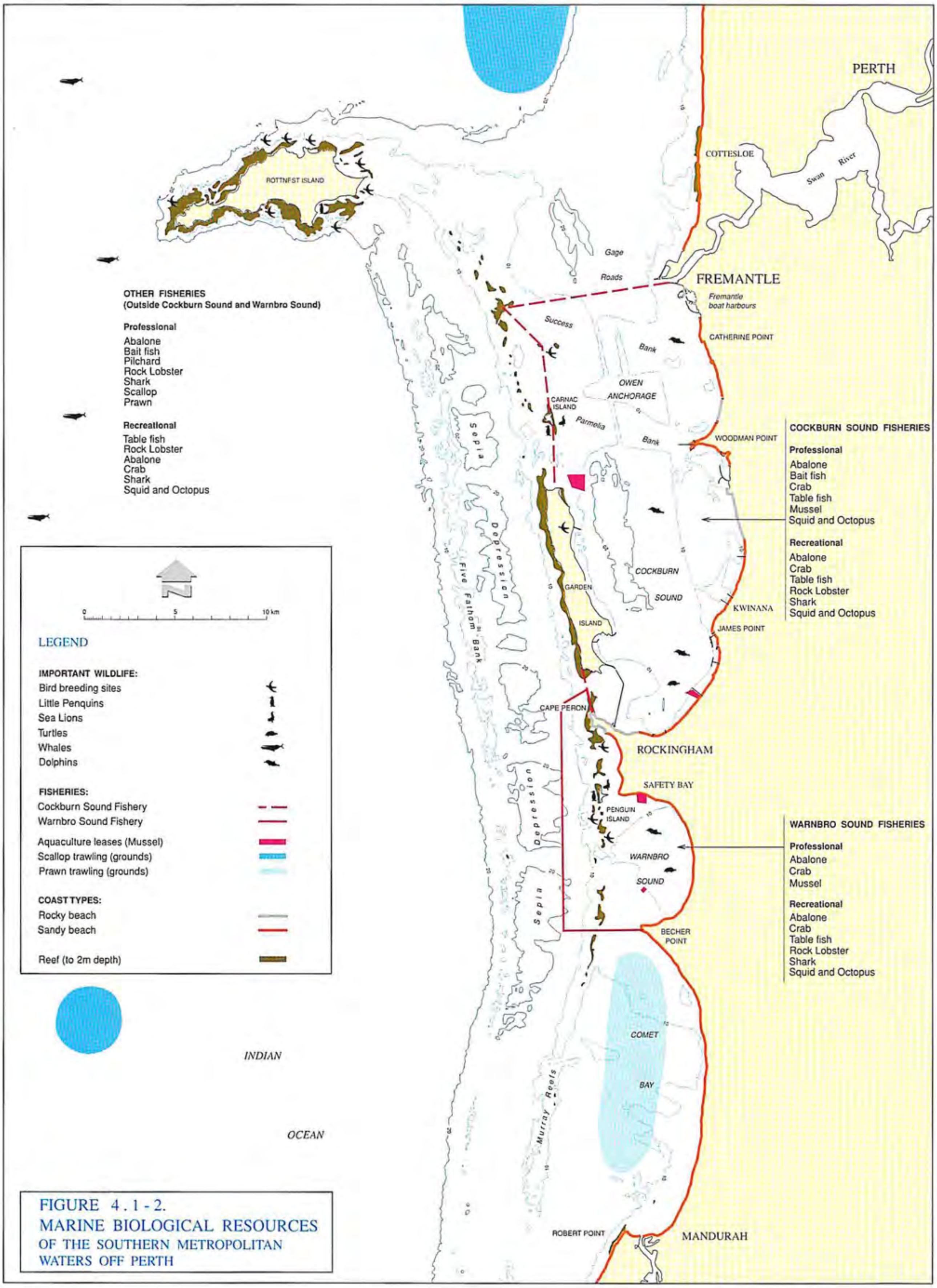


Figure 4.1-2. Marine biological resources of the southern metropolitan waters off Perth.

Marine reptiles

Turtles such as the Loggerhead (*Caretta caretta*), Leatherback (*Dermochelys coriacea*) and Green (*Chelonia mydas*) are occasionally sighted in the coastal waters off Perth (B Prince, personal communication).

Sea birds

The coastal islands and rock stacks in the study area, particularly in Shoalwater Bay, are important resting, breeding and nursery areas for a range of resident and migratory bird species (Figure 4.1-2). More than 45 species of bird use these islands for nesting, feeding and roosting (Department of Conservation and Land Management, 1990). At least 12 species are sea birds including waders and divers. The majority of the waders are migratory and do not breed locally, visiting the area between September to April each year, and some of these species are protected under international agreements (Department of Conservation and Land Management, 1990). By contrast, most of the divers are resident and use the offshore islands, particularly in the Shoalwater Bay, as nesting sites. Several migratory divers are also protected under international agreements.

Little Penguins (*Eudyptyla minor*) are found in the southern coastal waters of Perth. The islands off the metropolitan coast are the northern limit of the Little Penguin range. Penguin Island, in the Shoalwater Islands Marine Park, supports the largest breeding population of Little Penguins on the west coast of Australia (Department of Conservation and Land Management, 1990; Figure 4.1-2). There are roosting sites at Parker Point on Rottnest Island and on Seal and Bird Islands.

4.1.4 Recreational usage

In general, the recreational pressure on Perth's southern coastal waters reflects the distribution of the coastal population, coastal access for people and boats, and the presence of recreational attractions such as fishing and diving localities. Areas with the highest recreational pressure are Rottnest Island, Owen Anchorage, the Shoalwater Islands Marine Park and parts of Cockburn Sound. Recreational usage of the offshore waters and the more isolated areas, such as Comet Bay, is relatively low (Figure 4.1-5).

Coastal population

Since the late 1970s, the population of the coastal municipalities between Fremantle and Rockingham has increased by more than 60% (Figure 4.1-3) leading to an increased demand over that period for recreational facilities such as boat harbours, boat ramps, and recreation and conservation reserves along the coastal strip. Over the next 30 years the population of Perth's southern coastal municipalities is predicted to more than double (Figure 4.1-3).

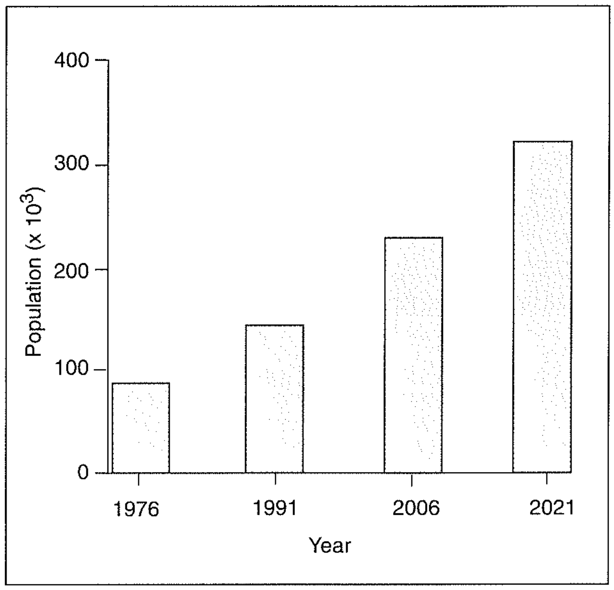


Figure 4.1-3. Population growth of coastal municipalities between Fremantle and Rockingham.

Boat usage

The number of registered recreational boats in Western Australia has increased from about 12 500 in 1966 to more than 53 000 in 1994 (Figure 4.1-4). With approximately 70% of the Western Australian population living in Perth, this suggests that there were about 37 000 registered recreational boats in the metropolitan area in 1994. The number of boat harbours and marinas has also increased over the same period (e.g. Challenger Harbour) and there have been several marina developments proposed for Owen Anchorage, Mangles Bay, Warnbro Sound and Comet Bay.

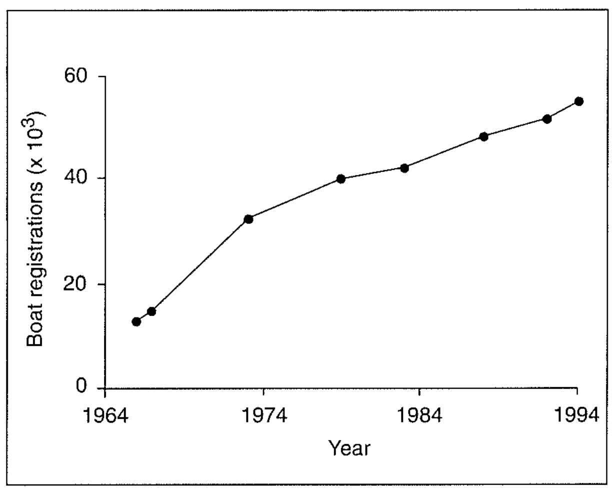


Figure 4.1-4. Boat registrations in Western Australia (data from Department of Transport).

Conservation and recreation reserves

Several coastal conservation and recreation reserves have been created since the late 1970s reflecting a substantial increase in demand for recreational amenities and facilities along the coastline over this period (Figure 4.1-5). The majority of Garden Island (other than areas set aside for defence purposes) is managed by the Commonwealth Government as a reserve for nature conservation and passive recreation. A concept and development plan for the Woodman Point regional recreation/conservation park was approved by the State Government in 1988 (Woodman Point Community Task Force, 1988). In 1993 an area of high conservation value was excised from the proposed Woodman Point regional park and vested with the National Parks and Nature Conservation Authority as an A Class Nature Reserve (Government Gazette, 1993). The remaining area is currently being managed by the Department of Youth, Sport and Recreation. The process for establishing the Beeliar Regional Park was initiated in 1986 and in 1992 the boundaries for the park were identified (Department of Planning and Urban Development, 1992). An area at Cape Peron was recommended in 1988 for a combination of conservation and recreation activities, including a marina (State Planning Commission, 1988). The Port Kennedy area has been proposed as a regional park for recreation and conservation (Department of Conservation and Environment, 1983; EPA, 1989).

There are two marine reserves in Perth's southern coastal waters. The waters and islands of Shoalwater Bay and Warnbro Sound were gazetted as the Shoalwater Islands Marine Park in 1990 (Department of Conservation and Land Management, 1990). The Rottnest Island Aquatic Reserve was declared in 1987 under the Rottnest Island Authority Act (1987) and includes the waters within approximately 1 km of the Island.

Beach usage

The last comprehensive survey of the recreational usage in Perth's southern coastal waters was undertaken by Feilman and Associates (1978) as part of the Cockburn Sound Environmental Study. This survey focussed mainly on the recreational use of Cockburn Sound. At the time Cockburn Sound was considered to be comparable to the Swan River in importance as a recreational resource. The peak day use of the beaches of Cockburn Sound and Owen Anchorage was estimated to be 8300 people, with a further 1200 in the Shoalwater Bay/Warnbro Sound areas. Additionally, on a typical hot summer weekend some 620 boats used these waters.

A recent study of recreational beach usage by Dielesen (1994) provides the opportunity to examine changes in beach usage since 1978. Swimming was the major recreational activity in both 1978 and 1994 at South Beach, Coogee, Kwinana, Bell Park, Rockingham and Palm Beach. Swimming was also the most popular activity in 1994 at beaches in Shoalwater Bay, Safety Bay and Warnbro Sound, however, no surveys were undertaken in 1978 for comparison. The most popular localities in 1994, for beach and jetty fishing, were Safety Bay, Challenger Beach, Kwinana Beach, Woodman Point, Coogee Beach and the North Mole. Since 1978 there have been significant increases in the number of people fishing at Challenger, Kwinana and Coogee beaches. The most used boat launching ramps were at Safety Bay, Palm Beach, Challenger Beach and Woodman Point in both 1978 and 1994.

4.1.5 Commercial usage

Port and industrial operations

The Fremantle Harbour is Western Australia's principal port; situated at the mouth of the Swan River, it services coastal, national and international vessels. Further south, the protected deepwater basin of Cockburn Sound provides a natural harbour to service the Kwinana industrial area located along the eastern shoreline (Figure 4.1-6). Industrial development in Kwinana began in 1952 with the establishment of an oil refinery; since then, development has continued and this area has become one of the most important industrial complexes in the State, with an annual production of more than $2 billion (Dames and Moore, 1993).

Shipping and related activities

The study area has the highest frequency of commercial shipping movements in the State. In 1993/94, 1700 ships used Fremantle Harbour and Cockburn Sound, with a total tonnage of 29 million tonnes (FPA, 1994). Since 1989/90 the number of ships visiting the port has increased by 8%. There are five marinas in Owen Anchorage and Cockburn Sound, providing facilities to smaller vessels such as trawlers, fishing boats and pleasure craft (Figure 4.1-6). The Jervoise Bay marina, at the northeast corner of Cockburn Sound, is the site of a large ship building and maintenance industry. HMAS Stirling in Careening Bay, at the southern end of Garden Island, is a significant naval facility, accommodating vessels from the Australian Navy Western Fleet.

Shellsand mining

Cockburn Cement Limited operates the largest lime manufacturing works in Australia, at South Coogee. Since 1972, Cockburn Cement has dredged shellsands from Parmelia Bank and Success Bank, to the north of Cockburn Sound (Figure 4.1-6), in accordance with a State Agreement Act (LeProvost Dames and Moore, 1994). The Act gives Cockburn Cement the legal right to extract shellsands within a radius of 8 km from Coogee Beach until the year 2011, with rights of extension to 2021. In 1994, 1.6 million tonnes of shellsand were dredged from these waters and it is forecast that, by the year 2001, approximately 2 million tonnes per year will be dredged (LeProvost Dames and Moore, 1994).

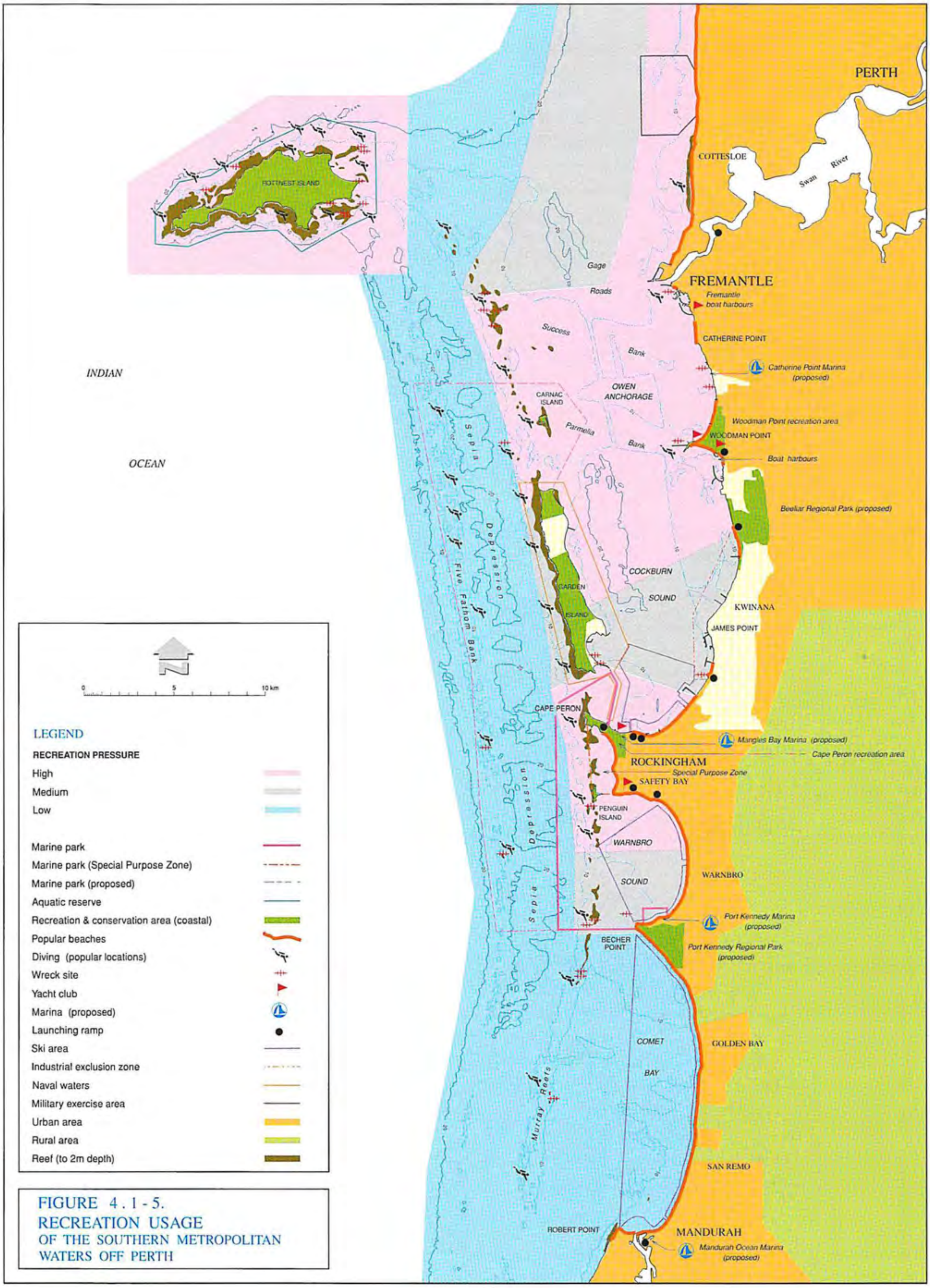


Figure 4.1-5. Recreational usage of the southern metropolitan waters off Perth.

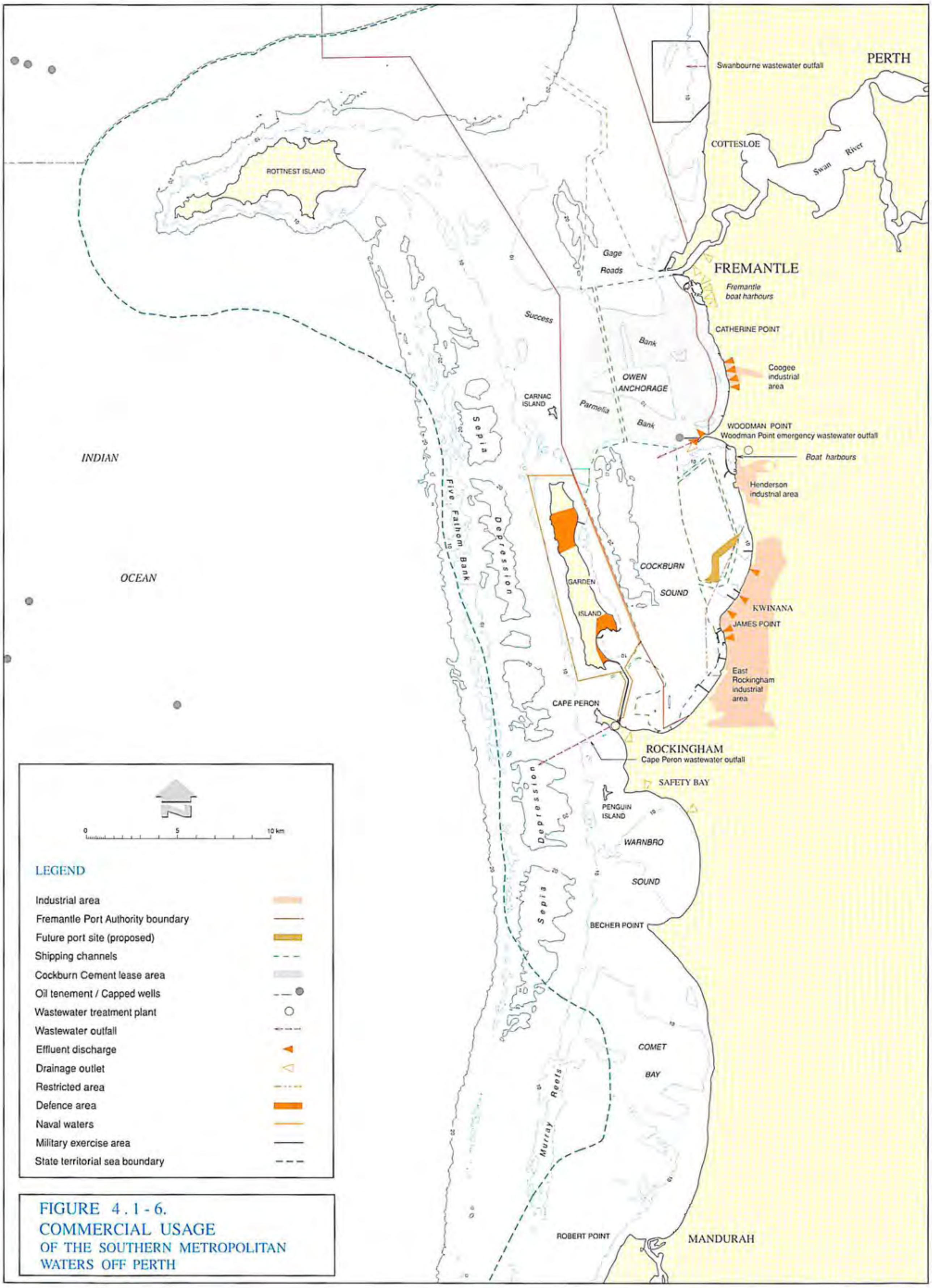


Figure 4.1-6. Commercial usage of the southern metropolitan waters off Perth.

The sands are mined by suction dredge, transferred to barges and shipped to a primary processing facility at Woodman Point. Shellsand is used to manufacture cement and quicklime, and this operation currently supplies approximately 70% of Western Australia's domestic requirements.

Oil exploration

Periodic seismic and exploratory drilling has been carried out during recent years in the Perth basin, primarily in offshore waters north and south of Rottnest Island, however there have also been wells drilled in nearshore waters, including two wells in Gage Roads and a well at the end of Woodman Point. To date no production wells have operated in these waters.

4.2 Current health of seagrass meadows

Seagrasses are important primary producers in the southern metropolitan coastal waters of Perth and occur as extensive meadows, particularly in relatively shallow, depositional environments moderately protected from ocean swells (Figure 4.1-1). Water circulation and flushing tend to be restricted in these areas and, consequently, seagrass communities are often more vulnerable to the effects of waste inputs, particularly nutrients, than many of the other biological communities that thrive in more open waters. One of the most common symptoms of nutrient enrichment in seagrass meadows is the presence of high abundances of epiphytic algae on the seagrass leaves and/or shifts in species composition of the epiphytes from encrusting coralline red algae to foliose and filamentous forms (Cambridge, 1986; Burt *et al.* 1995a). These attributes can form the basis of an 'epiphyte' index of seagrass health. When eutrophication is more advanced, reductions in the amount of seagrass leaf material present in the meadow and/or reductions in shoot density become apparent (Cambridge, 1979) resulting in a reduction in the canopy 'cover' index. Seagrass cover can also be affected by natural processes such as storm waves removing leaves or by sediment burial and, as such, changes in the 'cover' index may reflect either natural or anthropogenic influences.

There are anecdotal reports and observations by the Study team of large filamentous algae on seagrass leaves on the western side of Success Bank during late summer/early autumn. Occurrences of these algae are generally associated with nutrient enrichment.

The objectives of this study were to provide a broad overview of the health of seagrass communities in the study area and establish a semi-quantitative baseline to assess future changes. To assess the 'health' of seagrass communities in the study area, ten sites, considered to be broadly representative of the seagrass meadows in the 'core areas' (Figure 4.2-1), were surveyed using the seagrass community 'health' indices described above. Scores were made of these 'health' indices at each site and used to derive a relative 'cumulative stress level'. Further technical details can be found in Lavery (1994b). A companion study of changes in seagrass cover is described in section 4.3.

Findings

Least stressed sites

Becher Point (west), Southern Flats (east) and Woodman Point contained seagrass with the least evidence of stress. Of these three locations, Becher Point had clearly the least stressed seagrass, while Woodman Point and Southern Flats seagrass showed slight degrees of epiphyte-related stress, possibly linked to eutrophication. None of these sites showed any evidence of recent seagrass death in the form of old rhizome mat, though the possibility that dead mat has been eroded and dispersed from these sites cannot be ruled out.

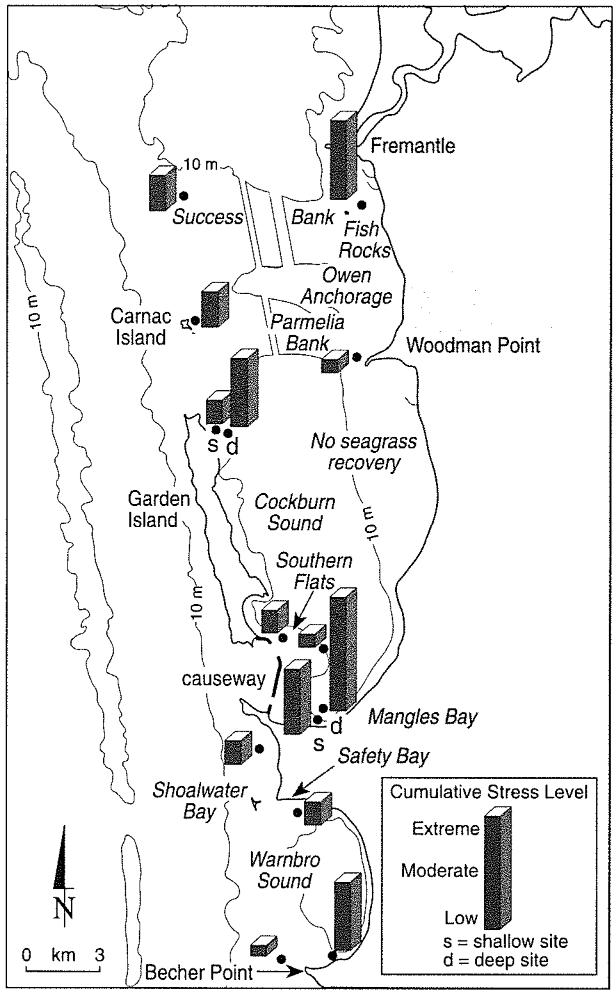


Figure 4.2-1. Cumulative 'stress' of seagrass meadows within the study area.

Slightly stressed sites

Success Bank, Carnac Island, Southern Flats (west), Garden Island (shallow), Shoalwater Bay and Safety Bay all showed some evidence of stress in the seagrass communities. The stress in seagrasses in the western Success Bank and Carnac Island area most likely reflects the energetic nature of this area as well as some possible eutrophication-related impacts and anchor-related damage. The stress at Southern Flats (west) appears to be the direct result of recent physical disturbance by anchors or some other mechanism, and is accompanied by the presence of rhizome mat in the sand areas and severely 'cropped' seagrass. At Garden Island, Shoalwater Bay and Safety Bay the stress appears to be linked to eutrophication.

Moderately stressed sites

Garden Island (deep), Mangles Bay (shallow) and Becher Point (east) all showed signs of moderate stress. Of these only the Mangles Bay site appears to have been clearly affected by anthropogenic impacts. At this site, the reduced cover and high epiphytic loads are clearly linked to eutrophication and the evidence of old rhizome mat is evidence of recent, widespread losses of seagrass. The other two sites are in locations with significant natural stresses, in the form of light limitation at the Garden Island site, and high sediment movement around Becher Point.

Extremely stressed sites

Mangles Bay (deep) and Fish Rocks showed evidence of extreme stress. In Mangles Bay there were clear indications of eutrophication-linked stress. At this site large accumulations of filamentous green and red algae, typical of nutrient-enriched conditions, were present on the seabed and on seagrass leaves. At Fish Rocks, there was moderate epiphytic growth, including filamentous species which suggest eutrophication effects. However, this site also appeared to be relatively mobile, being shallow and relatively exposed to wave action. The low percentage cover at this site probably also reflects a degree of natural erosion and the presence of colonising species is also indicative of the dynamic nature of the substratum in this area.

Conclusions

The results of this survey indicate that seagrass communities close to major sources of nutrient inputs are under eutrophication-related stress, particularly in southern Cockburn Sound and northeastern Owen Anchorage. The pattern of stress largely reflects the changes in seagrass cover outlined in the following section. These data, once again, emphasise the deleterious effects of excessive nutrient inputs on seagrass meadows. In some cases, for example Becher Point, the stress appears to be related to natural causes such as sedimentation effects and highlights the importance of also considering natural processes when assessing the 'health' of seagrass meadows.

4.3 Changes in seagrass cover

The temperate coastal waters of Western Australia contain one of the world's richest seagrass floras and seagrasses are important primary producers in the nearshore waters. Approximately 930 ha of seagrass occur in Warnbro Sound (Kirkman and Walker, 1989) and there is no evidence to suggest that the areal coverage has changed substantially for at least the past 30 years. In Cockburn Sound, seagrass meadows, predominantly the genus *Posidonia*, originally occupied approximately 4 000 ha and covered much of the seabed where water depths were less than about 8-10 m. The distribution and cover of seagrass within the Sound remained relatively unchanged between 1942 and 1957, but from 1957 through to 1968 there was a gradual retreat of meadows from the deeper margin of the eastern bank and thinning along portions of the adjacent shoreline. By 1972 most of the seagrass meadows on the eastern margin had disappeared and by the late 1970s only about 900 ha remained in the Sound, mostly along the Garden Island shoreline, Mangles Bay and across the northern barrier bank, Parmelia Bank (Department of Conservation and Environment, 1979). The seagrass losses were attributed to light starvation caused by a nitrogen-induced proliferation of algal epiphytes on seagrass leaves and phytoplankton in the overlying water.

Further losses occurred on Parmelia Bank in the early 1980s and have been attributed to the combined effects of eutrophication and limesand mining activities (LeProvost, Semeniuk and Chalmer, 1986). Accretion of grey Becher Sand on the northern shoreline of Woodman Point, evident since 1975 (LeProvost, Semeniuk and Chalmer, 1988a), indicates that the degradation of the seagrass meadow on east Parmelia Bank has resulted in a mobilisation of bank sediments causing an increased influx of organic-rich sand (LeProvost, Semeniuk and Chalmer, 1988b).

The areal extent of seagrass meadows on Success Bank, which lies to the north of Parmelia Bank, have also decreased over the last 20 years, both through direct removal during limesand mining and by other unknown causes. Mining has removed approximately 90 ha of seagrass meadow since 1972, mostly on eastern Success Bank. Clements (1994) investigated the rate of decline of dense seagrass meadow in a 450 ha portion of west Success Bank and found that 85 ha of the original 350 ha (24%) of seagrass meadow present in June 1986 had disappeared by the beginning of January 1993; this is equivalent to a rate of decline of over 13 ha y^{-1}. This seagrass loss is associated with an area of mobile sediment west of the FPA shipping channel through Success Bank.

Circular areas of bare sediment can be seen on aerial photographs of boat mooring areas located within seagrass meadows and this loss is attributed to scouring caused by mooring chains. A study conducted by Lukatelich *et al.* (1987) found that this process had resulted in the loss of 0.45 ha and 1.84 ha of seagrass meadow in Warnbro Sound and Cockburn Sound, respectively.

A small area of seagrass meadow (approximately 1 ha) in Luscombe Bay, on the eastern side of Garden Island, was denuded by sea urchin aggregations between 1981 and 1991 (Bancroft, 1992). Although the cause of this 'outbreak' is unknown, there is evidence of physical disturbance at the site before the 'outbreak' (e.g. submarine cable, mooring scars) suggesting an anthropogenic influence.

To assess the change in areal extent of seagrass in the study area since the late 1970s a study was undertaken in collaboration with Edith Cowan University, the Department of Land Administration and the CSIRO Division of Water Resources. Locations considered to be broadly representative of the main seagrass habitats in the 'core' study area were selected and suitable aerial photographs from different years were digitised, spatially rectified and compared to quantify changes in the areal extent of the seagrass meadows at sites within each location (Figure 4.3-1). Further technical details can be found in Lavery (1994a). A companion study on the current 'health' of these seagrasses is described in section 4.2.

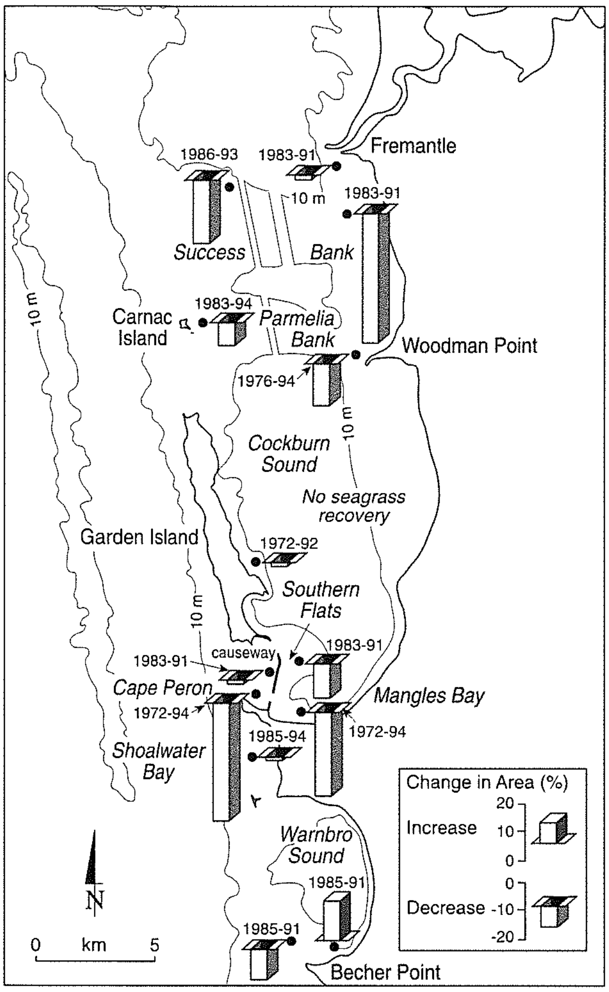


Figure 4.3-1. Changes in seagrass cover at selected sites within the study area (data periods are shown).

Findings

Local-scale changes

Changes in the areal extent of the seagrass meadows of the 'core' study areas are shown in Figure 4.3-1. Although the seagrass meadows have decreased at all locations apart from Becher Point (east), the decreases at Fremantle, Carnac Island, Garden Island causeway (east) and Shoalwater Bay are considered insignificant and within the error of the methodology. Moderate decreases have occurred on Southern Flats (west) and Becher Point (west). The greatest decreases occurred in Mangles Bay, near Fish Rocks, Cape Peron, Success Bank (west) and Woodman Point.

Possible reasons for these changes can be discerned when considered with the current 'health' of these meadows, as outlined in section 4.2. The seagrass meadows at Mangles Bay and near Fish Rocks were considered to be heavily stressed in 1994 suggesting past losses were caused primarily by eutrophication at Mangles Bay and a combination of eutrophication and natural processes at Fish Rocks. The large decrease in seagrass cover at the Cape Peron site between 1972 and 1994 is largely attributed to sediment inundation due to construction of a boat launching facility and current scouring associated with the trestle bridge of the Garden Island causeway. The seagrass 'health' data for Success Bank (west) and the infilling of the shipping channel in this area (Clements, 1994) suggest the seagrass losses at this location are possibly a result of combined anthropogenic and natural processes. Similarly sediment inundation at Becher Point (west) appears to be the most likely cause of the decrease in seagrass meadow area at this site.

Losses at four sites around Woodman Point, between 1976-1994, averaged about 15% with the greatest change occurring on the northern side, between the Cockburn Cement jetty and the tip of the point. On Southern Flats (west) there has been about a 12% reduction in seagrass cover between 1983 and 1991 and this is attributed to a combination of eutrophication effects and mechanical damage, presumably from boat anchors. It should be noted that the introduced polychaete worm *Sabella* cf. *spallanzanii* has established in this area with approximately 20 ha. of the Southern Flats known to be currently occupied by beds of this worm (see section 4.13) and, on aerial photographs, this infestation is indistinguishable from seagrass meadows. The apparent increase in seagrass cover at the Becher Point (east) location is thought to have been due to accumulations of macroalgal and seagrass wrack.

Broadscale changes in Cockburn Sound

In 1973 approximately 700 ha of the original 3 900 ha of seagrass meadow were estimated to be remaining in Cockburn Sound (excluding Parmelia Bank) based on manual digitising of aerial photographic images (Cambridge, 1979). This figure is in reasonable agreement with the 750 ha identified in the same area from high resolution digital images obtained in 1993 (section 4.1). Although direct comparisons should not be drawn between these estimates, due to the vast difference in resolution of the two techniques, these data and the data for individual locations in Cockburn Sound, suggest that the decline in seagrass meadows in Cockburn Sound has slowed considerably since the rapid, widespread losses that occurred in the late 1960s to early 1970s. Perhaps more importantly there has been no significant recovery of the seagrass meadows in Cockburn Sound over the past two decades (Figure 4.3-2).

Broadscale changes in Owen Anchorage

In 1994 approximately 2100 ha of seagrass were present in Owen Anchorage. The area present in the early 1970s could not be quantified accurately due to the absence of suitable aerial photographs. However, there is clear evidence that significant losses have occurred on east

Figure 4.3-2. Broadscale changes in the seagrass meadow area of Cockburn Sound. (Updated from Cambridge and McComb, 1984).

Parmelia Bank, since the late 1970s, and on portions of east and west Success Bank since the mid 1980s.
In addition about 90 ha of seagrass meadows have been lost as a result of limesand mining on these banks since 1972. These changes are represented schematically in Figure 4.3-3 and, in total, exceed several hundred hectares. Cockburn Cement Ltd. are currently conducting further detailed investigations of changes in seagrass cover in Owen Anchorage.

Conclusions

The results of this study indicate that seagrass losses have occurred throughout the 'core' areas since the early 1970s. In light of the findings of a companion study on seagrass health (section 4.2) these changes appear to be attributable to anthropogenic impacts (mainly eutrophication) and sediment inundation. It is not clear whether loss due to sediment inundation is a 'natural' process or related to anthropogenic activities. Losses of seagrass meadows in Cockburn Sound since the early 1970s have slowed considerably since the widespread losses that occurred in the late 1960s to early 1970s. No significant recovery has occurred in Cockburn Sound since this time.

Losses of seagrass in Owen Anchorage are attributed to limesand mining and the effects of eutrophication and sediment inundation. The results of studies which rely heavily on past aerial photography should be interpreted cautiously as they can give misleading results due to the inability to accurately distinguish between detached seagrass/macroalgal accumulations, infestations of the large worm *Sabella* and intact seagrass meadows. Similarly, it is not possible to distinguish between areas of bare sand where seagrass meadows have been lost and transient sand-sheets overlying viable seagrass rhizome mat, on the basis of aerial photography alone. Nonetheless the use of these techniques, particularly in conjunction with *in situ* studies, allows general trends to be detected adequately.

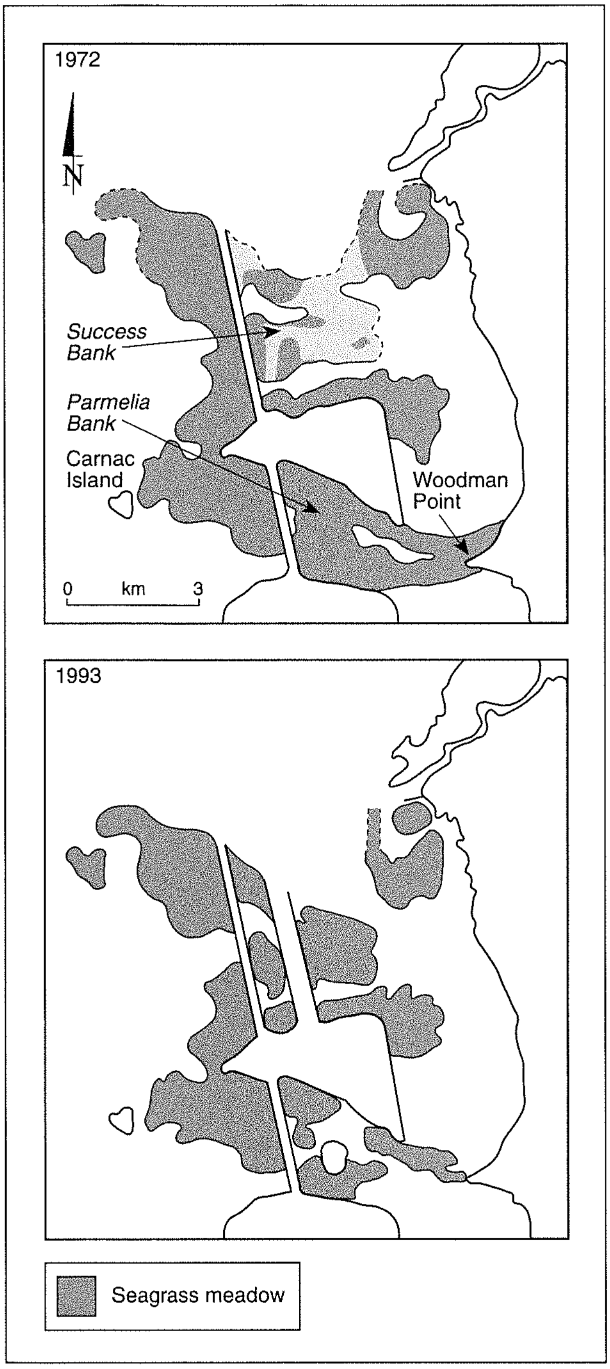


Figure 4.3-3. Schematic representation of broadscale changes in the seagrass meadow area of Owen Anchorage. Dotted lines and lighter shading indicate boundaries and areas of seagrass where interpretation of aerial photography is less certain.

4.4 Contaminant inputs

Contaminants such as nutrients, heavy metals, hydrocarbons, oil & grease, bacteria, pesticides, chemical oxygen demand and biological oxygen demand enter Perth's coastal waters from numerous sources including industrial and domestic wastewater outfalls, stormwater drains, groundwater inflow, deposition from the atmosphere and discharges from rivers and estuaries. Contaminants also enter these waters from ships, through controlled discharge (e.g. ballast waters, washdown wastes), accidental spillage and through leaching of toxic substances such as organotin compounds from the hulls of vessels.

The first comprehensive description of contaminant inputs to Cockburn Sound and Owen Anchorage was provided by Murphy (1979) who determined industrial and urban inputs into these waters from point and diffuse sources. A summary of the history of industrial development and associated waste discharges is presented in the Cockburn Sound Environmental Study report (Department of Conservation and Environment, 1979). Since then individual companies and government agencies discharging effluent into these waters have been required to report effluent characteristics and contaminant loadings. At the beginning of the SMCWS there was no comprehensive, current and accessible inventory of contaminant inputs to the southern metropolitan coastal waters. To address this deficiency a study was commissioned to compile all information on contaminant inputs, past, present and predicted, from point and diffuse industrial and urban sources to Perth's southern coastal waters (Martinick *et al.* 1993). An electronic database was developed for the contaminant loadings data. Estuarine inputs from the Swan-Canning and Peel-Harvey estuaries and inputs from shipping were also considered.

Some estimates of contaminant loadings presented in this report differ from the estimates compiled by Martinick *et al.* (1993) and quoted in the SMCWS Progress Report (Simpson *et al.* 1993). The discrepancies reflect revised estimates based on improved data. Details of revised estimates can be found in Muriale and Cary (1995). Estimates of projected loads from point sources are based on information received from the relevant organisations; projected loading estimates of diffuse sources are based on available information (Appleyard, 1994; DEP unpublished data).

Since the compilation of Muriale and Cary (1995) more recent contaminant inputs data have been received from organisations and, in addition, some companies have submitted reports outlining the basis for a revision of their earlier estimates of past contaminant loads. These data and estimates are not included in this study, however they will be considered as part of the annual process of updating the contaminant inputs inventory.

It is not intended to cover the entire range of contaminant inputs in this report. Rather, the discussion deals only with contaminants that were of historical concern or are considered to represent current/potential threats to the environmental quality of the study area, as identified by the contaminants surveys (see sections 4.5 and 4.6) or the contaminant inputs studies. Readers are referred to Martinick *et al.* (1993), Simpson *et al.* (1993), Muriale and Cary (1995), Deeley (in preparation) and Appleyard (1990, 1994) for further technical details.

4.4.1 Industrial and urban inputs

The contaminant inputs inventory provides annual estimates of contaminant loads to the coastal waters of the study area from all known industrial and urban sources for the years 1950 to 2021. Information on more than 50 contaminants is compiled under the general headings of nutrients, grease, biological oxygen demand, chemical oxygen demand, heavy metals, hydrocarbons, pesticides and bacteria. The inventory has recently been updated to include loadings to June 1994 (Muriale and Cary, 1995) and projected loads are based on information available at this time. The electronic database is currently maintained by the Kwinana Branch of the DEP.

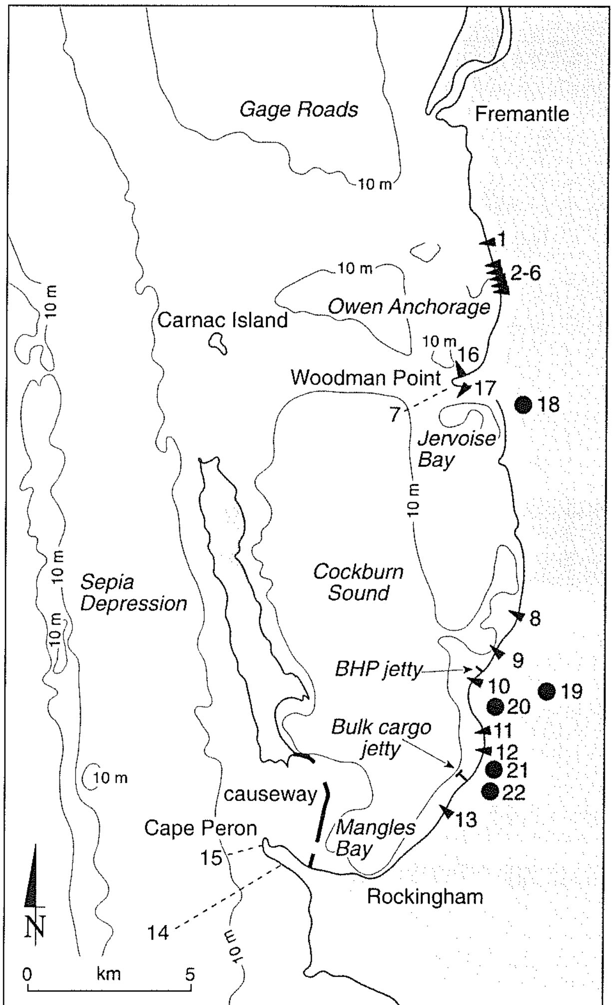
Findings

4.4.1.1 Cockburn Sound

Outfalls from the State Electricity Commission of Western Australia (SECWA), BP Refinery (Kwinana) Pty Ltd (BP), Tiwest Joint Venture (Tiwest) and CSBP & Farmers Ltd/Kwinana Nitrogen Company/Australian Gold Reagents (CSBP) are the major point source discharges into Cockburn Sound (Figure 4.4-1). The Water Corporation (formerly WAWA) also maintains two emergency wastewater outlets off Woodman Point which are considered a minor source of contaminants to these waters discharging for, on average, less than 10 hours per year over the past five years.

Nutrients

Nitrogen loading to Cockburn Sound was low until about the mid-1960s when loadings from industrial and domestic waste outfalls increased rapidly. Peak loadings occurred around 1978. Between 1978 and 1990, annual loads of nitrogen discharged to Cockburn Sound decreased substantially, from about 2000 to 900 tonnes (Figure 4.4-2). This was largely due to improvements in the treatment of industrial wastewaters, during the 1980s, and to the diversion, in 1984, of the domestic wastewater outfall in Cockburn Sound to an offshore site in Sepia Depression. Current (1994) annual nitrogen loads are estimated to be about 490 tonnes, with about 70% entering via contaminated groundwater inflow. Further planned reductions in industrial discharges to Cockburn Sound, with most industrial discharges declining



1 WA Meat Commission *(no longer in use)*
2 Kailis and France
3 Coogee Felmongers *(no longer in use)*
4 Gosh Leather Tannery *(no longer in use)*
5 George Weston Foods *(no longer in use)*
6 Fremantle Fishermans Co-operative
7 Woodman Point Domestic Wastewater Outfall *(1966-1984) now emergency overflow*
8 Western Power
9 BHP Steel *(no longer in use)*
10 BP Refinery (Kwinana) Pty. Ltd.
11 TiWest Joint Venture
12 Wesfarmers CSBP Ltd.
13 ICI Advanced Ceramics *(used by Hanwha Advanced Ceramics Australia Pty. Ltd. since 1995)*
14 Cape Peron Domestic Wastewater Outfall *(since 1984)*
15 Point Peron Domestic Wastewater Outfall *(not in use since 1984)*
16 Cockburn Cement Ltd.
17 Woodman Point Emergency Domestic Wastewater Overflow
18 Love Starches Pty. Ltd.
19 Nufarm Ltd.
20 BP Refinery (Kwinana) Pty. Ltd.
21 Wesfarmers CSBP Ltd.
22 Western Mining Corporation Ltd.
◄ Industrial outfall
------ Domestic wastewater outfall
● Major industrial sources of groundwater contamination

Figure 4.4-1. Location of existing and de-commissioned wastewater outfalls and major industrial sources of groundwater contamination in the study area.

to zero by 2021, are projected to reduce nitrogen loads to these waters to about 370 tonnes a year by the year 2021, with over 90% coming from contaminated groundwater inflows (Figure 4.4-2).

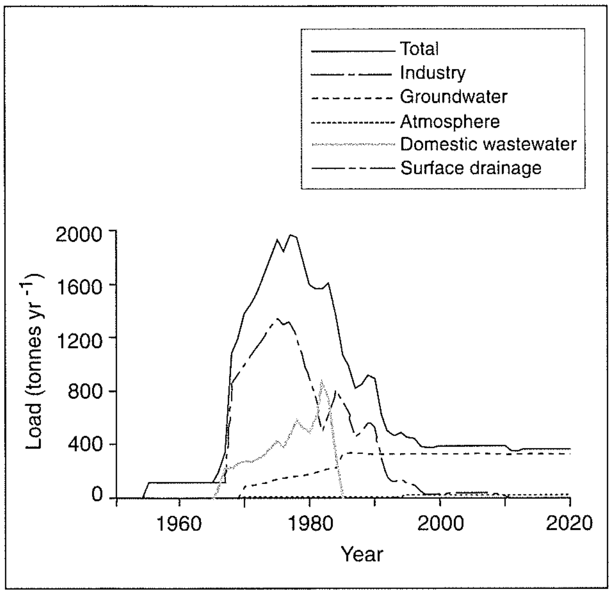


Figure 4.4-2. Annual loads of nitrogen to Cockburn Sound.

Most of the nitrogen entering Cockburn Sound in contaminated groundwater appears to be discharged from two short segments of coastline: one in southeast Cockburn Sound, next to Western Mining Corporation Ltd (WMC) and CSBP; the other in Jervoise Bay, north of the ship-building area (Appleyard, 1994; Figure 4.4-3). Current estimates indicate that the area adjacent to Western Mining Corporation (WMC) and CSBP accounts for about 80% of the annual total nitrogen contribution from groundwater and this is largely attributed to contamination from these industrial estates. A further 20% of the total enters the sound along the Jervoise Bay coastline and is probably due mostly to waste disposal by Love Starches via shallow-well injection. Appleyard (1994) also suggests that an additional source of contaminated groundwater may be the nearby Woodman Point Wastewater Treatment Plant.

A long-term relationship developed between nitrogen load into Cockburn Sound and chlorophyll *a* concentrations in the water column (see section 4.7.3) suggests that, since about 1991, there has been a 'missing' (i.e. not included in the inputs inventory) load of nitrogen into Cockburn Sound of about two tonnes per day. The progression of contaminated groundwater plumes toward the sound, coupled with the large error associated with estimating groundwater loads into Cockburn Sound (Appleyard, 1994) may account for this 'missing' load.

Phosphorus loadings over this period follow similar trends to those described above for nitrogen, for many of the same reasons. Total phosphorus loads reached a peak of about 1350 tonnes per year in 1978 and have since declined to current levels of about 55 tonnes per year (Figure 4.4-4).

Heavy metals

The historical trends in most heavy metal loadings to Cockburn Sound are similar to the trends for nutrient inputs. Annual loads from point sources for copper, lead, zinc, chromium and cadmium were generally low in the 1950s, increased sharply around the mid-1960s and decreased sharply between the late 1970s and 1990. The increases were due to industrial and domestic waste inputs and the reductions were due to improved industrial waste treatment and diversion of domestic wastes to Sepia Depression. An exception to this general trend is mercury loads (mainly from CSBP) which increased to about 450 kg in 1991, although this trend is based on limited data. Since then mercury loads decreased to about 15 kg a year in 1994 (Figure 4.4-5a). Copper loads increased from about 600 kg in 1992 to about 1400 kg in 1993 due to increased discharge from CSBP. Similarly zinc increased from about 3000 kg in 1991 to about 7000 kg in 1995 due to increased discharge from BP (Figure 4.4-5b). These zinc loads are approaching the loads entering Cockburn Sound in the 1970s and early 1980s. Currently, the major industrial sources of heavy metals to these waters are CSBP & Farmers Ltd. and, to a lesser extent, BP Refinery (Kwinana) Ltd. and TiWest Joint Venture. By the year 2021 direct inputs of most heavy metals to Cockburn Sound are predicted to decline to zero, or to early 1950s levels.

Hydrocarbons and phenol

Hydrocarbon and phenol loadings to Cockburn Sound are shown in Figure 4.4-6. Hydrocarbon loads, mainly from the BP outfall, were about 350 tonnes per year from the mid-1950s to 1979. They declined to about 80 tonnes per year by 1992 and then decreased further to the current level of about 30 tonnes per year. Hydrocarbon loading from the BP outfall is projected to be zero by 2011. Phenol loadings, also from BP, show a similar trend to hydrocarbon loadings and decreased from about 180 tonnes a year in 1978 to 5 tonnes in 1994. BP have an objective of achieving zero discharge of phenol by 2011.

Grease, fluoride, sulphate and total dissolved solids

Grease and fluoride inputs to Cockburn Sound are shown in Figure 4.4-7. Grease inputs declined from 700 tonnes in 1981 to zero in 1985 by which time the effluent from the Woodman Point Treatment Plant had been diverted from Cockburn Sound to Sepia Depression. The discharge of fluoride into Cockburn Sound, mostly from CSBP, reached a peak of over 2600 tonnes per year in 1977 and then decreased to the current level of just over 300 tonnes per year. Annual loads of sulphate from the Woodman Point Wastewater Treatment Plant reached 700 tonnes in 1983 and declined to zero in 1985. Current sulphate loading into the sound, mainly from Tiwest and to a lesser extent via contaminated groundwater from Western Mining, is about 3000 tonnes per year and is projected to remain at this level.

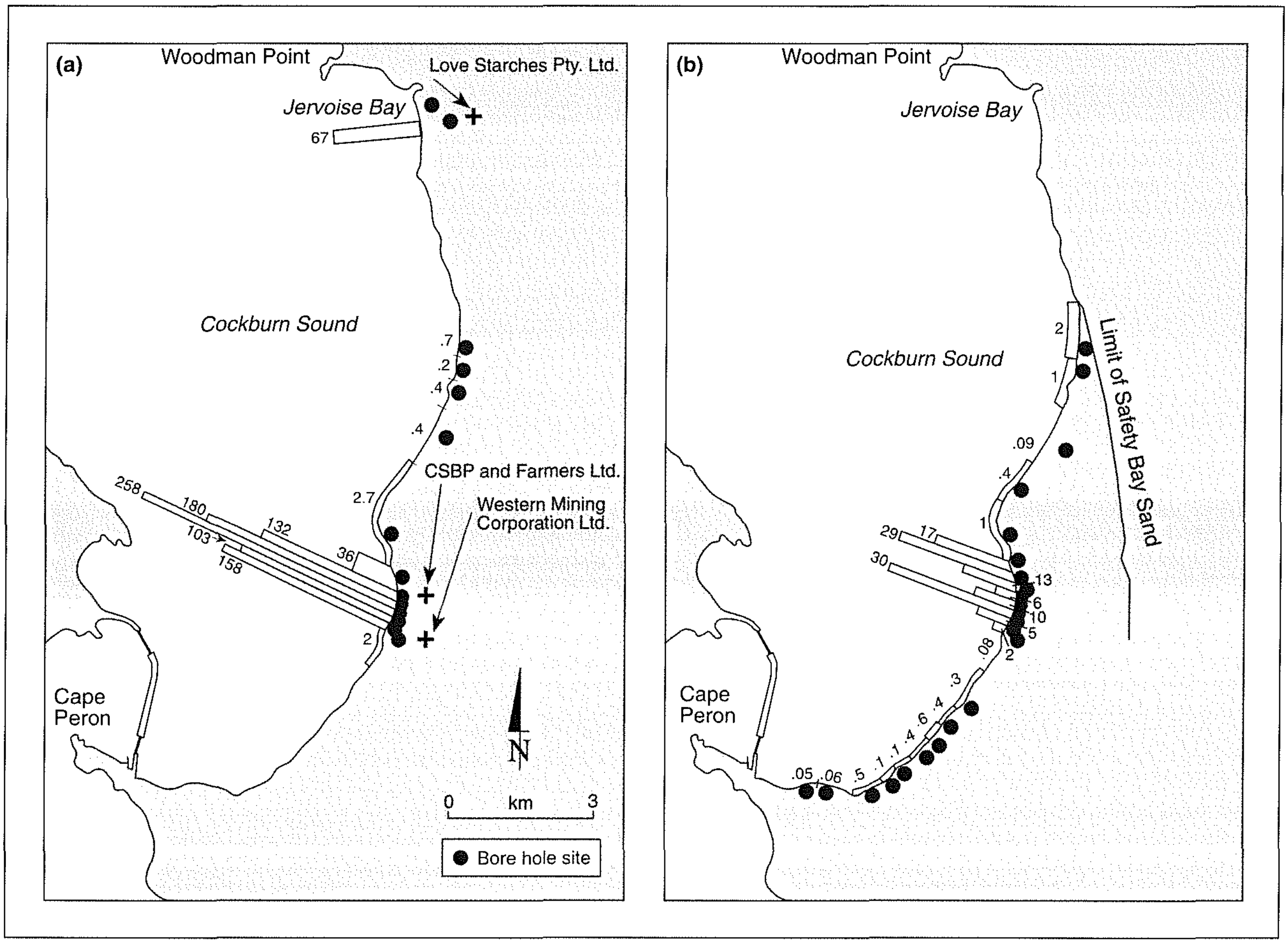


Figure 4.4-3. Groundwater nitrogen flux into Cockburn Sound from (a) the Tamala Limestone and (b) the Safety Bay Sand (redrawn from Appleyard, 1994). Loads are in tonnes yr^{-1} km^{-1}.

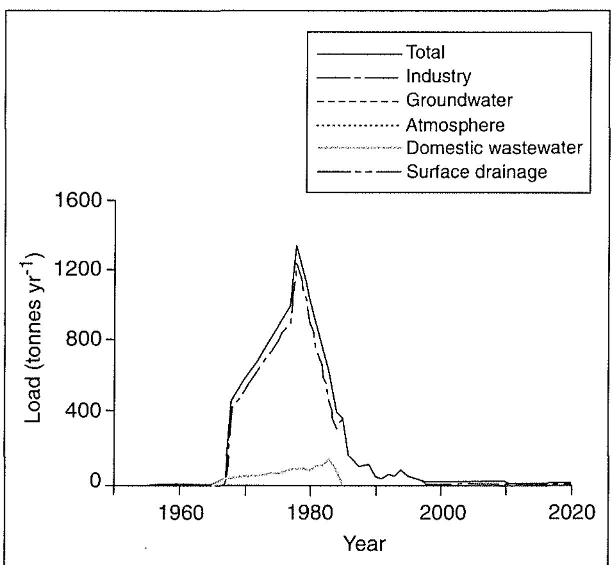


Figure 4.4-4. Annual loads of phosphorus to Cockburn Sound.

Total dissolved solids, also from Tiwest, show a similar pattern with annual loads in 1992 of about 34 000 tonnes. Current annual loads are 24 000 tonnes and are projected to remain at this level.

Radioactivity

Industrial discharges and nuclear-powered warships are the two potentially major sources of radionuclides to the study area. Tiwest's effluent contains four radionuclides: Radium 224, Radium 226, Radium 228 and Thorium 228. Discharge concentrations per annum between 1994 to 2021 will be 22 mBq l^{-1}, 75 mBq l^{-1}, 50 mBq l^{-1} and 22 mBq l^{-1}, respectively. Routine monitoring of the sediments, shellfish (mussels) and other biota show no presence of radionuclides from nuclear-powered warships, suggesting that there are no inputs from this source (Annual reports from 1976 by the Australian Nuclear Sciences and Technology Organisation).

4.4.1.2 Owen Anchorage

Point source discharges into the waters of Owen Anchorage have been mainly from a variety of animal processing plants in the South Fremantle/Coogee area. The waste streams of these industries have been typically characterised by high organic loads. However, the planned relocation of many of these industries to the North Coogee Redevelopment Project (Biotechnology Park) which incorporates a centralised waste treatment facility, and the connection of remaining industries' waste streams to the Cape Peron sewer by the mid-1990s is well advanced. As a result, all of these point

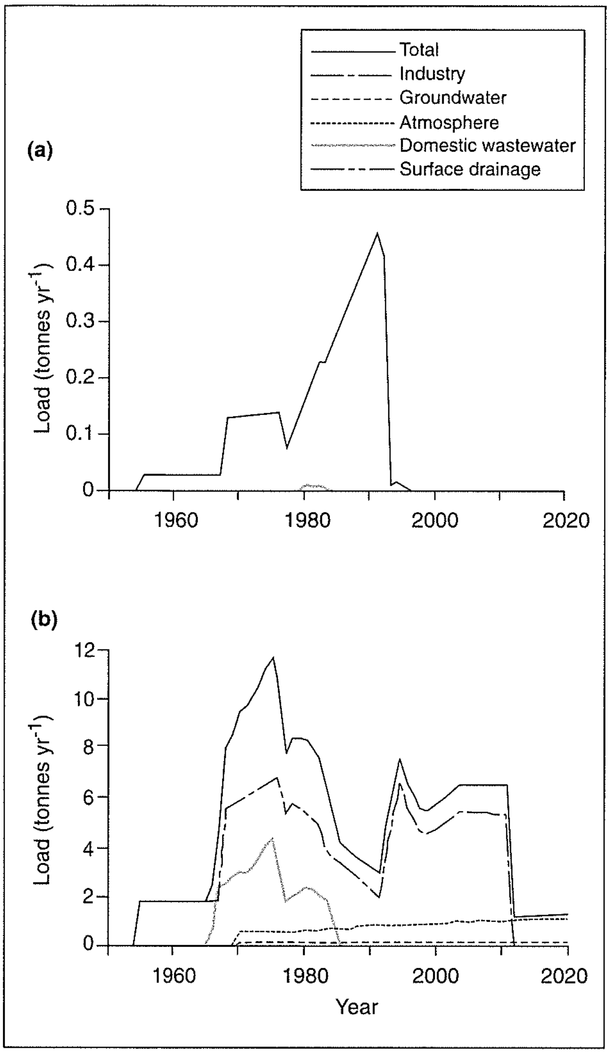


Figure 4.4-5. Annual loads of (a) mercury and (b) zinc to Cockburn Sound. Industrial discharges were the major (> 95%) source of mercury.

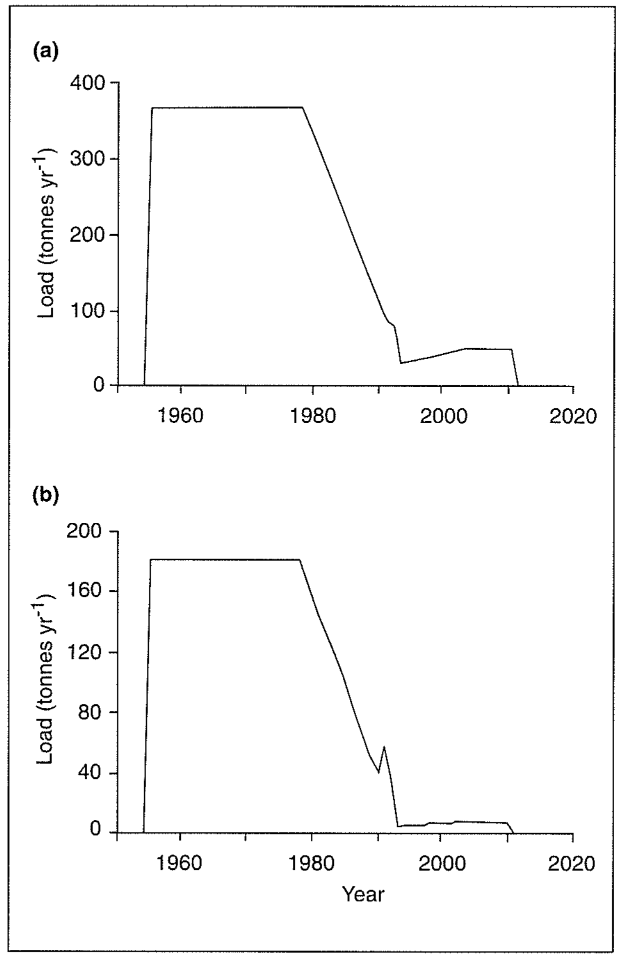


Figure 4.4-6. Annual loads of (a) hydrocarbon and (b) phenol to Cockburn Sound. BP was the major (> 95%) source of these contaminants.

source industrial waste discharges into Owen Anchorage should have ceased by about 1997, resulting in a major reduction in contaminant loads, particularly nitrogen, chromium, organic matter and bacteria, to these waters.

At the time of preparation of the Muriale and Cary (1995) report, the major remaining point source discharges into Owen Anchorage were Kailis and France Pty Ltd, Coogee Fellmongers, Fremantle Fisherman's Co-operative Pty Ltd, George Weston and Cockburn Cement Ltd. Since the late 1970s, the WA Meat Commission, the State Energy Commission power station at South Fremantle and Eagle West had closed, and the Gosh Leather tannery had relocated. Only the 'leather finishing' component of Gosh Leather is currently undertaken at Coogee and the treated wastewater from this process is discharged to the Woodman Point wastewater treatment plant.

More recently, the waste discharges to Owen Anchorage from Coogee Fellmongers and George Weston have also ceased.

Nutrients

Nutrient loads into Owen Anchorage are shown in Figure 4.4-8. In 1950, the annual nitrogen load to Owen Anchorage was about 60 tonnes. This increased sharply during the 1980s to about 170 tonnes in 1987, mostly as a result of waste discharges from the WA Meat Commission. Nitrogen loads have decreased since then to about 100 tonnes in 1993. With the expected relocation of industries and the connection of remaining industrial waste streams to sewer in the mid-1990s, nitrogen loads are projected to decrease to 50 tonnes a year in 2001, gradually increasing thereafter to about 70 tonnes by 2021. The main sources of nutrients to Owen Anchorage waters over this period (excluding estuarine inputs) are projected to be mainly from atmospheric deposition with some contribution from groundwater. Phosphorus loads show similar long-term trends to nitrogen.

Heavy metals

Apart from zinc and chromium, historical heavy metal loadings to Owen Anchorage from industrial discharges were low. Although industrial discharge of zinc to Owen Anchorage is projected to cease by the mid-1990s, annual loads, mainly due to groundwater and atmospheric inputs, will increase from about three tonnes per year in 1994 to

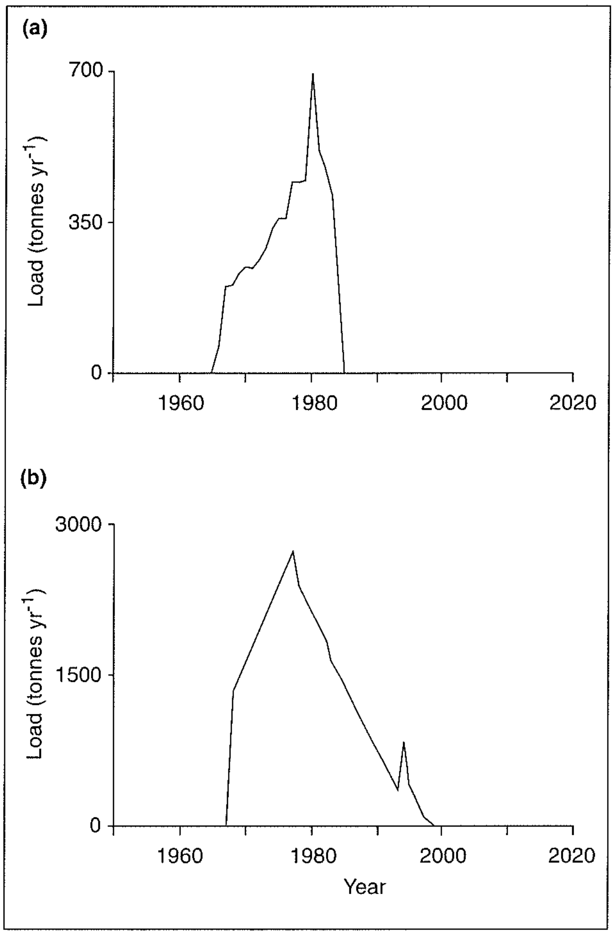


Figure 4.4-7. Annual loads of (a) grease and (b) fluoride to Cockburn Sound. Domestic wastewater was the major (> 95%) source of grease and CSBP was/is the major source of fluorine.

over 4.5 tonnes per year in 2021 (Figure 4.4-9a). The recent decrease in chromium discharge to these waters resulted from the waste stream from Gosh Leather being connected to sewer (Figure 4.4-9b).

Grease, total suspended solids, total dissolved solids, biological and chemical oxygen demand

Grease, total suspended solids, total dissolved solids, biological and chemical oxygen demand all show similar patterns, with high loadings throughout the 1950s to the late 1980s followed by a sharp decline to zero loading in the mid-1990s. Figure 4.4-10 exemplifies these trends for selected parameters. Industrial discharges were major sources of these contaminants.

4.4.1.3 Warnbro Sound

There are no industrial point source discharges to Warnbro Sound, and the main sources of contaminant inputs are via two surface drains at the northern end of the Sound, atmospheric deposition and minor contributions from groundwater. The cumulative contaminant loads from these sources are low.

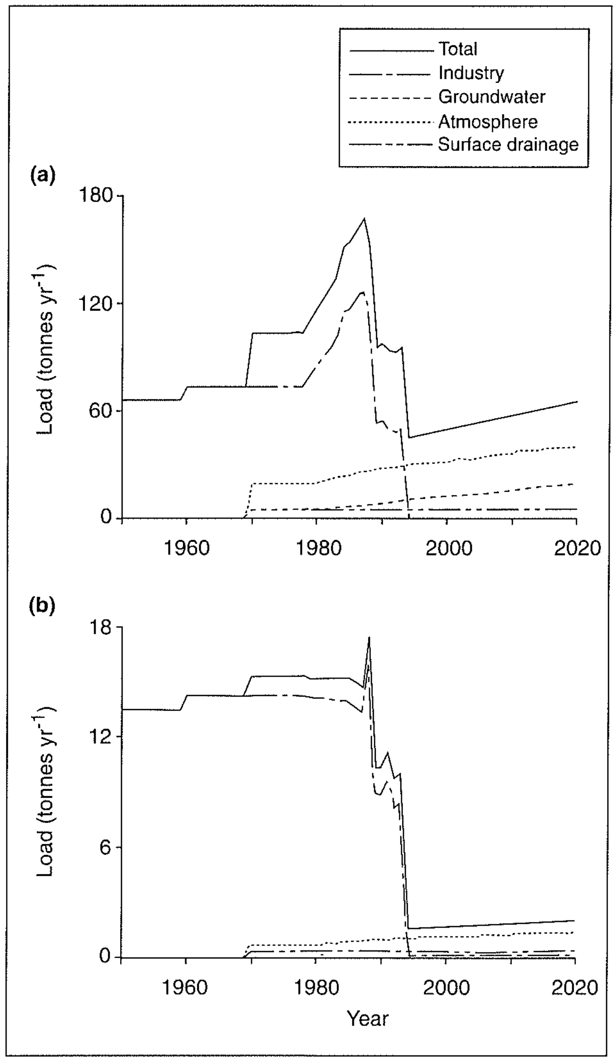


Figure 4.4-8. Annual loads of (a) nitrogen and (b) phosphorus to Owen Anchorage.

Nutrients

Current annual nitrogen loading to Warnbro Sound is 17 tonnes and is projected to increase to about 23 tonnes by 2021. Similarly, the annual load of total phosphorus, mainly in surface runoff, is about three tonnes, increasing to a projected 3.2 tonnes per year by 2021.

4.4.1.4 Sepia Depression

Before 1984 the only direct discharge of wastes to Sepia Depression was via a small domestic wastewater outfall off Cape Peron. In that year the domestic wastewater stream into the northern end of Cockburn Sound was diverted to Sepia Depression. Since then primary treated effluent from both the Point Peron and Woodman Point wastewater treatment plants has been discharged to Sepia Depression via a submarine outfall extending 4 km offshore. Treated wastewater is the largest source of contaminants to these waters. About 100 million litres per day of effluent were discharged in 1994 and it is projected that this will increase to over 250 million litres per day by 2021. Loads of nutrients, heavy metals, grease and other materials will increase in

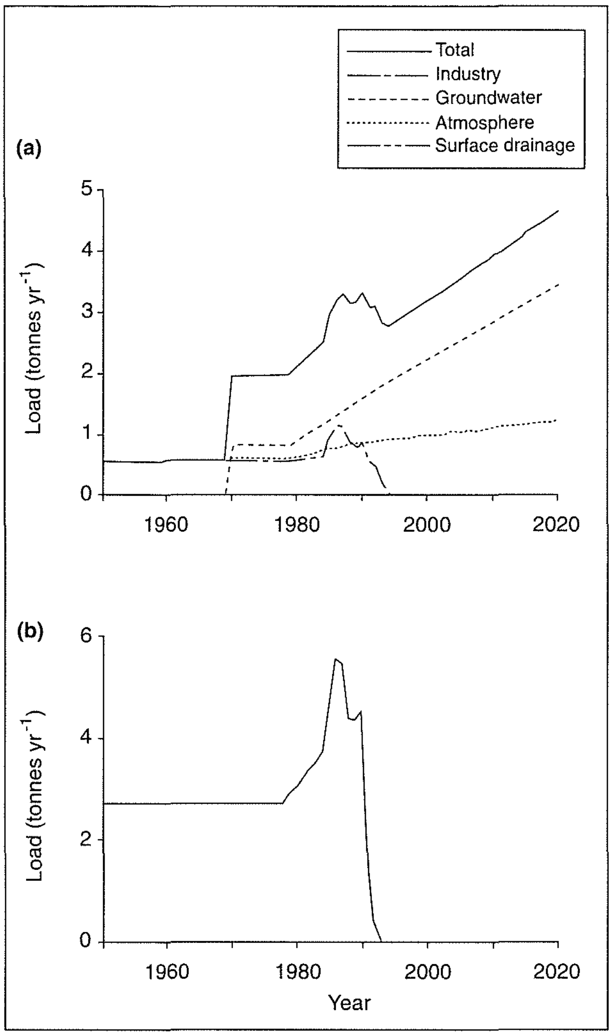


Figure 4.4-9. Annual loads of (a) zinc and (b) chromium to Owen Anchorage. Industrial discharges were the major (> 95%) source of chromium.

proportion with the flow rate if current treatment and disposal methods remain. Projected loads may also increase if industrial waste streams are discharged into the pipeline.

Nutrients

Historical, current and projected annual nutrient loads into Sepia Depression from 1970 to 2021 are shown in Figure 4.4-11. The total annual nitrogen load to these waters was about 15 tonnes in 1970, increased sharply to 537 tonnes in 1984 as a result of the diversion of effluent from Cockburn Sound, and was about 1900 tonnes in 1994, almost 80% as ammonium-N. Projected loads are estimated to reach over 5300 tonnes by the year 2021 (Figure 4.4-11), assuming the continuation of current treatment and disposal practices. Total phosphorus loads show similar trends and are approximately 25% of the loads for total nitrogen. The wastewater outfall is the predominant source of nutrients discharged to Sepia Depression.

Heavy metals

Heavy metal loads into Sepia Depression are shown in Figure 4.4-12. Over the period 1994 to 2021 annual loads of zinc are

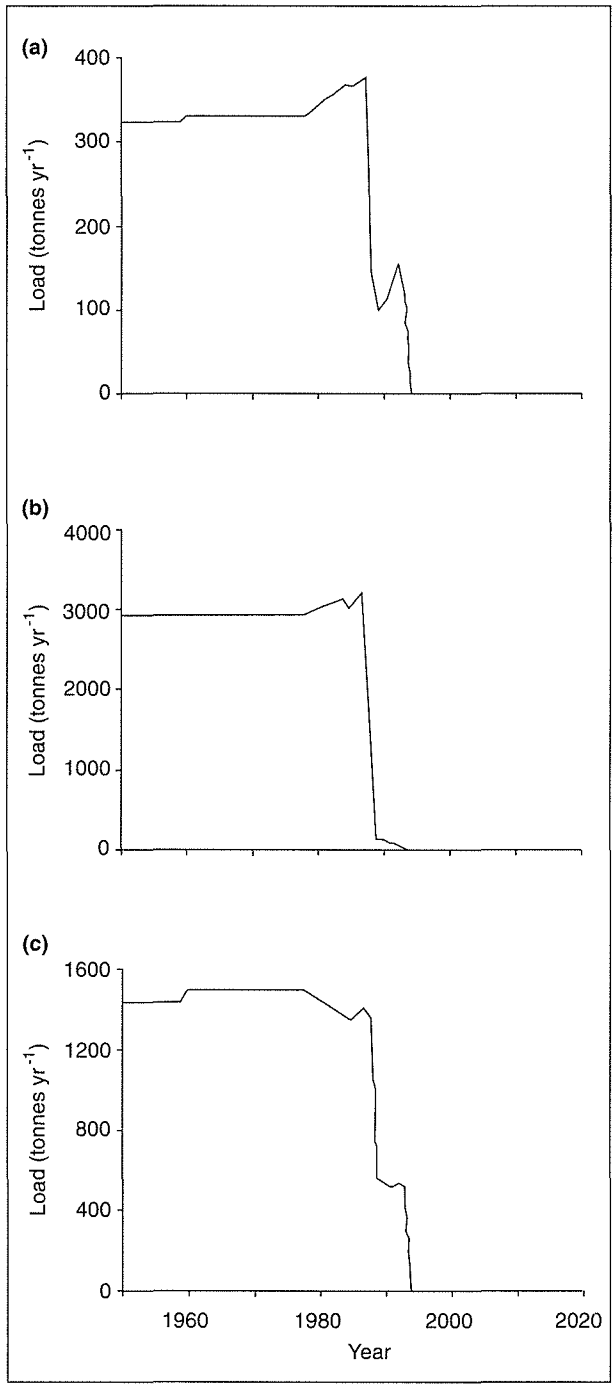


Figure 4.4-10. Annual loads of (a) grease, (b) total dissolved solids and (c) biochemical oxygen demand to Owen Anchorage. Industrial discharges were the major (> 95%) sources of these contaminants.

projected to increase from 15 to 37 tonnes, copper from 11 to 31 tonnes, cadmium from 0.8 to 2 tonnes, chromium from 3.6 to 10 tonnes, lead from 5.9 to 11 tonnes and mercury from 0.4 to 2 tonnes. Apart from lead, over 99% of these inputs are discharged via the wastewater outfall. Currently, atmospheric deposition of lead accounts for about 40% of the total loading to Sepia Depression whereas, by 2021, it will contribute less than 1% due to projected reductions in the lead content of petrol.

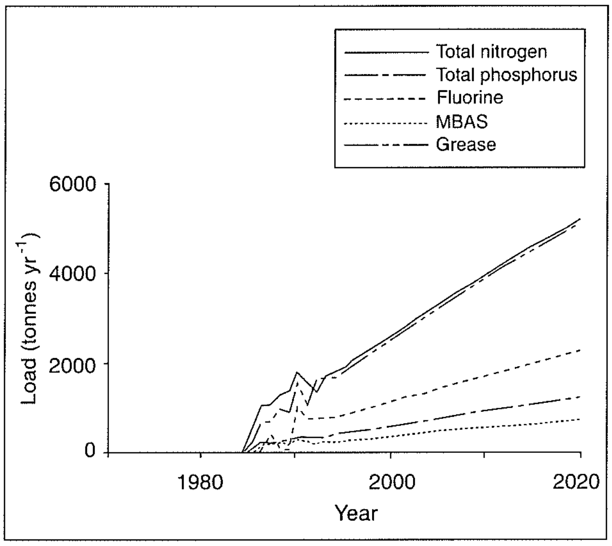


Figure 4.4-11. Annual loads of nitrogen, phosphorus, fluorine, MBAS (methylene blue active substances) and grease to Sepia Depression. The Cape Peron domestic wastewater outfall is the major (> 95%) source of these contaminants.

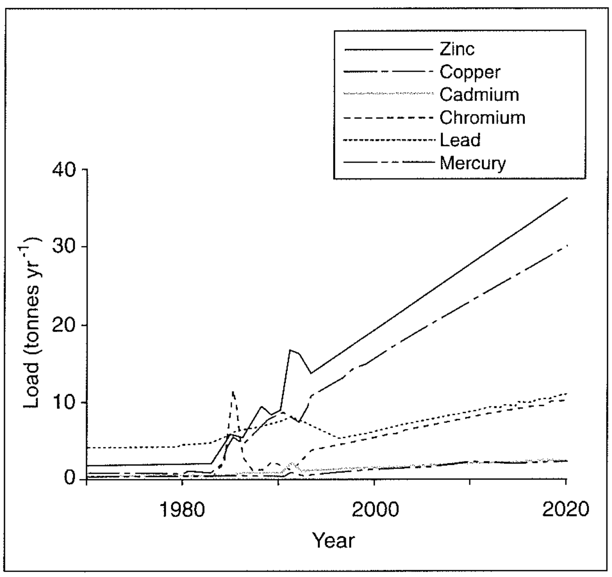


Figure 4.4-12. Annual loads of heavy metals to Sepia Depression. The Cape Peron domestic wastewater outfall is the major source of these contaminants.

Grease, fluorine, MBAS, sulphide, suspended solids and faecal coliforms

Current and projected loads of grease, fluorine and MBAS (methylene blue active substances) discharged into Sepia Depression are shown in Figure 4.4-11. Over the period 1994 to 2021 annual loads of grease are projected to increase from approximately 1800 tonnes to 5200 tonnes, Fluorine from about 800 tonnes to 2300 tonnes, MBAS from about 250 tonnes to 740 tonnes, sulphide from about 90 tonnes to 260 tonnes, and total suspended solids from about 3000 tonnes to 10 000 tonnes. These loadings enter Sepia Depression via the Cape Peron wastewater outfall. The major source of faecal bacteria to Sepia Depression is also from domestic wastewater and faecal coliform concentrations in this effluent exceed one million organisms per 100 millilitres.

Conclusions

Direct contaminant inputs into Cockburn Sound from industrial sources have generally declined significantly since the 1970s, whereas inputs from groundwater have increased; between 1994 and 2021 groundwater will be the major source of nitrogen to Cockburn Sound. The majority of contaminant inputs to Owen Anchorage have declined dramatically since the early 1990s and atmospheric deposition and groundwater will be the major generic sources of contaminants after the late 1990s due to the closure, relocation or connection to sewer of the industries which formerly discharged to Owen Anchorage. Contaminant loads to Warnbro Sound are small and have changed little since the 1950s. Contaminant inputs to Sepia Depression, mainly from urban (domestic wastewater) sources, are predicted to increase dramatically over the next 30 years, if current treatment and disposal practices are continued.

4.4.2 Estuarine inputs

The Swan-Canning and Peel-Harvey estuaries flow into Perth's coastal waters year round. Greatest flows occur from June to September, a period characterised by relatively high river flow and variable wind conditions. As the estuary outflow waters are more buoyant than the receiving coastal waters during this period, they form surface plumes which are readily moved by wind-stress during variable wind conditions. As a result, much of the nearshore southern metropolitan coastal waters can be affected by outflows during this period. Satellite images of discoloured surface waters, and physical, chemical and biological measurements, allow these plumes to be tracked, sometimes for over 100 km from their source (see sections 4.7.2, 4.8.1, 5.1.1). Estimates of nutrient loadings to the metropolitan coastal waters from estuarine sources were made by Deeley (in preparation) for the Swan-Canning and Peel-Harvey estuaries, and by Martinick *et al.* (1993) for the Swan-Canning estuary. Reliable estimates of nutrient loads discharged to the ocean from estuaries are difficult to make and further work is required in this area.

Findings

Currently the outflows from the Swan-Canning and Peel-Harvey estuaries are estimated to contribute a total of about 1300 tonnes of nitrogen and 160 tonnes of phosphorus annually to the coastal waters of Perth. On an annual basis, the estuaries are significant but not dominant sources of nutrients to the southern metropolitan coastal waters. Available estimates for the Swan-Canning estuary discharge into Perth's coastal waters range between 250 and 900 tonnes of nitrogen and between 30 and 100 tonnes of phosphorus, while estimates for the Peel-Harvey Estuary discharge range between 450 and 900 tonnes of nitrogen and between 60 and 120 tonnes of phosphorus. Approximately 30% of the total nitrogen load is in an inorganic nitrogen form. Blooms in the Peel-Harvey Estuary of the blue-green alga *Nodularia*, which have been occurring between October and February almost yearly since the 1980s, have been estimated to contribute an additional 500 tonnes of nitrogen annually to the coastal waters (Deeley, in preparation).

The long-term ecological effect of the Dawesville Channel is unknown, but preliminary data suggest that the majority of the nitrogen being discharged from the Peel-Harvey estuary enters the ocean via the Dawesville Channel, and that *Nodularia* blooms are likely to occur less frequently, due to the resultant change in the salinity regime of the Peel-Harvey estuary.

Since the 1960s, it is estimated that nitrogen loads discharged from the Peel-Harvey and the Swan-Canning estuaries have increased by more than 450% (1000% including *Nodularia* inputs) and 350%, respectively (Figure 4.4-13) (Deeley, in preparation). Future loads are unknown at this stage and will depend on the effectiveness of catchment management programmes and the effects of the Dawesville Channel.

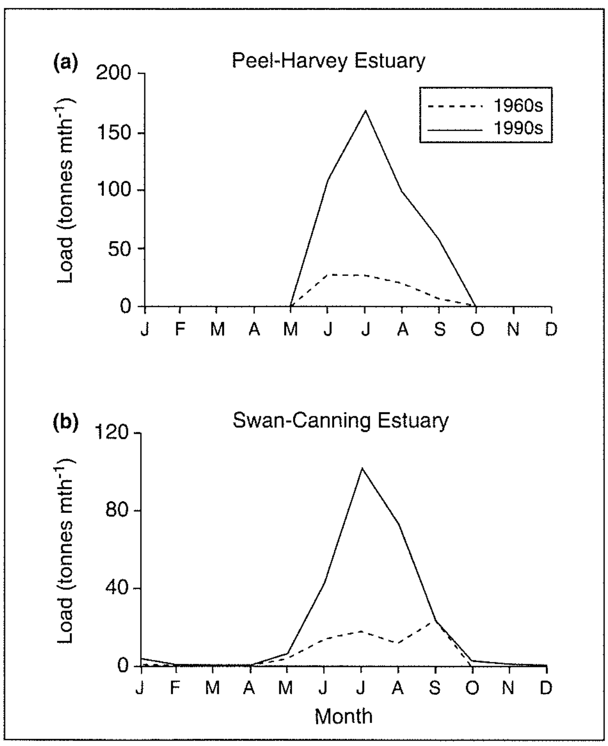


Figure 4.4-13. Estimated mean monthly loads of nitrogen to the coastal waters of Perth from (a) the Peel-Harvey and (b) the Swan-Canning estuaries in the 1960s and 1990s (data from Deeley, in preparation). Nitrogen loads to coastal waters from algal blooms are not included.

Conclusions

Contaminant inputs from estuarine discharges have increased significantly since the 1960s and are currently an important but not dominant source of nutrients to the southern coastal waters of Perth.

4.4.3 Shipping inputs

Over 1700 ships, averaging 17 000 tonnes, visited Fremantle Harbour and Cockburn Sound in 1993/94. The average duration of a visit was approximately two days. Ships introduce contaminants directly into coastal waters through accidental discharges, controlled discharge of sewage, ballast, engine coolant waters and bilge waters, and as a result of washdown procedures. Contaminants also enter coastal waters indirectly as a result of flaking or chemical leaching off the hulls of ships. Data have been provided by the Fremantle Port Authority and compiled by Martinick *et al.* (1993) and Muriale and Cary (1995).

Findings

4.4.3.1 Accidental spillages

Accidental spillages of both solid and liquid materials occur mainly during the loading and unloading of cargo. The majority of the data available on solid material spillages concern fertiliser, grains and bauxite spilled into Cockburn Sound during the unloading at the Fremantle Port Authority Bulk Cargo Jetty and the BHP jetty at Kwinana. A report by Meagher and Associates (1984) conservatively estimated annual losses of fertilisers to be in the range of 500-800 tonnes. Due to changes in offloading practices, annual loads of nitrogen and phosphorus (soluble) lost to the sound through spillage have declined dramatically and are currently about five tonnes each (C. Deans, personal communication). The fertilisers normally transported are listed below in approximate, diminishing order of annual tonnage of receival: rock phosphate, sulphur, diammonium phosphate, urea, muriate of potash, potassium sulphate, triple superphosphate and ammonium sulphate. Nitrogen and phosphorus loads from these fertiliser spills are therefore a minor nutrient source to Cockburn Sound.

The most common liquid spills are from petroleum-related products. There have been four reported spills between 1991 and 1994 discharging approximately eight tonnes of petroleum products into Cockburn Sound although this is probably underestimated by an order of magnitude (C. Deans, personal communication).

4.4.3.2 Controlled discharge

Controlled discharges are mainly liquid materials from ballast, sewage, engine coolant, bilge waters and washdown procedures. The regulations for the discharge of any wastes into the sea are bound by the Fremantle Port Authority Act in port waters and the International Maritime Organisation protocols MARPOL 73/78, Annexes I-V. The only quantitative data available for these inputs are on ballast water discharges and this is covered in section 4.13. Sullage from commercial vessels cannot be legally discharged into state waters, however discharges into Commonwealth waters just beyond the limit of the state waters have occasionally been reported. Recreational vessels cannot legally discharge sullage into estuarine or port authority waters. Sullage pump-out facilities are located at Barrack Street, Rous Head, HMAS Stirling and Mandurah, however there are no facilities at Hillarys Boat Harbour. Washdown procedures (of the storage hatch) apply to all vessels if the vessel changes its cargo, and to jetties/wharves after the vessel has loaded or unloaded its cargo. There were only two ships in 1994 that underwent hatch washdown procedures, discharging approximately 100 kg of nitrogen. Hatch washdowns occur in port unless the FPA believes that cargo in the hatch cannot be cleaned sufficiently and the ship hatch is washed down outside of state waters. However before 1992 most cargo ships underwent washdown procedures of their hatches at port and it is estimated that until 1992 up to one tonne of nitrogen and 10 tonnes of phosphorus were discharged into Cockburn Sound via this method (C. Deans, personal communication). The amount of fertiliser washed down from jetties and wharves has been taken into account in the accidental spillage section.

Bilge waters which can contain traces of oil are not normally pumped out into state waters however if this does occur then the water is passed through a separator to remove the oil. The contaminant input from bilge water is therefore considered to be small.

4.4.3.3 *Antifouling paints*

Antifouling paints are applied to the hulls of vessels to inhibit the build up of marine organisms that increase drag and slow the vessels down. These paints leach toxic ingredients into the water, killing marine organisms attached to the vessel. However, the leaching of some types of anti-fouling paints can cause harmful effects to a wide range of marine life other than the target organisms (see section 4.11). Since the 1960s the most effective and therefore widely used anti-fouling paint has contained organotin compounds and, in particular, tributyltin (TBT). In Western Australia, regulations which restrict the use of organotin-based antifouling paints became effective from November 1991. Vessels less than 25 m in length are not permitted to use organotin-based paints. Vessels greater than 25 m may still use organotin-based paints, but only if their long-term leaching rate is less than 5 μg organotin cm^{-2} day^{-1}. These regulations, however, do not apply to large vessels which visit Western Australia, but are registered and serviced elsewhere.

Conclusions

Ballast water (containing foreign marine organisms) and TBT leachate from anti-fouling paints are the contaminant discharges of major concern from shipping-related operations. Nutrient loads from accidental spillages during ship loading/unloading appear minor, but load estimates were based on limited data. Generally, more work is required to adequately quantify the inputs of contaminants from shipping-related operations.

4.4.4 Cumulative inputs

The current point source discharges of industrial and urban wastes into the southern metropolitan coastal waters of Perth consist of four heavy industry outfalls located along the eastern shoreline of Cockburn Sound, two emergency sewage outfalls off Woodman Point, three relatively minor outfalls discharging to Owen Anchorage and a major domestic wastewater outfall off Cape Peron. Industrial and urban wastes are not discharged directly into Warnbro Sound or Comet Bay. Contaminated groundwater contributes significant loads of contaminants to these waters, particularly to Cockburn Sound. Contaminants also enter these waters in surface drainage; there are four drains located near Fremantle, one in Mangles Bay and two in the northern part of Warnbro Sound. Atmospheric deposition of most contaminants to the southern coastal waters of Perth, apart from lead, is considered to be relatively minor.

Since the late 1970s the major changes in relation to waste discharges to these waters include the improved effluent treatment of industries discharging to Cockburn Sound, either the closure, relocation or connection to sewer of industries previously discharging to Owen Anchorage, and the diversion of the Woodman Point Wastewater Treatment Plant effluent stream from Cockburn Sound to Sepia Depression.

Findings

4.4.4.1 *Nutrient inputs*

Nutrient inputs to the southern metropolitan coastal waters of Perth, particularly from urban sources, are predicted to increase dramatically over the next 30 years. Trends for nitrogen and phosphorus are similar and have a relatively constant loading ratio of about 4:1 respectively (Figure 4.4-14). Excluding estuarine inputs (see section 4.4.2), about 2500 tonnes of nitrogen per year, mostly inorganic, are currently discharged to the southern coastal waters of Perth. By 2021, this will increase to approximately 6000 tonnes per year if current treatment and disposal practices continue. The main source (excluding estuaries) of nitrogen discharged to these waters has changed in recent times. In 1979, about 60% originated from direct industrial discharge. By 1990, this source was about 25% and, by 2021, is projected to contribute less than 5% of the total. By contrast, domestic wastewater discharge contributed about 60% in 1990 and will contribute over 90% of the total nitrogen load by 2021 (Figure 4.4-14a).

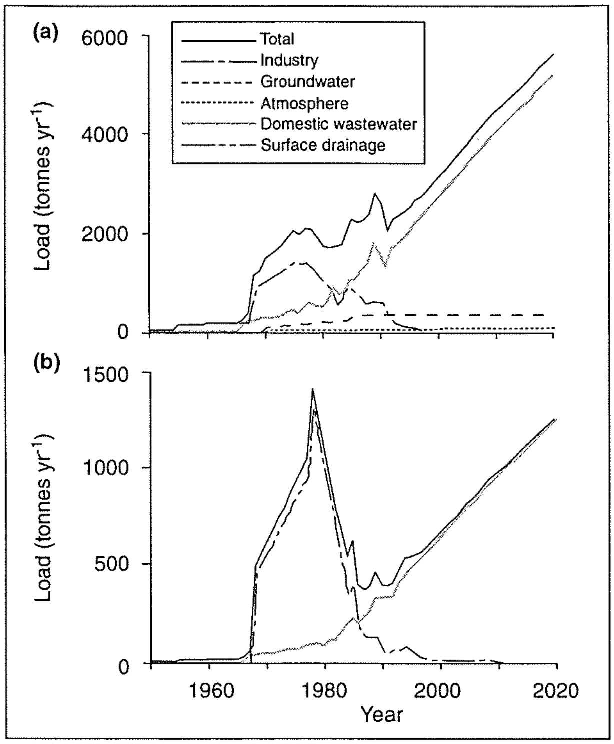


Figure 4.4-14. Annual loads of (a) nitrogen and (b) phosphorus to the southern metropolitan coastal waters of Perth, excluding estuarine inputs.

4.4.4.2 *Heavy metals*

Approximately 25 tonnes of zinc, 14 tonnes of copper, 4 tonnes of chromium, 1 tonne of cadmium, 0.5 tonnes of mercury and 9 tonnes of lead were discharged into the southern metropolitan coastal waters in 1994, with over 50% being discharged from the domestic wastewater outfall in Sepia Depression (Figure 4.4-15). These loadings are projected to increase by a factor of between two and three over the next 30 years with treated wastewater contributing about 80-90% of the total load by 2021. Lead loads declined by approximately 50% between 1990 and 1996 due to reduced lead content in motor vehicle fuel.

4.4.4.3. *Grease, total suspended solids, fluorine, sulphide and faecal coliforms*

Current and projected loads of grease and total suspended solids discharged into the southern metropolitan coastal waters are shown in Figure 4.4-16. Grease inputs in 1994 were estimated to be approximately 1800 tonnes with loads projected to increase to about 5200 tonnes by 2021. Total suspended solids were approximately 4000 tonnes in 1994 increasing to 13 000 tonnes in 2021. Fluorine loads to these waters were estimated to be about 1600 tonnes in 1994 and 2300 tonnes in 2021, and sulphide loads in 1994 were about 86 tonnes increasing to 261 tonnes in 2021. Over 95% of these loadings were discharged into Sepia Depression via the domestic wastewater outfall. The major source of faecal bacteria to these waters is from domestic wastewater discharged into Sepia Depression and faecal coliform concentrations in this effluent exceed one million organisms per 100 millilitres.

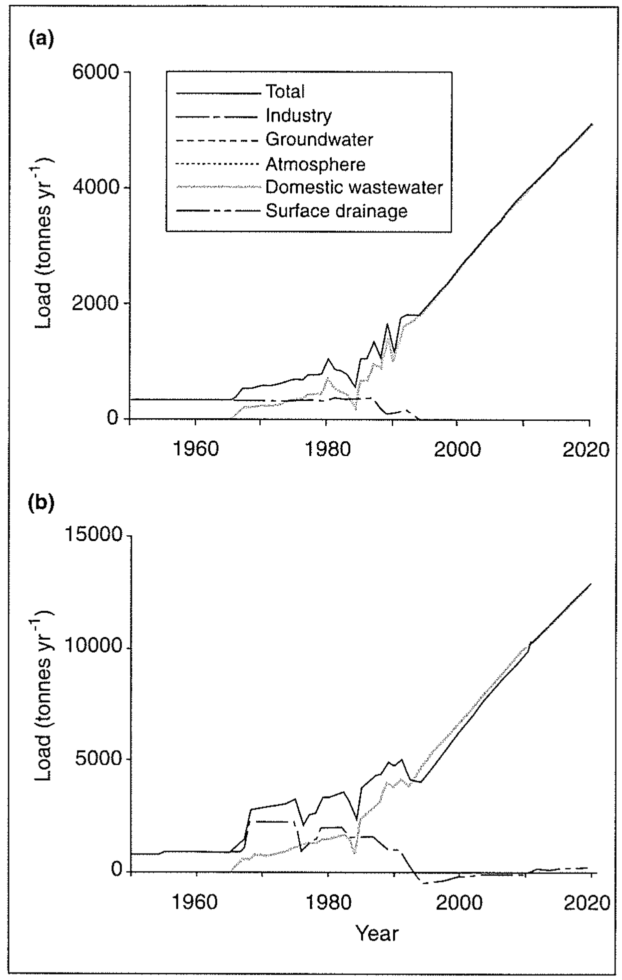


Figure 4.4-16. Annual loads of (a) grease and (b) total suspended solids to the southern metropolitan coastal waters of Perth.

Conclusions

Contaminant inputs to the southern metropolitan coastal waters of Perth, particularly from urban sources, are predicted to increase dramatically over the next 30 years unless wastewater treatment and disposal practices are changed. Domestic wastewater contributed about 60% of the total nitrogen load in 1990 and will contribute over 90% by 2021. Heavy metal loads will increase by a factor of between two to three by 2021 with treated wastewater contributing about 80-90% of the total load.

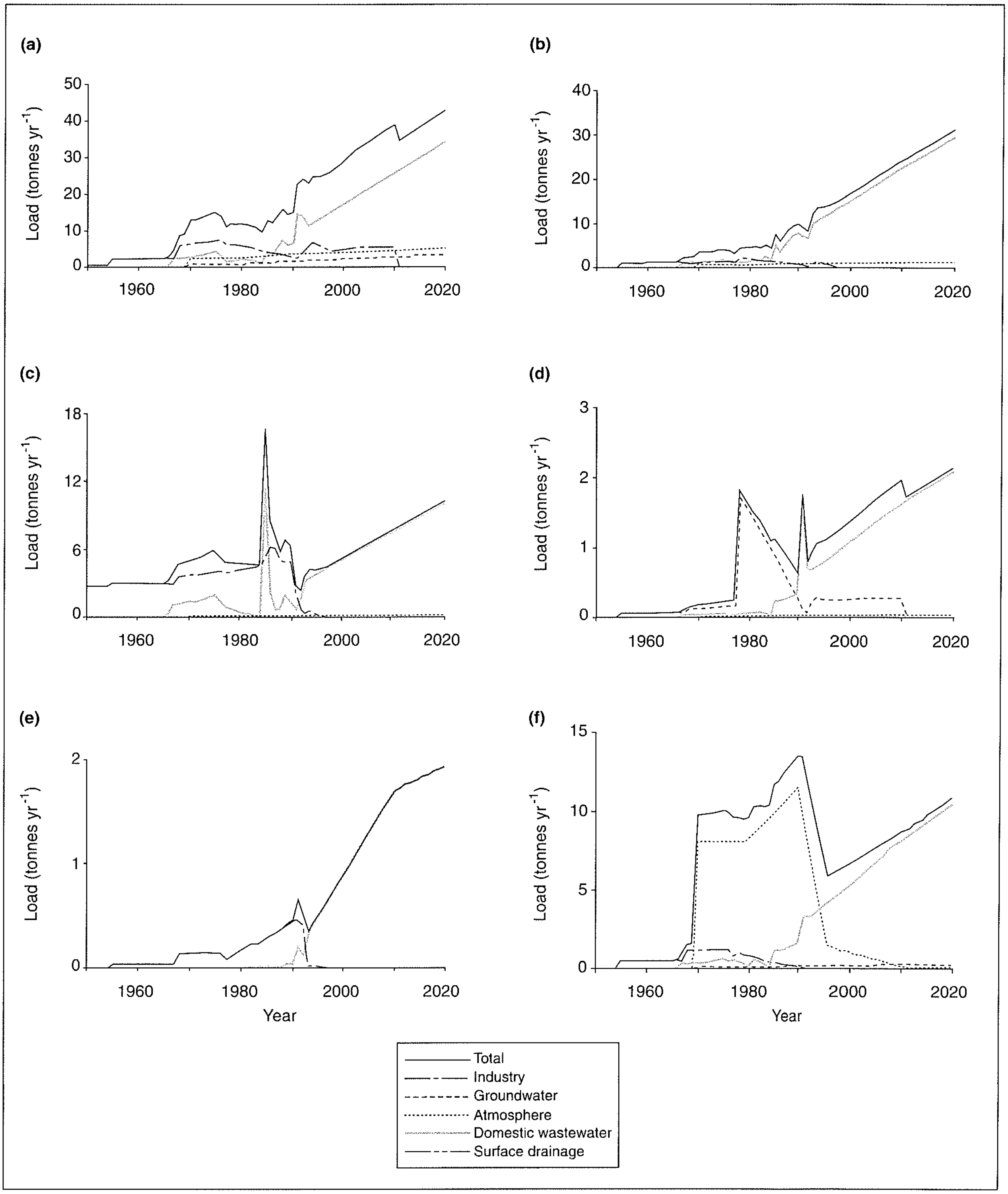


Figure 4.4-15. Annual loads of (a) zinc, (b) copper, (c) chromium, (d) cadmium, (e) mercury and (f) lead to the southern metropolitan coastal waters of Perth.

4.4.5 Potential future inputs

Toxic contamination of coastal groundwater occurs principally under the Kwinana industrial area, with contamination from substances such as pesticides, herbicides, heavy metals, hydrocarbons and phenolic compounds (DEP and Dames and Moore, 1994). The two major sources are the BP Refinery (Kwinana) Ltd. and Nufarm Ltd. (previously CIK). Sixty thousand tonnes of hydrocarbons are estimated to underlie the BP Refinery site and recovery has been occurring since 1988 at about five tonnes per day. The groundwater under the Nufarm site was contaminated with chlorinated phenolic compounds as well as the herbicides 2 4-D and 2 4 5-T, and the plume is slowly moving towards Cockburn Sound. The potential for both these contaminant sources to have a detrimental impact on the marine environment is high, and highlights the importance of the monitoring programmes to help develop appropriate remedial strategies.

4.5 Sediments

Marine sediments act as long-term integrators of contaminant inputs to coastal waters, particularly in sheltered depositional environments such as embayments and lagoons where sediments are relatively undisturbed by water movement from waves and currents. The re-suspension of contaminated sediments may also act as a source of contaminants to water and biota long after the cause of the initial contamination has ceased. A thorough knowledge of the mineralogy of marine sediments is generally necessary to interpret sediment contaminant and infauna distributions and to understand ecological processes such as sediment oxygen flux and the relationships between sediment characteristics and contaminant release rates.

The sediments at approximately 200 sites covering an area of about 1000 km^2 of the southern metropolitan coastal waters were sampled in November 1991 and February 1994. The 1994 survey was to provide more detailed information in the areas of highest contamination identified in the 1991 survey, to determine short-term trends for assessing the effectiveness of management initiatives, such as the introduction of restrictions on the use of organotin-based anti-fouling paints, to provide additional baseline information for 'non-core' areas such as Comet Bay and to provide information for other technical programmes of the SMCWS (see sections 4.10 and 5.6).

The sediment studies were conducted in collaboration with the Chemistry Centre of Western Australia with most of the analyses being undertaken at these laboratories. Organotin analyses were undertaken by the CSIRO Centre for Advanced Analytical Chemistry at Lucas Heights, New South Wales (Batley *et al.* 1988). Nutrient analysis was conducted at the Nutrient Analysis Laboratory, Murdoch University, Western Australia.

To provide an indication of the potential severity of contamination the survey results were compared with the ERL (effects range low) and ERM (effects range median) values (see Table 3.5-5) of the draft sediment quality criteria for EQO 2 (maintenance of ecosystem integrity).

The following sections refer primarily to the results of the 1994 survey (Burt *et al.* 1995d). Temporal comparisons have utilised paired site data from the 1994 survey and the results of a previous survey in Cockburn Sound in 1977 (Chegwidden, 1979) and the southern metropolitan coastal waters survey in 1991 (Burt *et al.* 1993b). Further technical details on analytical methodology and quality assurance protocols for the 1991 and 1994 surveys can be found in Burt and Ebell (1994) and Burt and Ebell (1995).

4.5.1 Mineralogy

Approximately 1 kg of surficial sediment (top 20 mm) was collected at each site and analysed for pH, Eh, grain size, organic, carbonate and refractory content, and the percentage of strontium, magnesium, sulphur and calcium, to describe the mineralogical characteristics of the marine sediments of the southern metropolitan coastal waters of Perth.

Findings

Coarse carbonate sands occurred in offshore areas (1.0-0.6 mm) and on the banks and sills (0.6-0.15 mm) east of the Garden Island Ridge (Figure 4.5-1). The sediments in the protected basins of Cockburn and Warnbro sounds and, to a lesser extent, Owen Anchorage consist principally of fine carbonate sands and silt (0.15-<0.038 mm). The refractory (i.e. inorganic, non-carbonate) fractions of the sediments were generally higher inshore than offshore (Figure 4.5-2) and organic content was highest in the basin sediments of Cockburn and Warnbro sounds. In Cockburn Sound the mean organic content of the deep basin sediments increased from approximately 4.5% at the northern entrance of the Sound to about 7% in the central basin to over 10% in the southern area of Mangles Bay. In Warnbro Sound organic content was highest in the central basin and generally decreased with distance offshore. The percentage of refractory material was highest in sediments near the entrances of the estuaries reflecting the terrestrial origin of much of this material. The refractory content of sediments was also high at a number of sites in the vicinity of the Swanbourne wastewater outfall suggesting that these outfalls can be a major source of this material.

In general, the pH of the sediments was slightly less than eight in the deep basins, and slightly greater than eight in bank and offshore sediments. Mean pH values of approximately 7.7 were found for both the Cockburn and Warnbro sound basin sediments. Eh values were generally highest and positive in bank and offshore sediments and lowest and negative in basin sediments. The concentrations of sulphur generally decreased with distance offshore and, in Cockburn and Warnbro sounds, were lower in bank sediments than in deep basin sediments. Relatively high concentrations of sulphur occurred along the eastern and southern edges of the Cockburn Sound deep basin, particularly near Woodman Point, Jervoise Bay, south of James Point, Mangles Bay and Careening Bay, and in the north-eastern area of the Warnbro Sound deep basin.

The general distribution of organic material in the coastal sediments off Perth in 1994 is similar to the late 1970s (Chegwidden, 1979; Chiffings, 1987). In 1977, the organic content of the sediments in Cockburn Sound was highest near the Woodman Point Wastewater Outfall and along the eastern shoreline, particularly in the vicinity of the CSBP outfall (Chegwidden, 1979).

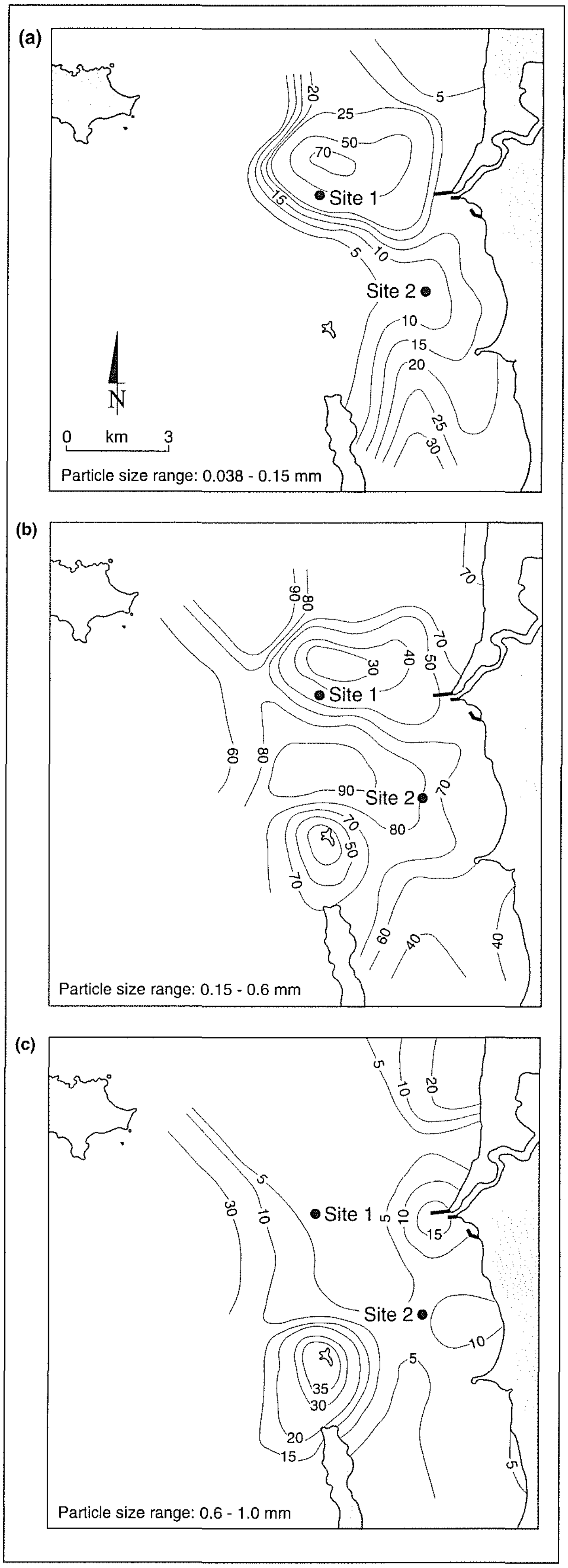


Figure 4.5-1. Relative percent composition of the surficial sediments of Owen Anchorage and surrounding waters in 1994: (a) silt and fine sand, (b) medium sand and (c) coarse sand.

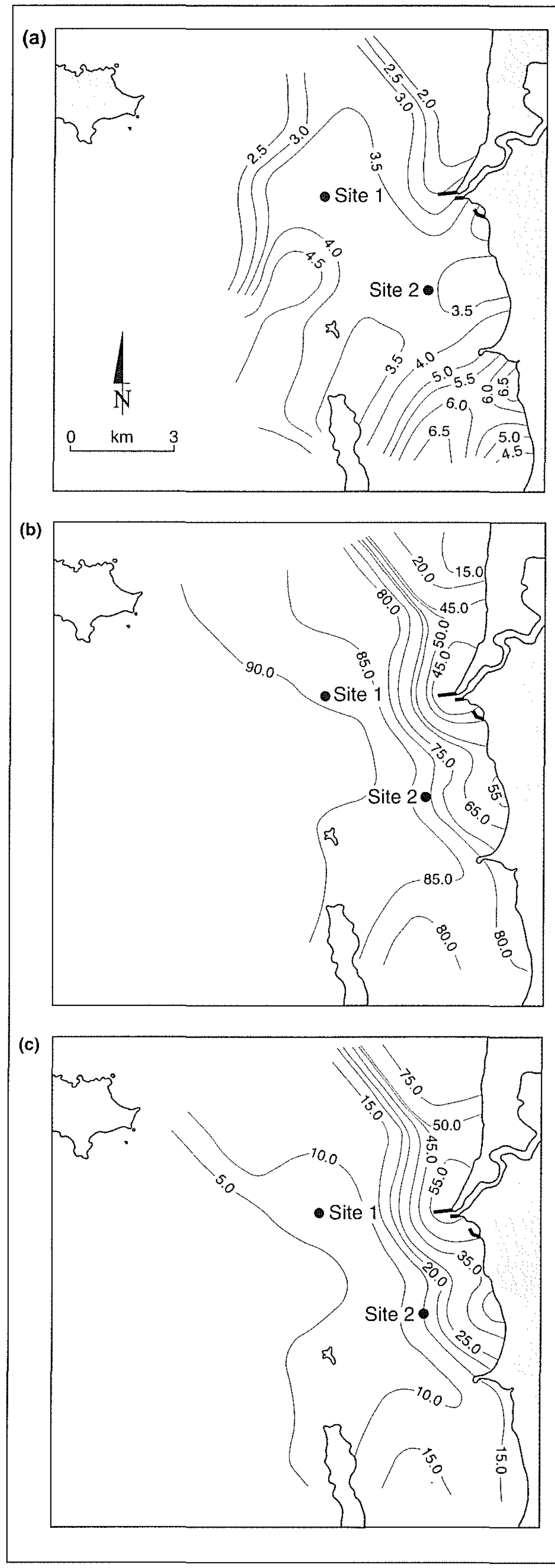


Figure 4.5-2. Relative percent composition of the surficial sediments of Owen Anchorage and surrounding waters in 1994: (a) organic, (b) carbonate and (c) refractory (inorganic, non-carbonate) material.

Since 1978, the mean organic content in the sediments of Cockburn Sound has decreased by approximately 50% (Chiffings, 1987). However in Owen Anchorage, the mean organic content in the sediments has decreased only by about 5% over the same period. Historical data of organic content in the sediments of Warnbro Sound do not exist for this period.

The pH values in the sediments of the study area in 1994 were generally higher in the offshore and bank sediments than in the basins, reflecting a pattern described by Chegwidden (1979) in the late 1970s. In 1977, the lowest pH values in Cockburn Sound occurred in the northwest of the deep basin and near the CSBP outfall. In Cockburn Sound, Warnbro Sound and Owen Anchorage, the mean pH values have remained unchanged since the 1970s (Chegwidden, 1979).

The overall pattern of Eh values in the sediments of the study area is currently similar to that of the late 1970s, however there has been a general decrease in Eh values over this period. In 1977, the Eh values in Cockburn Sound were generally positive, with higher values on the banks, except near the CSBP outfall, and lower values in the deep basin, especially in Mangles Bay and the northern end of the sound, near Woodman Point. Eh values at that time suggested that the sediments of Cockburn and Warnbro sounds were not anoxic but that they were relatively deficient in oxygen (Chegwidden, 1979). Generally, low concentrations of manganese in sediments occur in areas with chronic oxygen deficiency (Chegwidden, 1979). In 1977, the distribution of manganese in the sediments of Cockburn Sound also suggested that the sediments in the basin, particularly the southern area, were relatively deficient in oxygen. In Warnbro Sound, however, the concentrations of manganese in the sediments at that time suggested the sediments were neither oxygen deficient nor anoxic. Since then, the concentration of manganese in Warnbro Sound deep basin sediments has increased and, in 1994, the mean concentrations of manganese in Cockburn Sound and Warnbro Sound were similar, suggesting that the sediments in both embayments were deficient in oxygen.

Conclusions

The occurrence of fine organic sediments inshore and coarser carbonate sands offshore suggests that the relative exposure to wave energy is an important factor influencing the general distribution patterns of organic matter and grain size in the marine sediments of Perth's southern coastal waters. The high content of refractory material in sediments near the mouths of the Swan-Canning and Peel-Harvey estuaries indicates that these estuaries can have a significant local influence on sediment composition. In Cockburn Sound the organic content in the basin sediments increases from north to south, and this pattern has not changed since the mid 1970s. By contrast the mean organic content in the basin sediments of the sound has decreased by 50% over the same period.

Eh values in the sediments of Perth's coastal waters suggest that sediments in the Cockburn Sound basin, especially along the eastern margin, and to a lesser extent the Warnbro Sound basin are more oxygen deficient compared to inshore banks and offshore sediments. In Cockburn Sound, the level of anoxia in the sediments and the distribution of sediments with oxygen deficiency has been relatively stable since the mid 1970s. In Warnbro Sound, however, the data suggest that the level of oxygen deficiency in the sediments of the deep basin has increased over the same period.

4.5.2 Contaminants in sediments

Surficial (top 20 mm) sediments were analysed for over 50 individual contaminants, including organochlorine and organophosphate pesticides, polychlorinated biphenyls, organotin compounds, aliphatic and polycyclic aromatic hydrocarbons, heavy metals and nutrients. Unless otherwise stated, all results for contaminant concentrations in sediments are expressed in terms of dry weight.

Findings

Organochlorine and organophosphate pesticides and polychlorinated biphenyls

In 1991, pesticides and polychlorinated biphenyls (PCBs) were detected in sediments at six and two sites, respectively, of the 175 sites sampled during the broadscale survey. The concentrations at most sites were very low and near the levels of detection. In the 1994 survey, pesticides were detected in sediments at approximately 30% of the 210 sites sampled. DDT was the most common organochlorine (OC) pesticide detected and occurred at 30 sites with concentrations ranging from >1 to 110 $\mu g\ kg^{-1}$. The only other OC pesticide detected was dieldrin which occurred at 16 sites, mostly within harbours and marinas, with concentrations ranging from 1 to 2 $\mu g\ kg^{-1}$. An organophosphate pesticide (OP) fenamiphos was detected at one site, near the entrance to the Peel-Harvey Estuary, and had a relatively high concentration of 150 $\mu g\ kg^{-1}$. PCBs were detected in sediments at two sites. One site in the Jervoise Bay marina had a PCB concentration of 50 $\mu g\ kg^{-1}$, and the other site, in the southern basin of Warnbro Sound, had a PCB concentration of 150 $\mu g\ kg^{-1}$.

In 1994 the majority of sites in Cockburn Sound and Owen Anchorage where DDT was detected occurred within marinas, harbours and near industrial and municipal outfalls along the eastern shoreline. DDT in the sediments of Warnbro Sound and Comet Bay was more widespread, occurring at both inshore and offshore sites.

The concentrations of DDT in sediments throughout the study area were generally very low with 85% of the sites having concentrations of ≤1 µg kg^{-1} (i.e. limit of detection). The highest concentration of DDT of 110 µg kg^{-1} occurred west of the entrance to the Peel-Harvey Estuary.

The DDT data were compared with the ERL and ERM values (see Table 3.5-5) of the draft sediment quality criteria for EQO 2 (maintenance of ecosystem integrity). About 60% of the survey sites in the Warnbro Sound/Shoalwater Islands area had detectable levels of DDT, and at more than half of these the DDT concentrations exceeded the ERL value, but the ERM value was not exceeded at any site. In Owen Anchorage, DDT was detected at about 40% of the sites and at about a third of these the ERL value was exceeded, but the ERM value was not exceeded at any site. In Cockburn Sound (excluding harbours and commercial marinas), DDT was detected at about 10% of the 69 survey sites; the ERL value was exceeded at three sites, but the ERM value was not exceeded at any site.

These data indicate that most of the marine sediments of Perth's southern coastal waters are not significantly contaminated with pesticides or PCBs. However, the distribution of DDT and dieldrin in the sediments suggests that the source of these pesticides has been from domestic and agricultural usage and that they have entered the coastal waters mainly via stormwater drains and estuarine outflows. A total ban on the use of all OC pesticides in Australia came into effect on 31 July, 1995. Currently, there are no known active inputs of DDT to these waters, however DDT is known to be highly persistent in soils and marine sediments (ANZEC, 1991).

Organotin

Detectable concentrations of the organotin compounds monobutyltin, dibutyltin and tributyltin (TBT) were found in the sediments at 97% of the 190 sites surveyed in 1994. TBT, the most toxic of the organotin compounds, was detected at more than 80% of these sites, with concentrations ranging between 0.8 to 1560 µgTBT kg^{-1}. The general distribution of TBT in the sediments in 1994 (Figure 4.5-3) was similar to the results of the 1991 survey (Simpson *et al.* 1993; Burt and Ebell, 1995). Concentrations of TBT in sediments were generally below the limit of detection (i.e.< 0.8 µgTBT kg^{-1}) in offshore sediments and in nearshore areas with minimal shipping activity, such as Comet Bay. The mean concentration of TBT in the sediments of marinas and harbours and in Cockburn Sound generally, increased significantly between 1991 and 1994 and highest concentrations occurred in areas primarily associated with the operations of large vessels (e.g. berthing and maintenance facilities). Total shipping movements in the Port of Fremantle increased by about 10% during the same period (Fremantle Port Authority, 1994). In areas, such as Rottnest Island, which are visited primarily by small (< 25m) recreational boats, the mean concentration of TBT in sediments in 1994 was not significantly different from concentrations in 1991. These data suggest that the regulations, introduced in November 1991, prohibiting the use of TBT on boats under 25 m in Western Australia have been effective in preventing further contamination of sediments in some areas.

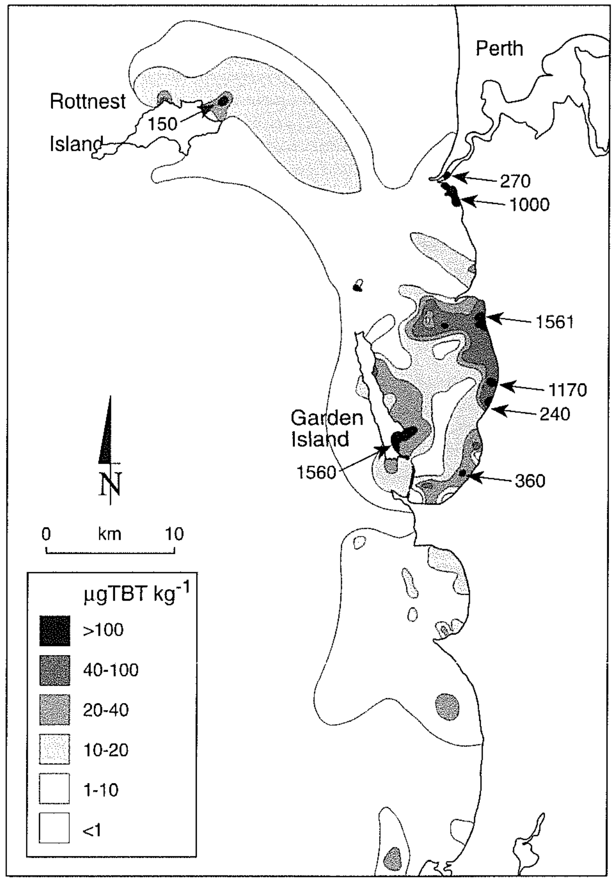


Figure 4.5-3. Tributyltin (TBT) in surficial sediments of the southern metropolitan coastal waters of Perth in 1994.

In relation to the draft criteria for EQO 2, the ERL value (Table 3.5-5) for TBT was exceeded at about 70% of the 158 sites where this substance was detected in the sediments of Perth's coastal waters. Furthermore, 33 sites had concentrations which exceed the ERM value (Table 3.5-5), and 23 sites had concentrations between 60 and 1000 µgTBT kg^{-1}, a range considered to represent moderate to high contamination according to the classification of Waite *et al.* (1991). Sites where concentrations exceeded 1000 µgTBT kg^{-1} occurred near the Alcoa Jetty in Cockburn Sound, the ship-lifting facility in Jervoise Bay and at two sites in Careening Bay. TBT was detected at over 90% of the survey sites in Cockburn Sound (excluding harbours and commercial marinas), and of these, about 75% and 20% had concentrations exceeding the ERL and ERM value, respectively.

Aliphatic and polycyclic aromatic hydrocarbons

Aliphatic hydrocarbons (C_9-C_{25}) were not detected ($\geq$ 1000 µg kg^{-1}) in sediments at any site. Polycyclic aromatic hydrocarbon (PAH) concentrations in sediments were detectable, but below the ERL value (Table 3.5-5), at all of the 205 sites sampled, with total PAH concentrations ranging from 1 to 2287 µg kg^{-1}. Naphthalene was the most frequently detected PAH and occurred at all but one site. The highest concentration of total PAHs was detected in Fremantle Harbour, however concentrations were also relatively high in the Jervoise Bay marina, Careening Bay (HMAS Stirling) and along the eastern shoreline of Cockburn Sound, particularly south of James Point. Figure 4.5-4 shows the distribution of total PAHs in the deep basin sediments of Cockburn and Warnbro Sound. Beyond these embayments the concentrations decreased and were generally less than 5 µg kg^{-1} in Comet Bay, along the western shoreline of Garden Island, in Sepia Depression, and about Rottnest Island.

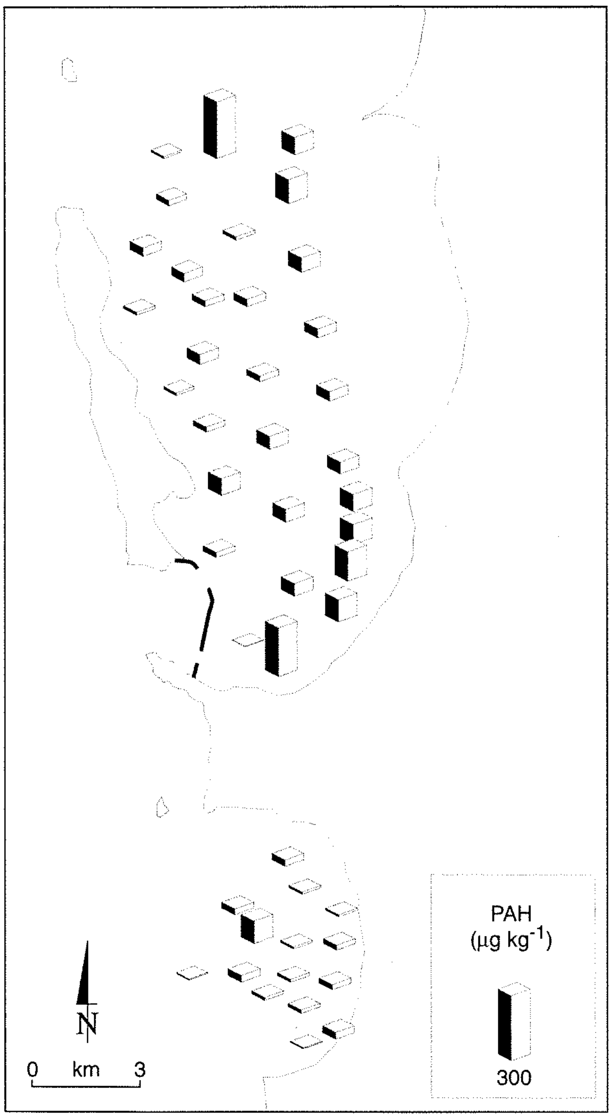


Figure 4.5-4. Polycyclic aromatic hydrocarbons (PAH) in surficial basin (> 10 m depth) sediments of Cockburn and Warnbro sounds in 1994.

There have been substantial reductions in the concentration and contractions in the distribution of aliphatic hydrocarbons in the sediments of Perth's southern coastal waters since the late 1970s (Alexander *et al.* 1979; Chegwidden, 1979). The concentrations of aliphatic hydrocarbons generally recorded in sediments from unpolluted waters elsewhere, range from 2000 to 26 000 µg kg^{-1} (Sleeter *et al.* 1980). From the SMCWS survey of 1991, the highest concentration of aliphatic hydrocarbons in sediments outside the confines of a harbour was 1000 µg kg^{-1}. In 1994 aliphatic hydrocarbons were not detected in sediments at any site in the study area.

In contrast to the reduction in aliphatic hydrocarbons, there has been no significant change in the concentration and distribution of total PAH in Owen Anchorage, Cockburn and Warnbro sounds since 1991, and trends over this period suggest that total PAH concentrations in the sediments of these embayments are increasing. In 1991, the most frequently detected PAHs were fluoranthene, pyrene and benzo(a)anthracene. These substances are generally produced as a result of the incomplete combustion of fossil fuels (Bates *et al.* 1984; Boehm and Farrington, 1984). Since 1991, there has been a change in the chemical profile of PAHs in sediments with a higher proportion of two-and three-ringed compounds (e.g. naphthalene) that are generally of petrogenic origins (e.g. diesel and fuel oil). The 1994 distribution of total PAHs in the sediments suggests that the primary sources of PAHs in the study area are from shipping activities, the outfall from the BP Refinery at James Point and runoff from industrial and municipal stormwater drains.

Although traces of PAHs are widespread in the marine sediments in Perth's southern coastal zone, the total PAH concentrations at all sites were well below the draft ERL sediment criterion for EQO 2 of 4000 µg kg^{-1} (Table 3.5-5).

Heavy metals

The survey data for heavy metal contamination in sediments are summarised in Table 4.5-1 for various sub-areas of the overall study area. Heavy metal concentrations in the sediments were generally highest near harbours, with concentrations decreasing with distance from these 'hot spots'. For most heavy metals the mean concentrations in the Cockburn Sound sediments were higher than in the other sub-areas. Mean concentrations of most heavy metals in the marine sediments of Rottnest Island, Warnbro Sound, Comet Bay and Sepia Depression were relatively low (Table 4.5-1). Excluding marinas and harbours, the mean concentrations of most heavy metals were below the ERL values (Table 3.5-5), with the exceptions of arsenic in Cockburn Sound and Warnbro Sound, and mercury in Owen Anchorage. None of these mean concentrations exceeded the ERM values.

The patterns of heavy metal distribution in the sediments of Cockburn Sound and Owen Anchorage have generally remained unchanged since the late 1970's.

Table 4.5-1. Mean heavy metal concentrations in sediments from selected localities in relation to draft criteria for the maintenance of ecosystem integrity (EQO 2; ERL and ERM) and in mussels in relation to the protection of aquatic life for human consumption (EQO 3). Standard errors are in parentheses. NC = No criterion.

Sediments ($\mu g\ g^{-1}$)											
Areas	Al	As♦	Cd	Cr	Cu	Fe	Mn	Ni	Pb	Zn	Hg♦
Owen Anchorage	1370 (55)	6 (0.3)	0.3 (0.01)	12 (0.8)	2.3 (0.13)	67 (1.8)	7.0 (0.1)	3.0 (0.08)	4.8 (0.2)	9.3 (1.0)	0.60 (0.05)
Cockburn Sound	4300 (35)	32 (0.4)	0.5 (0.01)	12 (0.2)	6.6 (0.12)	262 (2.6)	12.5 (0.1)	6.0 (0.05)	12.0 (0.1)	20.0 (0.5)	0.13 (0.01)
Warnbro Sound	2600 (78)	15 (0.7)	0.3 (0.01)	18 (2.0)	1.6 (0.04)	231 (7.0)	10.2 (0.2)	3.9 (0.10)	5.8 (0.2)	4.2 (0.2)	>0.05 (0.00)
Comet Bay	1310 (88)	7 (0.5)	0.5 (0.03)	9 (0.6)	0.8 (0.04)	136 (3.1)	11.8 (0.7)	2.7 (0.20)	3.2 (0.2)	1.9 (0.2)	0.06 (0.18)
Sepia Depression	530 (28)	2 (0.1)	0.3 (0.01)	13 (1.3)	0.6 (0.02)	85 (1.5)	9.4 (0.2)	2.1 (0.06)	3.6 (0.1)	1.6 (0.1)	0.09 (0.01)
Rottnest Island	700 (27)	2 (0.2)	0.4 (0.02)	5 (1.0)	0.6 (0.03)	28 (0.9)	3.5 (0.2)	2.6 (0.11)	3.3 (0.1)	2.3 (0.4)	<0.05 (0.00)
Harbours	4060 (128)	26 (0.9)	0.5 (0.01)	20 (1.0)	35.0 (2.11)	380 (22.5)	18.0 (0.6)	5.0 (0.11)	24 (1.0)	50.0 (2.2)	0.10 (0.01)
EQO 2 (ERL)	7140 *	8.2 #	1.2 #	81 #	34 #	NC	NC	20.9 #	46.7 #	150 #	0.15 #
(ERM)	17850 **	70.0 #	9.6 #	370 #	270 #	NC	NC	51.6 #	218 #	410 #	0.71 #

Mussels ($\mu g\ g^{-1}$)											
Area	Al	As♦	Cd	Cr	Cu	Fe	Mn	Ni	Pb	Zn	Hg♦
Owen Anchorage	3.0 (0.8)	3.7 (0.1)	<0.1 -	<0.1 -	1.1 (0.03)	12 (0.5)	0.79 (0.01)	<0.1 -	<0.5 -	33 (0.7)	<0.25 -
Cockburn Sound	4.8 (0.7)	4.3 (0.1)	<0.1 -	<0.1 -	1.0 (0.01)	20 (0.7)	0.79 (0.01)	<0.1 -	<0.5 -	32 (0.2)	<0.25 -
Warnbro Sound	3.4 (0.5)	9.6 (1.0)	<0.1 -	<0.1 -	0.8 (0.00)	16 (0.4)	0.79 (0.02)	<0.1 -	<0.5 -	28 (1.1)	<0.25 -
Harbours	5.9 (0.3)	4.5 (0.1)	<0.1 -	<0.1 -	2.1 (0.16)	20 (0.9)	0.96 (0.02)	<0.1 -	<0.5 -	44 (1.8)	<0.25 -
EQO 3	NC	NC•	2.0	NC	70	NC	NC	NC	2.5	150	0.5

Long *et al.* (1995)
* Based on twice background concentration (Pollution Research Pty Ltd, 1994) in Warnbro Sound sediments >10 m depth
** Based on five times background concentration (Pollution Research Pty Ltd, 1994) in Warnbro Sound sediments >10 m depth
♦ Total
• No criteria for total arsenic concentration

In Cockburn Sound heavy metal concentrations were generally highest in the southern half of the Sound, as illustrated in Figure 4.5-5 for the case of lead, and along the eastern margin next to the Kwinana industrial area. The distribution patterns of most heavy metals in the sediments of Cockburn Sound were similar (Figure 4.10-3), suggesting a common source or sources for many of these metals. Currently, the major industrial sources of heavy metals to these waters are CSBP & Farmers Ltd. and, to a lesser extent, BP Refinery (Kwinana) Ltd. and TiWest Joint Venture. The relatively high concentrations of copper, nickel, lead, zinc and mercury in sediments in the northeast of Cockburn Sound suggests that this area is a source of heavy metals to these waters. Residual contamination from the decommissioned (1984) WAWA domestic wastewater outfall, south-west of Woodman Point, may also be contributing to the concentrations of these metals in this area (Figure 4.10-3). In Owen Anchorage the sediments near the Explosives Jetty have relatively high heavy metal concentrations, especially of mercury (Figure 4.5-6) and arsenic. The source of this contamination is unknown. In Warnbro Sound the highest concentrations of heavy metals in sediments generally occurred in the north-east of the basin, near a storm water drain. Chromium concentrations were also relatively high at several sites in the southern ends of Cockburn and Warnbro sounds (Figure 4.5-7). Although there is no obvious explanation for the higher concentrations of chromium in these localities, similar results were also recorded in 1989 by Monk and Murray (1991). In Comet Bay the concentrations of aluminium, nickel, manganese, cadmium and chromium in sediments were highest at sites near the mouth of the Peel-Harvey Estuary suggesting that the estuary is a source of these metals.

Comparisons of mean heavy metal concentrations in the sediments of Cockburn Sound between 1977 and 1994, using sites common to these surveys, show there have been decreases in the mean concentrations of chromium, copper, iron, lead and zinc since the late 1970s (Figure 4.5-8a). A comparison of the annual loadings of these metals to the Sound over the same period suggests that these downward trends generally coincided with substantial reductions in direct loadings of these metals to these waters (see section 4.4). In contrast, the mean concentration of nickel in the sediments of Cockburn Sound has increased since 1977, coinciding with increased loadings of nickel to the Sound over the same period. The mean concentrations of cadmium and manganese in sediments of Cockburn Sound have changed little since 1977. Loadings of cadmium to the Sound over the same period have fluctuated greatly, however current loadings are similar to the early 1990s. Historical records of manganese loadings to these waters do not exist.

Figure 4.5-5. Lead in surficial basin (> 10 m depth) sediments of Cockburn and Warnbro sounds in 1994.

Comparisons of mean heavy metal concentrations in the sediments of Owen Anchorage between 1977 and 1994 show trends similar to Cockburn Sound with significant decreases in the mean concentrations of most metals since the late 1970s, with the exception of nickel, which increased slightly over that period (Figure 4.5-8b). The downward trend in concentrations of chromium and lead in the sediments coincided with substantial reductions in direct loadings of these metals to these waters. However, the lower copper and zinc concentrations are more difficult to explain as estimates of the annual loadings for these metals to Owen Anchorage were greater in 1994 than in 1977. There are no historical records of nickel loadings to these waters to account for the slight increase in nickel contamination detected in the latest survey. The mean concentrations of cadmium and manganese in the sediments of Owen Anchorage have not changed significantly since 1977 which, in the case of cadmium, is consistent with the similar loadings during the two periods. Historical records of manganese loadings to these waters do not exist.

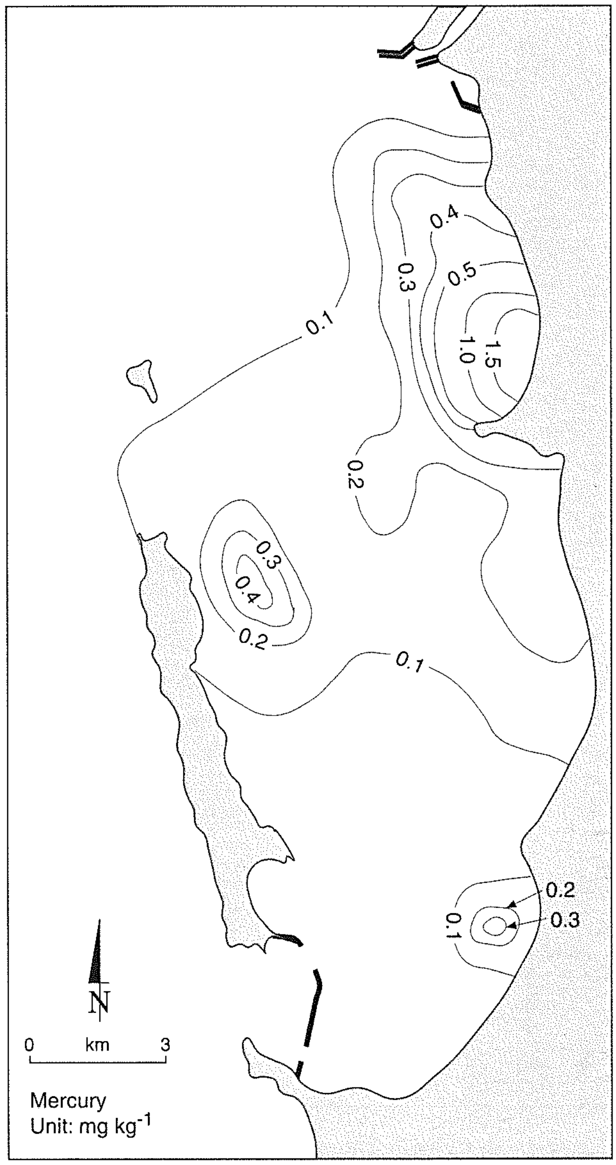


Figure 4.5-6. Mercury in surficial sediments of Owen Anchorage and Cockburn Sound in 1994.

In Warnbro Sound the mean concentrations of copper, zinc, manganese, lead and iron in the sediments decreased between 1977 and 1994. The mean concentrations of chromium and nickel were unchanged over the same period. There are no industrial or municipal point source discharges into Warnbro Sound (Muriale and Cary, 1995).

In general the concentrations of heavy metals in the sediments of Perth's southern coastal waters in 1994 were well below draft sediment criteria (ERL values) for EQO 2 (Table 3.5-5). However, the ERL value for arsenic was exceeded at about 70% of the sites in Cockburn Sound (excluding marinas and harbours), and at about 15% of these, the ERM value was exceeded. Mercury concentrations in sediments at a few sites near the CSBP outfall, and at sites distributed broadly across the northern portion of Cockburn Sound, exceeded the draft ERL criterion for EQO 2, but there were no broadscale exceedances of the ERM value.

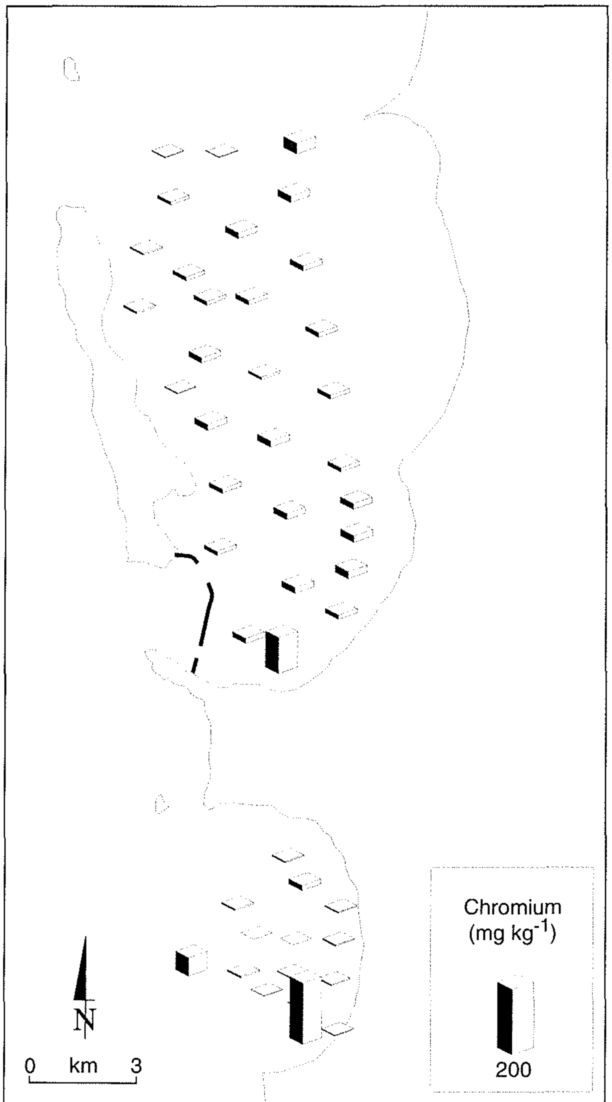


Figure 4.5-7. Chromium in surficial basin (> 10 m depth) sediments of Cockburn and Warnbro sounds in 1994.

Furthermore, aluminium concentrations in sediments at some sites in the Sound, particularly in the vicinity of the Alcoa Jetty, exceeded the draft criterion (the ERL value, but not the ERM value) for EQO 2. The concentration of mercury in sediments at about half of the sites in eastern Owen Anchorage, and arsenic at sites near the Explosives Jetty, exceeded the draft ERL criteria for EQO 2. The concentrations of arsenic in the sediments at most sites in the Warnbro Sound basin, and of chromium at one site in the Sound, exceeded the draft ERL criteria for EQO 2, but did not exceed the ERM values.

Nutrients

In 1991, the concentrations of total nitrogen (TN) and total phosphorous (TP) were determined in surficial sediments at 47 sites throughout the study area, including the offshore areas of Sepia Depression and Rottnest Island. The TN concentrations in sediments ranged from 80 to 2150 $\mu g\ g^{-1}$ (dry weight). The TN concentrations in the fine, organic rich sediments of the deep basins of Cockburn Sound and Warnbro Sound were higher (> 1000 $\mu g\ g^{-1}$) compared to the coarser carbonate sediments in the shallow inshore lagoons and the offshore areas (< 350 $\mu g\ g^{-1}$). The mean TN concentration in the basin sediments of Cockburn Sound (1000 $\mu g\ g^{-1}$) was higher than Warnbro Sound (650 $\mu g\ g^{-1}$). The mean TN concentration in basin sediments from the southern area of Cockburn Sound was higher (1100 $\mu g\ g^{-1}$) than the northern area (950 $\mu g\ g^{-1}$). Total phosphorous (TP) concentrations in the sediments of the study area ranged from 35 to 730 $\mu g\ g^{-1}$ (dry weight). TP concentrations were generally similar throughout the study area (350-500 $\mu g\ g^{-1}$) except for the lower concentrations in the sediments in the nearshore area between City Beach and Fremantle (35-280 $\mu g\ g^{-1}$). Outside this area, the differences in mean concentrations between inshore and offshore sediments or between banks and the deep basins were small. Mean TP concentrations were similar in the basin sediments of Cockburn (450 $\mu g\ g^{-1}$) and Warnbro sounds (420 $\mu g\ g^{-1}$) and there was no difference in mean TP concentrations between the northern and southern areas of the Cockburn Sound basin.

The spatial distribution of TN in the sediments of Cockburn Sound in 1991 was similar to that found in 1978 (Chiffings, 1987). Mean TN concentrations in the sediments of the Cockburn Sound (2140 $\mu g\ g^{-1}$) and Owen Anchorage (400 $\mu g\ g^{-1}$) in 1978 were more than twice the concentrations of TN in sediments from these areas in 1991. In the southern area of the Cockburn Sound basin the mean TN concentration in the sediments was more than three times higher in 1978 compared to 1991. The mean TP concentration in the sediments of Cockburn Sound decreased by more than 30% between 1978 and 1991. By contrast, in Owen Anchorage the mean TP concentration increased by 20% over the same period.

In the coastal sediments off Perth there is a general trend of increasing nitrogen and phosphorus concentrations from Ocean Reef, approximately 30 km north of Perth, south to Cockburn Sound (Rosich *et al.* 1994). The spatial distribution of nutrients in the sediments of Cockburn Sound, especially the higher concentrations in the southern basin, have been linked to the accumulation of detrital material, primarily from phytoplankton blooms stimulated by industrial loadings of nitrogen to these waters (Chiffings, 1987). Between 1978 and 1991, there were substantial reductions in total nutrient inputs to Cockburn Sound (see section 4.4) and over the same period the concentrations of TN and TP in the sediments of this area have decreased.

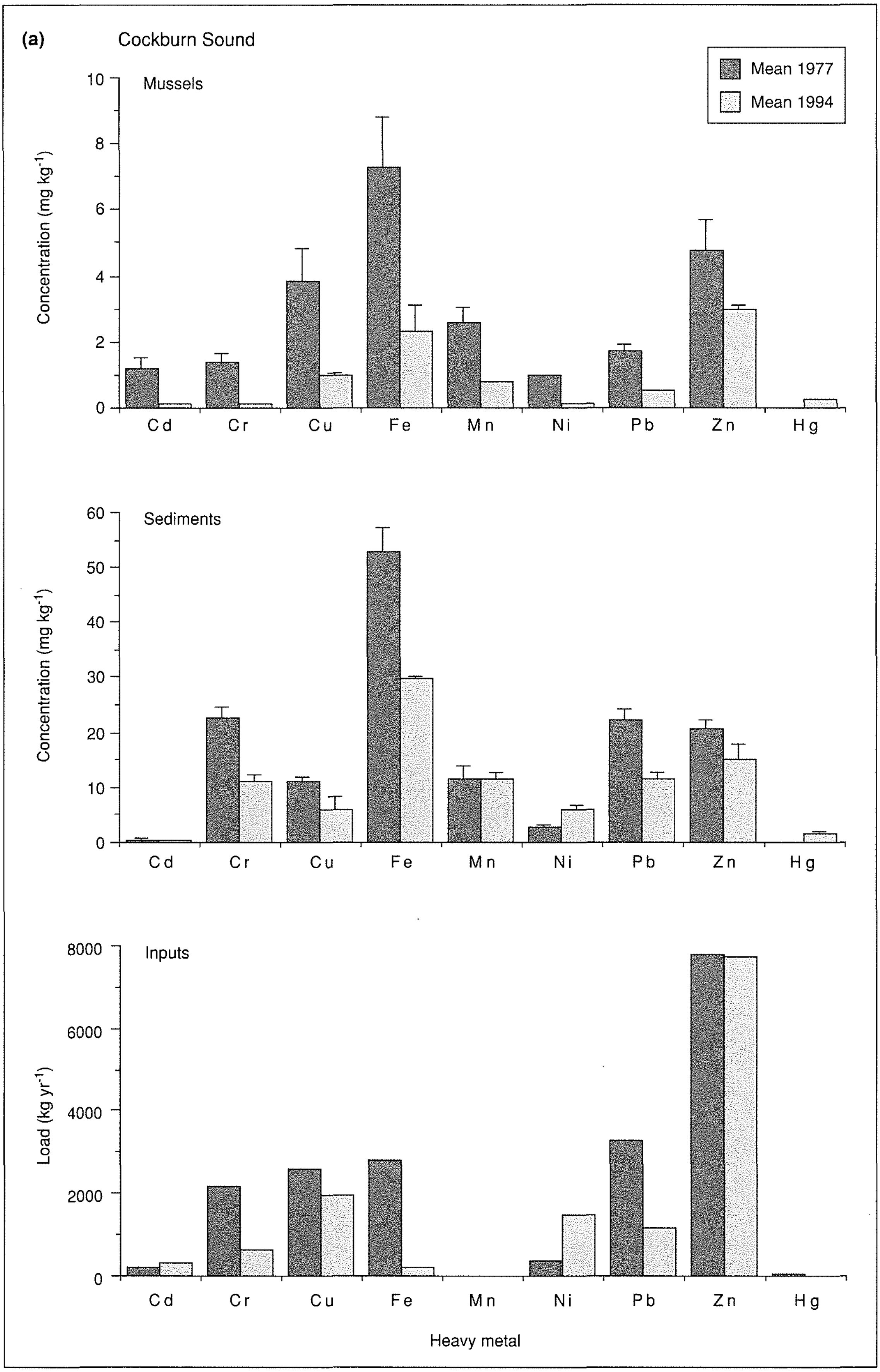
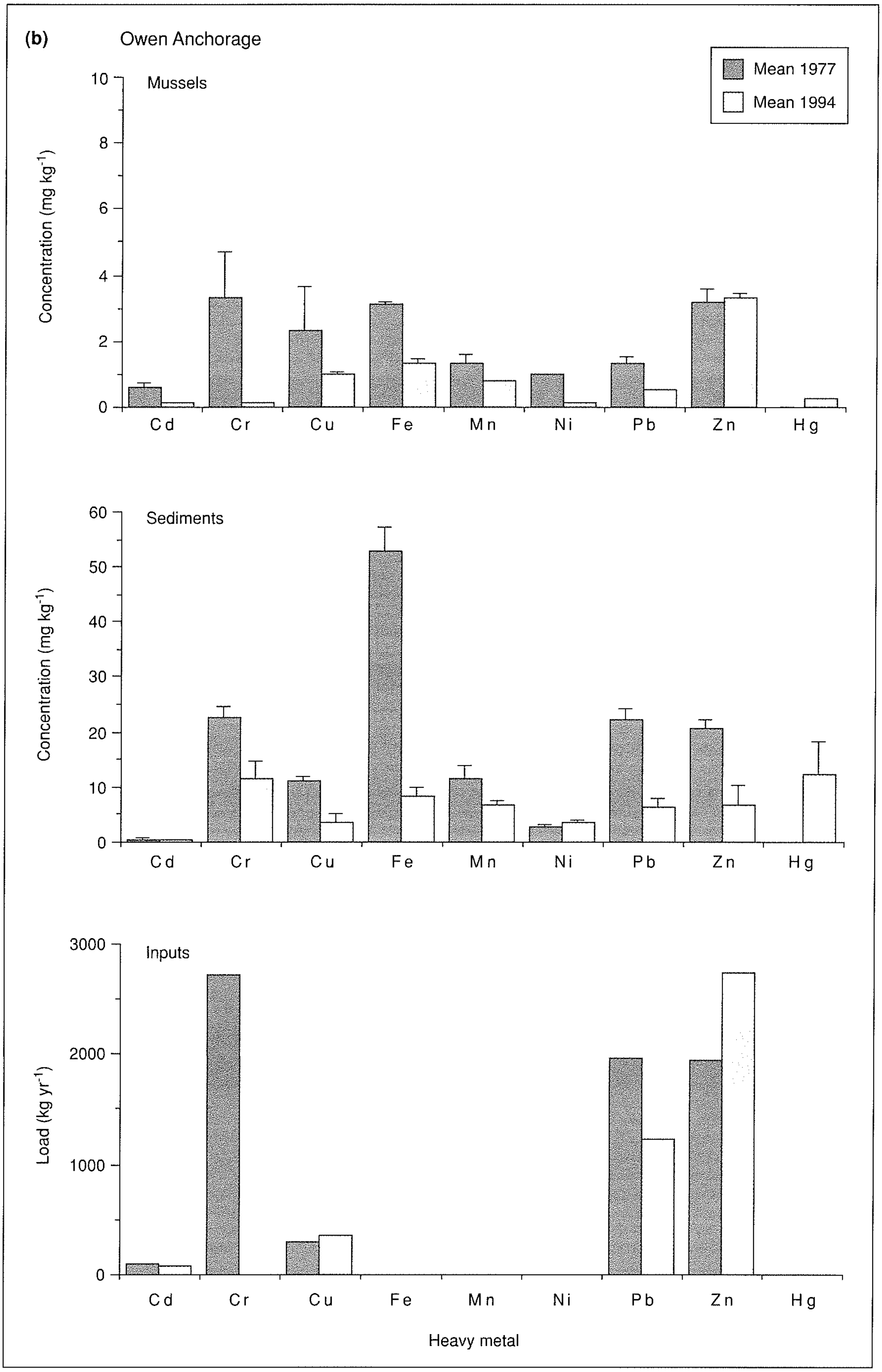


Figure 4.5-8. Comparisons of mean concentrations of heavy metals between 1977 and 1994 in mussel tissue and surficial sediments of (a) Cockburn Sound and (b) Owen Anchorage, and annual loads of heavy metals discharged to these waters (Fe and Zn concentrations in mussels were divided by 10 and concentrations of Fe and Hg in sediments were divided by 100 and multiplied by 10 respectively. Fe load to Cockburn Sound was divided by 100 and Cd and Hg loads to Owen Anchorage were multiplied by 10 and 100 respectively).


(b)
Owen Anchorage
Mean 1977
Mean 1994
Mussels
Concentration (mg kg-1)
10
8
6
4
2
0
Cd
Cr
Cu
Fe
Mn
Ni
Pb
Zn
Hg
Sediments
Concentration (mg kg-1)
60
50
40
30
20
10
0
Cd
Cr
Cu
Fe
Mn
Ni
Pb
Zn
Hg
Inputs
Load (kg yr-1)
3000
2000
1000
0
Cd
Cr
Cu
Fe
Mn
Ni
Pb
Zn
Hg
Heavy metal

Conclusions

The results of the contaminant surveys of the sediments in the southern metropolitan coastal waters of Perth indicate that, in general, this area is not significantly contaminated with pesticides, PCBs or hydrocarbons. Although polycyclic aromatic hydrocarbons are present in the sediments throughout Cockburn Sound, the levels recorded suggest that these contaminants, in isolation, are unlikely to be having a major ecological impact on the flora and fauna of the Sound.

In relation to draft EQO 2 (i.e. maintenance of ecosystem integrity), tributyltin concentrations in sediments exceed the ERL and ERM criterion values at approximately 70% and 18%, respectively, of the sites throughout the study area. The highest concentrations of TBT occur in harbours, boat mooring areas and near industrial wharves along the eastern shoreline of Cockburn Sound. Concentrations of TBT in the sediments of Perth's southern waters are among the highest in Australia.

TBT was the principal biocide used in anti-fouling paints on the hulls of boats and ships until the introduction of legislation, effective in Western Australia from November 1991, prohibiting the use of organotin-based anti-fouling paints on boats under 25 m, and restricting use to low-leaching paints on vessels over 25 m. Changes in the distribution patterns of TBT concentrations in sediments between 1991 and 1994 suggest that the TBT restrictions have generally been effective in minimising further contamination in areas used predominantly by small recreational boats (e.g. Rottnest Island). However the results of these surveys indicate that concentrations of TBT in sediments have continued to increase in areas associated with the use and maintenance of large vessels suggesting that the current source of TBT to the sediments of Perth's southern coastal waters is principally from the hulls of commercial ships, some of which is not subject to State legislative restrictions. This conclusion is similar to the results of comparable studies overseas (IMO, 1994; Minchin *et al.* 1995).

The high concentrations of TBT in the sediments of Perth's coastal waters and the high frequencies of the reproductive disorder imposex found in the mollusc *Thais orbita* throughout the study area, and linked to TBT contamination (see section 4.11), is cause for major concern and further emphasises the urgent need to address TBT contamination in these waters.

Regulations controlling licensable activities were amended recently (September 1996) to include ship-building and ship-maintenance facilities which use or remove organotin compounds. The DEP is identifying those premises and will progressively license them to ensure that waste materials containing organotin are disposed of in an environmentally acceptable way.

Heavy metal concentrations in the sediments of the study area are generally highest in Cockburn Sound and Owen Anchorage, particularly near harbours, with concentrations decreasing with distance from these 'hot spots'. Concentrations of most heavy metals in the sediments of Rottnest Island, Warnbro Sound, Comet Bay and Sepia Depression are relatively low. In Cockburn Sound the concentrations of heavy metals are generally highest in the southern half of the Sound and along the eastern margin, next to the Kwinana industrial area, reflecting the location and proximity to the major current and historical industrial sources of heavy metals to these waters. In Owen Anchorage the sediments near the Explosives Jetty have relatively high heavy metal concentrations, especially of mercury and arsenic.

In Cockburn Sound and Owen Anchorage the mean concentrations of most heavy metals have decreased significantly since the late 1970s as a result of substantial reductions in direct discharges of heavy metals to these waters. This trend suggests that management measures implemented during this period, such as relocation of the Woodman Point domestic wastewater outfall to Sepia Depression and improved industrial wastewater treatment processes, have generally been effective in reducing the concentrations of heavy metals in sediments of Cockburn Sound and surrounding waters.

In general the concentrations of heavy metals in the sediments of Perth's southern coastal waters in 1994 are well below draft sediment criteria (ERL values) for EQO 2. In Cockburn Sound however, the concentration of arsenic in the sediments exceeded the draft ERL and ERM criteria for EQO 2 at about 70% and 10% of the sites, respectively. Mercury concentrations in sediments at a few sites near the CSBP outfall, and at sites distributed broadly across the northern portion of Cockburn Sound exceeded the draft ERL criterion for EQO 2. Furthermore, aluminium concentrations in sediments at some sites in the Sound, particularly in the vicinity of the Alcoa Jetty, exceeded the draft ERL criterion for EQO 2. In eastern Owen Anchorage, the concentration of mercury in sediments at the majority of sites, and arsenic at sites near the Explosives Jetty, exceeded the draft ERL criterion for EQO 2. In the basin (> 10 m) of Warnbro Sound, particularly the northern area, the concentrations of arsenic in sediments generally exceeded the draft ERL criterion for EQO 2, but did not exceed the ERM value.

The ecological significance of these findings needs to be determined as a matter of priority; the development of sediment quality criteria to be undertaken as part of the consultative EQO/EQC finalisation process (see section 3.6) will help address this issue.

4.6 Contaminants in biota

Some marine animals, particularly filter-feeders, bioaccumulate contaminants that occur in seawater, and therefore contaminant concentrations in their tissues provide a time-integrated measure of the concentrations of these substances in the water column. Using filter-feeders in this way overcomes the inherent difficulties in achieving reliable results from analysing these substances in seawater. Contaminants such as hydrocarbons and heavy metals generally occur in seawater at very low concentrations and their spatial distributions are often highly variable with time. This variability, coupled with the lack of reliable, rapid and high resolution *in situ* measurement techniques for many contaminants, and the general high cost of trace analyses, severely limits the use of seawater contaminant concentrations for monitoring trends. The blue mussel, *Mytilus edulis*, is common throughout Cockburn Sound and Owen Anchorage, and concentrations of contaminants in the tissue of this filter-feeder have been used in the SMCWS as an indicator of the recent history of contaminant inputs to these waters. An additional benefit is that the results also provide an indication of the level of human health risk associated with the consumption of seafood. This species is harvested for human consumption both from wild stock and from commercial 'mussel farms'. Since *Mytilus edulis* has been used in contaminant studies in coastal waters at numerous other locations around the world, the scientific literature provides a wealth of data for comparison with the findings of the SMCWS mussel surveys.

Mussels were collected from 34 sites in November 1991 and 54 sites in February 1994 and analysed for over 50 individual contaminants, including organochlorine and organophosphate pesticides, polychlorinated biphenyls, organotin compounds, aliphatic and polycyclic aromatic hydrocarbons, and heavy metals (1994 only). Mussels were collected from 1 m below low water as a previous comparison of contamination in mussels collected from the top and bottom metre of their vertical distribution indicated that there was no significant depth effect on contaminant concentrations in these waters (Burt and Scrimshaw, 1993). The 1994 survey was undertaken firstly, to provide more detailed information in the areas of highest contamination identified in the 1991 survey, secondly, to determine short-term trends for assessing the effectiveness of management initiatives, such as the introduction of restrictions on the use of organotin-based anti-fouling paints, and thirdly, to further resolve current sources of contamination.

The surveys were conducted in collaboration with the Chemistry Centre of Western Australia, with most of the analyses being undertaken at these laboratories. Organotin analyses were undertaken by the CSIRO Centre for Advanced Analytical Chemistry at Lucas Heights, New South Wales (Batley *et al.* 1988). All results for contaminant concentrations in mussels are expressed here in terms of wet weight.

In relation to the suitability of seafood for human consumption, the mussel contamination data were compared with the Australian Food Standard A12 and other published guidelines (see Table 3.5-6). Furthermore, Chegwidden (1979) proposed a set of values for heavy metal concentrations in mussels for Cockburn Sound that he considered would be indicative of 'contamination'. These values (see section 3.5), together with values for mercury (Neilson and Natham, 1975) and tributyltin (Page and Widdows, 1991), were used as reference points against which the 1994 mussel contamination data were compared to identify areas that could be considered to be 'contaminated'.

Unless otherwise stated, this section refers to the results of the 1994 survey (Burt *et al.* 1995d). The results of previous broadscale surveys in 1977 (Chegwidden, 1979) and 1991 (Burt *et al.* 1993b) were used to assess trends since the mid 1970s. Temporal comparisons have utilised paired site data from the surveys. The results of the 1989 survey of heavy metals in mussels in Cockburn Sound and surrounding waters can be found in Burt and Scrimshaw (1993). Further technical details on analytical methodology and quality assurance protocols for the 1991 and 1994 surveys can be found in Burt and Ebell (1994) and Burt and Ebell (1995).

Findings

Organochlorine and organophosphate pesticides and polychlorinated biphenyls

In 1991, organophosphate pesticides (OP) and polychlorinated biphenyls (PCBs) were not detected in mussels at any site in the study area and organochlorine pesticides (OC) were detected at two sites adjacent to the Fremantle small boat harbours. In the 1994 survey, organophosphate pesticides and polychlorinated biphenyls were not detected in any mussel sample. The only organochlorine pesticide detected in the tissue of mussels was DDT, which was found at 20 of the 54 sites, but was above the level of detection at only 9 of the sites sampled. These sites were situated along the shoreline of Owen Anchorage and the northern end of Cockburn Sound, particularly near the Jervoise Bay marina. The highest DDT concentrations (up to 4 $\mu g\ kg^{-1}$) were found in the Fremantle Fishing Boat harbour and near the Explosives Jetty in Owen Anchorage. DDT was not detected in mussels from Warnbro Sound.

All OC compounds have been banned in Western Australia for agricultural use since 1987 (ANZEC, 1991). Since then, restricted use of aldrin, chlordane and heptachlor was approved for termite control. However, a total ban on the use of all OC pesticides in Australia came into effect on 31 July, 1995. The use of PCBs has been restricted in Australia since 1975 (ANZEC, 1991).

Currently, there are no known inputs of DDT to these waters. However, DDT is highly persistent in marine sediments (ANZEC, 1991) and some of the highest concentrations of DDT in sediments in 1994 occurred in small boat harbours along the mainland shore (see section 4.5), suggesting that residual contamination of these sediments, and inputs of residual DDT through drainage discharging into these areas, are the most likely sources of DDT contamination in mussels.

According to Riseborough *et al.* (1983), DDT concentrations in the tissue of mussels in uncontaminated waters range between 1 to 10 μg kg^{-1} (wet weight). ANZECC (1992) published a guideline of 500 μgDDT kg^{-1} for the Protection of Food for Wildlife in relation to toxicants that can accumulate along the foodchain. On the basis of these values, mussels in Perth's southern coastal waters are not significantly contaminated with this pesticide.

Organotin

Organotin compounds were detected in the tissue of mussels at 42 of the 43 sites sampled. Tributyltin (TBT), the most toxic of these compounds, was detected at 40 sites and concentrations ranged from < 0.8-732 μgTBT kg^{-1} (wet weight). The highest concentration of TBT in mussels was found in the Fremantle Fishing Boat Harbour. Relatively high concentrations (> 200 μgTBT kg^{-1}) also occurred near the Explosives Jetty and the Cockburn Cement Jetty in Owen Anchorage, within the marina and boat harbour in Jervoise Bay, near the industrial wharves along the eastern side of Cockburn Sound, and in the vicinity of the naval facilities at Careening Bay.

In 1994, the concentrations of TBT in mussels from Perth's coastal waters had a distribution similar to the pattern in 1991, described by Burt and Ebell (1995). High concentrations of TBT in mussels were found in areas with relatively high shipping activity and near boat maintenance facilities. Excluding sites in boat harbours and marinas, the mean concentrations of TBT in mussels from Cockburn Sound and Owen Anchorage increased significantly over the 1991-1994 period.

The concentrations of TBT in mussels in Perth's southern coastal waters are high in comparison with the results of similar Australian and overseas studies (Burt and Ebell, 1995) indicating that these waters, particularly Cockburn Sound and Owen Anchorage, are significantly contaminated with organotin compounds. The concentrations of TBT in mussels at approximately 40% of the sites are at levels (> 0.1 μgTBT kg^{-1} wet weight) likely to cause physiological stress in mussels (Page and Widdows, 1991). These results, in addition to the widespread occurrence of imposex in *Thais orbita* in the study area, which has been linked to TBT contamination (Field, 1993), suggest that the current input of TBT to Perth's coastal waters is having a significant impact on local marine biota (see section 4.11). Furthermore, the concentrations of TBT in mussels from 14 sites, primarily in harbours/marinas and near the industrial wharves along the eastern shoreline of Cockburn Sound, exceed levels recommended by the World Health Organisation (146 μgTBT kg^{-1} wet weight) for the safe consumption of food containing this substance (WHO, 1990), indicating that the consumption of mussels from these sites poses a risk to human health.

Aliphatic and polycyclic aromatic hydrocarbons

In 1994, aliphatic hydrocarbons (C_9-C_{25}) were not detected in mussels collected from any of the 54 survey sites. Polycyclic aromatic hydrocarbons (PAHs) were detected in mussels from all 49 sites and concentrations ranged from 2 to 81 μg kg^{-1} (wet weight). Naphthalene, the most frequently detected PAH, occurred at all sites. The total concentrations of PAHs in mussels were highest in the Fremantle Fishing Boat Harbour and were relatively high in the Jervoise Bay marina, Careening Bay and the eastern margin of Cockburn Sound.

Risebrough *et al.* (1983) suggested that the concentrations of aliphatic hydrocarbons in mussels from uncontaminated waters should range between 800 and 2000 μg kg^{-1}. Since the late 1970s the concentrations of aliphatic hydrocarbons in mussels from Perth's coastal waters have decreased dramatically (Alexander *et al.* 1979; Chegwidden, 1979). By 1991, concentrations of aliphatic hydrocarbons were found in mussels at only two sites in the study area, on the eastern shoreline of Cockburn Sound. In 1994, mussels from Perth's southern coastal waters were not contaminated with aliphatic hydrocarbons.

In 1991, low concentrations of PAHs were found in mussels from 27 sites (87%) spread across the study area. The highest total concentration of PAHs was 12 μg kg^{-1} and concentrations greater than 1 μg kg^{-1} occurred at only two other sites. Naphthalene was again the most frequently detected PAH.

The general distribution of total PAH concentrations in mussels from the study area remained unchanged between 1991 and 1994 with the highest concentrations found along the eastern shoreline of Cockburn Sound, particularly next to the Kwinana industrial area. Over the same period there was a slight increase in total PAH concentrations in mussels.

Rainio *et al.* (1986) suggested that total PAH concentrations in mussels from uncontaminated waters should range from about 50 to 140 μg kg^{-1}. In 1994, the concentrations of total PAHs in mussels exceeded 50 μg kg^{-1} at only one site in Perth's southern coastal waters, suggesting that mussels from these waters are not significantly contaminated with PAHs. The most frequently detected PAH in 1991 and 1994 was naphthalene, suggesting that PAH contamination of mussels from these waters is of petrogenic origins such as diesel and fuel oil (Bates *et al.* 1984; Boehm and Farrington, 1984). The spatial distribution of PAHs in mussels in the study area suggests that the primary source is associated with shipping activities, particularly in and near to the Fremantle Harbour and the adjacent small boat harbours, Careening Bay and along the eastern shoreline of Cockburn Sound.

Heavy metals

Heavy metal concentrations in mussels in 1994 were generally low throughout the study area. Concentrations of cadmium, chromium, lead, nickel and mercury (total and organic) were below the limits of detection at all sites in the study area. The concentrations of aluminium, arsenic (total), copper, iron, manganese and zinc were generally highest in the vicinity of harbours/marinas and to a lesser extent along the eastern shorelines of Cockburn Sound and Owen Anchorage.

In 1977, the concentrations of heavy metals in mussels were generally highest in Cockburn Sound, especially along the eastern shoreline next to the Kwinana industrial area and, to a lesser extent, along the eastern shoreline of Owen Anchorage near the Coogee industrial area (Chegwidden, 1979). Since then, there have been substantial changes in the distribution of heavy metals in the study area, reflecting the major reductions in loadings of most metals to these waters from industrial and municipal point sources along these eastern shorelines. In 1989, there were no significant differences in the mean concentrations of most heavy metals in mussels between Cockburn Sound and Owen Anchorage (Burt and Scrimshaw, 1993). By 1994, the mean concentrations of most heavy metals in mussels from Owen Anchorage, Cockburn Sound and Warnbro Sound (based on limited data) were similar (Table 4.5-1). The exceptions include relatively high concentrations of zinc along Coogee Beach in Owen Anchorage, iron along the eastern shoreline of Cockburn Sound and arsenic in Warnbro Sound.

Comparisons of mean heavy metal concentrations in mussels calculated using the same survey sites in Cockburn Sound and Owen Anchorage in 1977 and 1994 indicate there have been significant decreases in the mean concentrations of cadmium, chromium, copper, iron, manganese, nickel and lead in mussels from both these areas, and of zinc in Cockburn Sound (Figure 4.5-8). A comparison of the annual heavy metal loadings to the Sound over this period indicates that the downward trend in concentrations in mussels coincided with a substantial reduction in the direct loading of these metals to these waters. Notable exceptions were zinc and nickel. Increased zinc loads to Owen Anchorage were reflected in increased concentrations of zinc in mussels whereas, although nickel loads to Cockburn Sound have increased in recent years, concentrations in mussels have decreased.

In 1977, the concentrations of copper, zinc, cadmium, manganese, lead, chromium and iron in mussels from most sites in Cockburn Sound, and the concentrations of cadmium, lead and chromium at most sites in Owen Anchorage, exceeded values proposed by Chegwidden (1979) as indicative of 'contamination'. In 1989, the concentrations of zinc and chromium, and to a lesser extent copper, exceeded these indicative values at most sites along the eastern shoreline of Cockburn Sound. In 1994, the concentrations of heavy metals in mussels, apart from zinc, were below the indicative values throughout the study area. Concentrations of zinc exceeded the values proposed by Chegwidden (1979) at the majority of sites in Cockburn Sound and eastern Owen Anchorage.

In 1977, the concentrations of cadmium and lead in mussels at several sites along the eastern shoreline of Cockburn Sound exceeded the levels recommended by the Western Australian Department of Health for metals in seafood and the draft criteria for EQO 3 (i.e. the maintenance of aquatic life for human consumption). The concentrations of zinc at some of these sites, and chromium at several sites in Owen Anchorage, were also cause for concern (Chegwidden, 1979). In 1989 and 1994 the concentrations of heavy metals in mussels were below the draft criteria for EQO 3 at all sites in the study area.

Conclusions

Organophosphate pesticides, polychlorinated biphenyls and aliphatic hydrocarbon concentrations in mussels are below the limit of detection at all sites in the study area. Traces of DDT, the only organochlorine pesticide detectable in the tissue of mussels, are present at some sites in Owen Anchorage and Cockburn Sound. The highest concentrations of DDT are detected in areas within or immediately next to harbours, marinas and jetties. Concentrations of these substances are well within the range found in mussels in similar Australian and overseas studies for areas considered to be uncontaminated. The low concentrations of DDT in mussels, coupled with the nationwide ban on all OC pesticides, suggests that management action has been effective in addressing the issue of DDT contamination in Perth's southern coastal waters.

Since the late 1970s the mean concentrations of most heavy metals in mussels from Cockburn Sound and Owen Anchorage have decreased substantially. The concentrations of cadmium, chromium, lead, nickel and mercury in mussels in 1994, are below the limits of detection at all sites in the study area. The concentrations of aluminium, arsenic, copper, iron, manganese and zinc in mussels are generally highest near harbours/marinas and to a lesser extent along the eastern shorelines of Cockburn Sound and Owen Anchorage. Concentrations of zinc exceed the values proposed by Chegwidden (1979) as indicative of 'contamination' at the majority of sites in Cockburn Sound and Owen Anchorage. The concentrations of heavy metals surveyed in 1994 are below the draft criteria for EQO 3 (i.e the maintenance of aquatic life for human consumption) at all sites sampled.

Mussels with high concentrations of organotin compounds, particularly TBT, are widespread throughout the study area with the highest concentrations generally occurring in harbours, boat mooring areas and along the eastern shoreline of Cockburn Sound. The concentration of TBT in mussels at 92% of the sites exceed the levels proposed by Page and Widdows (1991) to indicate possible contamination and 40% of the sites are at levels likely to cause physiological stress in mussels. At 14 sites, primarily in the harbours and wharves in Cockburn Sound, the concentrations of TBT in mussels exceed the draft criterion for EQO 3, suggesting that the consumption of mussels from these sites, at least, is a human health risk. TBT was the principal biocide used in anti-fouling paints on the hulls of boats and ships until the introduction of legislation, effective in Western Australia from November 1991, prohibiting the use of organotin antifouling paints on boats under 25 m, and restricting use to low-leaching paints on vessels over 25 m. The distribution of TBT in mussels generally agrees with the findings of the sediment surveys (see section 4.5) and supports the conclusion that the primary source of this contamination is associated with the use and maintenance of large vessels. The high frequency of the reproductive disorder *imposex* that has been found in the intertidal whelk *Thais orbita* throughout the study area (see section 4.11) and, which has been linked to TBT contamination, is cause for major concern and further emphasises the urgent need to address this problem.

4.7 Water quality

The physical, chemical and biological quality of the coastal waters off Perth is important in relation to the maintenance of environmental values, such as the ecological integrity of local ecosystems, recreational pursuits (e.g. swimming, diving and fishing) and commercial activities (e.g. professional fishing, aquaculture and ecotourism). All of these values depend on water quality being maintained at or near 'natural' levels. To assess the effects of anthropogenic point- and diffuse-source inputs of nutrient contaminants on ambient water quality it is necessary, firstly, to characterise 'natural' or background water quality levels, and secondly, to understand the relative importance of regional and local scale factors that influence these conditions.

Satellite images of sea surface temperature and colour clearly indicate that river outflows and the Leeuwin Current are seasonally important shelf-scale influences on the quality of Perth's coastal waters (Simpson *et al.* 1993). Industrial and domestic wastewater discharges, surface runoff and contaminated groundwater inflows are important influences at a basin-scale (Martinick *et al.* 1993).

4.7.1 Background conditions

The seaward boundary of coastal waters off the Perth region can be arbitrarily defined as the distance offshore to which the currents are wind-dominated. To the west of this boundary is a zone where the Leeuwin Current (Cresswell and Golding, 1980) is commonly encountered, flowing southward from the tropics throughout the year in opposition to the predominantly northward wind stress. In summer, when the mean northward wind stress is strong, a wind-driven northward flow occupies most of the continental shelf off Perth (Cresswell, 1991). When the mean northward wind stress weakens, for example in winter, the seaward boundary of the coastal waters moves eastward toward the coast (Figure 6.2-2; Cresswell, 1991; Mills *et al.* 1996). Hence, the boundary of the coastal waters is located typically 10-30 km offshore in winter and 30-50 km offshore in summer. In terms of bathymetry, the boundary of the coastal waters generally lies between the 30 and 50 m contours in winter, and between the 50 and 200 m contours in summer. The coastal waters can be divided into 'offshore' and 'nearshore' coastal waters, the latter being defined as waters east of the Garden Island Ridge and consisting of partially-sheltered embayments and lagoons (Plate 4.1-1).

Background conditions for summer and winter were characterised (see Cary *et al.* 1995b) using data from areas considered to be uninfluenced by anthropogenic inputs and estuarine outflows (Buckee *et al.* 1994; Cary *et al.* 1995a; Cary and D'Adamo, 1995). Background levels are based on the median and 90th percentile values. As estuarine outflows can influence much of the study area during June to September (Simpson *et al.* 1993; Figure 4.7-2), background conditions were not often encountered in the winter surveys. Consequently, background water quality conditions for winter have been determined on the basis of limited data.

Findings

Indicative background conditions for the 'nearshore' and 'offshore' coastal waters of southern Perth in both summer and winter are shown in Table 4.7-1 and indicate that the southern coastal waters off Perth are oligotrophic by world standards (Kirkman, 1981; Codispoti, 1983; Pearce, 1991; Ignatiades *et al.* 1992). Inorganic N:P ratios (mass) are generally much less than 7 suggesting that nutrient limitation of primary productivity is determined by the availability of total inorganic nitrogen. Chlorophyll *a* concentrations and light attenuation of the water column are also low by world standards (Pearce, 1991) and decrease in an offshore direction. These results are supported by water quality data collected in Geographe Bay, near Busselton (Walker *et al.* 1994a) and from 30 years of data from the CSIRO site off Rottnest Island (Pearce *et al.* in preparation), and are similar to values found by Rochford (1980) for offshore waters off southwest Australia.

Table 4.7-1. Median and 90th percentile (in parentheses) background levels for selected water quality parameters for the 'nearshore' and 'offshore' coastal waters in summer and winter.

Water quality parameter	Nearshore coastal waters		Offshore coastal waters	
	summer	winter	summer	winter
Total inorganic nitrogen (μg l^{-1})	< 6 (9)	< 4 (8)	< 6 (9)	< 5 (8)
Total inorganic phosphorus (μg l^{-1})	< 5 (6)	< 4 (6)	< 4 (4)	< 4 (6)
Chlorophyll *a* (μg l^{-1})	< 0.5 (0.8)	-	< 0.2 (0.4)	-
Light attenuation coefficient (m^{-1})	< 0.08 (0.09)	-	< 0.04 (0.05)	-

Conclusions

The southern metropolitan coastal waters of Perth are oligotrophic and chlorophyll *a* concentrations and light attenuation are low by world standards.

4.7.2 Shelf-scale water quality

Prior to the SMCWS, water quality data for the coastal waters of the Perth region, west of the 30 m depth contour, were limited to data collected by the CSIRO since 1951 at a site about 5 km west of the western end of Rottnest Island. To address this deficiency regional water quality surveys were carried out during 'typical' winter and summer conditions to characterise the waters of the study area, particularly in relation to nutrient and chlorophyll *a* concentrations and water clarity. An additional objective was to identify and determine the spatial scales of any major regional influences on the water quality of the area. Surface and bottom waters were sampled in August 1991 and March 1992 at over 100 sites. These sites were located between the mouth of the Peel-Harvey Estuary in the south and Mullaloo Point in the north; and across the continental shelf to 80 km offshore (Figure 4.7-1). The regional surveys complemented an intensive 12-month (March 1991 to February 1992) survey in Cockburn Sound, Warnbro Sound and a site in Sepia Depression (see section 4.7.3)

Further technical details can be found in Cary *et al.* (1995a and 1995b) and Simpson *et al.* (1993).

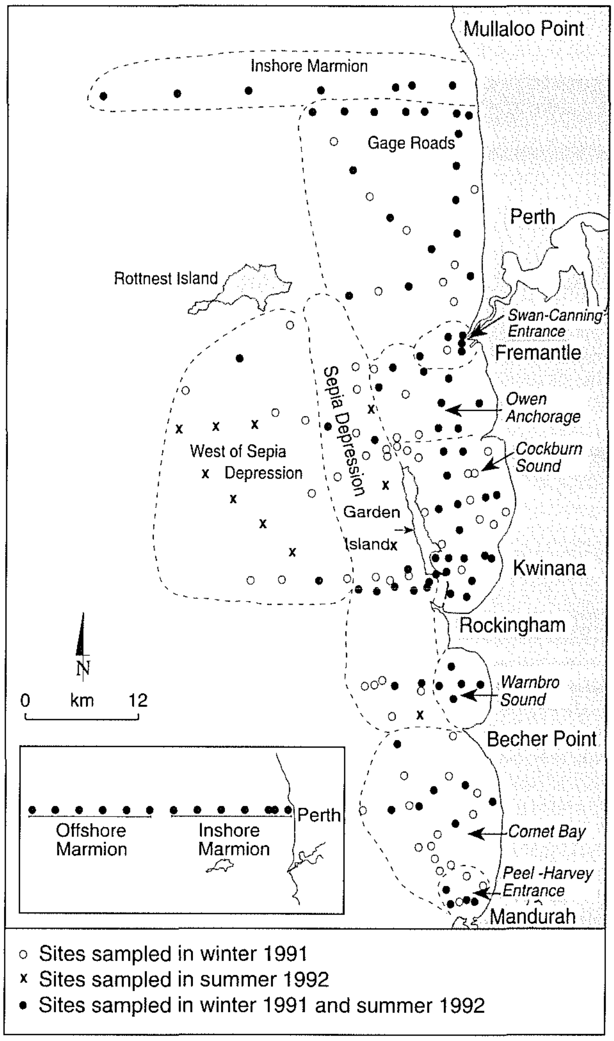


Figure 4.7-1. Location of sampling sites and 11 sub-regions for the shelf-scale water quality surveys.

Findings

4.7.2.1 *Temporal variation*

Chemical and biological water quality parameters were significantly higher during winter, while the physical parameters (including secchi depth) were lower in winter than summer, indicating a strong seasonal variation in the water quality of the region (Tables 4.7-2 and 4.7-3). The regional water quality survey in August 1991 identified two generic seasonal influences on the water quality of Perth's coastal waters: the outflows of the Peel-Harvey and Swan-Canning estuaries (Figure 4.7-2; see section 4.4.2 for nutrient loads from estuaries) and the incursion of water from the Leeuwin Current over the mid- and inner-shelf (Figure 4.7-3; Plate 5.1-1d and f). The elevated nitrate-N concentrations in winter, in surface and bottom waters throughout the southern coastal waters of Perth, when compared with similar data in summer, clearly illustrate the influence that the estuarine inputs, particularly the Peel-Harvey Estuary outflow, can have on these waters during winter (Figure 4.7-2).

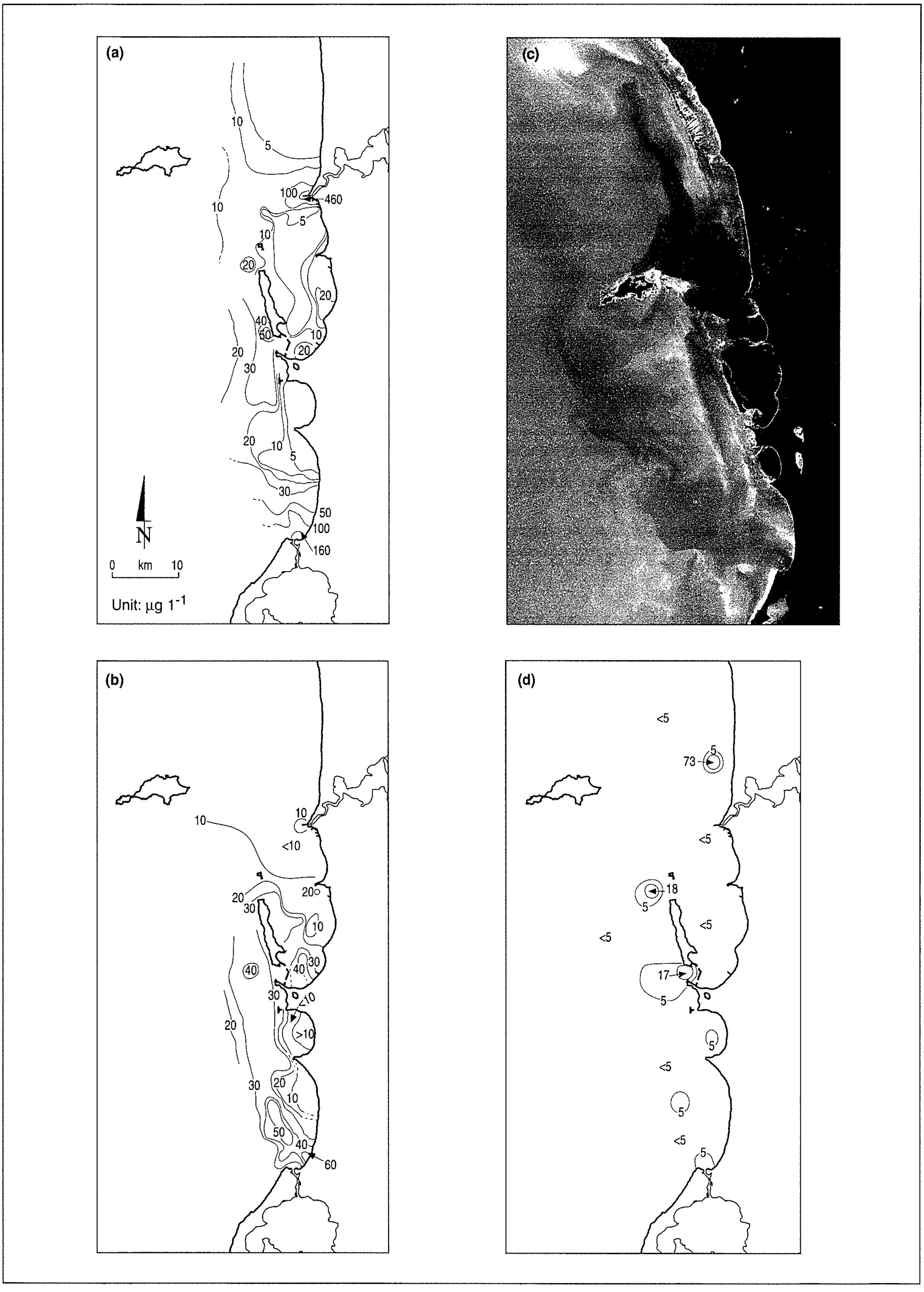


Figure 4.7-2. Distribution of nitrate-N in (a) surface and (b) bottom waters during 13-14 August 1991 and (d) surface waters during March 1992; (c) satellite image of estuarine plumes on 14 August 1991.

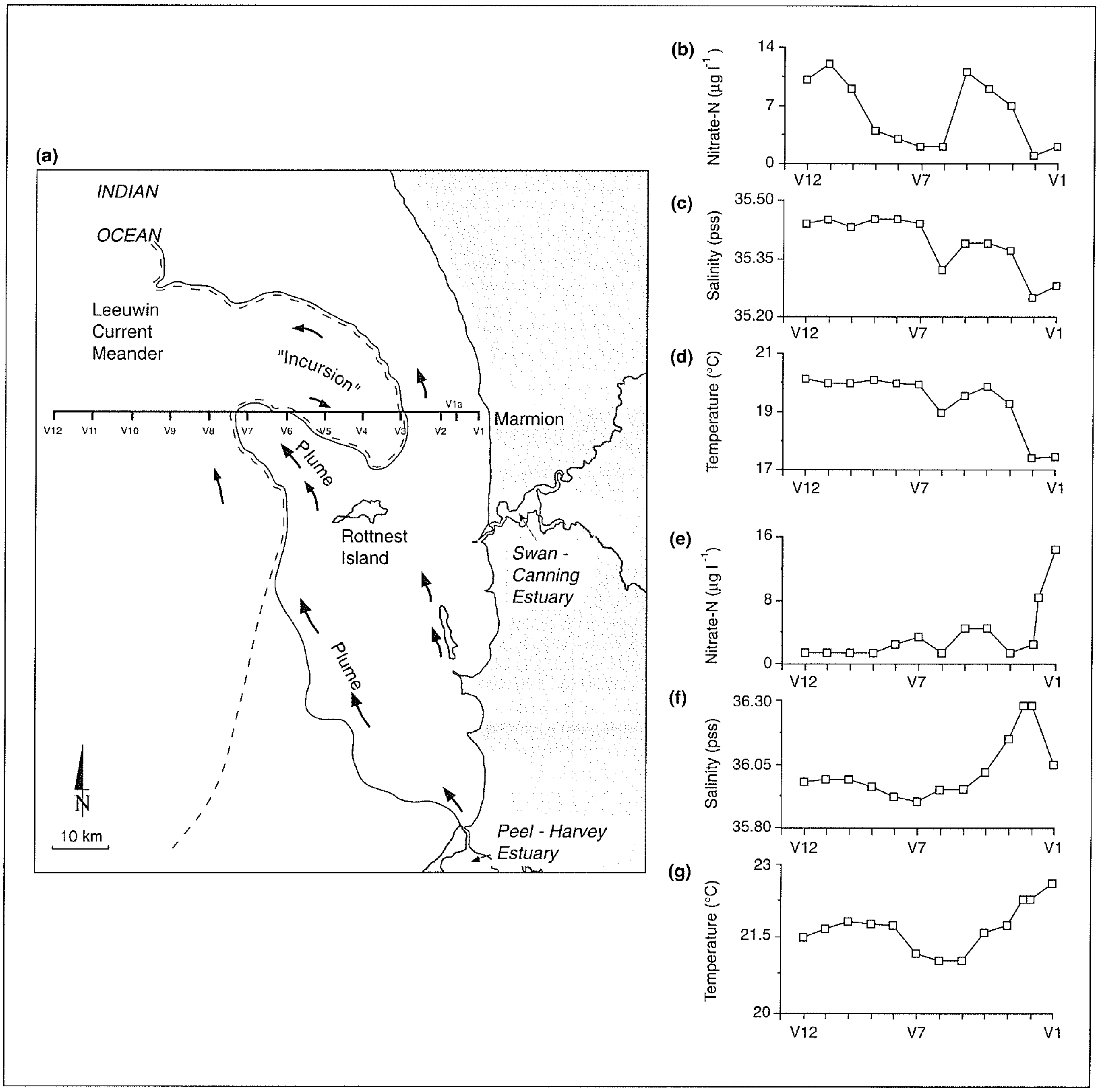


Figure 4.7-3. Cross-shelf measurements of selected surface water quality parameters along (a) a transect north of Rottnest Island. Data in (b), (c) and (d) are from 13- 23 August 1991 and pass through the Leeuwin Current and an estuarine 'plume' shown in (a). Data in (e), (f) and (g) are from 9- 27 March 1992.

The data indicate that these plumes can affect the quality of Perth's coastal waters, on occasions, for distances of over 100 km (Figure 4.7-2c; Plate 5.1-1). The hydrodynamic modelling of estuarine plumes (Chapter 6) indicated that they can spread both to the north and south, depending on the wind direction, which varies within synoptic weather systems.

Although the winter regional surveys of 1991 showed widespread elevations above background levels (see Figure 4.7-2a, b), comparable data in the winter of 1994 were significantly lower (Buckee *et al.* 1994). These differences can be explained in terms of the intra-annual and inter-annual variability of estuarine outflows (Deeley, in preparation), the meteorological conditions preceding the surveys and, to a lesser extent, the intermittent presence of the Leeuwin Current over the mid-shelf (see for example Plate 5.1-1).

4.7.2.2 *Spatial variation*

In the winter survey, chlorophyll *a* and nutrient levels were elevated near the entrances of the estuaries and declined with distance away, while for temperature and salinity this pattern was reversed (Table 4.7-2). South to north and nearshore to offshore decreasing gradients of chlorophyll *a* and nutrient concentrations and light attenuation were observed. In the summer survey, nutrient concentrations were generally near background levels apart from elevations about domestic wastewater outfalls.

Table 4.7-2. Summary of 'winter' shelf-scale water quality data from August 1991. Sub-regions are shown in Figure 4.7-1.

WINTER Sub-region	Number of sites	Kjeldahl-N ($\mu g\ l^{-1}$)		Nitrate-N ($\mu g\ l^{-1}$)		Ammonium-N ($\mu g\ l^{-1}$)		Phosphate-P ($\mu g\ l^{-1}$)		Inorganic N:P		Chlorophyll *a* * ($\mu g\ l^{-1}$)		Temperature (°C)		Salinity (pss)		Secchi depth (m)
		S	B	S	B	S	B	S	B	S	B	S	B	S	B	S	B	
Swan - Canning Entrance	6	284	98	163	7	28	1	23	3	9	3	6.4	5.9	15.9	16.8	28.7	35.0	4
Owen Anchorage	10	110	110	11	6	1	1	2	2	5	5	4.6	4.4	16.2	16.3	34.5	34.7	7
Cockburn Sound	30	176	176	7	14	3	9	2	4	10	9	5.8	4.3	16.0	16.0	34.2	34.6	6
Gage Roads	25	157	-	7	-	1	-	2	1	6	-	8.3	-	17.0	16.8	35.0	35.0	6
Inshore Marmion	7	174	-	5	-	2	-	2	-	4	-	2.8	-	18.5	18.5	35.3	35.3	8
Offshore Marmion	6	134	-	7	-	1	-	1	-	8	-	1.0	-	20.1	17.3	35.4	35.6	13
Peel-Harvey Entrance	7	422	258	74	34	4	3	47	18	3	2	17.7	6.4	16.6	16.4	26.8	35.0	2
Comet Bay	20	118	148	24	24	1	1	11	10	2	3	7.5	4.0	16.6	16.6	33.0	35.1	4
Sepia Depression	26	117	89	21	24	2	2	5	3	5	3	2.5	2.4	16.8	16.9	35.0	35.1	6
Warnbro Sound	5	79	89	3	18	1	1	3	4	2	3	4.1	3.6	15.4	15.4	34.5	34.6	5
West of Sepia Depression	8	117	-	6	6	1	-	1	1	6	-	1.9	1.2	18.7	18.2	35.4	35.4	8

* Chlorophyll *a* was determined fluorometrically and should not be compared to other chlorophyll *a* values (derived spectrophotometrically) elsewhere in the report.

Table 4.7-3. Summary of 'summer' shelf-scale water quality data from March 1992. Sub-regions are shown in Figure 4.7-1.

SUMMER Sub-region	Number of sites	Nitrate-N ($\mu g\ l^{-1}$)		Chlorophyll *a* * ($\mu g\ l^{-1}$)		Temperature (°C)		Salinity (pss)		Secchi depth (m)
		S	B	S	B	S	B	S	B	
Swan Canning Entrance	5	5	3	1.2	0.5	23.7	22.7	35.2	35.5	6
Owen Anchorage	8	3	3	0.6	0.7	23.6	22.9	36.3	36.4	7
Cockburn Sound	17	3	4	1.4	1.8	24.0	22.8	36.2	36.4	7
Gage Roads	17	5	2	0.5	0.9	23.4	22.3	36.2	36.1	8
Inshore Marmion	7	4	8	0.5	1.0	21.7	21.7	36.0	36.0	10
Offshore Marmion	6	3	-	0.1	-	21.7	15.1	36.0	35.7	12
Peel-Harvey Entrance	4	3	3	1.3	1.2	23.6	23.2	36.6	36.6	5
Comet Bay	7	3	3	1.2	3.0	23.5	23.0	36.5	36.5	7
Sepia Depression	10	7	4	0.8	0.5	23.4	22.7	36.3	36.4	10
Warnbro Sound	5	3	6	0.6	0.8	24.1	23.3	36.3	36.5	9
West of Sepia Depression	9	2	6	0.2	0.5	21.6	21.4	36.1	36.1	12

* Chlorophyll *a* was determined fluorometrically and should not be compared to other chlorophyll *a* values (derived spectrophotometrically) elsewhere in the report.

Elevated chlorophyll *a* levels were found near the entrances to the estuaries and in Cockburn Sound and Comet Bay (Table 4.7-3). There was also a nearshore to offshore decreasing gradient in chlorophyll *a* concentrations and light attenuation. In general salinity and temperature were higher inshore.

Conclusions

Estuarine outflows and to a lesser extent the Leeuwin Current are significant shelf-scale influences on the quality of Perth's coastal waters. In particular, the Peel-Harvey Estuary can elevate nutrient and chlorophyll *a* concentrations over much of the southern metropolitan coastal waters in winter. These influences, particularly estuarine outflows, need to be considered in relation to the monitoring and management of Perth's coastal waters.

4.7.3 Local-scale water quality

The physical, chemical and biological status of Cockburn Sound and, to a lesser extent, Owen Anchorage has been regularly monitored during summer (December to March) every 2-3 years since the Cockburn Sound Environmental Study 1976-1979 (CSES). In contrast, limited data have been collected on the water quality of other parts of the southern metropolitan coastal waters. One site in Warnbro Sound, which is the only other sheltered marine embayment in Perth's coastal waters and which now forms part of the Shoalwater Islands Marine Park, and a site in Sepia Depression, were monitored as 'control' sites during the CSES. Industrial and domestic wastes have not been discharged directly into Warnbro Sound and, as a result, the waters of this embayment are still considered to be largely 'pristine' and, as such, to provide an important natural 'baseline' for Cockburn Sound. Domestic wastes have been discharged into Sepia Depression since 1984 when the Cape Peron outfall was commissioned. Since then, the quality of these waters has been surveyed at least annually by the Water Corporation (formerly the WAWA), and by the DEP during the SMCWS.

Water quality monitoring at four sites in Cockburn Sound, six sites in Warnbro Sound, and at a site in Sepia Depression (due west of central Warnbro Sound) was undertaken every two weeks for 12 months, from March 1991 to February 1992. Two sites in Owen Anchorage were monitored 45 times between June 1992 and June 1993 (Burt *et al.* 1995b). A further survey was undertaken weekly for 16 weeks from December 1993 to February 1994 at four sites in Cockburn Sound, five sites in Warnbro Sound, two sites in Sepia Depression (4-5 km north and south of the Cape Peron outfall) and at two sites, 8 km apart, 7 km west of Sepia Depression. The objectives of these surveys were to quantify the water quality status of the nearshore 'core' areas over an annual cycle, to identify the major influences on the water quality, to determine long-term trends by comparing these surveys with historical data, to investigate causes for the long-term trends, and to update the water quality status of these waters toward the end of the SMCWS.

Nutrient and chlorophyll *a* analyses were undertaken at the Nutrient Analysis Laboratory at Murdoch University. Further technical details can be found in Simpson *et al.* (1993), Cary and D'Adamo (1995) and Cary *et al.* (1995a; 1995b).

Findings

4.7.3.1 *Annual patterns*

Annual patterns of physical, chemical and biological water quality parameters in Cockburn Sound, Warnbro Sound and Sepia Depression are presented in Figure 4.7-4. Water quality data for Owen Anchorage are presented separately in Figure 4.7-5, as these data are not contemporaneous with the data for the other three areas.

Physical parameters

Annual temperature patterns in Cockburn Sound and Warnbro Sound were closely linked and mean values ranged from about 16 °C to 24 °C. In contrast Sepia Depression was warmer, by about 2 °C, than the two basins during autumn and winter and cooler during spring and summer. These differences can be explained by the enhanced warming and cooling of the relatively 'trapped' waters in the two basins and the moderating influence of offshore waters on Sepia Depression. Mean salinities also show a pronounced annual pattern with highest values in summer and lowest in winter; the greatest annual salinity range occurs in Cockburn Sound and the least in Sepia Depression. The mean salinity of Cockburn Sound was lower than that of Warnbro Sound during late autumn and winter and this difference can be explained by the relative influence of freshwater outflows from the Swan-Canning and the Peel-Harvey estuaries on the salinity of these embayments. The two basins had similar salinity values in summer, which were higher than the salinity of Sepia Depression, and this difference can be explained by the differential evaporation of the relatively 'trapped' water in the two basins. Further details of the annual patterns in salinity and temperature are given in section 5.1.

In contrast to the unimodal annual patterns of temperature and salinity, those of light attenuation in the water column were bimodal in the embayments, with a primary maximum in winter and a secondary maximum in summer. Minima occurred in mid-autumn and mid-spring/early summer. The annual pattern in light attenuation was similar to the pattern for chlorophyll *a*, but other studies in Owen Anchorage/Gage Roads (Burt *et al.* 1995b) found that, in addition to chlorophyll *a*, light attenuation during winter

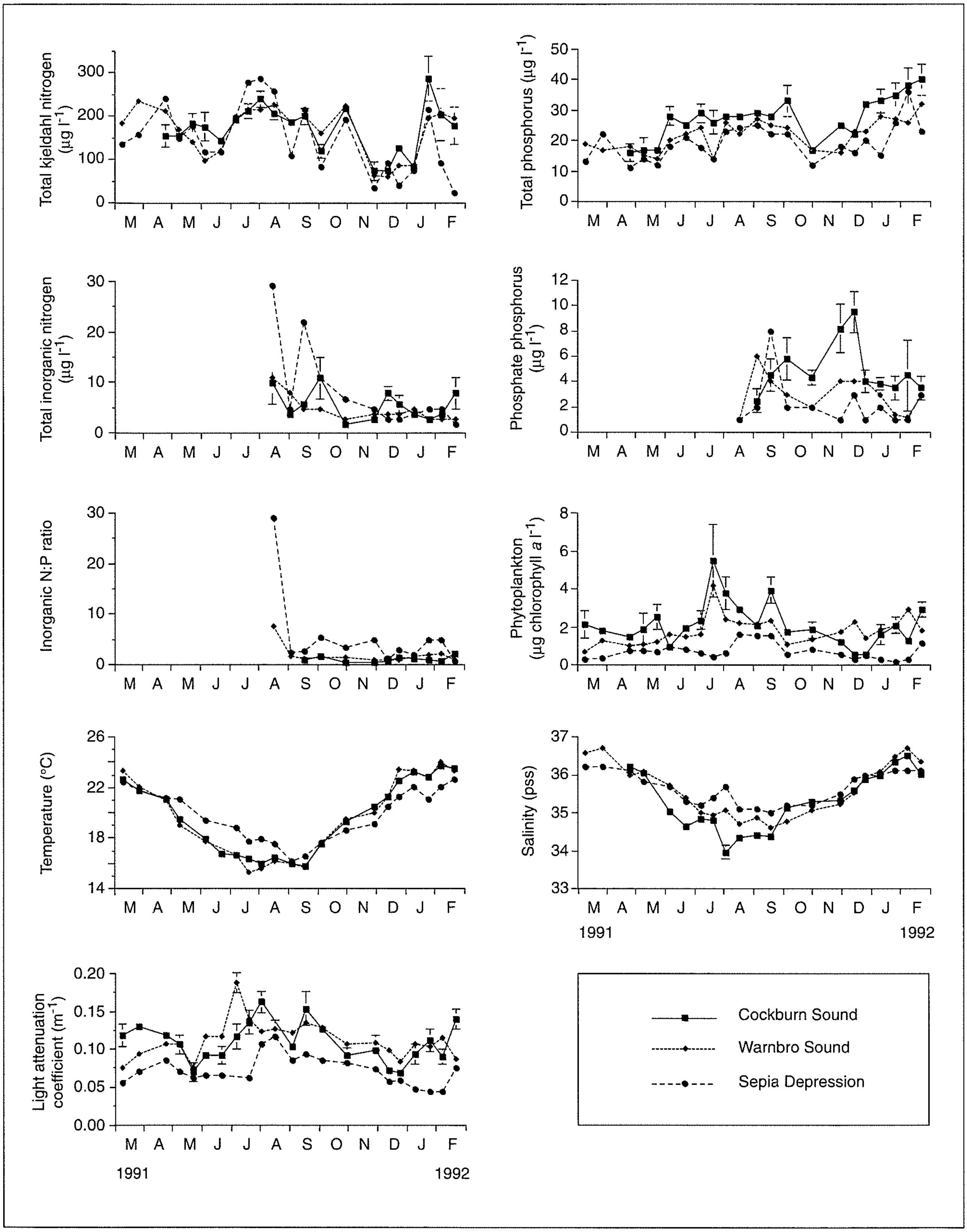


Figure 4.7-4. Annual patterns of selected water quality parameters for Cockburn Sound, Warnbro Sound and Sepia Depression.

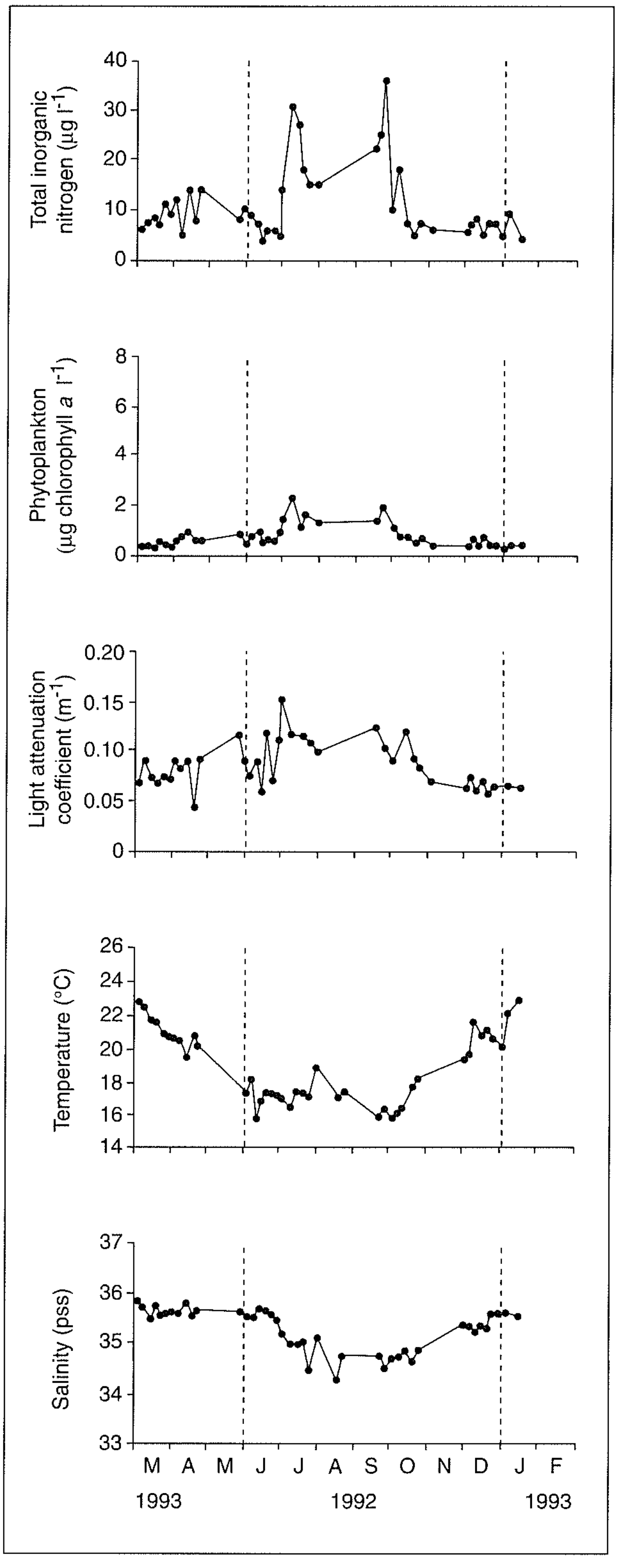


Figure 4.7-5. Composite annual patterns of selected water quality parameters for Owen Anchorage. Data from June 1992 to June 1993.

was correlated to particulates resuspended by waves, and that estuarine outflows also have intermittent effects on the water clarity of the area. During autumn and spring the resuspension of sediments by swell and wind waves is less, estuarine outflows are reduced and the phytoplankton biomass is relatively low, conditions resulting in high water clarity. Apart from the winter months, light attenuation in Sepia Depression is relatively constant throughout the year. The winter maximum appears to be due to increased phytoplankton biomass resulting from nutrient enrichment of these waters and from direct transport of phytoplankton via the estuarine outflows (see sections 4.7.2 and 4.8.1).

Chemical parameters

Mean total kjeldahl nitrogen (TKN) levels were similar in Cockburn Sound, Warnbro Sound and Sepia Depression and ranged from 40 to 287 µg l^{-1} with most values occurring in the range of 150 to 200 µg l^{-1}. As 200 to 300 µg l^{-1} is considered the background concentration in these waters (Chiffings, 1987) and 200 µg l^{-1} the limit of analytical precision, further discussion of the apparent coherence in the seasonal trends of TKN concentrations within and between these waterbodies is meaningless. Mean total phosphorus concentrations ranged from 11 to 40 µg l^{-1} with most values occurring between 15 - 30 µg l^{-1}. There was no clear seasonal pattern in the data. In general, values were highest in Cockburn Sound and lowest in Sepia Depression. Mean total inorganic nitrogen ranged from 4 to 10 µg l^{-1} in the Cockburn and Warnbro sounds, from 2 to 40 µg l^{-1} in Owen Anchorage, and from 2 to 30 µg l^{-1} in Sepia Depression. The high values recorded in winter at Owen Anchorage and Sepia Depression are associated with the nutrient-enriched outflows from the Swan-Canning and Peel-Harvey estuaries. Mean orthophosphate-phosphorus levels followed no seasonal trend and ranged from 1 to 10 µg l^{-1} with concentrations generally higher in Cockburn Sound than in either Warnbro Sound or Sepia Depression. Mean inorganic N:P ratios (mass) were generally less than seven, apart from in Sepia Depression, in winter, when levels reached approximately 30. These data indicate that primary productivity in these waters is generally limited by the availability of inorganic nitrogen, except for some periods during winter, when large volumes of nitrogen-enriched water from the Swan-Canning and Peel-Harvey estuaries flow into the southern coastal waters of Perth.

Biological parameters

Chlorophyll *a* concentrations in all waterbodies peaked in winter and spring, when outflows from the estuaries were relatively high. During this period mean chlorophyll *a* concentrations in the four waterbodies ranged from 0.2 to 5.5 µg l^{-1} with the highest levels in Cockburn Sound, followed by Warnbro Sound, Owen Anchorage and the lowest levels in Sepia Depression. Mean chlorophyll *a* concentrations in all four waterbodies were relatively constant during summer and autumn, with Cockburn Sound again having the highest concentrations, and Owen

Anchorage and Sepia Depression having the lowest concentrations.

4.7.3.2 *Current water quality status*

Water quality surveys in Cockburn Sound, Warnbro Sound, Sepia Depression and in waters further offshore, were undertaken between December 1993 to March 1994 (Cary and D'Adamo, 1995). The results of these surveys and comparable data for Owen Anchorage from December 1992 to March 1993 (Burt *et al.* 1995b) are presented in Table 4.7-4 to determine the 'current' status of selected aspects of the quality of the nearshore and offshore southern coastal waters of Perth.

Direct inputs of nutrients to Warnbro Sound are minimal and, consequently, the water quality data from Warnbro Sound can be considered to be largely representative of 'pristine' conditions in the nearshore basin environment of the study area in summer. These data therefore provide a 'benchmark' for comparisons with Cockburn Sound. Similarly, as the offshore coastal waters are 'source' waters for Sepia Depression, water quality data from these areas can be used to provide a 'benchmark' for comparison with measured Sepia Depression water quality.

The mean inorganic N:P ratios were less than four, providing further evidence that these waters are nitrogen limited. Although volume adjusted nitrogen loadings to Cockburn Sound were 5-6 times the loading to Warnbro Sound inorganic nitrogen concentrations were similar. This is likely to be due, in part, to greater nitrogen uptake in Cockburn Sound by phytoplankton, where the biomass, when expressed as chlorophyll *a* concentrations, was about four times higher than in Warnbro Sound. Inorganic phosphorus concentrations in Cockburn Sound were almost double the concentrations in Warnbro Sound, a ratio similar to that of the volume adjusted loading. This further supports the conclusion that inorganic phosphorus uptake in these waters is low compared with the uptake of inorganic nitrogen. A comparison of mean light attenuation coefficients indicates that light reaching 10 m depth in Cockburn Sound was about 40% of the intensity reaching the same depth in Warnbro Sound. The current pattern of chlorophyll *a* distribution in Cockburn Sound, with highest levels along the eastern margin, particularly near Jervoise Bay and Mangles Bay, and lowest levels in the northwestern part of the Sound, has not changed since the 1970s.

Mean inorganic nitrogen concentrations at the Sepia Depression sites were 2-3 times those of the offshore coastal waters sites, three to four kilometres west of Five Fathom Bank. These data, coupled with the higher inorganic N:P ratio in Sepia Depression, suggest a large source of nitrogen in the area resulting in measurably higher nutrient and phytoplankton concentrations and light attenuation in these waters. The time-series of the two offshore coastal sites and the two Sepia Depression sites further support these conclusions (Figure 4.7-6). The major source of nitrogen to Sepia Depression is the Cape Peron wastewater outfall and the impact of this outfall on the water quality of Sepia Depression is discussed further in the following section.

4.7.3.3 *Long-term trends*

Using the concentrations of nutrients to establish trends in the trophic status of waterbodies can be misleading, as nutrients may be taken up by plankton or macroalgae, and nutrient-enrichment of waters may therefore be manifest as increased primary production, rather than as higher nutrient concentrations in water. Hence, phytoplankton concentrations and light attenuation coefficients are used here as the primary indices of water quality status, with nutrient data used as secondary indices. These primary indices are preferred to nutrient concentrations, as increased phytoplankton biomass is often a direct response to nutrient enrichment and is strongly correlated with light attenuation in the water column during summer (Cary *et al.* 1995a;

Table 4.7-4. Water quality parameters for the 'core' areas of the southern metropolitan coastal waters of Perth. Data are for December to March 1993/94 except for Owen Anchorage which are for 1992/93.

	Cockburn Sound			Warnbro Sound			Owen Anchorage			Sepia Depression			Offshore waters		
	n	Mean	Range	n	Mean	Range	n	Mean	Range	n	Mean	Range	n	Mean	Range
Nitrate-N ($\mu g\ l^{-1}$)	192	3	1-26	240	2	1-15	60	2	1-4	96	6	1-18	96	2	1-9
Ammonium-N ($\mu g\ l^{-1}$)	192	3	1-16	240	3	1-15	60	4	1-12	96	5	1-24	96	3	1-10
Phosphate-P ($\mu g\ l^{-1}$)	48	7	2-14	60	4	2-9	-	-	-	24	3	3-5	24	3	2-4
Inorganic N:P ratio (mass)	48	2	1-12	60	1	1-4	-	-	-	24	3	2-14	24	2	1-4
Chlorophyll *a* ($\mu g\ l^{-1}$)	192	1.9	0.2-7.3	240	0.5	0.2-1.3	60	0.4	0.3-0.9	96	0.3	0.1-1.0	96	0.2	0.1-1.2
Light attenuation coefficient (m^{-1})	192	0.11	0.06-0.19	144	0.07	0.04-0.12	60	0.07	0.06-0.12	96	0.06	0.04-0.07	96	0.04	0.02-0.06

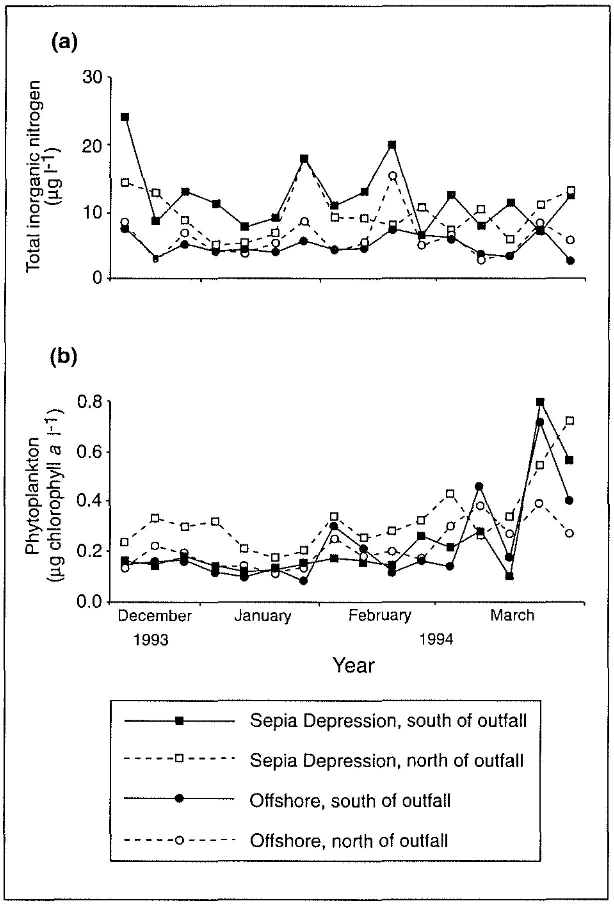


Figure 4.7-6. Time series of weekly (a) total inorganic nitrogen and (b) chlorophyll a concentrations approximately four kilometres north and five kilometres south of the Cape Peron domestic wastewater outfall in Sepia Depression and in offshore coastal waters (seven kilometres west of Sepia Depression) between December 1993 and March 1994.

Figure 4.7-7). Light attenuation in the water column, being inversely related to light reaching benthic plant communities, is an ecologically significant parameter. In using these parameters to assess long-term trends it is necessary to acknowledge that other factors also affect phytoplankton biomass (e.g. zooplankton grazing) and light attenuation (e.g. resuspension of sediments by waves).

Two data sets were used to examine long-term trends. Composite annual patterns were constructed from data collected during the periods 1977-1981 and 1991-92. Mean monthly values were then compared statistically. Weekly data collected during regular summer (December to March) monitoring programmes since the late 1970s were also examined.

Cockburn Sound

Composite annual patterns for various water quality parameters for Cockburn Sound are shown in Figure 4.7-8. In general chlorophyll *a* concentration and light attenuation throughout the year in 1991/92 were slightly lower than the 1977/81 period. Total kjeldahl nitrogen, total phosphorus, total inorganic nitrogen and phosphorus were all markedly lower in 1991/92 than during the 1977/81 period reflecting a marked reduction in point source industrial and domestic waste discharges into the Sound (see section 4.4.1).

The water quality of Cockburn Sound in summer, expressed as chlorophyll *a* and light attenuation, has varied considerably since the late 1970s, when the sound was in its poorest recorded state (Figure 4.7-9). The poor water quality in the late 1970s was associated with high nitrogen loads, mainly from industrial point sources (Department of Conservation and Environment, 1979; see section 4.4.1).

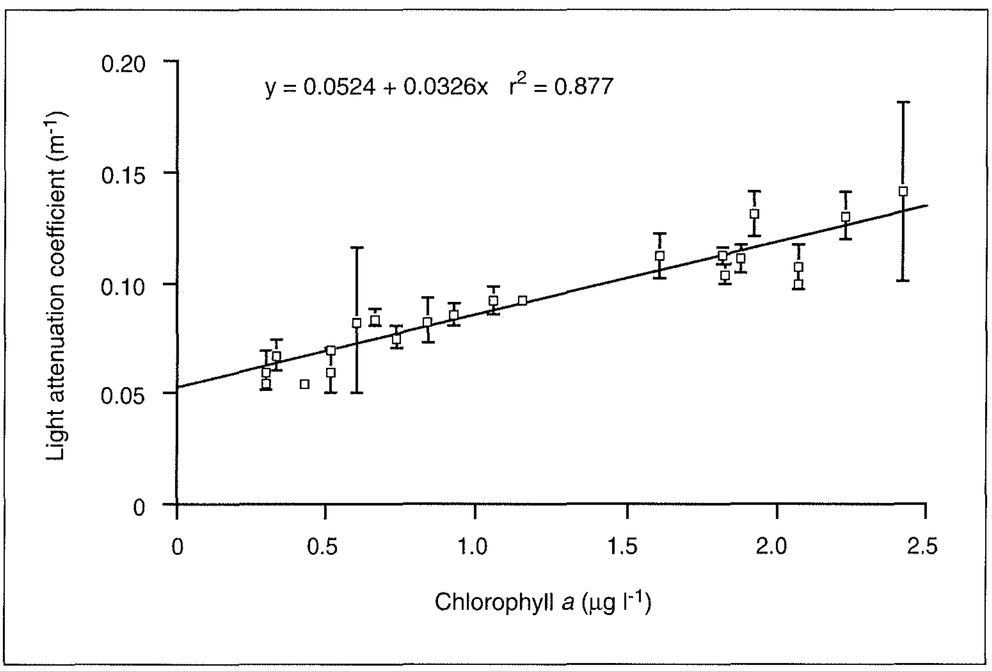


Figure 4.7-7. Relationship between mean chlorophyll a concentration and light attenuation coefficient using data (see Figure 4.7-9) collected during the summer period (December to March) in Cockburn Sound, Warnbro Sound and Sepia Depression. Error bars are standard errors.

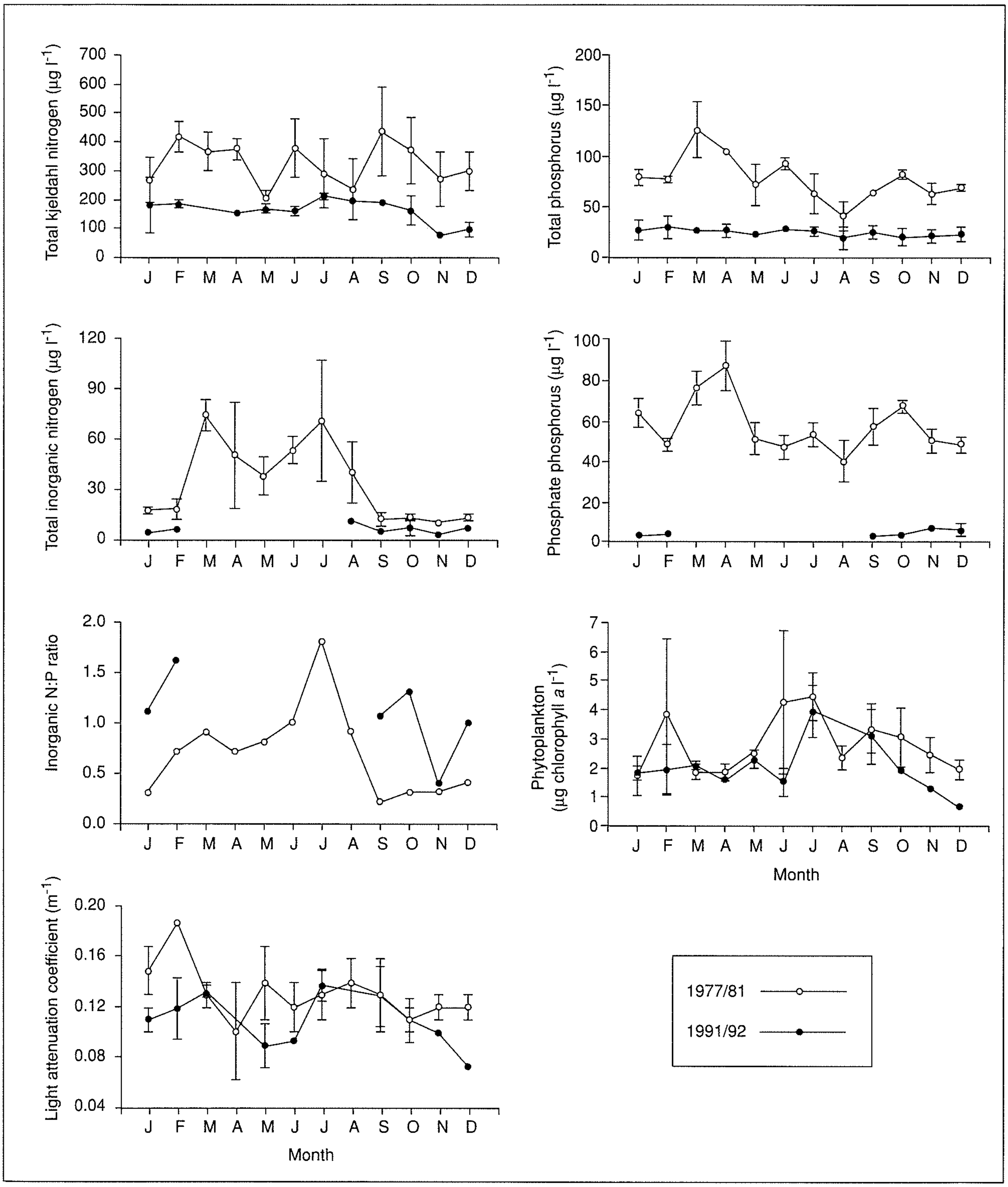


Figure 4.7-8. Composite annual patterns of selected water quality parameters in Cockburn Sound for 1977/81 and 1991/92. Data for each month have been averaged.

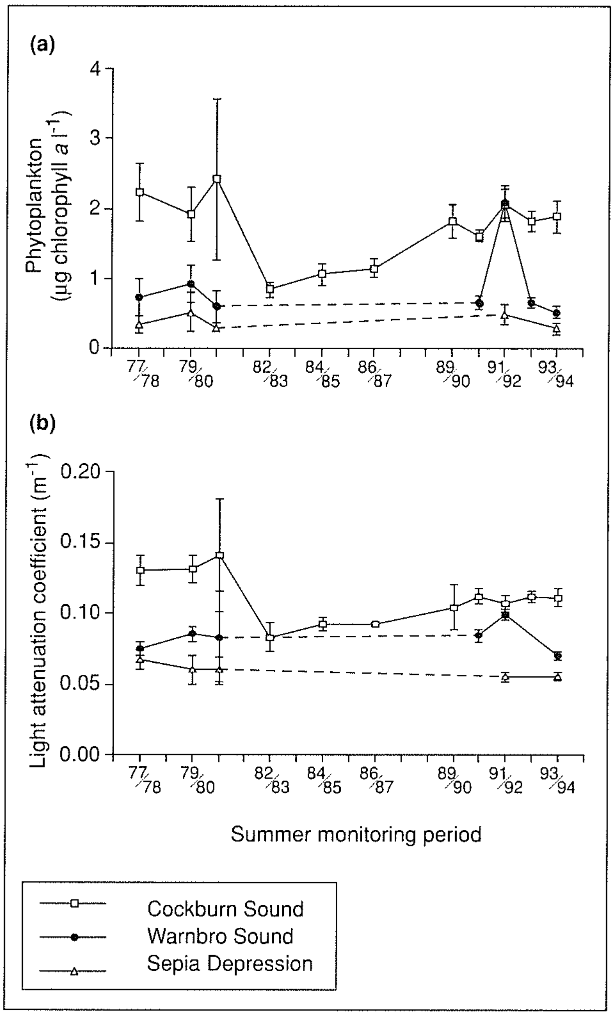


Figure 4.7-9. Mean summer (a) chlorophyll a concentrations and (b) light attenuation coefficients in Cockburn Sound, Warnbro Sound and Sepia Depression (for areas not influenced by the Cape Peron outfall) between 1977/78 and 1993/94.

The marked improvement in the water quality between 1980/81 and 1982/83 coincided with a 40% reduction in nitrogen loading, mainly from industrial sources, and resulted from improved waste management practices implemented as a result of the recommendations of the Cockburn Sound Environmental Study (Department of Conservation and Environment, 1979). The gradual decline in the water quality from the early to late 1980s was again associated with an increase in nitrogen loads, again mainly from industrial point sources. The 1993/94 water quality is only marginally better than in the late 1970s.

A significant relationship was established between nitrogen loads into Cockburn Sound and mean chlorophyll *a* concentrations for the summer monitoring periods between 1982/83 and 1989/90 (Cary *et al.* 1991). An extended data set, covering summer monitoring periods between 1977/78 and 1989/90, was used to further refine this relationship (Cary *et al.* 1995a). These findings support the earlier conclusion of the Cockburn Sound Environmental Study that nitrogen loads are a major determinant of water quality in the sound. However, summer monitoring data collected since 1989/90 show an apparent, significant departure from this relationship, with the water quality, expressed as chlorophyll *a* and light attenuation, remaining at about 1989/90 levels even though nitrogen loading estimates have declined by over 50% (Figure 4.7-10).

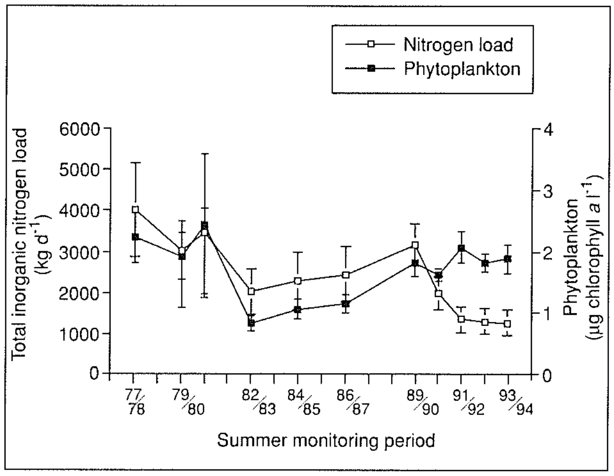


Figure 4.7-10. Total inorganic nitrogen loads from all known sources (excluding the Woodman Point outfall) and mean chlorophyll a concentrations for Cockburn Sound during summer between 1977/78 and 1993/94. Vertical bars are standard errors except for nitrogen loads in 1979/80 and 1980/81 where the bar indicates the range.

The recent departure from the nitrogen loading versus chlorophyll *a* response relationship can possibly be explained in several ways: either the original correlation between loadings and mean chlorophyll *a* concentrations was coincidental, or some new ecological process has dominated the nutrient dynamics of Cockburn Sound since 1990, or current estimates of total nutrient load to the Sound are less accurate than in the past (see section 6.3.5 for further discussion). The probability that the correlation was due to chance is less than 5% and currently there is no evidence to suggest that key ecological processes of Cockburn Sound have changed significantly since 1990. In relation to the accuracy of loading estimates, no new direct discharges have occurred since 1990 (Muriale and Cary, 1995), however estimates of groundwater nitrogen flux to Cockburn Sound (Appleyard, 1990; 1994), currently thought to be about 70% of the total loading, may be much higher due to the considerable error associated with estimating the total mass of nitrogen entering Cockburn Sound from this source (Appleyard, 1994). The uncertainty associated with identifying and accurately quantifying groundwater sources and nitrogen loads is supported by Martinick *et al.* (1993) in their assessment of the quality of the data used to estimate groundwater flux to the Sound, by the 'discovery' in 1990/91 of a major groundwater source near the WMC industrial estate (Figure 4.4-3), and by the subsequent doubling of original nitrogen loading estimates in 1993, after additional investigations (Western Mining Corporation Limited, 1993; 1995).

Based on mean chlorophyll *a* values for the whole of Cockburn Sound during summer, the nutrient-related criteria for draft Environmental Quality Objective 2 (i.e. maintenance of ecosystem integrity) were clearly exceeded in 1993/94 and the four previous summer periods. On this basis, it is clear that a nutrient management strategy is needed to ensure that Environmental Quality Objective 2 is met in Cockburn Sound.

Warnbro Sound

The composite annual pattern of chlorophyll *a* concentrations and light attenuation coefficient in Warnbro Sound indicates that these parameters were generally higher, particularly in the latter half of 1991 than in 1977/81, suggesting that the water quality of Warnbro Sound had declined (Figure 4.7-11). The high chlorophyll *a* concentrations were associated with a winter 'bloom' of

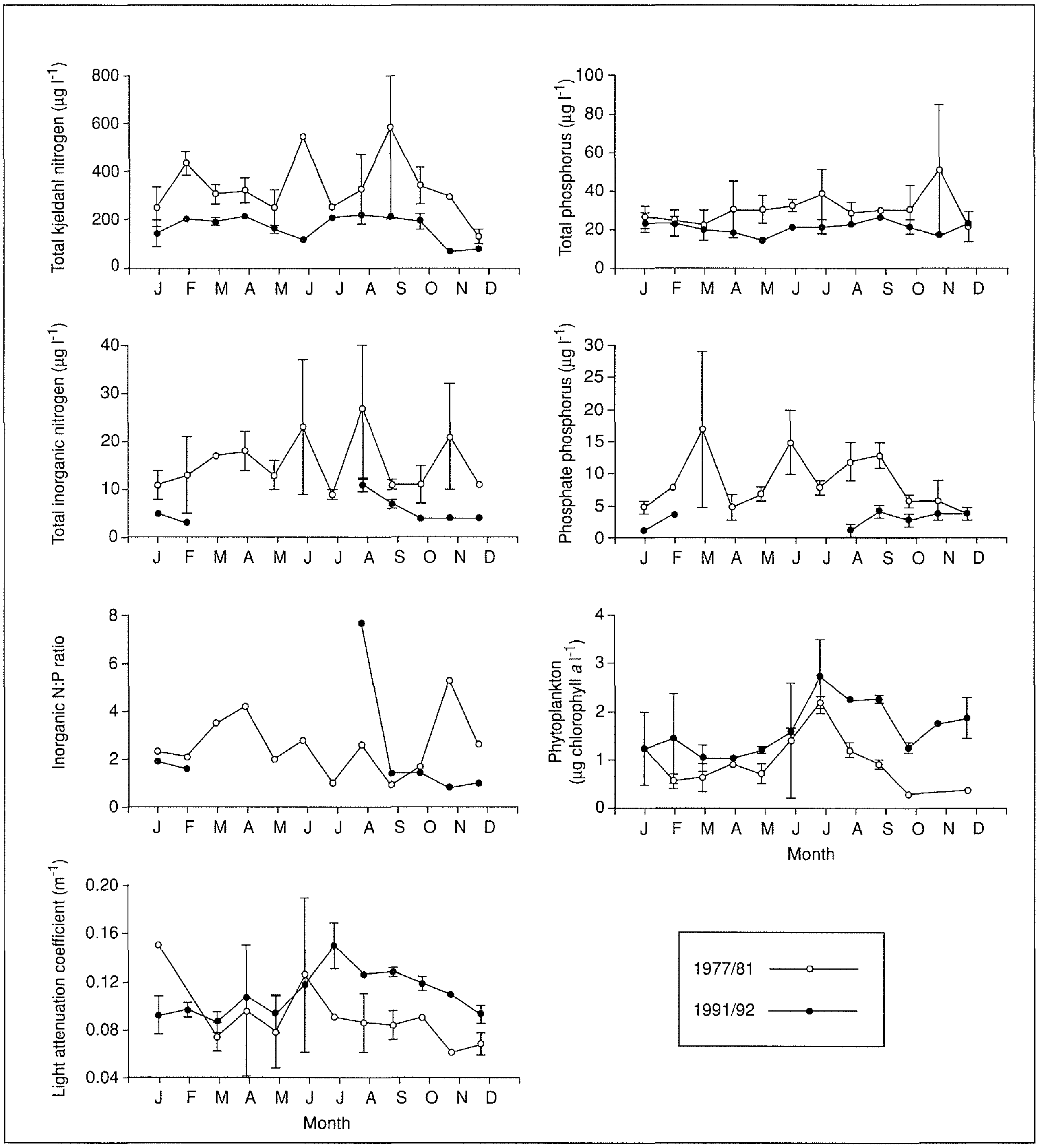


Figure 4.7-11. Composite annual patterns of selected water quality parameters in Warnbro Sound for 1977/81 and 1991/92. Data for each month have been averaged.

silicoflagellates that continued through to the summer (see section 4.8). These 'blooms' have been recorded each year during winter since 1991 in the southern coastal waters of Perth (Cousins, 1991; Helleren and John, 1995) but only once during summer. These data suggest that the high chlorophyll *a* concentrations in Warnbro Sound during the summer of 1991/92 were 'atypical' and unlikely to be indicative of a general decline in the water quality of the Sound.

The water quality of Warnbro Sound in summer in the 1990s remains largely unchanged from the levels of the 1970s (Figure 4.7-9). As mentioned above, the high mean chlorophyll *a* concentrations and associated light attenuation recorded in 1991/92 were associated with a prolonged silicoflagellate bloom (through winter, spring and summer) which, since that year, has been confined to the winter-spring period of the year.

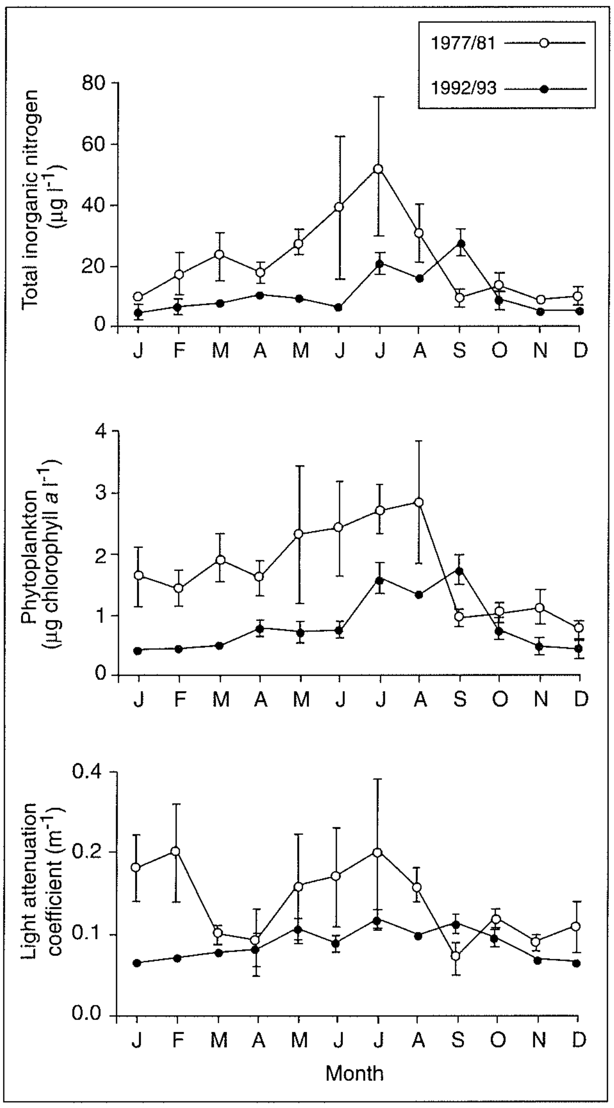


Figure 4.7-12. Composite annual patterns of selected water quality parameters in Owen Anchorage for 1977/81 and 1992/93. Data for each month have been averaged.

Owen Anchorage

Comparisons of composite annual patterns of chlorophyll *a* and total inorganic nitrogen concentrations and light attenuation coefficients suggest that the water quality of Owen Anchorage has improved since the late 1970s and early 1980s (Figure 4.7-12).

Similarly, a time-series of the summer water quality of Owen Anchorage shows that it has improved significantly since the 1970s, when the area was in its poorest recorded state (Figure 4.7-13). Nitrogen loads from industrial sources have reduced dramatically since the 1970s as industries employed better waste management practices, connected to deep sewer or closed down.

In recent years several water quality surveys have recorded high concentrations of nutrients in the northern Garden Island to west Success Bank area (Burt *et al.* 1993a; Bastyan *et al.* 1994; Cary *et al.* 1995b).

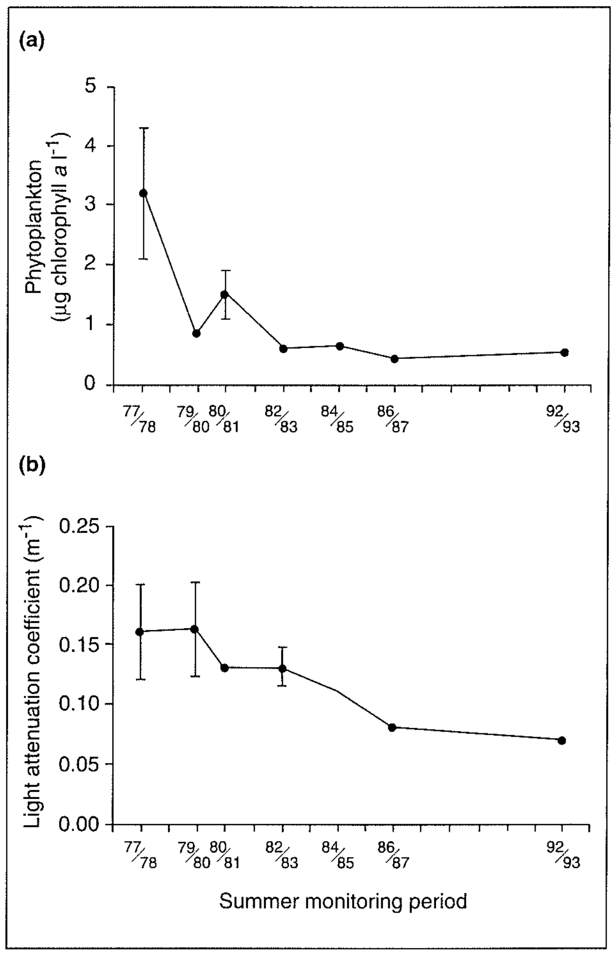


Figure 4.7-13. Mean summer (a) chlorophyll a concentrations and (b) light attenuation coefficients in Owen Anchorage between 1977/78 and 1992/93.

Sepia Depression

Long-term comparisons of water quality parameters were made between summer data collected in Sepia Depression west of Cape Peron in 1977-1981, (i.e. before the commissioning of the Cape Peron domestic wastewater outfall in 1984), and summer data collected in 1991/92 and 1993/94 from a site west of Warnbro Sound and approximately 5 km south of the outfall. The southern-most site was considered to be not significantly influenced by the outfall in summer due to the predominantly northward direction of flow in Sepia Depression during this season (Steedman and Associates, 1981; Halpern Glick Maunsell, 1992). The data suggest that the water quality (measured as chlorophyll *a* concentration) of Sepia Depression in summer, in areas not influenced by the Cape Peron outfall, remains unchanged from conditions in 1977/81 (Figure 4.7-9). However, a comparison of composite annual patterns of chlorophyll *a* concentrations in Sepia Depression for 1977/81 and 1991/92 shows that mean monthly chlorophyll *a* concentrations were generally higher in 1991/92 than in 1977/81, suggesting that the water quality of Sepia Depression has declined since the late 1970s (Figure 4.7-14). Nutrients entering Sepia Depression from the estuaries in winter (particularly the Peel-Harvey Estuary; see section 4.7.2) and from the Cape Peron outfall, are possible explanations for this decline.

Comparisons of composite annual patterns of chlorophyll *a* concentrations from data collected in Sepia Depression within about 5 km of the outfall during pre- and post-commissioning monitoring programmes show a significant rise in chlorophyll *a* concentrations after discharge commenced in 1984 (Figure 4.7-15). Furthermore, water quality data collected during the summer of 1993/94 at two sites, approximately 4 and 5 km north and south of the outfall respectively, indicate that chlorophyll *a* concentrations were consistently higher north (i.e. 'downstream') of the outfall during summer when prevailing winds are from the south (Figure 4.7-16). These data suggest that the Cape Peron outfall is having a significant measurable influence on the water quality of Sepia Depression during the non-winter months of the year. These findings are supported by recent WAWA surveys which found elevated (above background) concentrations of ammonium-N, total phosphorus and chlorophyll *a* over an area of up to 10 km^2 around the Cape Peron outfall (Claudius and Nener, 1995a, b). During winter, the influences of the Cape Peron outfall and the estuarine outflows on the water quality of Sepia Depression cannot be readily disentangled.

Conclusions

Ecological and analytical considerations indicate that chlorophyll *a* concentrations and light attenuation coefficients are better indicators of the 'health' of oligotrophic marine waters than ambient nutrient concentrations. Inorganic nitrogen to phosphorus ratios in the water suggest that nitrogen is the principal macro-nutrient limiting primary production in these waters.

The nutrient-related water quality of Cockburn Sound is only marginally better now than in the late 1970s when it was in its poorest recorded state. The departure since 1990 from the historical relationship between summer nitrogen loads and water quality in Cockburn Sound suggests that current estimates of nitrogen loads to the Sound are inaccurate. Much of this inaccuracy is associated with estimates of nitrogen flux via groundwater (Appleyard, 1994), which is now the major nitrogen pathway into the sound.

The water quality of Warnbro Sound has remained largely unchanged since the late 1970s. The large inter-annual variations in some water quality parameters underline the importance of understanding natural variation when assessing long-term trends.

The water quality of Sepia Depression, west of central Warnbro Sound, has not changed significantly since the late 1970s. However there has been a measurable decline since 1984 in the summer water quality of Sepia Depression, north of the Cape Peron wastewater outfall, suggesting that the outfall is having a significant effect on the water quality of this area.

4.7.4 Microbiological water quality

Faecal bacteria levels of the coastal waters of Perth have been routinely monitored by the Health Department of Western Australia since the 1960s. More recently the WAWA has monitored the Cape Peron wastewater outfall at least once a year and, with the Rockingham Shire, the southern metropolitan beaches every two weeks. Data collected between 1991 and 1994 were obtained from these organisations and used to assess the current microbiological status of these waters (Figure 4.7-17). The data were then compared against water quality guidelines for direct contact recreation and the taking of fish and other aquatic organisms for human consumption (Table 3.5-7; ANZECC, 1992; EPA, 1993) to assess the potential public health risks.

Findings

Faecal coliform concentrations for all mainland beaches monitored in the southern metropolitan coastal waters did not exceed the draft water quality criterion for EQO 4 (i.e. maintenance of recreational values), indicating that there was no significant risk to human health from faecal pollution. Faecal coliforms at these beaches did not exceed the draft water quality criterion for EQO 3 (i.e. maintenance of aquatic life for human consumption), except at Palm

Beach and at the beach near the Safety Bay jetty. These are the only two mainland beaches in the study area close to recreational and commercial boat mooring areas. As water quality monitoring for faecal coliforms is not undertaken near boat mooring areas at Garden Island or Rottnest Island, there are no data available to assess potential health risks in these areas.

Faecal coliform concentrations above the draft water quality criterion for EQO 4 (i.e. maintenance of recreational values) were found in Sepia Depression, on several occasions extending up to 2 km from the Cape Peron outfall. These data indicate that a potential public health risk exists in these waters. Similarly, the draft water quality criterion for EQO 3 (i.e. maintenance of aquatic life for human

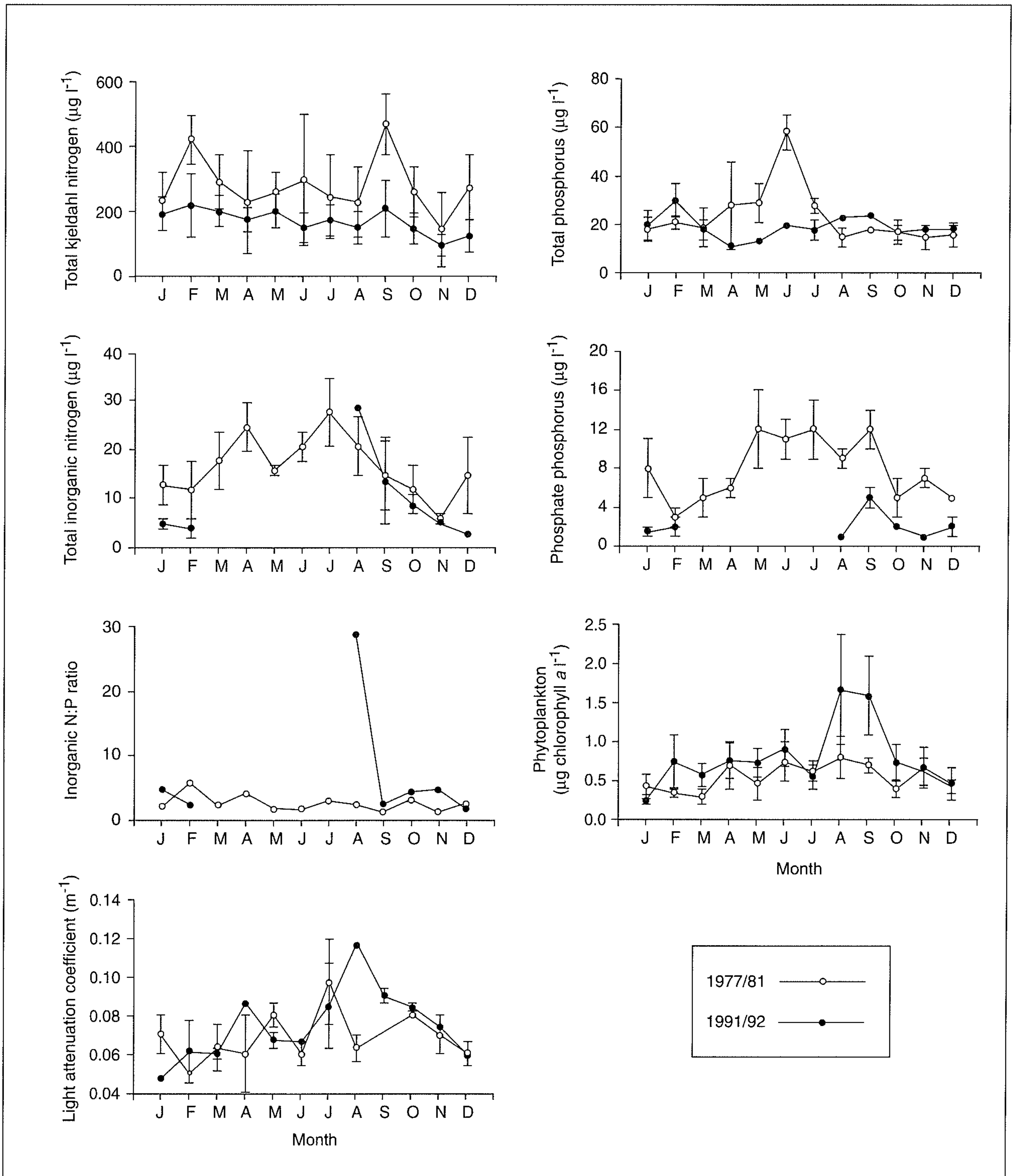


Figure 4.7-14. Composite annual patterns of selected water quality parameters in Sepia Depression in the immediate vicinity of the Cape Peron outfall for 1977/81 and 1991/92. Data for each month have been averaged.

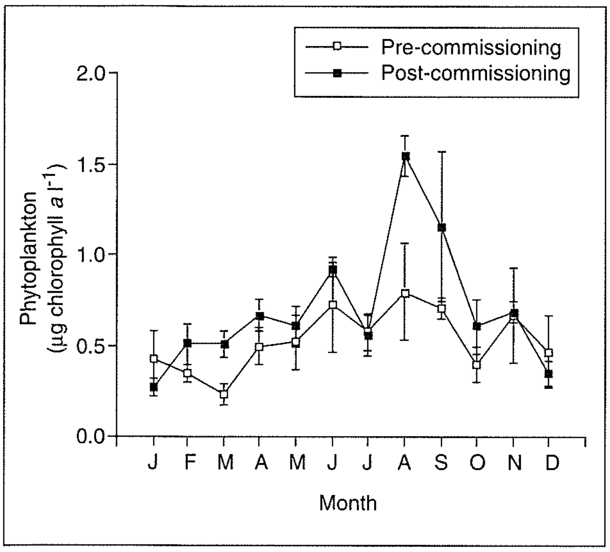


Figure 4.7-15. Composite annual patterns of chlorophyll a concentrations in Sepia Depression pre- and post-commissioning of the Cape Peron domestic wastewater outfall. Data for each month have been averaged.

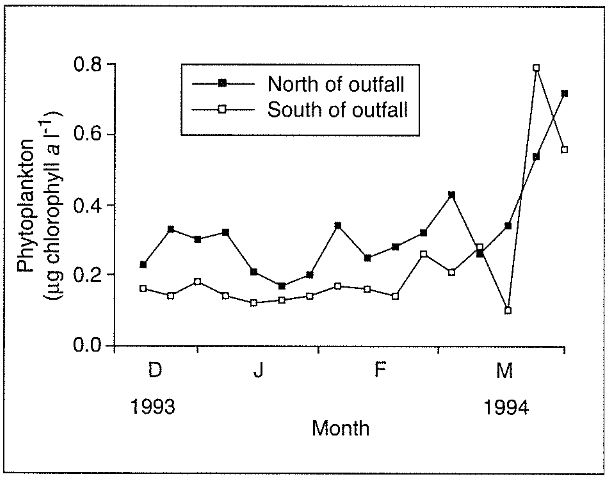


Figure 4.7-16. Weekly depth-averaged chlorophyll a concentrations at sites approximately 4 km north and 5 km south of the Cape Peron domestic wastewater outfall in 1993/94.

consumption) was found to be exceeded, on occasions up to 4 km from the outfall, suggesting a potential for contamination of shellfish living on reefs of the Shoalwater Islands Marine Park. However, limited data on bacteria levels in mussels, approximately 2 km from the outfall, indicate levels that were below those in the Australian Food Standards Code (National Food Authority, 1994) and that those mussels were safe for human consumption.

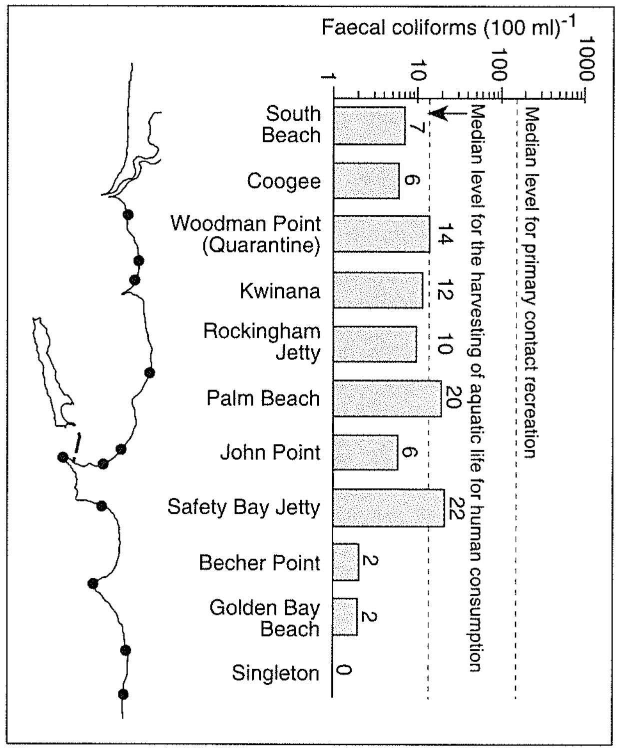


Figure 4.7-17. Median faecal coliform levels for Perth's southern metropolitan coastal beaches between 1991 and 1994.

Conclusions

Water quality data from Sepia Depression for faecal coliforms suggest a potential health risk exists within 2 km of the Cape Peron outfall, in relation to direct contact recreation, and within 4 km for the consumption of fish and other aquatic organisms. The consumption of shellfish collected from two beaches in the southern coastal waters poses potential health risks.

4.8 Phytoplankton

Phytoplankton are important primary producers in aquatic environments. In the offshore coastal waters of Perth (outside the embayments and lagoons) phytoplankton biomass is generally low by world standards and this is the result of the generally oligotrophic (i.e. nutrient-poor) status of the waters that occur off Western Australia (Pearce, 1991). Furthermore, the mediterranean climate of southwestern Australia results in seasonal, relatively low river flow and, therefore, low input of terrigenous material to Perth's coastal waters. Combined with the low phytoplankton abundance, this characteristic allows high light penetration through the water column during most of the year. As a consequence, benthic plant communities in these waters exist at greater depths than would be possible if the phytoplankton biomass and sediment input were higher.

The species composition and biomass of phytoplankton populations are sensitive to changing environmental conditions, both natural (e.g. seasonal) and unnatural (e.g. anthropogenic impacts). Short-term increases in nutrient loadings to coastal waters can result in rapid changes in phytoplankton populations, with many species able to double their biomass in less than 24 hours under favourable conditions. Similarly, the diversity and species composition of the phytoplankton can change as a result of long-term alterations in environmental conditions caused by chronic waste inputs (Hallegraeff, 1994). Furthermore, the phytoplankton population characteristics directly affect the penetration of light through the water column. Thus the biomass and species composition of phytoplankton populations may be useful, firstly, as short- and long-term indicators of environmental change, and secondly, as indicators of potential impacts on benthic plant communities as a consequence of light limitation.

Data on the species composition of phytoplankton communities of Perth's coastal waters are sparse. Chaney (1978) undertook an intensive study of the phytoplankton communities in Cockburn Sound during the Cockburn Sound Environmental Study (Department of Conservation and Environment, 1979). Phytoplankton biomass, expressed as chlorophyll *a* concentrations in seawater, has been regularly monitored in past water quality surveys, particularly in Cockburn Sound.

To gain a better understanding of the phytoplankton communities throughout the wider southern metropolitan coastal waters of Perth, and the factors that control these populations, intensive local-scale and complementary regional-scale studies were undertaken.

The objectives of these studies were to characterise the seasonal and inter-annual variation in the phytoplankton assemblages in Cockburn and Warnbro sounds over the period February 1991 to August 1994 and provide a more regional perspective of phytoplankton populations during 'typical' summer and winter conditions. These studies were undertaken in collaboration with the School of Environmental Biology at Curtin University.

The phytoplankton studies and complementary zooplankton data (see sections 4.9 and 5.5) also provide the information required to develop a phytoplankton species succession and growth model which is described in section 6.3.2. Further technical details can be found in Cousins (1991), Cousins and Masini (1992), Simpson *et al.* (1993), Helleren and John, (1994, 1995) and Masini and Cousins (1995).

Findings

4.8.1 Shelf-scale phytoplankton

Seasonal characteristics

A total of 130 phytoplankton taxa were initially recorded during the winter regional survey (13-17 August, 1991) of 44 sites (a sub-set of the sites shown in Figure 4.7-1) spread longshore between Mandurah and Marmion, and offshore to the continental shelf break. Upon further scrutiny, however, the number of phytoplankton taxa identified was reduced to 95, comprising two silicoflagellates (Chrysophyta), 20 dinoflagellates (Dinophyta), and 73 diatoms (Bacillariophyta). Of these taxa, only 32 (two silicoflagellates, four dinoflagellates and 26 diatoms) were considered to be abundant (i.e. comprised > 5% of total cell numbers at any site on any sampling occasion). One species, the silicoflagellate, *Dictyocha octonaria*, was dominant (i.e. most abundant of all taxa present) at 29 sites. A dinoflagellate, *Prorocentrum micans*, was relatively abundant at 25 sites, but was dominant at only two of these sites.

During the summer survey conducted on 9 March, 1992, a total of 85 taxa were found (two silicoflagellates, 20 dinoflagellates, 63 diatoms) and 48 of these were termed 'abundant', in that they individually comprised more than 5% of the total abundance (one silicoflagellate, 17 dinoflagellates and 30 diatoms). Dinoflagellates were common during the summer survey, with *Ceratium furca* being dominant at 30 of the 44 sites. *Prorocentrum micans* was abundant at 28 of the 44 sites but was dominant at only one site. Phytoplankton species richness was higher during winter than during summer, however the abundance was more evenly distributed across the species in summer, with over 50% of the total number of species being abundant, compared with 33% in the winter survey. Phytoplankton abundance in offshore Marmion waters (Figure 4.7-1) was an order of magnitude lower, but the diversity was greater, than in the other sub-regions of the study area during both summer and winter. The highest cell numbers were recorded near the mouth of the Swan-Canning Estuary during winter and in Cockburn Sound during summer (Table 4.8-1). These

(a) Relative Composition

Sub-region	Dinoflagellates (%)		Silicoflagellates (%)		Diatoms (%)	
	Winter	Summer	Winter	Summer	Winter	Summer
Swan-Canning Entrance	3.3	3.3	7.5	0.2	89.2	96.5
Cockburn Sound	36.2	5.4	36.6	0.0	27.2	94.6
Gage Roads/Owen Anchorage	22.2	76.2	35.4	0.7	42.4	23.1
Inshore Marmion	12.3	57.0	38.2	0.6	49.5	42.4
Offshore Marmion	13.8	82.0	26.5	0.0	59.7	18.0
Peel-Harvey Entrance	39.0	42.2	7.5	1.1	53.5	56.7
Comet Bay	22.0	8.8	40.6	1.6	37.4	89.6
Sepia Depression	8.8	24.7	69.5	0.5	21.7	74.8
Warnbro Sound	14.0	30.0	77.0	13.5	9.0	56.5
West of Sepia Depression	12.2	85.1	68.1	1.5	19.7	13.4
Mean	17.6	9.4	47.6	0.4	34.8	90.2

(b) Abundance

Sub-region	Dinoflagellates (cells l^{-1})		Silicoflagellates (cells l^{-1})		Diatoms (cells l^{-1})	
	Winter	Summer	Winter	Summer	Winter	Summer
Swan-Canning Entrance	3916	5580	8917	279	106083	165726
Cockburn Sound	15176	9196	15344	0	11448	161244
Gage Roads/Owen Anchorage	2754	1215	4387	10	5256	368
Inshore Marmion	1500	532	4676	6	6064	395
Offshore Marmion	77	123	148	0	334	27
Peel-Harvey Entrance	16009	936	3082	26	21992	1257
Comet Bay	4737	830	8738	150	8045	8428
Sepia Depression	4719	1445	37218	27	11648	4382
Warnbro Sound	9813	1186	54072	536	6353	2238
West of Sepia Depression	1936	746	10766	13	3111	117
Mean	4587	1338	12393	54	9080	12768

Table 4.8-1. Major phytoplankton groups in sub-regions of the study area during summer and winter: (a) relative composition and (b) abundance. Sub-regions are shown in Figure 4.7-1.

abundances are relatively high for coastal waters in Australia but are an order of magnitude lower than typically found in nutrient-enriched areas in other parts of the world (Hallegraeff, 1994).

Spatial distribution

The spatial distribution patterns of total phytoplankton abundance were quite different during summer and winter. During winter, the zone of high phytoplankton abundance (e.g. > 10 000 cells l^{-1}) extended offshore for about 30-40 km; during summer, however, equivalent concentrations were largely restricted to the embayments and the nearshore zone, to a maximum distance of about 15 km offshore (Figure 4.8-1a). The most striking seasonal difference in phytoplankton assemblage composition was in the abundance and distribution of the silicoflagellate *Dictyocha octonaria* (Table 4.8-1). During winter, the abundance of this species was 2 000 cells l^{-1} or more over the majority of the study area and over 50 000 cells l^{-1} in Warnbro Sound and in Sepia Depression offshore from Cape Peron (Figure 4.8-1b). In contrast, the abundance of this silicoflagellate was much lower during summer (maximum of 930 cells l^{-1}) and appeared to be linked to estuarine discharges from the Swan-Canning Estuary in the northern portion of the study area and either the Peel-Harvey or Leschenault estuaries (see section 4.8.2) in the south (Figure 4.8-1b).

Correspondence analysis identified groups of sites with similar phytoplankton assemblages within each season. Distinct 'inshore' and 'offshore' phytoplankton populations occurred to the north of the Swan-Canning Estuary mouth during winter. South of the Swan-Canning Estuary mouth, the phytoplankton populations were relatively uniform due to the dominance and high abundance of silicoflagellates. The influence of the Peel-Harvey Estuary was clearly evident as a gradient of phytoplankton species assemblage from the Peel-Harvey Estuary mouth at Mandurah, in the south, to Garden Island in the north, with adjacent sites having the greatest similarity in phytoplankton composition.

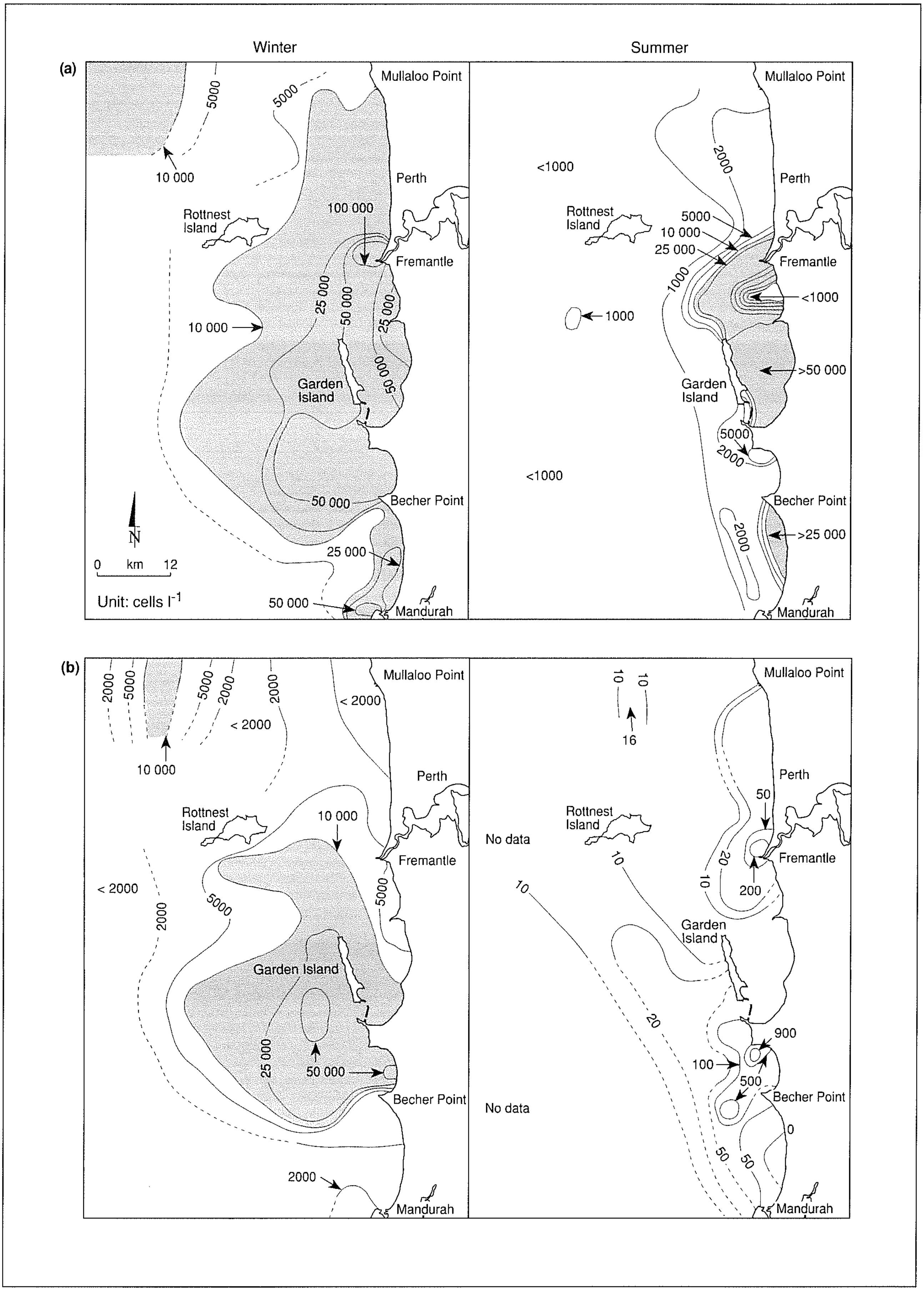


Figure 4.8-1. Distribution of (a) total phytoplankton abundance and (b) silicoflagellate abundance in surface waters during August 1991 and March 1992. Shading shows areas where cell numbers exceeded 10 000 cells l^{-1}.

By contrast, three main phytoplankton populations were identified in summer and were centred around the Swan-Canning Estuary entrance/Cockburn Sound/Carnac Island area, the inshore Marmion area and, to a lesser extent, the Comet Bay/Warnbro Sound/Garden Island area. The phytoplankton assemblage at the entrance to the Swan-Canning Estuary was very different from the majority of the other sites but showed affinities with the phytoplankton of Cockburn Sound (mainly due to common species of *Chaetoceros*) and also to an area west of Carnac Island due to the presence of *Skeletonema costatum*. The general similarity in phytoplankton of the Comet Bay/Warnbro Sound/Garden Island area was predominantly due to the presence of common *Rhizosolenia* species, suggesting a common linkage between these areas and the Peel-Harvey Estuary, as found during winter.

Bio-indicators of estuarine influences

The diatom *Skeletonema costatum* is the most common phytoplankton species blooming in the Swan-Canning Estuary during winter (John, 1987) and was abundant at the estuary mouth during the winter survey. This species has been recorded in the Peel-Harvey Estuary (J John, personal communication) but was not present near the entrance to this estuary during the winter survey. For these reasons *Skeletonema costatum* was used as a bio-indicator of the spatial influence of the Swan-Canning Estuary outflow in the coastal waters of Perth. The distribution of this species during the winter regional survey extended as far south as Warnbro Sound (Figure 4.8-2a) and closely matched simulations of the buoyant outflow of the Swan River under similar wind conditions (see section 6.2.4.5). Similarly, the distribution of *Peridinium oblongum* provided an indication of the spatial extent of influence of the Peel-Harvey Estuary outflow in the coastal waters at that time (Figure 4.8-2b).

The dominance of the phytoplankton populations by a single silicoflagellate species during winter, and by diatoms in general during summer, is consistent with the findings of the local scale surveys conducted over several annual cycles, and discussed below.

4.8.2 Local-scale phytoplankton

Assemblages and seasonality

The phytoplankton of Cockburn Sound and Warnbro Sound is comprised of over 300 taxa and has a relatively high diversity for temperate coastal waters (Helleren and John, 1995). Four major groups of phytoplankton were identified: Bacillariophyta (diatoms), Dinophyta (dinoflagellates),

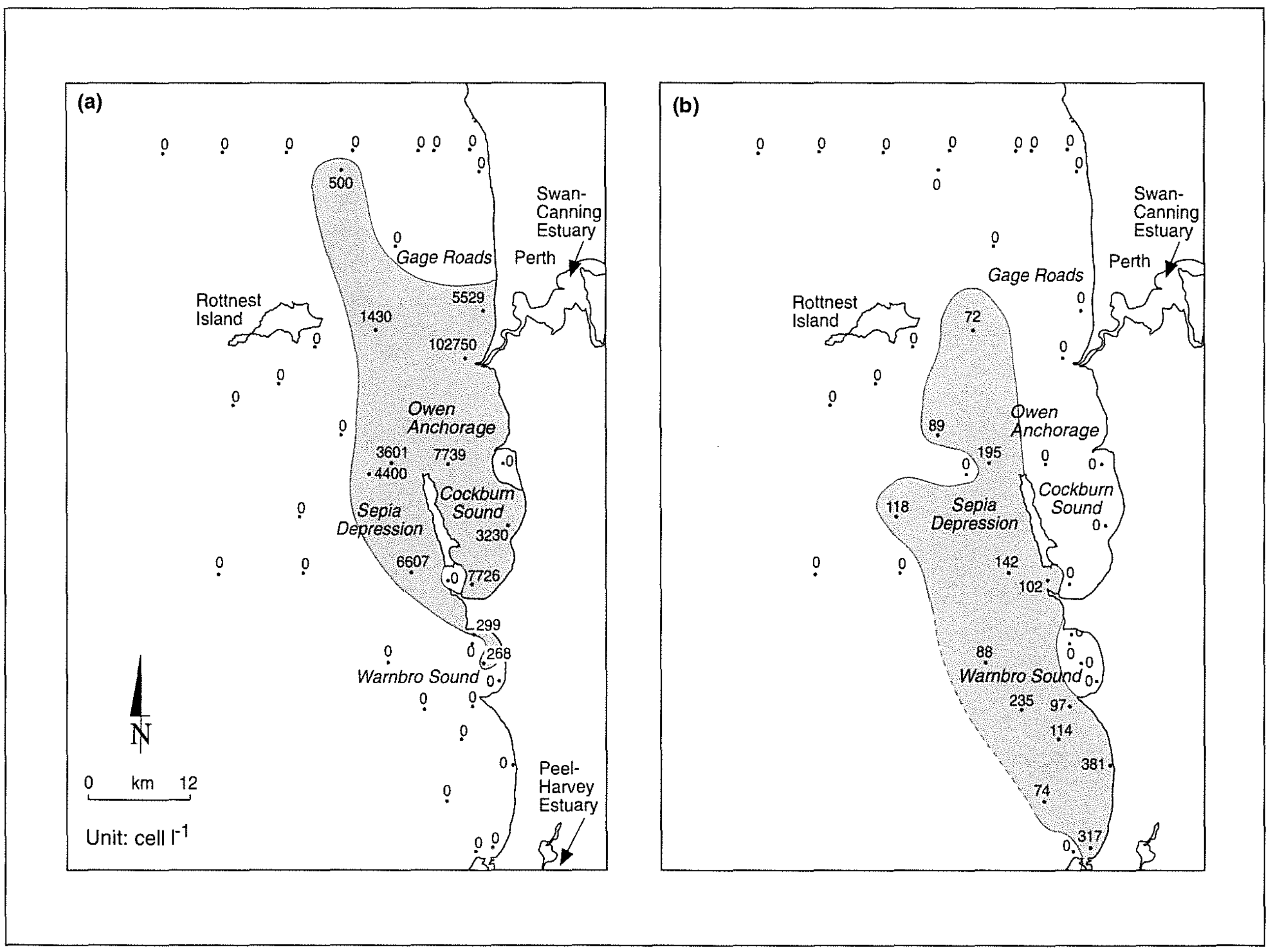


Figure 4.8-2. Distribution of (a) Skeletonema costatum and (b) Peridinium oblongum in surface waters during August 1991. Shading shows areas where these species occurred.

Chrysophyta (silicoflagellates) and the Cyanophyta (cyanobacteria). Diatoms were the most diverse group, consisting of over 200 taxa, and generally dominated by the genera *Chaetoceros*, *Leptocylindrus*, *Nitzschia* and *Rhizosolenia*. Dinoflagellates were less diverse, with approximately 45 taxa recorded, and were dominated by the genera *Ceratium*, *Dinophysis*, *Mesoporos*, *Scrippsiella* and *Prorocentrum*. The silicoflagellates were dominated by a single species, *Dictyocha octonaria*. The cyanobacteria were abundant at times; the dominant species, *Oscillatoria erythraea (=Trichodesmium)*, is widely distributed along much of the Western Australian coastline in both nearshore and offshore waters (Creagh, 1985). Benthic microalgae were usually present in bottom water samples, especially at shallow sites, but apart from this, there was no consistent vertical distribution pattern of phytoplankton species or assemblages at any site.

A broad pattern of phytoplankton assemblage succession, common to Cockburn Sound and Warnbro Sound, was observed over the period 1991-1994 (Figure 4.8-3). Dinoflagellates and diatoms were present throughout the year. Diatom blooms occurred during summer and autumn. During winter and spring, the phytoplankton assemblages were characterised by a monospecific silicoflagellate bloom. In 1991/92 the silicoflagellate bloom in Warnbro Sound re-initiated in December and continued through until late summer. Extensive surface slicks of the tropical blue-green planktonic alga *Oscillatoria erythraea* were common during late summer and early autumn. The phytoplankton of Sepia Depression were monitored over only one annual cycle, but the assemblage succession pattern was almost identical to that in the embayments, suggesting that this pattern is common to Perth's southern metropolitan coastal waters.

Spatial and temporal variability

Multivariate analyses indicated a greater similarity between sites within embayments than between embayments. The differences between embayments are partly due to subtle differences in the species composition, but were primarily due to differences in phytoplankton abundance, which was greater in Cockburn Sound than Warnbro Sound, and lowest in Sepia Depression.

There was some spatial heterogeneity in phytoplankton abundance within the embayments (see Figure 5.5-1a).

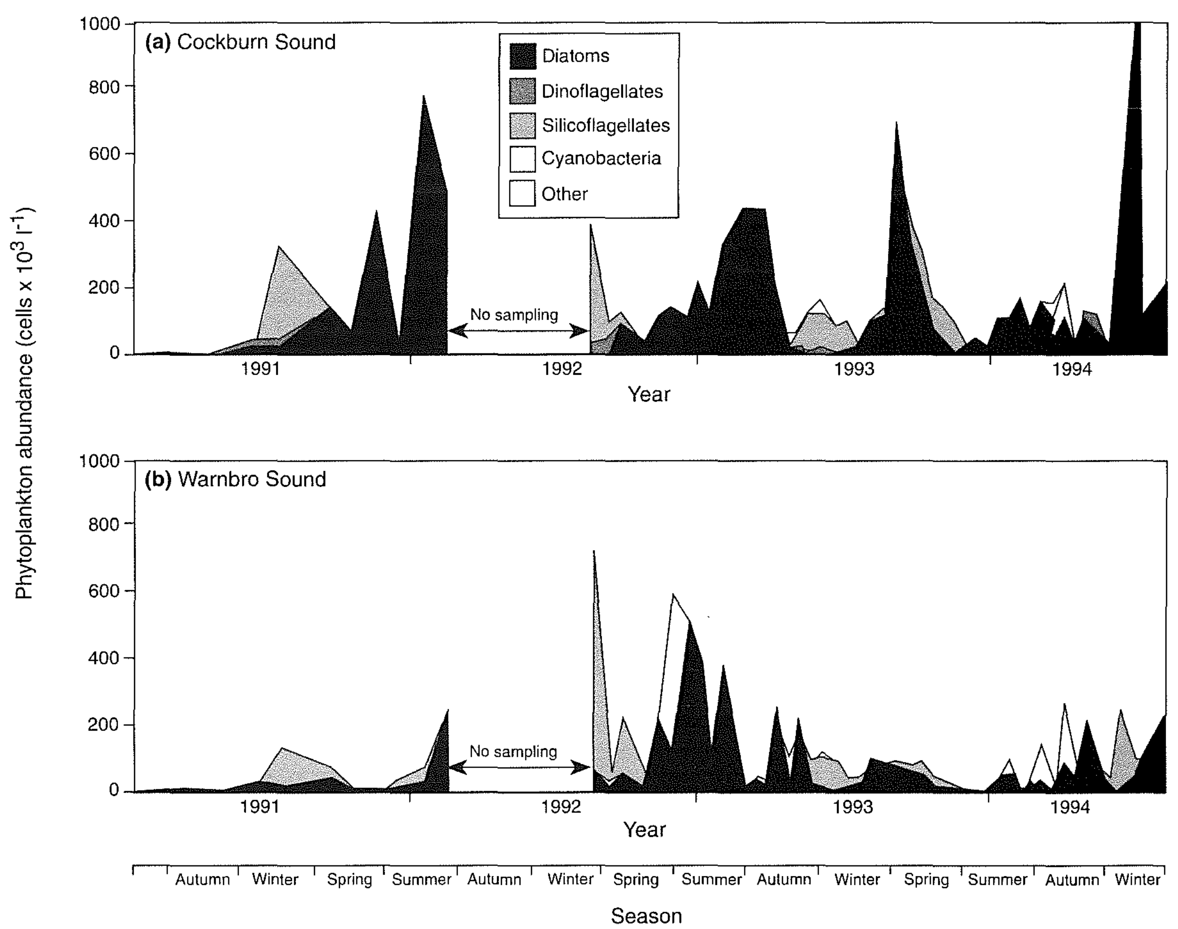


Figure 4.8-3. Seasonal succession of phytoplankton assemblages and abundance in (a) Cockburn Sound and (b) Warnbro Sound.

In Cockburn Sound the highest abundance was typically found in Mangles Bay, which is close to major sources of nitrogen and is poorly flushed with oceanic waters. The next highest abundance was found in Jervoise Bay, followed by the area off James Point. The lowest abundance was typically found in the northwest corner of the Sound, which is distant from major sources of nitrogen and relatively well-flushed with oceanic waters. In Warnbro Sound, the most northerly site near Mersey Point had a consistently lower abundance (~0.5 times) and had species that were not present at other sites in either Cockburn or Warnbro sounds. An estuarine species of cyanobacteria *Nodularia spumigena*, which has been observed to bloom in the Peel-Harvey Estuary during spring and summer, was found in high concentrations in southwestern Warnbro Sound during November 1992. This provides supporting evidence of a hydraulic connection between this estuary and Warnbro Sound (see sections 5.1.2 and 6.2.4.6) and highlights again the usefulness of certain phytoplankton species as indicators of the area of influence of estuarine outflows.

Phytoplankton biomass, measured as chlorophyll *a* concentration, was generally higher in winter than summer, although high chlorophyll *a* concentrations were occasionally found in association with summer blooms of diatoms. The maximum chlorophyll *a* concentration recorded was 17.3 $\mu g\ l^{-1}$ during early spring at mid-depth in Jervoise Bay and the lowest was 0.11 $\mu g\ l^{-1}$ during mid-summer in a surface sample from inshore central Warnbro Sound. In general however, the annual minima and maxima ranged from about 0.3 to 4.0 μgchl *a* l^{-1} in Warnbro Sound and from about 0.5 to 7.0 μgchl *a* l^{-1} in Cockburn Sound. These ranges are slightly smaller than the seasonal ranges of chlorophyll *a* reported for Sydney's coastal waters (0.1-15.0 μgchl *a* l^{-1}), Sydney Harbour (1.8-19.0 μgchl *a* l^{-1}) and Port Phillip Bay (0.1-30.0 μgchl *a* l^{-1}) but higher than the range 0.1-1.1 μgchl *a* l^{-1} expected in southeast Indian Ocean waters (Hallegraeff, 1994). Spatial patterns of phytoplankton biomass were very similar to patterns of phytoplankton abundance described above for both embayments.

Long-term changes

Successional patterns of phytoplankton species assemblages recorded by Chaney (1978) in Cockburn Sound between December and April of 1977/78 were similar to those found during the same period in the present study. However, unlike the silicoflagellate dominance characterising the late autumn/winter period in recent years, the phytoplankton assemblage of Cockburn Sound in the 1977 autumn/winter period was dominated by unidentified 'Chlorophytes' (likely to have been Prymnesiophytes). A bloom of Prymnesiophytes, similar to Chaney's descriptions, occurred in Cockburn Sound and Warnbro Sound during June 1993, but was far less persistent than the blooms recorded in 1977/78. In the spring of 1977 Cockburn Sound was dominated by a different species of 'Chlorophyte' and a species presumed to be a dinoflagellate (possibly *Gyrodinium*). In contrast, the spring period in the present studies was typically dominated by diatoms.

The silicoflagellate species *Dictyocha octonaria*, which dominated the winter assemblage between 1991 and 1994, has a distinctive morphology in its skeletal stage (Plate 4.8-1) which makes it conspicuous and unlikely to

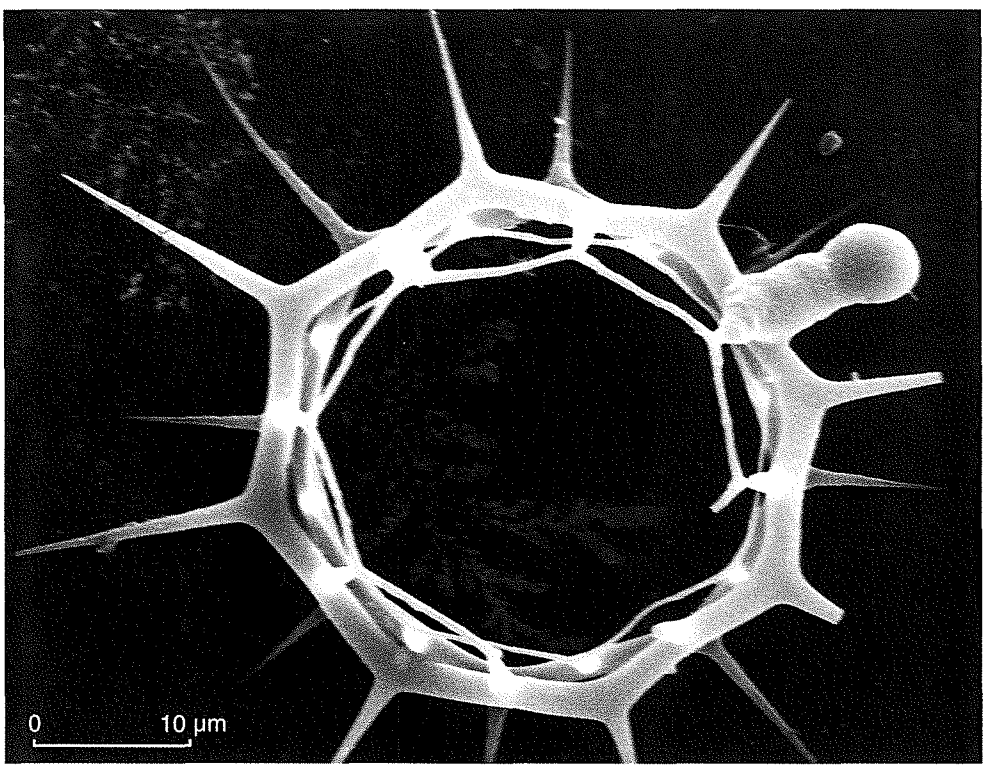


Plate 4.8-1. Scanning electron micrograph of the silicoflagellate, Dictyocha octonaria. (Photograph: J John).

have been overlooked or mis-identified by Chaney. In fact, this species was recorded sporadically through autumn 1978 but comprised < 5% of the phytoplankton abundance, which is consistent with levels typically found in other coastal waters around Australia (Jeffrey and Vesk, 1990). The marked dominance of silicoflagellates during winter (up to 95% of cells present) and the widespread distribution and high abundances (up to 8.5×10^5 cells l^{-1}) found in recent years, suggest that there has been a significant change in the winter phytoplankton assemblages in the study area compared with the patterns described in the late 1970s.

A closely related species, *Dictyocha speculum*, is common in the southern ocean and in northern European waters, where it usually occurs at densities of about 1×10^4 cells l^{-1}, but blooms at densities of up to 1.5×10^7 cells l^{-1}. One such bloom was reported to have been triggered by nutrient enrichment following abnormally high rainfall, and to have caused the death of fish in mariculture enclosures by physical damage to the gills (Moestrup and Thomsen, 1990). There is some evidence to suggest that the abundance of *D. speculum* has been slowly increasing in northern European waters and this may be a result of gradual and broadscale eutrophication (Moestrup and Thomsen, 1990). *D. octonaria* (listed as *D. speculum*) has also been recorded in Oyster Harbour, and in the Leschenault and Peel-Harvey estuaries, three Western Australian estuaries that are suffering from different levels of nutrient enrichment (Hosja and Deeley, 1994).

External influences

The majority of the phytoplankton taxa recorded during the present study were cosmopolitan temperate marine species, but several tropical species which are generally characterised by their large size and/or elaborate structure were also recorded (Helleren and John, 1995). The symbiotic ciliate *Vorticella oceanica* was found attached to the diatom *Chaetoceros coarctatus*, a relationship common in tropical waters. Similarly, the endosymbiotic cyanobacterium *Richellia intracellularis* was found within the diatom *Rhizosolenia clevei* during March 1994 in Warnbro Sound. During this time free-living *Richellia* filaments were observed in large numbers in Warnbro Sound and co-dominated the phytoplankton assemblage with the coccoid cyanobacterium *Synechococcus* sp.. *Richellia* was also recorded in Cockburn Sound at the same time but in much lower numbers. The large tropical species recorded in both sounds included *Coscinodiscus* (visible with the naked eye), *Bacteriastrum*, *Rhizosolenia*, *Eucampia* and *Chaetoceros* and these were most abundant during autumn. A tropical *Ceratium* species was consistently recorded during winter, but only in Cockburn Sound. The timing of occurrence of these species coincides with the period of strongest flow of the Leeuwin Current (Cresswell, 1991) indicating that this current influences the phytoplankton ecology of Perth's nearshore waters.

Two freshwater phytoplankton genera were found in the embayments, *Pediastrum* on the eastern margin of Warnbro Sound during June/July 1993 and *Scenedesmus* on the eastern margin of Cockburn Sound in August 1994, indicating freshwater runoff into these areas.

Potentially harmful species

Some phytoplankton are capable of producing powerful neurotoxins which affect humans who consume shellfish that have ingested these algae (Hallegraeff, 1993). Many of these phytoplankton species are cosmopolitan and some were found in the present study, but generally at relatively low abundances. The dinoflagellates *Alexandrium* and *Gymnodinium* were found in Cockburn Sound and Warnbro Sound and some strains are known to cause Paralytic Shellfish Poisoning (PSP) (Hosja and Deeley, 1994). *Alexandrium* occurred at all sites in Warnbro Sound and in Cockburn Sound (except Mangles Bay) and was most common during spring; the maximum abundance (~9 000 cells l^{-1}, 2.2% of total abundance) was recorded in bottom samples from Jervoise Bay in early spring 1993. *Gymnodinium* species were found at all sites and were most common during autumn and winter, but abundances rarely exceeded 1 000 cells l^{-1}.

The dinoflagellates *Prorocentrum micans* and *Dinophysis acuminata* were found in both embayments and are known to be responsible for Diarrhetic Shellfish Poisoning (DSP). The average abundances of *Prorocentrum micans* were 800 and 400 cells l^{-1} in Cockburn Sound and Warnbro Sound, respectively, during the 1992-1994 sampling period. The maximum abundance (24 000 cells l^{-1}; about 3.5% of total abundance) was recorded near James Point in late winter 1992. *Dinophysis acuminata* was commonly found throughout the year at average densities of 230 and 140 cells l^{-1} in Cockburn Sound and Warnbro Sound, respectively, and maximum densities of 11 600 cells l^{-1} (or about 1.2% of total abundance) were recorded in surface waters near Mangles Bay during mid-winter 1994.

Some strains of the cyanobacterium *Nodularia spumigena* produce hepatotoxins that are potentially harmful to animals and can accumulate in shellfish (Falconer *et al.* 1992). Although *Nodularia spumigena* originating in the Peel-Harvey Estuary has been recorded in high densities (up to 1.6×10^6 cells l^{-1}) in Warnbro Sound, it is considered unlikely that this brackish water species can survive for long in waters of oceanic salinity (J. John, personal communication).

Outbreaks of PSP and DSP have had serious repercussions on the viability of shellfish harvesting and farming industries in Europe, Asia and America (e.g. Hallegraeff, 1993), but the low abundances of the causative phytoplankton species and the absence of any reported poisoning of humans in local waters (M. Jackson, personal communication) suggest that there is no immediate cause for concern; however, the presence of these species is indicative of the potential for a problem to occur in the future.

Other species that were also recorded in the present study, such as the dinoflagellates *Dinophysis caudata*, *D. rotundatum*, *Scrippsiella*, *Amphidinium*, and the cyanobacterium *Oscillatoria erythraea*, have been implicated as the cause of fish mortalities or have been the cause of public health concerns elsewhere, but their toxicity status remains largely unresolved (Hosja and Deeley, 1994).

Conclusions

The presence of distinct seasonal phytoplankton assemblages and the recurring, cyclical patterns of succession of these assemblages allows diverse phytoplankton populations to be characterised and described simply, facilitating the development of efficient and focussed biological monitoring programs. This information is a prerequisite for the development of a phytoplankton species succession growth model.

The commonality in phytoplankton assemblages and the seasonal successional patterns in the two embayments and Sepia Depression suggest that factors common to the region control species assemblages and the successional pattern of these assemblages. The differences in phytoplankton abundance and biomass in the sub-regions of the study area suggest that factors specific to sub-regions control abundance and biomass. Further, consistent spatial patterns of highest phytoplankton abundance and biomass near the areas of major nutrient discharge within Cockburn Sound suggest a direct relationship between nitrogen discharge and phytoplankton biomass.

Certain estuarine phytoplankton species can be useful as biological indicators of the scale of influence of estuarine discharges. The timing of occurrence and the distribution of tropical phytoplankton species can provide information on the influence of the Leeuwin Current in Perth's coastal waters.

There are two possible explanations for the apparent recent (< 15 years) dominance of silicoflagellates during winter in Perth's southern metropolitan coastal waters. The literature suggests that high concentrations of dissolved heavy metals can inhibit algal growth and that diatoms, dinoflagellates and cyanobacteria are the most sensitive, and the green flagellates the least sensitive (Hallegraeff, 1994). During the late 1970s, loadings of heavy metals to Cockburn Sound were generally higher than current loadings (see section 4.4) and the apparent dominance by flagellates during 1977 may reflect a competitive advantage given to them as a result of heavy metal pollution. Alternatively, the current dominance and broad-scale distribution of silicoflagellates may be an indication of chronic, low-level anthropogenic influences operating at a larger scale which either stimulate silicoflagellate growth or reduce grazing pressure, but data from outside of Perth's coastal waters are required to establish whether this is actually the case.

The observations of large surface-slicks of species such as *Oscillatoria erythraea*, apparently advected into the embayments from offshore, indicates that the presence of high abundances of phytoplankton in a particular environment does not necessarily imply that the phytoplankton grew in that area in response to local stimuli. These advected species, when present, raise chlorophyll *a* concentrations, complicating the interpretation of water quality monitoring programmes, and can lead to a false conclusion that water quality has deteriorated due to local factors (e.g. waste inputs) unless the monitoring is conducted over the appropriate spatial and temporal scales.

4.9 Zooplankton

Zooplankton are primary grazers of phytoplankton and can therefore indirectly influence the amount of light reaching benthic plant communities. The species composition of zooplankton populations is also sensitive to changing environmental conditions. Hence the zooplankton are potentially useful indicators of environmental change.

There have been no intensive studies of the zooplankton populations of Perth's coastal waters although some work was conducted during the early 1970s (Environmental Resources of Australia, 1971a,b) with the intent of using the abundance of selected species as indices of water residence times in Cockburn Sound. These studies found a higher abundance of zooplankton in the waters of Cockburn Sound than adjacent offshore waters and that the degree of vertical stratification in abundance was small and diel migration was limited.

To better understand the nature of zooplankton communities in the southern metropolitan coastal waters of Perth and the factors that control these populations, intensive local-scale studies have been undertaken as part of the SMCWS. The objectives of these studies were to characterise the seasonal and inter-annual variation in the zooplankton assemblages in Cockburn and Warnbro sounds over the period August 1992 to August 1994. These studies, with complementary phytoplankton data (see sections 4.8 and 5.5) also provide information to help develop a phytoplankton species succession and growth model for these waters.

Further technical details can be found in Simpson *et al.* (1993); Helleren and John (1994); and Helleren and John (1995).

Findings

Characteristics

A total of 124 zooplankton taxa were identified in the samples collected from Cockburn Sound and Warnbro Sound between 17 August 1992 and 15 August 1994. These taxa represent the invertebrate phyla Annelida, Arthropoda, Bryozoa, Chaetognatha, Ciliophora, Cnidaria, Ctenophora, Echinodermata, Mollusca, Nematoda, Platyhelminthes, Sarcodina and the vertebrate phylum Chordata. In addition, 22 unidentified invertebrate taxa were present.

The most common invertebrate phylum, Arthropoda, included 90 taxa of crustaceans which accounted for over 72% of all the taxa recorded. The crustaceans included cladocera, ostracods, copepods, cirripedes, decapods, mysids, isopods and amphipods. Of these, the copepods were the most abundant and diverse. The decapods had the next highest diversity but they were never as abundant as the copepods. Twenty-five copepod taxa were recognised and, for counting purposes, the individuals were grouped into five size classes, but their taxonomy was not resolved beyond the Order level. A comparison of the taxa found in both sounds, excluding copepods, indicated a high degree of commonality in composition, with 71% of the taxa common to both Cockburn Sound and Warnbro Sound. Fourteen per cent were exclusive to Cockburn Sound and 15% were exclusive to Warnbro Sound.

Abundance

Copepods numerically dominated the zooplankton fauna. Three copepod families (cyclopoida, calanoida, harpacticoida) were present in both Cockburn and Warnbro sounds, and of these, the calanoids were most abundant (Figure 4.9-1a,b). Over 80% of adult copepods were in the two smallest size classes (< 0.75 mm) and this pattern was similar for each of the copepod families, however the abundance of each size class in Warnbro Sound was generally about half of that found in Cockburn Sound (Figure 4.9-1c).

Cladocerans were the second most abundant crustaceans in both sounds and were comprised predominantly of *Penilia avirostris* and seven taxa of *Podon* (Figure 4.9-2a,b). The abundance of *Penilia* adults in Cockburn Sound was twice that found in Warnbro Sound (Figure 4.9-2c); the same pattern as that found for the copepods.

The third major zooplankton group was the protozoa, and radiolarians comprised over 80% of this group. Of the remaining zooplankton groups, molluscs (usually mussel veliger larvae), tunicates and polychaetes were common, but were minor components of the zooplankton assemblage, comprising less than 10% of the total.

The abundance of zooplankton was typically greater in Cockburn Sound than in Warnbro Sound, however extremely high abundances of radiolarian protozoans (up to 2.7×10^5 m^{-3}) were encountered throughout Warnbro Sound (excluding the northernmost site) on 15 August 1994. The maximum abundance recorded in Cockburn Sound on the same day (1.2×10^4 m^{-3}) was an order of magnitude lower. This sampling day was considered to be atypical (see Figure 4.9-3) and to heavily bias calculations of average zooplankton abundance. When this sampling date was excluded, the basin-scale average of zooplankton abundance in Cockburn Sound was 2.3 times higher than in Warnbro

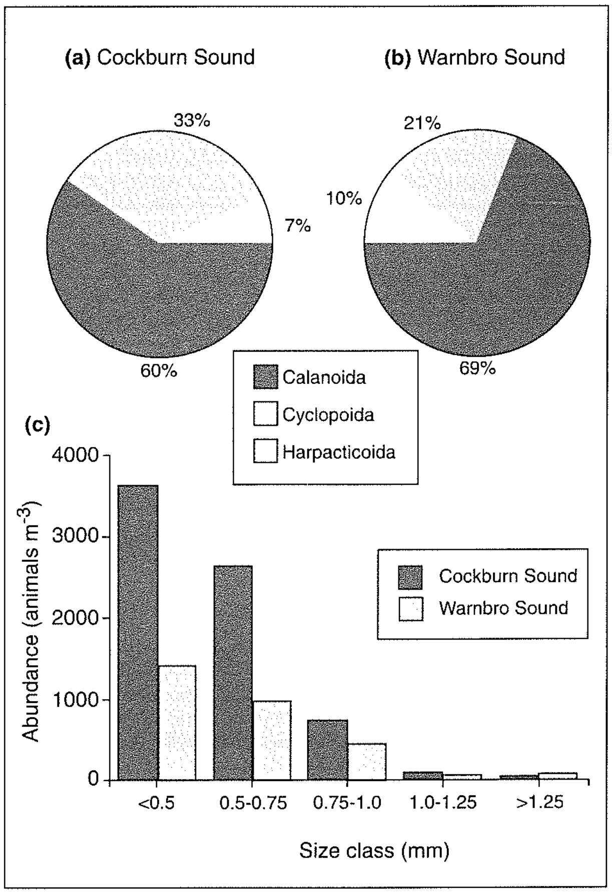


Figure 4.9-1. Relative abundance of the major copepod families in (a) Cockburn Sound and (b) Warnbro Sound and (c) comparative abundance of the five size classes of copepods.

Sound. The lowest zooplankton abundance in Cockburn Sound was typically found in the northwestern corner and, when this site was excluded, the average zooplankton abundance in Cockburn Sound was calculated to be 2.6 times that in Warnbro Sound. Within Cockburn Sound, zooplankton abundance tended to be highest in Jervoise Bay, and was approximately two times higher at this site than at the northwestern corner of the sound (see Figure 5.5-1b).

Seasonality

The seasonal abundance of individuals (including adults and juveniles) of each of the major zooplankton groups is shown in Figure 4.9-3 for both embayments. Peaks in copepod abundance tended to occur during spring and summer in both embayments however, in Cockburn Sound, an additional peak occurred during autumn. The copepod abundance during these 'blooms' was highest in Cockburn Sound and typically double that found in Warnbro Sound. Cladocerans were never as abundant as copepods but showed similar seasonal trends, including an additional peak during autumn in Cockburn Sound. Radiolarian blooms tended to be more seasonally defined, starting in mid-winter and peaking in late winter/early spring. In two out of the three blooms recorded, the peak radiolarian abundances in Warnbro Sound were an order of magnitude higher than equivalent peaks in Cockburn Sound. These peaks tended to

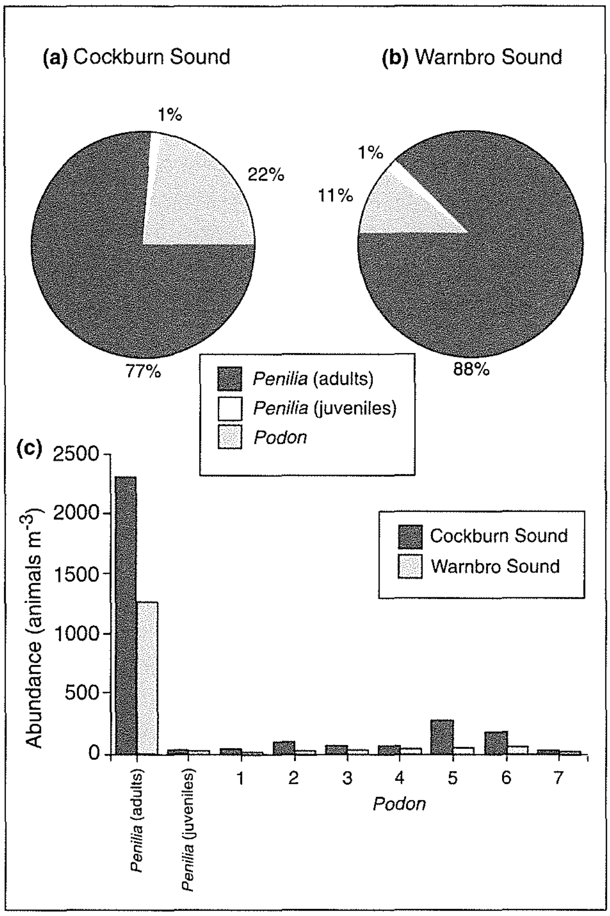


Figure 4.9-2. Relative abundance of the major cladoceran taxa in (a) Cockburn Sound and (b) Warnbro Sound and (c) comparative abundance of Penilia adults and juveniles and Podon taxa 1-7.

coincide with the decline in the winter silicoflagellate blooms (see sections 4.8 and 5.5).

Long-term changes

Complete species lists of zooplankton collected during the early 1970s are not available. This precludes an assessment of long-term variation, but the limited data that are available suggest that the zooplankton of the sound was probably dominated by copepods, as it is now (Environmental Resources of Australia, 1971a,b). The copepods *Acrocalanus/Paracalanus* and the cladoceran *Penilia* were the dominant species present during May 1971 and these typically occurred at densities of approximately 4 000 m^{-3} and 3 000 m^{-3}, respectively, in surface waters in Mangles Bay and the southern end of the Cockburn Sound deep basin, and at densities of approximately 1 300 m^{-3} and 500 m^{-3}, respectively, in the northern deep basin and eastern edge of the basin. Eight other species were each recorded at densities of approximately 100 m^{-3} at the same time. In the present study, *Penilia avirostris* was also found at lower densities in the northern part of the sound (190 m^{-3}) than in the southern end (481 m^{-3}) during May 1993/94, however the abundances were lower than found during 1971. The average abundance of *Penilia* in Cockburn Sound over the study period was 2 300 m^{-3} and was consistently higher along the eastern margin (2 860 m^{-3}) than at the northwestern end (990 m^{-3}) which is more consistent with both the patterns and abundances recorded during the early 1970s. Data from the present study were generated from 24 monthly cruises and provide a more reliable estimate of abundance compared with the 1970s data which were based on the results of two cruises conducted two days apart.

During the present study, sampling was only conducted during daylight hours and there were no significant differences in abundance between surface, middle and bottom waters at sites in either embayment. The low relative zooplankton abundance in the northwest of Cockburn Sound (see Figure 5.5-1b) where exchange with offshore waters is greatest, is consistent with the conclusion (Environmental Resources of Australia, 1971a,b) that the highest zooplankton abundances were associated with waters that had the longest residence times in the Sound.

Egg and larval abundance and seasonality

The seasonal timing of peak abundance of the eggs and larvae in the two sounds are presented in Table 4.9-1. The timing of occurrence of eggs and larvae varied among the zooplankton groups and also between embayments for some groups. Mollusc, polychaete and copepod larvae were very common, occurring at all sites on practically all sampling occasions. Cirripede (barnacle) nauplii, crab zoea and echinoderm eggs and larvae were encountered at all sampling locations on most occasions. Cirripede cypris, penaeid nauplii and juveniles of *Penilia* occurred at most sites but on less than half of the sampling occasions, and crab megalopa were rare, being encountered at mid-depth at one site in each embayment, but at different times of the year (Table 4.9-1).

The abundance of larvae and eggs in Cockburn Sound was higher and more than twice that found in Warnbro Sound for most groups apart from echinoderm larvae. The average number of echinoderm eggs was higher in Cockburn Sound than in Warnbro Sound whereas the larvae were more commonly encountered in Warnbro Sound and occurred in similar numbers on average in both embayments. A recent study comparing the benthic fauna of the deep basins of Cockburn and Warnbro sounds found higher abundances of echinoderms in Warnbro Sound than in Cockburn Sound and additionally, a north-south gradient of decreasing abundance of echinoderms within Cockburn Sound (Cary *et al.* 1995c).

Comparisons with other regions

With the exception of the large radiolarian blooms during late winter and early spring, the zooplankton fauna of Cockburn and Warnbro Sounds were not unusual by world standards. Both holoplankton (full planktonic life cycle) and meroplankton (partial planktonic life cycle) components are similar to those found in other temperate coastal regions of the world and off the New South Wales coast near Sydney (Kingsford, 1995).

The zooplankton succession off Perth is generally characterised by the following stages: overall low abundances in winter, increasing toward the end of winter; highest larval abundance during spring; a rapid increase in cladoceran numbers during summer; and abundant cladocerans and copepods in autumn. Similar trends to these have been recorded from Sydney's coastal waters (Kingsford, 1995) and other temperate regions, such as the Baltic Sea, Narragansett Bay and Chesapeake Bay, USA (Martin 1965, 1970; Kivi *et al.* 1993; White and Roman, 1992).

Penilia avirostris, the most abundant cladoceran in Cockburn Sound and Warnbro Sound, is also regarded as being the most abundant, cosmopolitan cladoceran world-wide. It is frequently abundant in warm and productive nearshore waters of the tropics and subtropics (Wong *et al.* 1992). In temperate waters, *P. avirostris* exists seasonally in large numbers in polluted estuaries (Yoo and Kim, 1987).

In coastal areas off Sydney, *P. avirostris* is present at concentrations of over 50 000 m^{-3} during summer and early autumn but it is rarely found during winter (Kingsford 1995). In Tolo Harbour, Hong Kong, abundances of *P. avirostris* averaged 4 900 m^{-3} during early summer and 4 500 m^{-3} during late autumn (Wong *et al.* 1992). In Cockburn and Warnbro sounds (present study), *P. avirostris* was also most abundant during summer and autumn, but was present for most of the year. The average annual abundances of *P. avirostris* adults in Cockburn Sound and in Warnbro Sound were 2 300 m^{-3} and 1 300 m^{-3}, respectively. These are considerably lower than values recorded in the coastal waters off Sydney, but comparable to those recorded in Hong Kong.

Planktonic radiolarians are widely distributed throughout most of the world's oceans but there is little available information on their ecology in nearshore coastal waters anywhere in the world. In highly productive, offshore oceanic regions, abundances of radiolarians may vary from several dozen to more than 10 000 m^{-3} (Anderson, 1993). In contrast, the abundances of radiolaria in the nearshore coastal waters off Perth (the present study) were about 10-25 times higher and reached more than 270 000 m^{-3} in Warnbro Sound. Analysis of available data on spatial and vertical distribution of radiolarians in the open ocean suggests an association with cooler waters, as they are found throughout the water column in high latitudes and at

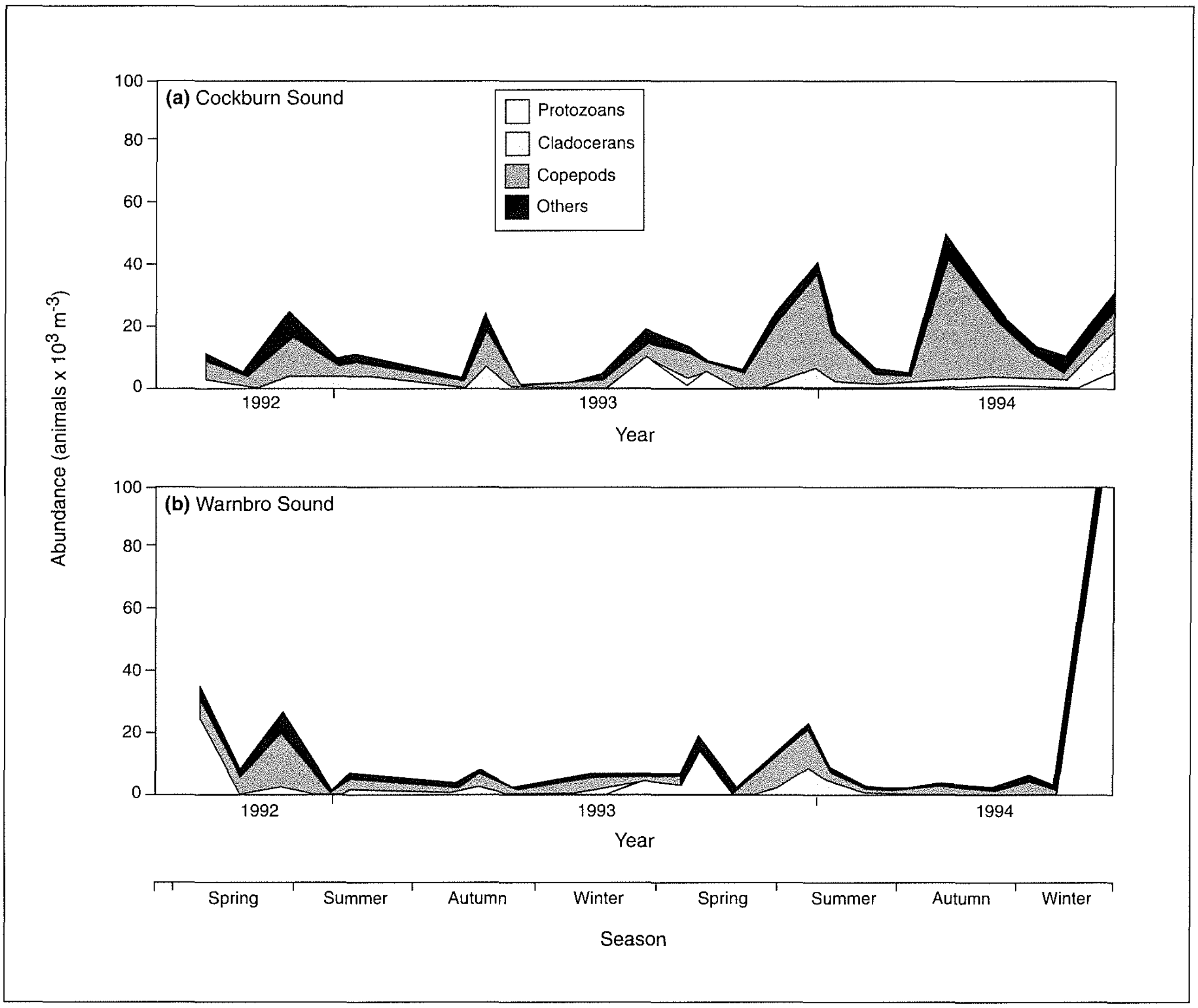


Figure 4.9-3. Seasonal succession of zooplankton assemblages and abundance in (a) Cockburn Sound and (b) Warnbro Sound.

Table 4.9-1. Seasonal timing of peak abundance of planktonic eggs and larvae, mean abundance when present and frequency of occurrence in Cockburn Sound and Warnbro Sound between August 1992 and August 1994.

Group	Seasonal Peaks		Mean abundance when present (individuals m^{-3})		Comments and frequency of occurrence (times present/times sampled)
	Cockburn Sound (CS)	Warnbro Sound (WS)	CS	WS	
cirripede nauplii	autumn/winter	autumn/winter	404	29	common, found at all sites, CS $^{24}/_{24}$, WS $^{21}/_{24}$
cirripede cypris	winter	winter	136	12	uncommon, most sites, CS $^{9}/_{24}$, WS $^{4}/_{24}$
crab zoea	early summer	early summer	108	52	common, all sites, CS $^{18}/_{24}$, WS $^{20}/_{24}$
crab megalopa	mid autumn	early summer	182	13	rare, found on only one occasion
echinoderm larvae	late spring/late winter	autumn	76	79	common, all sites, CS $^{15}/_{24}$, WS $^{22}/_{24}$
echinoderm eggs	spring/autumn	spring/autumn	106	67	common, most sites, CS $^{17}/_{24}$, WS $^{16}/_{24}$
penaeid nauplii	summer	summer	31	15	uncommon, only found in north CS, central/east WS, CS $^{10}/_{24}$, WS $^{7}/_{24}$
Penilia juveniles	autumn	late winter/early spring	200	91	uncommon, all sites, CS $^{11}/_{24}$, WS $^{8}/_{24}$
lamellibranch velligers	late spring/early winter	late spring/early winter	652	395	very common, all sites, all occasions
polychaete larvae	autumn/summer	winter spring	191	58	very common, all sites, CS and WS $^{23}/_{24}$
copepod nauplii	late winter/early spring	late winter/early spring	1127	639	very common, all sites, all occasions

progressively greater depths as latitude decreases (Anderson, 1993). Although similar information is not available for radiolaria in nearshore coastal waters, the high abundances during winter and the low abundance or absence during the remainder of the year, found during the present study, also suggests an association with lower temperatures.

Conclusions

Apart from the radiolarian 'blooms' during the winter months, zooplankton species composition and seasonal succession patterns are not unusual by world standards. Zooplankton abundance was about 2.5 times higher in Cockburn Sound than Warnbro Sound and this is presumably in response to greater food (i.e. phytoplankton) availability. Similar differences between embayments were found for the abundance of juveniles of most groups.

The abundance of radiolarians recorded in the present study were more than an order of magnitude greater than those considered normal for productive oceanic areas. Considering their apparent association with silicoflagellates, these radiolarians may be indicators of broadscale environmental change. Whether these radiolarians are responding to the same changes as the silicoflagellates or responding to the silicoflagellates themselves (see section 5.5) is not known.

Information on the timing of larval recruitment of the major zooplankton groups, including the commercially important groups such as crabs and mussels, is useful for management. Information on the timing of larval recruitment may also assist in combating the spread of nuisance species such as the exotic fan worm, *Sabella* cf. *spallanzanii*, which has spread over much of Port Phillip Bay (Bonny, 1995) and has been recently discovered in Cockburn Sound (see section 4.10.3).

4.10 Benthic invertebrate fauna

4.10.1 Deep basin fauna

The fine, deep basin sediments of Cockburn Sound and Warnbro Sound occupy about 50-60% of the benthic habitat areas of these embayments and are not found elsewhere along the central coast of Western Australia (Wilson *et al.* 1978). The benthic faunal communities associated with these sediments are considered unique because the area is a zone of overlap between the tropical fauna from the north and the temperate fauna from the south (Wilson and Gillett, 1971; Wilson and Stevenson, 1977). Pollutants tend to accumulate in the fine organic-rich sediments that characterise relatively deep, sheltered basins and, as a result, these substances have the potential to affect benthic communities. A study of the benthic invertebrate fauna of the deep (> 10 m) basins of Cockburn Sound and Warnbro Sound was undertaken in collaboration with the WA Museum. The study provides a quantitative description of the species composition, abundance and biomass of the benthos of these basins, determines if the distribution of the benthos is related to sediment characteristics, contaminant concentrations and/or ecological process changes, determines if temporal changes in species composition and abundance of selected taxa in Cockburn Sound have occurred since earlier studies (Wells, 1978; Wells and Threlfall, 1980), and provides baseline data for future reference.

A pilot study was conducted in March 1993 to assess the small-scale spatial patchiness of the benthic invertebrate fauna in Cockburn Sound in relation to sampling design. A pilot trial determined that four replicates would sample 70-80% of the fauna at a site. This was later supported by the study of Chalmers (1993). The field programme for the main study was undertaken in early April 1993 to coincide approximately with the seasonal timing of the study of Wells (1978). Forty-five sites were distributed evenly throughout the two embayments and specific site locations were based on the deep basin sites of the contaminant surveys of sediments undertaken in 1991 and 1994 (Burt and Ebell, 1994; Burt *et al.* 1995e). Taxa of the phyla Chordata, Echinodermata, Arthropoda (Class: Crustacea), Annelida (Class: Polychaeta), Mollusca and Cnidaria were identified to species level by taxonomists from the WA Museum, Australian Museum and the University of Western Australia.

Further technical details can be found in Cary *et al.* (1995c, 1995d).

Findings

Over 40 000 individuals consisting of 222 species from the six major invertebrate phyla were recorded. These numbers are comparable with the soft substrate deep basin habitat of Port Phillip Bay, Victoria (Poore and Rainer, 1979). The benthic invertebrate fauna in Warnbro Sound is distinct from Cockburn Sound with only 54% of the species recorded common to both basins. A distinct difference also occurs within Cockburn Sound, where only 45% of the species recorded in the northern half of the Sound were found in the southern half. Species richness and diversity indices were similar at sites throughout Warnbro Sound. In contrast, a decreasing southward trend in species richness and diversity occurred within Cockburn Sound, with the northern half of the sound being significantly higher than the southern half (Figure 4.10-1). Species richness and diversity indices divided the study area into three regions; Warnbro Sound, north Cockburn Sound and south Cockburn Sound, with Warnbro Sound having the highest species richness and diversity and south Cockburn Sound having the lowest.

Multivariate statistical analyses indicate that the benthic invertebrate fauna in these basins occurs in four distinct groups (Groups I - IV, Figure 4.10-2) and the spatial arrangement of these groups is consistent with the major spatial differences in species richness and diversity, described above. Based on species composition and abundance, the communities that occur in the sediments of Warnbro Sound and closest to the northern 'opening' of Cockburn Sound (Groups I and II) were most dissimilar to the species-poor, less abundant communities of the southern half of the Sound (Group IV). A comprehensive range of sedimentological and contaminant data obtained at each site were examined to assess whether these distribution patterns were statistically related to sediment characteristics, such as grain size, and/or the concentration of toxic contaminants in the sediments. The results of these analyses suggest that the broad difference between Warnbro Sound and Cockburn Sound, and the difference between the northern and southern halves of Cockburn Sound (Group III & IV) were related to the differential contamination of sediments by toxic substances (Figure 4.10-3). In Warnbro Sound benthic faunal communities appear to be more related to the differences in sediment mineralogy.

Univariate analyses supported these conclusions. Heavy metal contamination of the sediments in Cockburn Sound were inversely related to the species richness of the invertebrate fauna living in these sediments, indicating a possible cause-effect relationship (Figure 4.10-4). Many of these contaminants were co-related with each other and with sediment characteristics such as organic carbon, water and sulphur content and particle size. Furthermore, direct evidence suggested that the sediments in the far southern end of Cockburn Sound can be anoxic (Bastyan and Paling, 1995; section 5.6). Such associations confound attempts to identify specific cause-effect relationships between contaminants and the community structure of the benthic fauna in the sound. However, similar relationships between heavy metal contamination and sediment characteristics were not apparent in Warnbro Sound and the sediment characteristics of the deep basins of Warnbro and Cockburn Sounds are similar. These data suggest that the 'cocktail' of toxic contaminants in the sediments influence the structure of benthic fauna communities of Cockburn Sound. Episodic anoxia events may be an additional influence, particularly in the southern part of the Sound (see section 5.6).

Comparisons between the benthic molluscan communities in the deep basin of Cockburn Sound recorded in the 1978 survey by Wells, and in the 1993 study, indicated that significant changes have occurred over this period. Twenty-four species of molluscs were recorded in Cockburn Sound in 1978 compared to 40 species in 1993, with approximately 30% of the species recorded in 1993 being common to both surveys. *Tellina cockburnensis* and the sulphur reducing bivalve, *Solemya* sp were abundant in the 1993 study, particularly in the southern end of the sound, but were uncommon or significantly lower in numbers in 1978 (Figure 4.10-5). Species richness and diversity indices were higher in 1993 than 1978 in the northern half of Cockburn Sound, but lower in the southern half, indicating that there has been an improvement in benthic fauna species richness in the northern half of the sound and a decline in the southern half (Table 4.10-1). It is suggested that the cessation of routine domestic wastewater discharge into the northeastern corner of the Cockburn Sound basin has contributed to an increase in species numbers in the northern end of the sound. The decline in species richness in the southern end of the sound is most likely due to historical and/or current concentrations of toxic substances in the sediments combined with episodic anoxia events.

Conclusions

The benthic faunal communities of Warnbro Sound and the northern and southern halves of Cockburn Sound are dissimilar, with Warnbro Sound having the highest species richness and the southern half of Cockburn Sound having the lowest. It is suggested that a suite of toxic contaminants in the sediments have influenced the structure of benthic

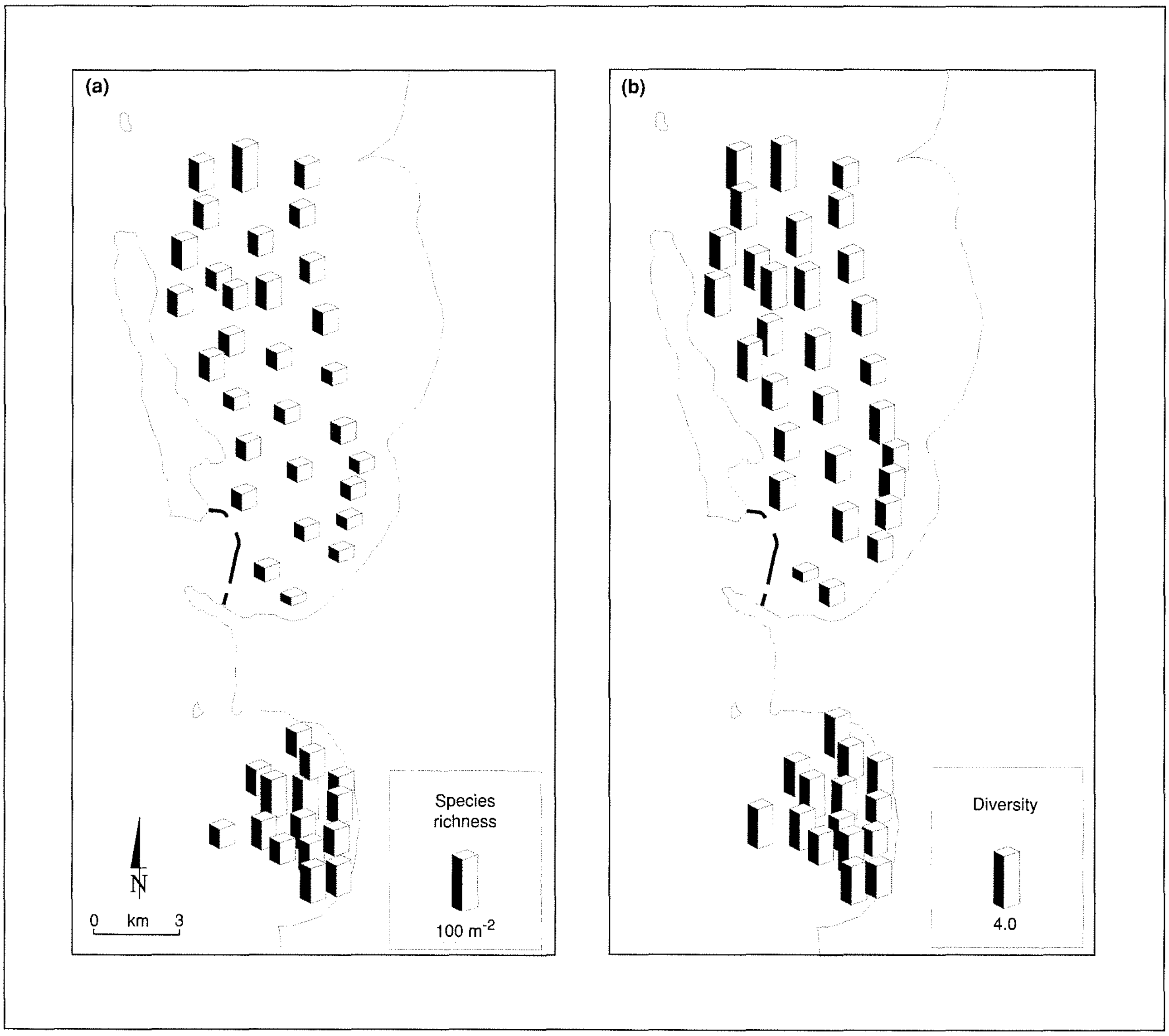


Figure 4.10-1. Benthic invertebrate faunal (a) species richness and (b) diversity in the basins (> 10 m depth) of Cockburn and Warnbro sounds.

faunal communities of Cockburn Sound and that episodic anoxia may be an additional influence, particularly in the southern end of the sound.

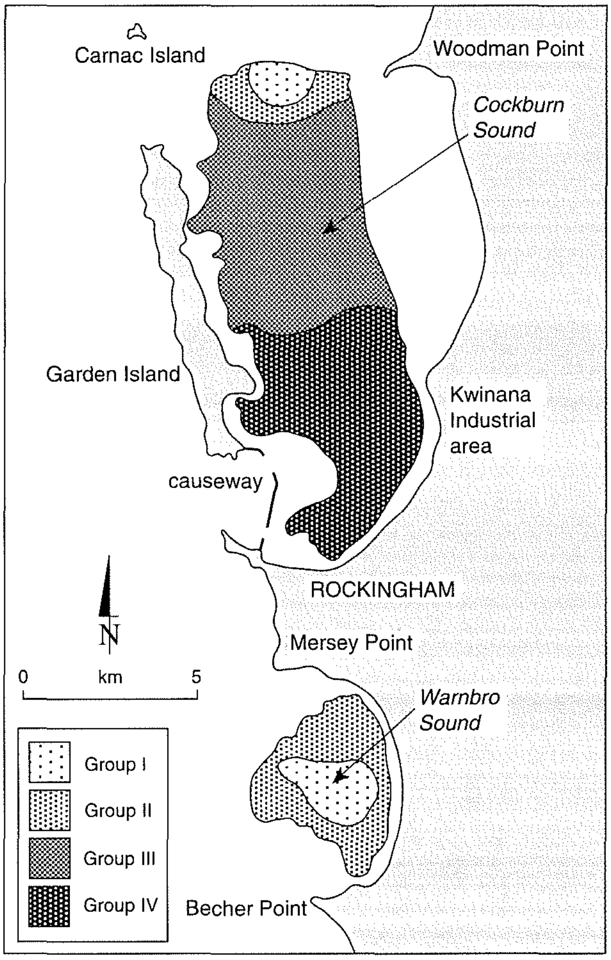


Figure 4.10-2. Classification of benthic invertebrate fauna in the basins (> 10 m depth) of Cockburn and Warnbro sounds.

Significant changes have occurred to the benthic molluscan communities between 1978 and 1993. Species richness and diversity indices suggest that the southern end of the sound was more disturbed in 1993 than in 1979, while the northern end was less disturbed. It is suggested that the diversion of a routine domestic wastewater stream which formerly entered the northeastern corner of the Cockburn Sound deep basin has contributed to the increase in species richness in the northern end of the sound, and further, that the suite of toxic contaminants in the sediments, combined with episodic anoxia events, has led to a decline in species richness in the southern end of the sound.

4.10.2 Disturbance of deep basin fauna

The level of disturbance of the benthic fauna at individual sites is important in determining the well-being of the community being studied. To determine the level of disturbance of the deep basin benthic fauna of Cockburn Sound and Warnbro Sound, a study was undertaken by the Geography and Zoology Departments of the University of Western Australia in collaboration with the Department of Environmental Protection. This study examined the relative level of 'disturbance' of the benthic invertebrate fauna at three sites in Cockburn Sound and three in Warnbro Sound by using abundance biomass comparison (ABC) curves (Warwick, 1986). In order to assess anthropogenic effects, Warwick (1986) adapted the graphical method of Lambshead *et al.* (1983) to generate ranked curves for both the abundance and biomass of individual species within a sample. A shift from low numerical and high biomass dominance to a situation of high numerical and low biomass dominance, as short lived and smaller-sized 'opportunistic' species become favoured over the fewer larger-sized but less tolerant species, signals a change from undisturbed (e.g. unpolluted) to disturbed (e.g. polluted) conditions, with the cross-over point of the respective curves representing an intermediate stage.

This study was intended to complement the more detailed investigation of the benthic fauna of Cockburn Sound and Warnbro Sound described in Section 4.10.1. Further technical details of the study can be found in Chalmers (1993).

Findings

In Cockburn Sound, the level of disturbance of the benthic fauna showed a strong trend from north to south, with no disturbance at the northern end of the Sound, and moderate disturbance in the central and southern end of the basin (Figure 4.10-6). In Warnbro Sound, the two most northern sites were in an undisturbed state and the southern site showed moderate disturbance. These results were also supported by a comparison of the diversity and evenness indices generated at the study sites. The study suggested that, in Cockburn Sound, the combined effect of a range of toxic substances could explain the pattern of relative 'disturbance' and, in Warnbro Sound, the build up of organic material in the southern end of the deep basin could explain the greater level of 'disturbance' in this area. The definition of 'disturbance' includes both anthropogenic and natural 'disturbances'. This study supported the general findings, outlined in section 4.10.1, of the investigation by Cary *et al.* (1995c).

Conclusions

The findings of this study suggest that the site in the northern end of Cockburn Sound is 'undisturbed', while the sites in the centre and southern end of the sound are 'moderately' disturbed, probably due to the combined effect of a range of toxic contaminants. In Warnbro Sound, the two most northern sites are 'undisturbed' while the southernmost site was 'moderately' disturbed, probably due to a build up of detrital organic material. The ABC curves can be used to identify the relative 'disturbance' at the study sites, but cannot directly identify the cause of the 'disturbance'.

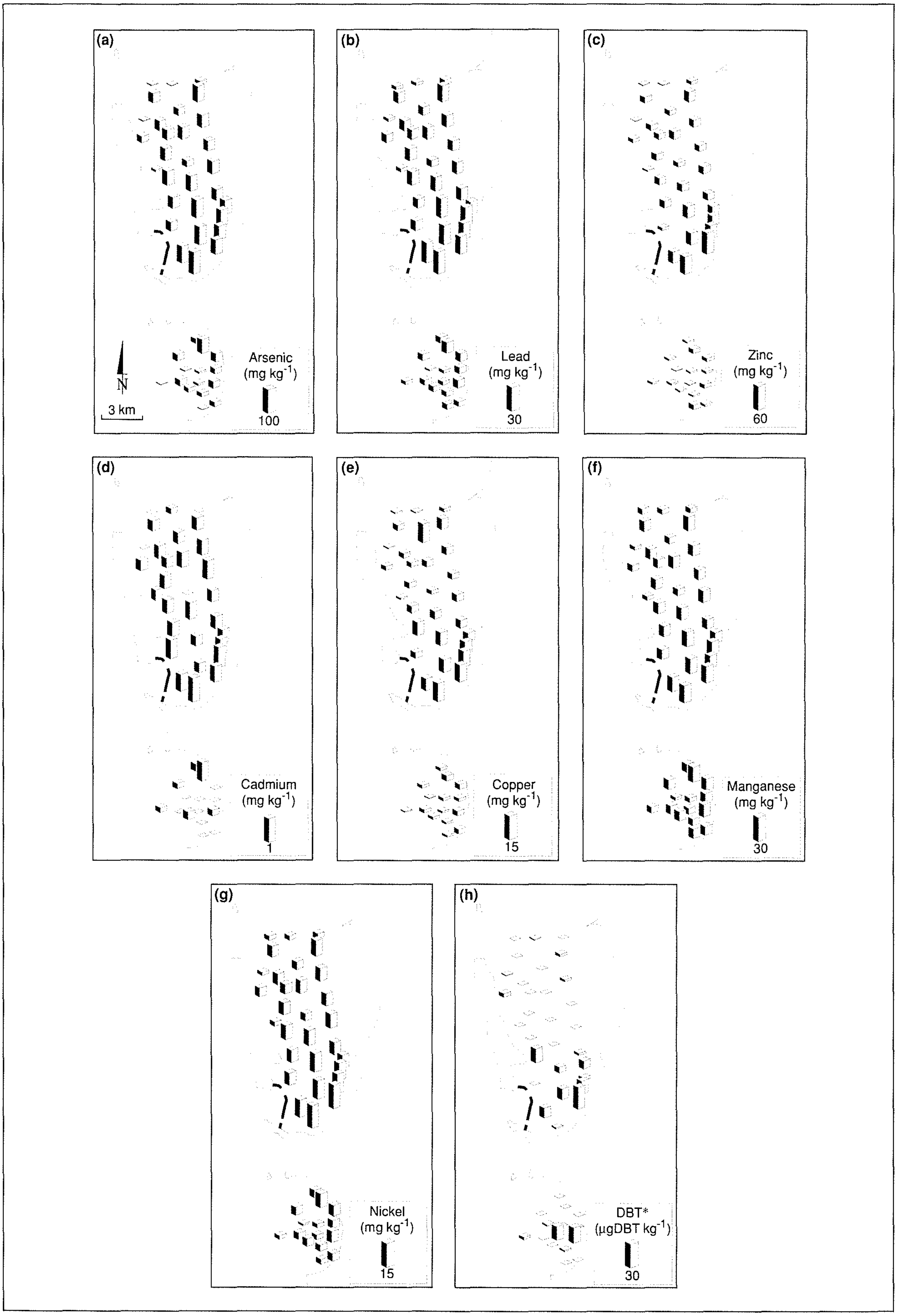


*Figure 4.10-3. Contaminant concentrations in the surficial basin (> 10 m depth) sediments of Cockburn and Warnbro sounds. *DBT= dibutyltin, a breakdown product of tributyltin.*

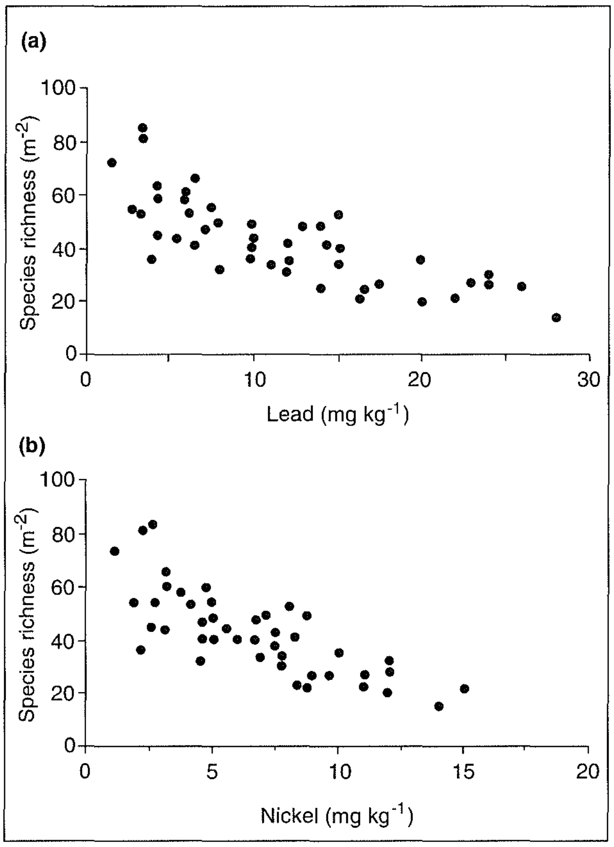


Figure 4.10-4. Relationships between benthic invertebrate faunal species richness and concentrations of (a) lead and (b) nickel in the surficial basin (> 10 m depth) sediments of Cockburn and Warnbro sounds.

For example, either small scale physical effects, such as anchor damage, or 'broadscale' pollution could induce 'disturbance' at a particular site. Therefore, consideration should be given to the various cultural activities that occur in the area and cause-effect hypotheses should be tested using all available means.

4.10.3 Benthic filter-feeders in seagrass meadows

Filter feeders or suspension feeders are organisms which filter suspended material, usually plankton, from the surrounding water. Filter feeders have the capacity to filter large quantities of water and they therefore exert a control on phytoplankton biomass (Loo and Rosenberg, 1989; Petersen and Riisgard, 1992) and improve light penetration through the water column. This pilot study was undertaken by Edith Cowan University in collaboration with the DEP and CSIRO, as part of a much larger collaborative CSIRO-DEP study of filter-feeder communities in eutrophic and oligotrophic waters, within CSIROs Coastal Zone Programme. The study examined the relationship between macro and micro filter-feeder invertebrate assemblages and phytoplankton biomass in Cockburn Sound. As food supply is thought to be the limiting factor in the biomass of benthic filter feeders, biomass is likely to increase with increased food supply.

Further technical details can be found in Clapin (1994), Greenway (1994) and Lemmens *et al.* (in press).

Findings

The spatial distribution of filter-feeders in Cockburn Sound is characterised by higher biomass at sites along the eastern and southern shorelines, than the western shoreline, with chlorophyll *a* levels showing a similar spatial pattern. Filter-feeder biomass and diversity was higher in *Posidonia* seagrass meadows than other seagrass genera and bare sediment, indicating that *Posidonia* meadows are the most important seagrass habitat for filter-feeders in the sound. *Posidonia* and *Amphibolis* meadows in Cockburn Sound support sufficient numbers of suspension feeders per square metre of meadow to filter the volume equivalent of a 10 m water depth in one day, while *Heterozostera* meadows and unvegetated substrate would need several weeks. An introduced filter feeder worm, *Sabella* cf. *spallanzanii*, was found in large numbers on the Southern Flats in southern Cockburn Sound (see section 4.13). It is suggested that Cockburn Sound is ideally suited for the invasion of such pest species, as it offers large areas of disturbed habitat with no vegetation, and a large food source (i.e. phytoplankton).

Conclusions

Compared with filter feeders in *Heterozostera* meadows and unvegetated substrate, the filter feeders in *Posidonia* and *Amphibolis* meadows in Cockburn Sound are far more abundant, and appear more capable of controlling the densities of suspended organic matter, thereby improving light conditions and assisting in sustaining healthy seagrass meadows. Cockburn Sound appears to be ideally suited for the invasion of pest species as it offers large areas of disturbed habitat with no vegetation and a large food source.

Table 4.10-1. Temporal comparison of species richness, diversity and evenness indices for benthic invertebrate fauna in north and south Cockburn Sound.

	North			South		
Parameter	1978	1993	p	1978	1993	p
Species richness	4.7	8.5	< 0.01	6.1	4.4	< 0.01
Diversity	1.0	1.4	> 0.05	1.2	0.9	< 0.01
Evenness	0.7	0.7	> 0.05	0.7	0.6	> 0.05

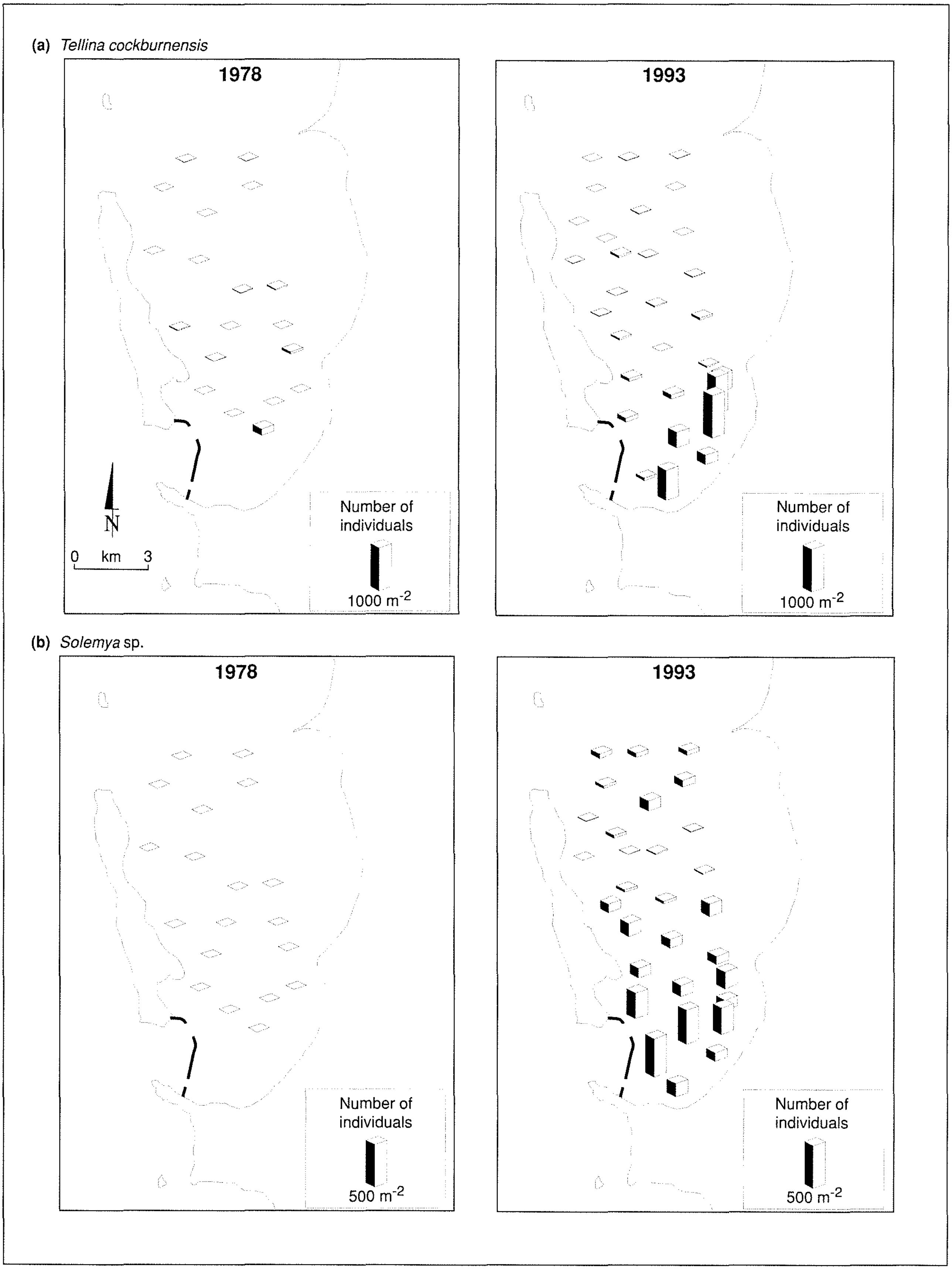


Figure 4.10-5. Comparative abundance of (a) Tellina cockburnensis and (b) Solemya sp. in surficial basin (> 10 m depth) sediments of Cockburn Sound in 1978 and 1993.

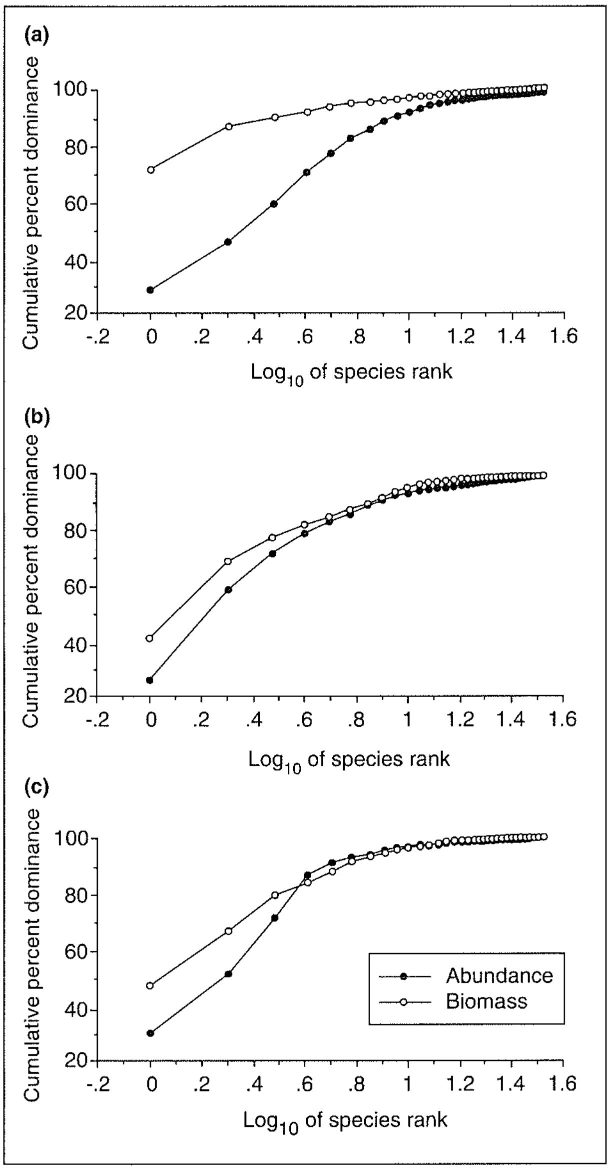


Figure 4.10-6. Abundance-biomass curves of benthic invertebrate fauna in the surficial basin (> 10 m depth) sediments of Cockburn Sound showing (a) an 'undisturbed' site in the northern end, (b) a 'moderately disturbed' site in the centre and (c) a 'disturbed' site in the southern end.

4.11 Thais orbita as a bioindicator of TBT contamination

Tributyltin (TBT) is the most toxic component of organotin-based antifouling paints and has been used widely around the world on the hulls of boats and ships since the 1960s. Since then, the recorded incidence of certain deformities in marine animals living near marinas and boat harbours has increased, and this has been linked to TBT contamination (World Health Organisation, 1990). Local evidence of the reproductive disorder *imposex* (imposed sexual characteristics) was found in *Conus* (cone shells) collected from Rottnest Island in 1991 (Kohn and Almasi, 1993). An examination of preserved museum specimens of the same genus collected from Rottnest Island in 1975 showed no such abnormalities (F. Wells, personal communication).

A 1991 survey of contaminants in the marine environment of the southern metropolitan coastal waters of Perth indicated that contamination of organotin compounds in sediments and mussels was widespread, and at levels likely to be causing significant biological effects (Burt *et al.* 1993b; Simpson *et al.* 1993; sections 4.5 and 4.6). In response to these findings, the DEP funded a post-graduate study of the biological impact of TBT on an indicator species common in Perth's coastal waters (Field, 1993). The study was undertaken by the Zoology Department of the University of Western Australia in collaboration with the DEP and the WA Museum. The main objective of the study was to determine the frequency and spatial extent of imposex in the mollusc, *Thais orbita*, a common neogastropod on intertidal reef platforms throughout the metropolitan coastal waters, including Rottnest Island. This animal is also known to develop the reproductive disorder imposex when exposed to TBT (Wilson *et al.* 1993).

Further technical details of this study can be found in Field (1993).

Findings

Imposex in *Thais orbita* was widespread throughout the metropolitan coastal waters of Perth, with the frequency of imposex being highest in the Fremantle and Cockburn Sound region, at Hillarys Boat Harbour, and in Thompsons Bay and Geordie Bay at Rottnest Island. This disorder was also high on reefs at Cottesloe and Trigg, and at Straggler Reefs, due west of Fremantle. The lowest frequency of imposex (4%) was recorded on The Sisters reef in the southern end of Warnbro Sound (Figure 4.11-1). The distribution of imposex throughout the study area is generally consistent with the distribution of TBT in both sediments (section 4.5.2) and mussels (section 4.6) which, in turn, reflects the general location of major ship and boat mooring areas in the metropolitan coastal waters of Perth.

Exceptions to this general rule were the reefs at Cottesloe and Trigg, and at Straggler Reefs. The results of hydrodynamic modelling provide a possible explanation for these apparent anomalies in that these reefs are periodically exposed to the Swan River outflow (section 6.2) which is contaminated with TBT, as shown by the high concentrations of TBT in mussels from Fremantle Harbour (Burt *et al.* 1995e).

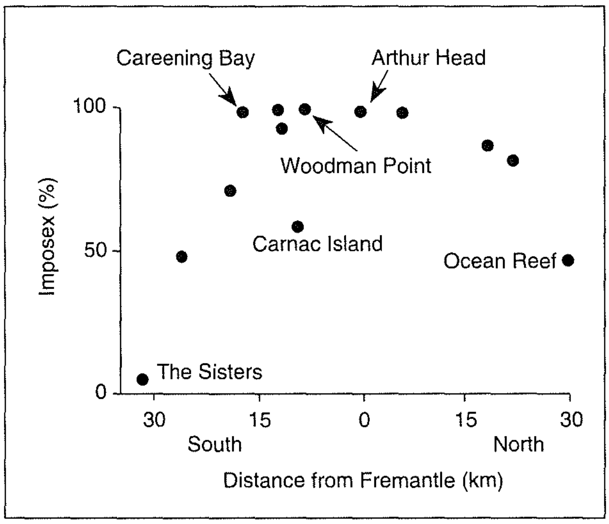


Figure 4.11-1. Frequency of imposex in Thais orbita in 1993 in the coastal waters of Perth.

The relationship between the frequency of imposex in *Thais orbita* and the concentration of TBT in mussels from the same or adjacent sites is shown in Figure 4.11-2. TBT concentration in mussel tissue is used as a relative index of TBT concentrations in seawater and is used to infer relative exposure of *Thais* to TBT contamination. The relationship illustrates the extreme toxicity of TBT in causing a high incidence of imposex at all but very low levels of exposure. Reproductive disorders caused by exposure to TBT have been documented for a range of marine life (Maguire, 1987; World Health Organisation, 1990; Bryan and Langston, 1992; Burridge *et al.* in press) suggesting that TBT is likely to be affecting many other local species in Perth's coastal waters with unknown long-term ecological implications.

Imposex can be induced in some marine animals by natural 'stressors', such as high water temperatures that sometimes occur in rock pools (Nias *et al.* 1993), but at frequencies generally less than about 5%. The generally good agreement between the frequency of imposex and the spatial distribution of TBT in sediments and mussels suggests that TBT contamination is causing the observed reproductive disorders. The decrease in the frequency of imposex with increasing distance from major boat mooring sites at Rottnest Island provide further support for this conclusion (Figure 4.11-3) as does experimental data from this study where imposex was induced in *Thais orbita* in aquaria by exposing 'unaffected' animals to TBT (Field, 1993).

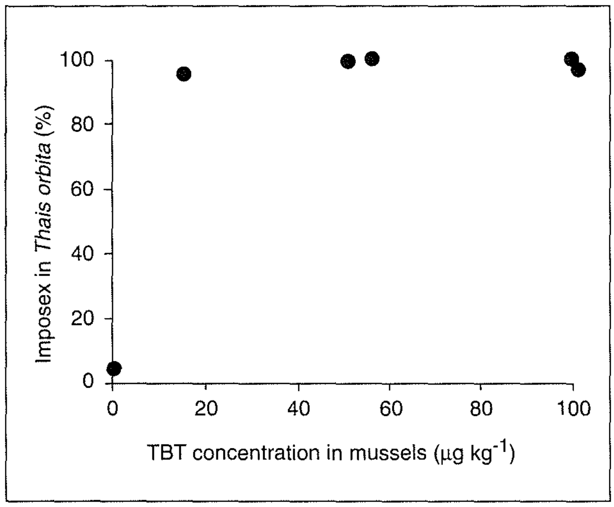


Figure 4.11-2. Relationship between the frequency of imposex in Thais orbita in 1993 and the concentration of tributyltin (TBT) in mussel tissue from the same or nearby localities in November 1991.

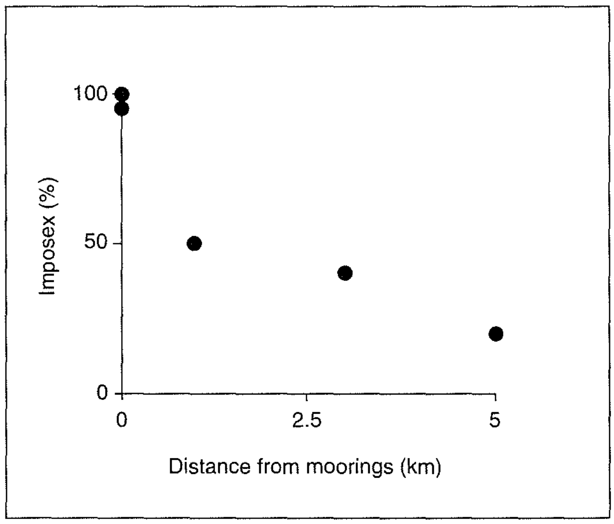


Figure 4.11-3. Relationship between the frequency of imposex in Thais orbita in 1993 at Rottnest Island and distance from boat mooring areas.

The frequency of imposex exceeded the criterion for draft Environmental Quality Objective 2 at all sites throughout Perth's coastal waters, apart from one site in southern Warnbro Sound.

Conclusions

The reproductive disorder, imposex, in the mollusc *Thais orbita* is widespread in the metropolitan coastal waters. This disorder is related to tributlytin contamination of the local marine environment, presumably from antifouling paints on the hulls of ships and boats that use these waters. This substance is known to affect a wide variety of marine plants and animals and is, therefore, likely to be affecting many other marine species living in these waters. The long-term ecological implications of tributyltin contamination in Perth's coastal waters are presently unknown but are cause for serious concern.

4.12 Larval fish assemblages in healthy and degraded seagrass meadows

Meadow-forming seagrasses of the genus *Posidonia* occur in the more sheltered parts of the southern metropolitan coastal waters of Perth (see section 4.1). These plant communities are key primary producers; they provide important habitat and nursery areas for fish and other marine animals, and they bind sediments and reduce wave-induced currents at the sediment-water interface, thereby protecting submarine banks and shorelines from erosion (Larkum *et al.* 1989). Little quantitative information exists on how these attributes are affected in degraded seagrass meadows. To assess the relative ecological importance of 'healthy' and 'degraded' seagrass meadows in Cockburn Sound as fish nursery areas, a post-graduate study was undertaken by Murdoch University, in collaboration with the DEP, to examine the species composition and abundance of juvenile fish and fish larvae occurring over seagrass meadows at two locations in Cockburn Sound over a summer breeding season.

Fish larvae were sampled within the seagrass meadows at two sites along the central eastern shoreline of Garden Island and at two sites in southern Mangles Bay. The Garden Island sites were considered as 'healthy' and the Mangles Bay sites as 'degraded' on the basis of historical and current data (Cary *et al.* 1991; section 4.2). Sampling was conducted each month between September 1992 and February 1993. Conical and beach seine plankton nets were used resulting in catches of both pelagic and settled fish larvae and juvenile fish.

Further technical details of this study can be found in Jonker (1993).

Findings

The diversity and concentration of fish larvae in 'healthy' seagrass meadows off east Garden Island and 'degraded' meadows in Mangles Bay are shown in Figure 4.12-1. As expected the concentration of fish larvae at both locations was higher over the summer months. Although family diversity was similar at both locations (Figure 4.12-1a) the concentration of fish larvae at Mangles Bay was significantly higher than at Garden Island, particularly in summer (Figure 4.12-1b).

Jonker (1993) suggested that the higher concentration of fish larvae in the 'degraded' seagrass meadows in Mangles Bay could be attributed to higher availability of food (phytoplankton) resulting from higher nutrient enrichment of these waters, lower predation due to increased shelter provided by higher epiphyte loads on the seagrass leaves and greater retention of fish larvae due to lower dispersion of the waters of Mangles Bay.

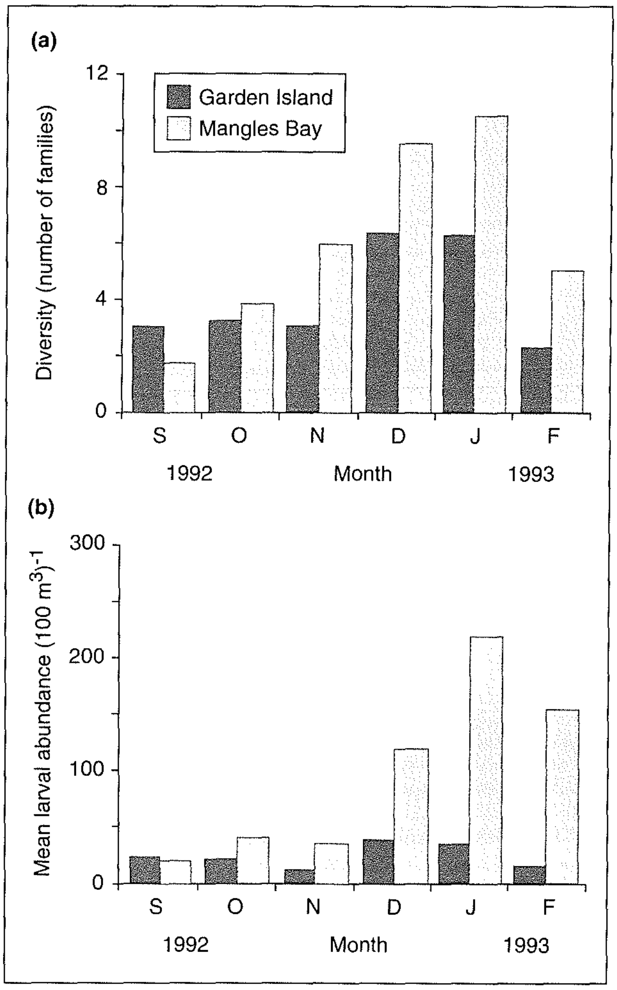


Figure 4.12-1. Fish larvae in seagrass meadows along the eastern side of Garden Island and in Mangles Bay: (a) diversity and (b) abundance.

Conclusions

This study showed that both 'healthy' and 'degraded' seagrass meadows in Cockburn Sound contain significant numbers of fish larvae and, therefore, continue to be important nursery grounds for fish in these waters. These findings also illustrate the important ecological role of seagrass meadows regardless of their 'health', within the range encompassed in this study.

4.13 Exotic organisms

The introduction of foreign organisms in Australian coastal waters has been reviewed by Jones (1991) and Gibbs (1993). More recently, the CSIRO Centre for Research on Introduced Marine Pests released a guide to the introduced marine species in Australian waters (Furlani, 1996). Currently, over 60 foreign species have successfully established in Australian coastal waters following introduction via discharges of ships' ballast waters and associated sediments or by dislodgement from the hulls of ships (ANZECC, 1995; Furlani, 1996). At best, established exotic species can displace indigenous species and disturb ecosystems. At worst, exotic species can cause massive disruption to flora and fauna, extinction of indigenous species and damage or loss to marine-based industries amounting to millions of dollars. A loss of US$5000 million has been estimated as the cost of the introduction of the zebra mussel to US and Canadian waters (Australian Quarantine Inspection Service, 1993).

Although there are voluntary guidelines covering the discharge of ballast waters in Australian waters (see Gibbs, 1993) compliance is not complete. Therefore the potential for exotic species to be introduced via ballast water remains wherever shipping occurs. In 1993/4, over 1700 ships, including 63 naval vessels, totalling nearly 30 million gross tonnes used Fremantle Harbour and Cockburn Sound (FPA, 1994). This level of shipping is estimated to have discharged approximately 3.5 million tonnes of ballast waters (Martinick *et al.* 1993; Muriale and Cary, 1995) indicating a significant potential for the introduction of foreign organisms to these waters.

The occurrence of foreign organisms in the southern metropolitan coastal waters of Perth has not been a primary focus of the SMCWS as this potential impact was considered, in the short-term, to be secondary to the potential impacts of chronic industrial, municipal and rural waste discharges on coastal ecosystems off Perth. However some component studies of the SMCWS indirectly provide some information on this issue. The species lists of the benthic invertebrate fauna (see section 4.10), phytoplankton (see section 4.6), zooplankton (see section 4.7) and fish larval (see section 4.12) studies have all been examined for the presence of exotic or unusual species.

Findings

Examination of the benthic fauna, phytoplankton, zooplankton and fish larvae species lists did not identify any exotic species. However the study of the filter-feeders in Cockburn Sound (see section 4.10.3) recorded the presence of the highly visible polychaete worm, *Sabella* cf. *spallanzanii*. Further surveys have confirmed that this species is common throughout the sound, and have found that it is mainly attached to hard substrata, including wharf and jetty pylons. This polychaete forms extensive aggregations on the Southern Flats, covering approximately 20 ha. *Sabella* is also found in harbours at Fremantle, Bunbury and Albany (Clapin and Evans, 1995). This worm has reached plague proportions in Victoria's Port Phillip Bay where it threatens the local scallop industry (Bonny, 1995). The recognition of foreign species requires a comprehensive inventory of the native flora and fauna of the area in question and a thorough understanding of the geographical range of these species. Many aspects of the flora and fauna of the coastal waters of Perth are poorly described and, for this reason, only the most visible foreign organisms, such as *Sabella*, are likely to be detected.

Twenty one species are known to have been introduced into Perth's coastal waters, presumably via discharge of ships' ballast water and associated sediment, or dislodgement from the hulls of ships. Of these, eighteen are known to have established and the remainder are presumed to have established based on available information. The species that have established include the fishes, *Tridentiger trigonocephalus* and *Sparidentex hasta*, the crabs, *Pyromaia tuberculata* and *Carcinus maenas*, the molluscs *Musculista senhousia*, *Theora lubrica* and *Crassostrea gigas*, the polychaete worm *Sabella* cf. *spallanzanii* and the toxic dinoflagellate *Alexandrium minutum* (ANZECC, 1995; Furlani, 1996).

Conclusions

The presence of at least 18 introduced species in Perth's coastal waters and the large quantities of ballast water discharged annually into these waters indicates a significant potential for the introduction of foreign organisms by visiting ships. The absence of a comprehensive database of local marine flora and fauna will continue to hinder any attempt to seriously address the issue of the introduction of foreign species in ballast water discharge. A coordinated approach is required at individual port, state, national and international levels to implement management strategies to minimise the risk of introducing foreign marine organisms.

4.14 Other studies

This section briefly refers to other recent or ongoing studies that provide baseline or inventory data relevant to the physical oceanography and ecology of the southern metroplitan coastal waters.

4.14.1 Physical oceanography

The CSIRO Division of Oceanography is undertaking a long-term research programme on the influence of the Leeuwin Current on the life cycle of the western rock lobster (Hutchins and Pearce, 1994). This work involves an analysis of the variability of the Current over time scales ranging from days to years. One aspect of the work is an investigation of the occurrence of meso-scale meanders of the Current which sometimes become detached, forming oceanic warm-core eddies. Another aspect deals with the occurrence of smaller scale 'billows' and incursions of Leeuwin Current water over the continental shelf zone off Perth. The hydrodynamic influence of such meanders, incursions and 'billows' of the Leeuwin Current over the shelf zone off Perth is discussed in section 5.1.1.

Since the early 1970s, the CSIRO Division of Oceanography has monitored salinity, temperature, nitrate and silicate levels at about three-weekly intervals at a site about 5 km west of the western end of Rottnest Island (Johannes *et al.* 1994; Pearce *et al.* in preparation). These data form the longest time series of marine water quality measurements taken in Western Australia and are being analysed to investigate the seasonal and inter-annual changes in these parameters in relation to meteorological forcings and the Leeuwin Current. Since the early 1980s the surveys have included sampling of the above parameters at nine equi-spaced sites between the 'Rottnest' station and Fremantle.

The CSIRO Division of Oceanography is researching the 'climatology' of the Leeuwin Current off southwest Australia. This work aims to characterise the seasonal variation in the cross-shelf sea-surface temperature structure in response to air-sea heat transfer and the Leeuwin Current. The results of the study are not yet available but will be of relevance to the understanding of the seasonal dynamics of the shelf region (A F Pearce, personal communication).

In December 1994 the CSIRO Division of Oceanography conducted cross-shelf hydrographic surveys off Perth and Cape Mentelle, southwest Australia, from the research vessel RV Franklin (G R Cresswell, personal communication). The transects went beyond the shelf edge and included vertical profiling of salinity, temperature, currents and biological parameters. The current data (from Acoustic Doppler Current Profiling) confirmed the presence of a northward flowing current over the shelf. This flow has been previously identified in summer sea-surface temperature satellite images (C B Pattiaratchi and A F Pearce, personal communication) and called the "Capes Current". It is believed to be driven by the prevailing summer south-southwesterly winds. Early analyses of satellite tracked drifter buoys by Cresswell and Golding (1980) highlighted the occurrence in summer of northward tracks of buoys off the Cape Leeuwin-Cape Naturaliste block and off Perth, Western Australia. The results provide additional insight into the interaction of the southward flowing Leeuwin Current and northward flowing wind-driven coastal flows during summer. The RV Franklin surveys are also valuable in terms of providing data on background nutrient levels for southwest Australian waters.

The Centre for Water Research (University of Western Australia) conducted a series of oceanographic measurements throughout 1993 as part of the WAWA Perth Coastal Waters Study (Pattiaratchi *et al.* 1995). Time-series data of current speed and direction were obtained at sites over the shelf and also within the Ocean Reef lagoon and Sepia Depression, in the vicinity of wastewater outfalls, with meters positioned at various depths throughout the water column. At deep water sites (>100 m) current meters were located at four depths and at shallow water sites meters were located at near surface and near bottom. These data have provided information on the statistics of flow regimes over the Perth shelf in response to meteorological and oceanographic conditions throughout an annual cycle. Pattiaratchi *et al.* (1995) also analysed a time series of 142 sea-surface temperature images, captured by the NOAA-AVHRR satellite, for the period September 1991 to December 1993, including 63 images from 1993. This provided a means of interpreting the large-scale circulation patterns over the shelf region of southwest Australia, including the waters off Perth. This analysis of the satellite imagery helped to identify a relatively narrow and cooler northward flowing coastal current during summer, called the Capes Current. As part of the PCWS, Pattiaratchi *et al.* (1995) conducted shelf scale and localised CTD surveys, approximately monthly, in order to gain an understanding of the seasonal characteristics of the cross-shelf salinity-temperature and density structure out to about the 100-140 m depth zone, west of the Marmion Lagoon. In addition to these parameters Pattiaratchi *et al.* (1995) also collected year long time series of meteorological data. The results of the shelf scale studies, conducted for the PCWS, have been utilised in the SMCWS to assist in the interpretion of more focussed nearshore studies.

The WAWA has conducted several effluent plume tracking surveys about the Cape Peron domestic wastewater outfall since it was commissioned in 1984 (e.g. Halpern Glick Maunsell, 1992; Claudius and Nener, 1995a, b). These data have provided spatial information about the spread of the plume water, within the limits of detection for various parameters (such as salinity, nutrients and bacteria). These data indicate that the plume is frequently transported in directions approximately parallel to the Sepia Depression channel alignment. However, the data also indicate that, on occasions, the plume can move to the Shoalwater Islands Marine Park, to the waters off southern Garden Island, or toward the reefs of the Five Fathom Bank. The field measurement programme of the SMCWS took these data into account when structuring grid layouts for oceanographic and water quality measurements near the Cape Peron outfall.

Researchers from Murdoch University (Bastyan *et al.* 1994) have recently conducted field surveys of the vertical salinity, temperature and dissolved oxygen structure of Cockburn Sound during autumn. Bastyan *et al.* (1994) conclude that there is the potential for significant de-oxygenation of the bottom waters in southern Cockburn Sound during calm autumn conditions. This work has been used as reference material for further investigations of first, the relationships between vertical density stratification and vertical dissolved oxygen stratification in the Cockburn Sound basin during autumn conditions (D'Adamo and Mills, 1995c; Coleman, 1995), and second the potential for sediment nutrient release by the action of sulphate-reducing bacteria during stratified conditions in Cockburn Sound (Barry, 1995).

4.14.2 Ecology

Other studies that have recently been conducted in Perth's coastal waters and which assist with the biological or chemical characterisation of these waters are briefly described.

Water quality monitoring surveys of Cockburn Sound were conducted by the Nutrient Analysis Laboratory at Murdoch University during the summer of 1990/91 and 1992 (Bastyan and Paling, 1992), and 1993 (Bastyan *et al.* 1994). These studies were a continuation of previous summer monitoring surveys conducted since 1982/3 by the Department of Environmental Protection (Cary *et al.* 1991) and form part of the results included in section 4.7.3 on local water quality. The 1993 survey (Bastyan *et al.* 1994) included measurements of dissolved oxygen and nutrient concentrations in the water column, and estimates of sediment nutrient release rates from a site in the southern basin of Cockburn Sound. Bastyan and Paling (1995) compared the nutrient release rates and nutrient content of sediments from Cockburn Sound and Warnbro Sound.

Regional water quality surveys of Perth's metropolitan coastal waters, offshore to the 100 m depth contour, were conducted in the winter of 1993 and the summer of 1994 (Buckee *et al.* 1994). The aim of these surveys was to provide a synoptic view of the nutrient concentrations in these waters. Johannes *et al.* (1994) conducted an intensive three-year study characterising the seasonal and spatial variation in the nutrient regime of Perth's coastal waters.

In 1991, a survey was conducted of meiofauna assemblages (e.g. amphipods and nematodes) in the surficial marine sediments (top 10 cm) of beaches on Garden Island (B Knott, personal communications). The abundance and species diversity of meiofauna from these beaches were compared.

In recent years there have been numerous field and laboratory studies conducted on various aspects of seagrass biology. The primary focus of these studies has been on seagrass rehabilitation trials (Kirkman, 1989a; Nelson, 1992; Hancock, 1992; Walker, 1994), seagrass productivity (Carruthers, 1994; Paling and McComb, 1995), nutrient cycling (Walker *et al.* 1988; Paling and McComb, 1994), and the effects of eutrophication (Kirkman and Manning, 1993). There are also numerous ongoing university studies that are examining aspects of seagrass genetics, ecophysiology and reproduction (D I Walker, M Borowitzka, E I Paling, personal communications).

The ecology of epiphytes growing on seagrass leaves, particularly in relation to the effects of nutrient enrichment, has been the subject of numerous studies (Trautman, 1990; Clapin *et al.* 1993; Hollyock, in preparation). Studies of benthic macroalgae have included their role in nutrient cycling (Paling, 1988), distribution and composition (Kirkman, 1989b, Kendrick, 1993) and recruitment (Kendrick and Walker, 1994). The ecological significance of benthic microalgae in relation to their productivity was examined by Masini (1990).

The relative diversity of fish and macro-faunal assemblages living in seagrass meadows, and in adjacent habitats, has been studied in Cockburn Sound (Chalmers, 1993; Jonker, 1993; Vanderklift, 1994), Warnbro Sound (Walker, in press) and Rottnest Island (Edgar and Shaw, 1993). The distribution and abundance of filter-feeders in seagrass meadows from Perth's coastal waters have been reported by Clapin (1994), Greenwood (1994), Kirkpatrick (1994), Lemmens and Edgar (in preparation) and Lemmens *et al.* (in press). On a broader scale, Ayvazian and Hyndes (in press) compared the species composition of fish assemblages in the surf zone at sites in the temperate waters along the southwest coast of Western Australian, including sites at Rottnest Island, Cockburn and Warnbro sounds. The primary focus of this study was to examine the local and regional differences in species composition in relation to local (e.g. habitat type) and regional influences (e.g. Leeuwin Current).

There have been some studies of fish biology and stock assessments that are relevant to the SMCWS. Fletcher *et al.* (in preparation) described the distribution and abundance of whitebait in the metropolitan coastal waters as part of a regional stock assessment of whitebait along the southwestern coast of Western Australia. A central focus of this study was to determine the life history of whitebait, especially the reproductive cycle, which involved characterising the planktonic assemblages of whitebait larvae in these waters. Hutchins and Pearce (1994) examined the influence of the Leeuwin Current on the recruitment of tropical reef fish at Rottnest Island.

5. PROCESS STUDIES

This chapter describes studies of key physical, chemical and biological processes that were undertaken to develop an understanding of the impacts of waste inputs to Perth's southern metropolitan coastal waters. Many of these processes were identified and expressed as key pathways in conceptual models constructed at the outset of the SMCWS (Masini *et al.* 1991; Simpson *et al.* 1993). The process studies have been directed at quantifying the forcing factors and responses, and determining the functional relationships and rate functions which would simulate key processes linking nitrogen inputs to the 'health' of benthic plant communities. The Coastal Ecology Model (COASEC) and the Benthic Site Model (BSM), which were developed in collaboration with the WAWA Perth Coastal Waters Study, provide frameworks for the integration of these findings. The following summary of the SMCWS process studies should be considered together with complementary process studies undertaken as part of the PCWS. Some of these studies are specific to the southern metropolitan coastal waters (e.g. local oceanography) and some are of a more generic nature (e.g. regional oceanography, photosynthesis versus irradiance relationships of seagrasses). The previous chapter described the inventory and baseline studies. Process information has been derived from many of these studies and therefore there is overlap with chapters 4 and 5.

5.1 Oceanography

The hydrodynamic flushing characteristics of receiving waters are important in determining the relationships between pollutant loadings and environmental impacts. Water circulation, density stratification and the energetics of mixing affect the horizontal and vertical distribution of contaminant concentrations throughout a waterbody. In addition, an understanding of the relative importance of regional and local forcings and the pathways of materials transport into and out of the different areas of interest is critical in determining potential impacts and helping define the appropriate temporal and spatial scales of environmental management. A brief summary of oceanographical aspects of the region is presented in Section 2.2. The oceanographic programme of the SMCWS was designed to build on past studies and address critical gaps in existing information, many of which had been identified in the review phase of the SMCWS (Hearn, 1991, D'Adamo, 1992). This programme involved both shelf-scale and local-scale components.

The physical oceanography programme had as broad objectives: to characterise the hydrodynamic response of the southern metropolitan coastal waters to both local forcings (e.g. local wind stress, and local air-sea heat transfer) and larger scale influences (e.g. the Leeuwin Current); to characterise the time and space scales of materials transport (e.g. contaminants, phytoplankton); to identify potential transport paths (linkages) between contaminant sources and areas of ecological significance; to implement, calibrate and validate hydrodynamic and transport models of the study area, and to supply the necessary oceanographic inputs to an ecological nutrient-effects model.

5.1.1 Shelf-scale oceanography

Water currents over the continental shelf off metropolitan Perth are driven principally by wind stress, regional-scale poleward pressure gradients associated with a drop in sea level along the west coast of Australia, and differences in barometric pressure across meso-scale weather systems. Southward propagating shelf waves are produced on the west coast by disturbances further north, and these also generate currents over the shelf. Near the shelf break there is a southward flow called the Leeuwin Current, which consists of warm, low salinity water (Cresswell and Golding, 1980). The Leeuwin Current is driven by a difference in steric height between the waters off North West Cape and the southwest coast of Western Australia. Smith *et al.* (1991) suggest that the greater predominance of southerly winds in summer leads to a weaker Leeuwin Current in summer compared to winter.

In shallow water, the wind stress is the dominant forcing; in deeper offshore water, the long-shelf pressure gradient dominates. Where wind stress opposes the pressure gradient force, the shore-parallel current may reverse at some distance off the coast, with downwind current nearshore, and reversed current offshore. The bathymetric distribution of the area (including the shape of the coastline and the presence of islands, reefs and banks, as shown in Figures 1.1-1 and 1.1-2 and Plate 4.1-1) guides the water flow. The influence of the earth's rotation is important, particularly in introducing a component of movement and exchange of water in the cross-shelf direction.

Pearce and Church (submitted) and Hodgkin and Phillips (1969) have demonstrated that the salinity and temperature properties of nearshore and shelf waters differ due to various factors such as differential heating and cooling, differential exposure to the Leeuwin Current, differential salinity increases due to evaporation, and the influence of freshwater runoff. In turn, these salinity and temperature differences determine water density differences. The interaction and response of different water masses with distinct temperature-salinity-density characteristics to forcings, such as wind and pressure gradients, depends on the density differences between these water masses, and the energy available for mixing.

The salinity-temperature-density structure of the southern metropolitan coastal waters was investigated to determine the annual cycle of offshore and nearshore water properties, to document the transport of estuarine plumes, and to help characterise hydrodynamic processes and regimes, particularly in relation to the mixing and flushing of the coastal embayments. Routine conductivity-temperature-density (CTD) surveys were undertaken every 1-2 months during 1991-93 to achieve this objective. The key oceanographic processes identified in these surveys were examined in detail during intensive CTD field exercises in winter (August 1991), summer (March 1992) and autumn (March and May 1994), which were supported by current meter, satellite, meteorological and hydrological data acquisition. The findings of these studies provided the basis for the implementation of the Princeton Oceanographic Model (POM) to simulate the circulation of these waters.

Further technical details can be found in Simpson *et al.* (1993), Mills *et al.* (1996), D'Adamo *et al.* (1995a, b), D'Adamo and Mills (1995a, b, c).

Findings

5.1.1.1 *Seasonal hydrodynamic regimes*

A series of 17 cross-shelf CTD surveys (August 1991-February 1993, and May 1994) of the southern coastal waters off Perth described an annual cycle in the vertical and horizontal density structure caused by seasonal variation in the salinity-temperature stratification. These results add to the findings of earlier studies undertaken in the wider Perth coastal region (Pearce and Church, submitted).

The annual cycle of wind speed and direction is presented in Figure 5.1-1 and the relatively greater occurrence of strong winds is highlighted for the spring-summer period. The autumn-winter period is generally characterised by periods of lighter and more variable winds, however storms cross the study area at about 7-10 day intervals as part of cyclonic low pressure synoptic systems (Steedman and Craig, 1979; Breckling, 1989).

The series of measurements of the salinity, temperature and density characteristics of the offshore southern metropolitan coastal waters and Cockburn Sound waters provided the basis for determining the annual cycles in the cross-shelf salinity, temperature and density differences (Cockburn Sound minus mid-shelf), as shown in Figure 5.1-2 (D'Adamo and Mills, 1995b). The cross-shelf density difference was subsequently identified as a key factor determining the nature of the major hydrodynamic regimes in the nearshore southern metropolitan waters. The cyclic plot in Figure 5.1-2 is used to identify three main regimes which have been named the 'winter-spring', 'summer' and 'autumn' regimes, respectively (D'Adamo and Mills, 1995b). The 'winter-spring' regime is from about mid-winter to spring (August to October), with the exact period of this regime depending largely on the timing of freshwater discharge from rivers and on the

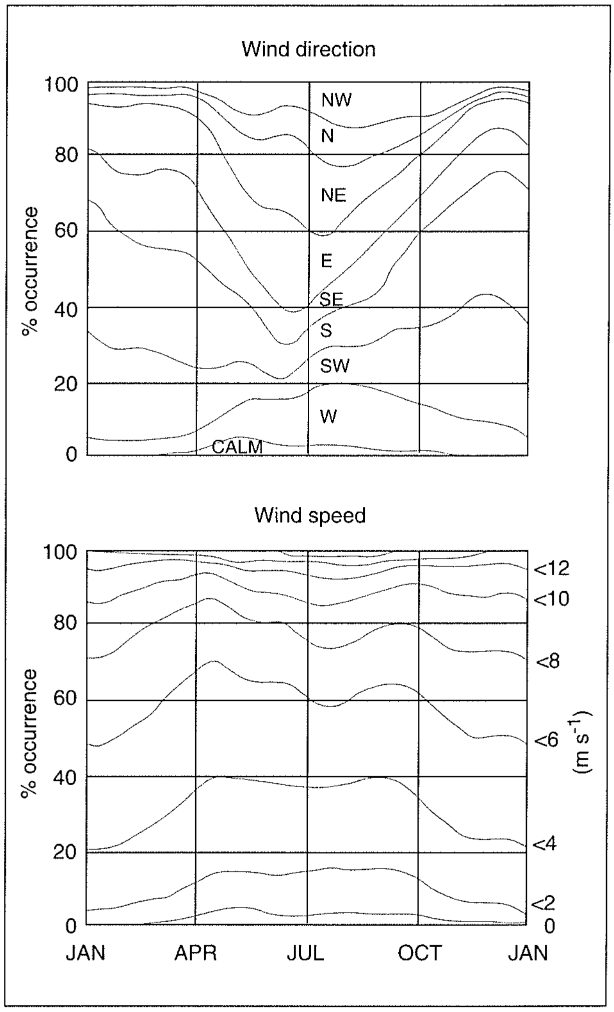


Figure 5.1-1. Cumulative percent occurrence of wind direction and speed derived from seven years of data from Fremantle (re-drawn from Steedman and Associates, 1979).

strength of the warming of nearshore waters in spring. During the 'winter-spring' regime basin waters are relatively buoyant in comparison to offshore waters, due at first to local river discharges which create a *Region of Freshwater Influence* (Simpson and Rippeth, 1993), and then to enhanced warming of nearshore waters; strong wind mixing events occur every 7-10 days, with relatively calm intermittent periods (D'Adamo *et al.* 1995b). The 'summer' regime (December to March) is characterised by strong penetrative convection and strong wind-mixing events which occur every 1-2 days, and nearshore-offshore density differences diminish (D'Adamo and Mills, 1995a). The 'autumn' regime (April to July) begins in autumn and extends into early winter. During this period basin waters are relatively dense, due firstly to high salinities, caused by evaporation over summer and autumn, and secondly to differential cooling from late autumn to early winter, which results in relatively low water temperatures within the nearshore embayment. Penetrative convection is moderate during the autumn regime, and strong mixing events (i.e. mixing to the bottom) occur at 7-10 day intervals, on average, although they can be up to three weeks apart if there are prolonged periods of weaker winds (D'Adamo and Mills, 1995c). The change

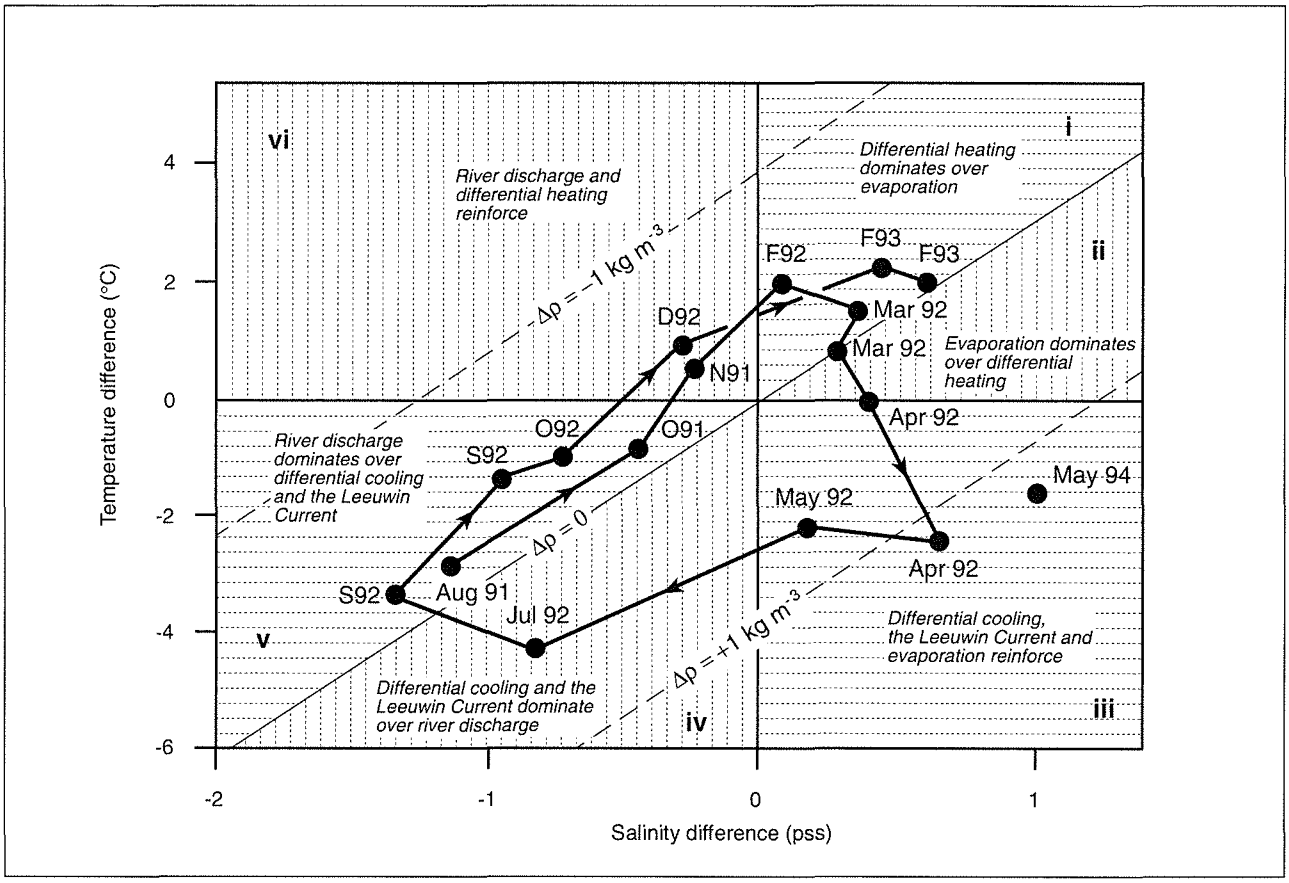


Figure 5.1-2. The annual cycle in the salinity and temperature differences (at 10 m depth) between Cockburn Sound and the mid-shelf, southwest of Rottnest Island. The influence of major forcings on the cross-shelf differences are indicated for each of the six categories (I to VI) within the cycle. The diagonal lines are contours of density difference and data above/below the central contour indicate that Cockburn Sound is less/more dense than mid-shelf water.

between the 'winter-spring' and 'autumn' regimes generally occurs as a gradual transition. However the transition that follows the 'autumn' regime can be quite rapid if large quantities of low salinity water from rivers enter the coastal waters over a short period in early winter.

The strengths of vertical salinity, temperature and density gradients also undergo seasonal variation in response to the seasonal variability in the stratifying influences and strength of vertical mixing processes. Vertical profile data from the Cockburn Sound basin, Sepia Depression and the mid-shelf region south of Rottnest Island have been used in Figure 5.1-3 to plot characteristic vertical density differences (measured from profiles collected since 1981) for the different seasons (D'Adamo and Mills, 1995b). The data were generally collected during the period between mid-morning and mid-afternoon, prior to strong mixing events, thereby representing the strength of vertical density stratification that vertical mixing processes have to overcome in order to fully mix the upper 21 m of the water column (which is the maximum depth in the nearshore basins). The strongest vertical differences occurred in Cockburn Sound in winter and spring, reflecting the stratifying influence of buoyant estuarine discharges and solar heating. Interestingly, Sepia Depression is strongly stratified in autumn, and this is due to the influence of buoyant Leeuwin Current water that

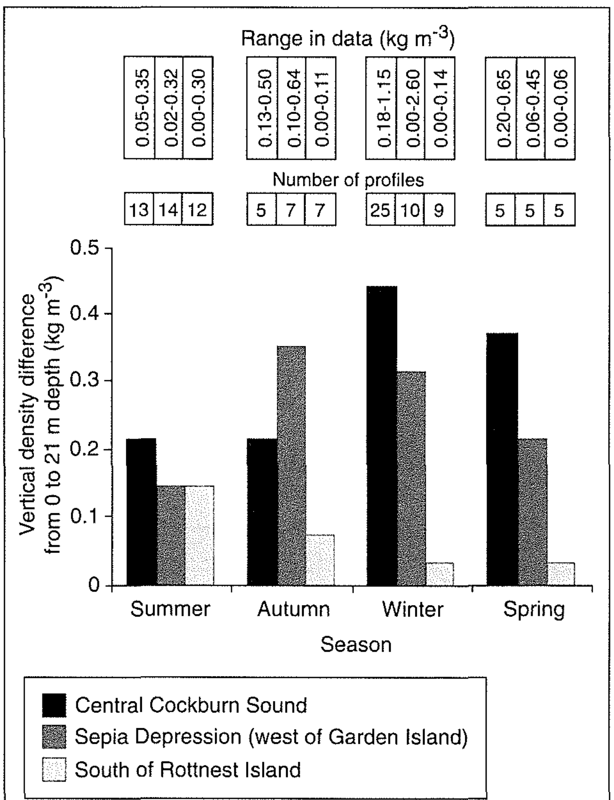


Figure 5.1-3. Seasonal mean vertical density differences for selected locations in the study area.

spreads over the mid-shelf region. Nearshore vertical gradients are relatively weak in the 'summer' regime, due to the absence of significant sources of low salinity water; the Leeuwin Current is generally located beyond the shelf break during this period, and discharges from the estuaries are typically low.

Throughout the year, vertical mixing is due primarily to wind stress with assistance from penetrative convection caused by surface heat losses at night. A methodology to predict the mixing potential of wind and penetrative convection was taken from Imberger (1994) and a vertical mixing analysis for these two mechanisms was performed to determine the potential of bursts of wind (< 12 hours), such as sea-breezes and winter storms, in conjunction with night-time penetrative convection, to fully-mix the water column during a typical diurnal cycle (D'Adamo and Mills, 1995 a, b and c; D'Adamo *et al.* 1995b). The results of the mixing analysis suggest that vertical mixing to the bottom of Cockburn Sound (21 m) is expected on average 10-15 times per month during the 'summer' regime, and this is in contrast to about 4-5 times per month during the 'winter-spring' and 'autumn' regimes. The result for the mid-shelf region indicates more regular vertical mixing of the upper 21 m of the water column, reflecting the reduction in intensity of characteristic vertical density gradients with distance offshore. Exceptions to this can occur in the 'autumn' and 'winter-spring' regimes, for example, when fronts between the Leeuwin Current and nearshore waters are present over the mid shelf; under these circumstances relatively strong vertical gradients can form, and inhibit vertical mixing, when buoyant Leeuwin Current water moves over denser nearshore waters (Mills *et al.* 1996).

The difference between the nearshore and offshore annual density cycles is important in that it sets up horizontal gradients of density which, together with wind forcing and vertical mixing processes, are key factors determining the nature of exchange between the nearshore basins, such as Cockburn Sound, and adjacent oceanic waters. In 'winter-spring', although there are relatively few full-depth mixing events, they are typically followed by renewal of the deep basin waters by inflow of surrounding dense shelf water. In 'autumn', however, the few full-depth mixing events that do occur are followed by near-surface renewal of basin water, because the mean density of the basin is relatively high and hence shelf waters tend to be driven in as surface flows. Hence, the potential for near-bottom waters to be vertically mixed through the water column is less in 'autumn' compared to 'summer'. In 'summer', horizontal density differences are small and vertical gradients are regularly eliminated by wind mixing and penetrative convection, hence flushing is dominated by wind-driven barotropic circulation with density effects only playing a relatively minor role in the mean hydrodynamic behaviour of the basin during summer.
The hydrodynamics of these three regimes are elaborated upon in section 5.1.2. They have also been investigated with the aid of a numerical model (see chapter 6).

5.1.1.2 Winter

In winter, atmospheric high pressure systems and intervening cold fronts (as part of low pressure systems) migrate eastward over southwest Australia. Winter wind patterns are typically cyclic (period 7-10 days) with episodic northwest to southwest gales associated with passing cold fronts, followed by longer periods of moderate and weak winds swinging through the south and the east as extensive high pressure systems move through. The wind vector plot in Figure 5.1-4a is from the intensive survey of August 1991 and shows one such typical winter wind cycle. These winds drive predominantly longshore coastal currents at speeds of order 0.1 m s^{-1}, with current direction reversals (accompanying major wind shifts) occurring at intervals of about 3-5 days. This current pattern is typified by the current vector plot in Figure 5.1-4b, obtained from a moored current meter at mid-depth in central Sepia Depression, west of Cape Peron. The coastal water therefore tends to flow longshore several tens of kilometres before reversing upon itself. This is a small distance compared to the length of the metropolitan coastline. Hence, in winter, the potential for longshore flushing of pollutants from Perth's coastal waters can be limited. It is therefore important to understand to what extent flushing of coastal waters can occur in the cross-shelf direction.

The Leeuwin Current advects warm tropical water poleward along the edge of the continental shelf. The August 1991 intensive oceanographic survey clearly identified the presence of this current (as a 160 m deep, well-mixed, warm water core) located over the outer continental shelf and slope (Figure 5.1-5a) (Mills *et al.* 1996). Relatively cool, fresh coastal water is formed by enhanced cooling in the shallows and mixing with freshwater sources (rivers and groundwater). Generally, the cool coastal water is denser than the warm Leeuwin Current water (Smith *et al.* 1991), except in regions of freshwater influence, near estuarine outlets, where the coastal waters can be more buoyant than the shelf waters. The cross-shelf exchange of these waters is governed to a large degree by the interaction between wind stress, sea-level slope, the effect of the earth's rotation, and the density differences (both horizontal and vertical) between these adjacent water masses.

While the bulk of the Leeuwin Current flows southward beyond the edge of the continental shelf, the influence of the earth's rotation is to cause a secondary transport of warm Leeuwin Current water onto the shelf. The earth's rotation also causes near-surface currents that are driven by northeasterly to northwesterly winds to be deviated counterclockwise towards the coast in what is termed surface Ekman flow (Csanady, 1982). The shoreward component of advection of buoyant Leeuwin Current water tends to be balanced by offshore flow in a frictional boundary layer near the seabed. This situation leads in winter to episodes of layered cross-shelf exchange, with denser coastal water

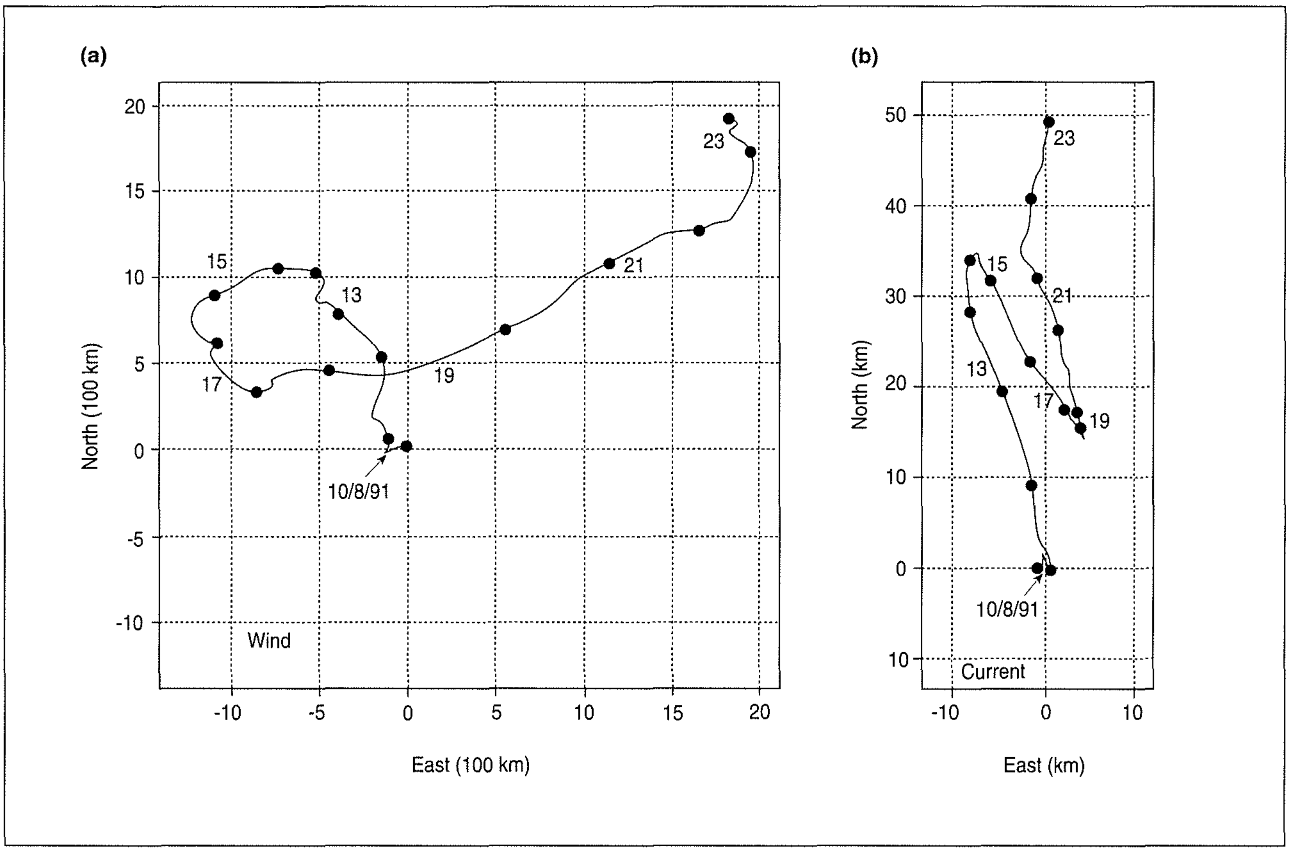


Figure 5.1-4. Progressive vector plots of (a) wind at Kwinana and (b) mid-depth current in Sepia Depression for 10-23 August 1991 showing current reversals associated with changes in wind direction.

moving obliquely offshore beneath incoming lighter (warmer) water (Mills *et al.* 1996). One such event occurred between 14 and 21 August 1991, and the resultant change in density, temperature and salinity structure is shown in Figure 5.1-5a and b. The details of this layered cross-shelf circulation are influenced by mixing due to wind and bottom frictional stresses, and by density differences between the coastal and offshore waters. Because of the depth (~ 150 m) of the Leeuwin Current surface mixed layer, and the general absence of strong, sustained southerly winds during winter, wind-induced upwelling of deeper waters onto the mid-shelf is unlikely to occur in winter (and was not observed).

Mesoscale (several hundred km) instabilities of the Leeuwin Current (i.e. meanders of the current growing seaward) are common. Meanders present offshore in mid August 1991 and their inferred water circulation patterns are shown in Plate 5.1-1a and b. Such meanders induce anticlockwise circulation of warm Leeuwin Current water, which moves away from the continental shelf as it enters and begins to circulate about the meander. On the inner side of the meander, northward flow over the continental shelf and slope may be generated, and may persist for a week or more. Such a flow was observed in mid August 1991 to induce sub-surface "upwelling" of deeper water (> 160 m) over the outer shelf (but not up to the mid-shelf) and to result in steeply-inclined density contours over the mid and inner shelf (Figure 5.1-5a), even although the data were collected toward the end of a prolonged period (about 10 days) of light, variable winds (Mills *et al.* 1996). Under these conditions, northward moving, denser coastal waters, including nutrient-rich estuarine plumes, were confined to the shelf for about one week, during which time mixing and dilution rates of these plumes were low. The long-shore extent of this behaviour was probably governed by the corresponding north-south size (about 100 km) of the offshore Leeuwin Current meander.

Plate 5.1-1 presents satellite images of the study region from the August 1991 survey period and highlights the spatial variability of features associated with the Leeuwin Current, and with mixing processes over the shelf. Plates 5.1-1a and b show the regional sea-surface temperature (SST) structure associated with the Leeuwin Current for 15 August 1991. Two large mesoscale meanders off Perth are clearly evident. Plates 5.1-1c and d are higher resolution images of water colour and SST, respectively, from the Landsat TM satellite pass of 0930 on 14 August 1991. These images also highlight the range of spatial scales at which interleaving and mixing occurs between the Leeuwin Current and the coastal waters, over the continental shelf region.

A major incursion of Leeuwin Current water onto the mid and inner shelf off Trigg was observed to persist for about 10 days during August 1991 (Plates 5.1-1d and f). A marked temperature-density front was located at the boundary between this warm water incursion and cooler coastal water (Mills *et al.* 1996). This front was generally located within

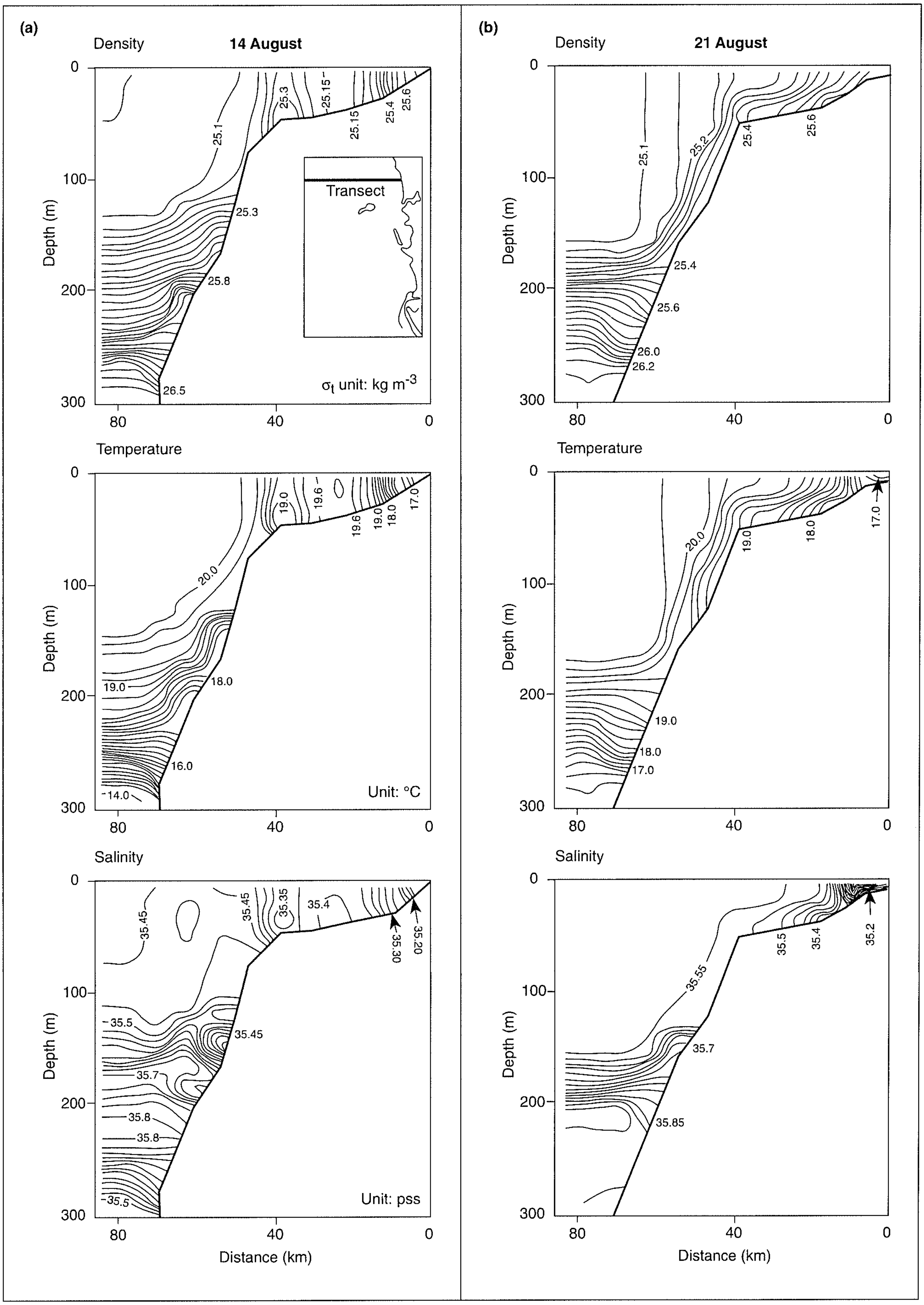


Figure 5.1-5. Cross-shelf vertical sections off Trigg (a) before and (b) after a storm in August 1991 (wind data shown in Figure 5.1-4). Inset shows transect location.

16 km of the coast and moved cross-shelf in response to changing meteorological and oceanographic conditions, as shown by the sequence of cross-shelf density contours in Figure 5.1-6. It was observed to migrate shoreward during conditions of predominantly northerly winds, and subsequently to reach the coast following the onset of strong westerly winds and longshore coastal current reversal. The warm water moved shoreward and displaced denser coastal water offshore as a lower layer. During southerly winds and northward coastal currents, the surface front again moved to an offshore position, accommodating northward flow of coastal water. Under these conditions, the frontal structure became almost vertical and limited the tendency for offshore movement (Ekman drift) of coastal water.

Estuarine plumes from the Peel-Harvey and the Swan-Canning estuaries have been identified using in-situ salinity, temperature, nitrate and phytoplankton measurements, and remotely-sensed Landsat TM imagery (Plates 5.1-1c and d (14 August 1991) and 5.1-1e and f (23 August 1991)). During mid August 1991 the plume from the Peel-Harvey Estuary extended up to about 100 km northward over the inner and mid continental shelf. The data suggest that inshore embayments and basins such as Comet Bay, Warnbro Sound and Sepia Depression can be significantly influenced by the Peel-Harvey Estuary plume (D'Adamo *et al.* 1995a).

The Swan-Canning Estuary plume was traced as far as 60 km north of the source in mid August 1991 (Plates 5.1-1c, d), but under winds from the northerly quadrants enters Owen Anchorage and Cockburn Sound (D'Adamo *et al.* 1995b). The injection of buoyant, less saline water to these water bodies in winter has a marked effect on their hydrodynamics (see section 5.1.2). This issue, and that of the general transport of estuarine plumes over the shelf under different wind conditions, has been the subject of numerical modelling, as discussed in chapter 6.

5.1.1.3 Summer

In summer, wind-stress is the principal factor influencing water circulation in the coastal and mid-shelf waters off Perth. This has been demonstrated by past oceanographic studies and by the analysis of more recent current meter data from these waters (Hearn, 1991; D'Adamo, 1992; D'Adamo and Mills, 1995a; Pattiaratchi *et al.* 1995). Strong south-southwesterly sea-breezes and night-time easterly to southeasterly winds result in a net northward drift. Tracking of isohalines within the relatively saline nearshore zone in late summer revealed significant northward coastal transport of water driven by sea-breeze winds. For example, a hypersaline pulse of Peel-Harvey Estuary outflow was tracked and found to travel northwards into Comet Bay, along Sepia

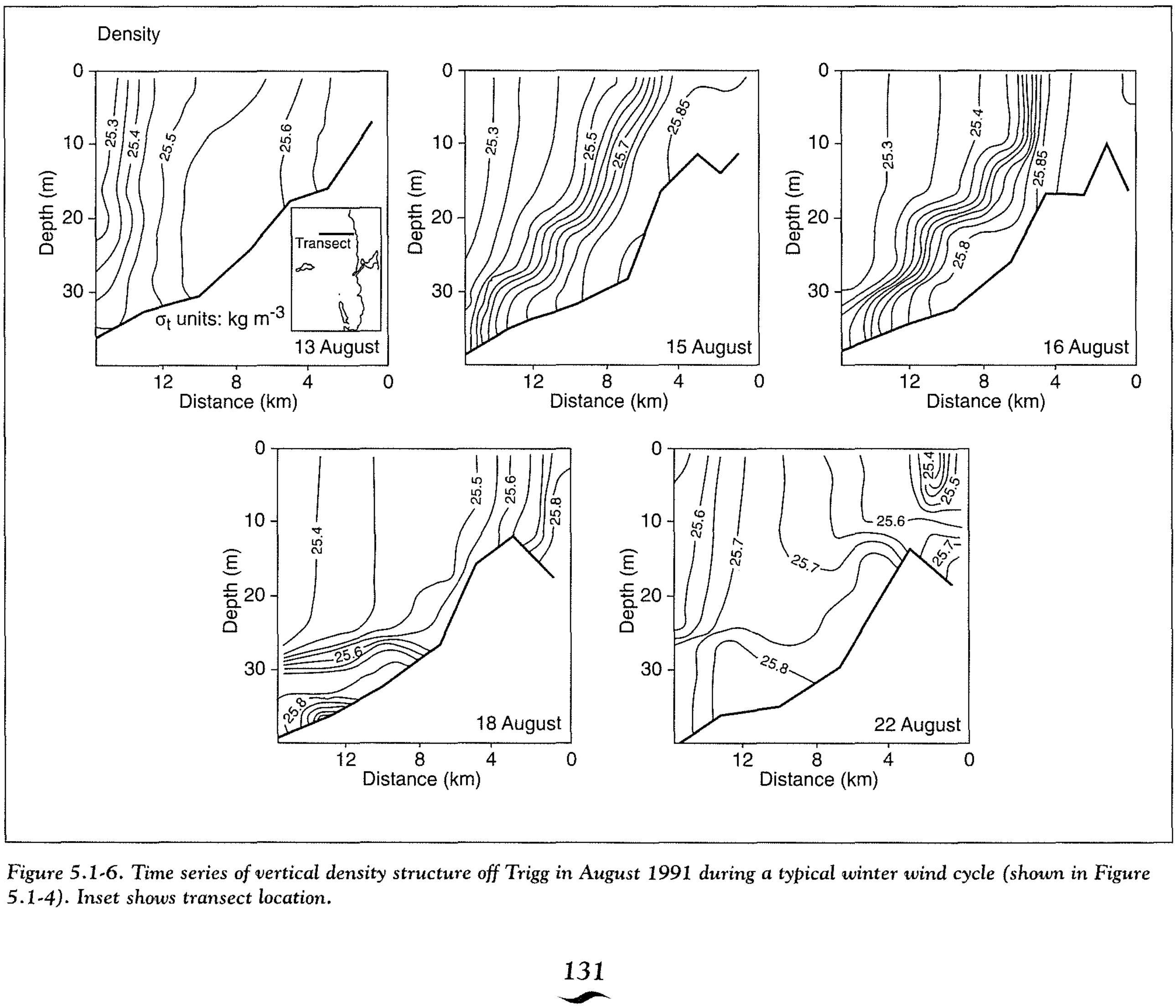


Figure 5.1-6. Time series of vertical density structure off Trigg in August 1991 during a typical winter wind cycle (shown in Figure 5.1-4). Inset shows transect location.

Depression and into Warnbro Sound. Numerical modelling also suggests that the balance between the north-south steric height gradient, the northward wind-stress and bottom friction, results in wind-stress being the dominant force in the coastal zone and surface waters of the mid-shelf region in summer (see chapter 6). The boundary defining the western extent of the wind-dominated coastal region moves perpendicular to the coast according to the relative magnitude of the principal forces. In summer, strong sea-breezes, of order 10 m s^{-1}, result in this boundary residing to the west of the 50 m contour (see chapter 6). Typical current speeds recorded by meters deployed nearshore and over the mid-shelf during January 1994 indicate that during typical summer south-southwesterly wind conditions of 5-10 m s^{-1}, northward currents of order 0.1 m s^{-1} were common.

During the field survey of 9-27 March 1992 the Leeuwin Current was found to be present over the outer shelf and continental slope, albeit in a weaker, shallower form than observed during winter by Mills *et al.* (1996). Sea-surface NOAA-AVHRR satellite imagery and complementary CTD profile data collected across the shelf on 23 March 1992 (D'Adamo and Mills, 1995d) revealed that a layer of about 50 m thickness of relatively low salinity and high temperature water was located over the shelf edge (Plate 5.1-2).

Cross-shelf CTD data suggested that, during 'summer' conditions, transient upwelling of cooler water from depths greater than 60 m over the outer shelf occurred as a result of strong southerly winds. For example, between 11 and 13 March 1992, upwelled water was observed to move onto the mid-shelf and to within about 10 km of the coast, where it was overlain by warmer, more buoyant coastal water, which tended to spread offshore as it drifted northward under the influence of the predominantly southerly winds. Figure 5.1-7 presents the cross-shelf density structure along a transect from Trigg to 80 km offshore on the 11 and 13 March 1992. The upwelling is indicated for example, by noting the shoreward displacement of the 25.5 kg m^{-3} isopycnal which moved vertically upwards from the 80 m to the 20 m depth contour, and shoreward from 45 km offshore to within about 10 km of the coast at Trigg. The upwelling occurred following strong (> 10 m s^{-1}) sustained south-southwesterly winds suggesting wind as the principal cause. The separation of flow about Rottnest Island can also influence the density structure to the north of the island (Pattiaratchi *et al.* 1995).

Additional data collected south of Rottnest Island (Figure 5.1-8) show a case of low salinity, low temperature bottom water extending shoreward as a bottom layer from the 40 m contour to the Five Fathom Bank, and into Sepia Depression. There was no apparent source of this water from the nearshore zone. Hence, transient upwelling of deep water onto the mid-shelf appears to be a cross-shelf transport feature both north and south of Rottnest Island.

Intensive measurements of the vertical density structure throughout synoptic meteorological cycles between the coast and the shelf break indicates that vertical stratification is established by diurnal heating, but that at night penetrative convection can mix the water typically to depths of 10-15 m. Hence, this establishes a base level for diurnal mixing. Wind mixing during strong sea-breezes has a dominant influence on vertical mixing and acts in concert with penetrative convection over diurnal time scales. Strong sea-breeze winds (about 10 m s^{-1} or greater) result in surface mixed layer depths of 20-30 m or more over the mid-shelf and nearshore zone. Such winds occur every day or two throughout the summer months.

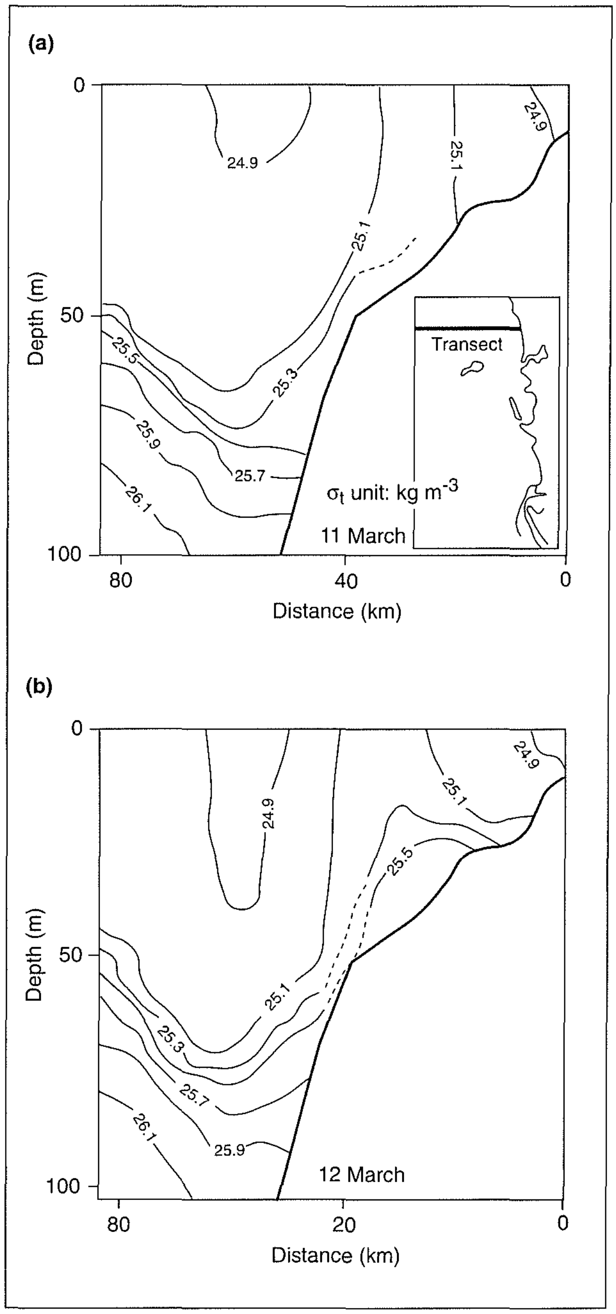


Figure 5.1-7. Cross-shelf vertical density structure off Trigg in March 1992 (a) before a strong sea-breeze and (b) following a strong sea-breeze showing upwelling of shelf waters into the nearshore zone. Inset shows transect location.

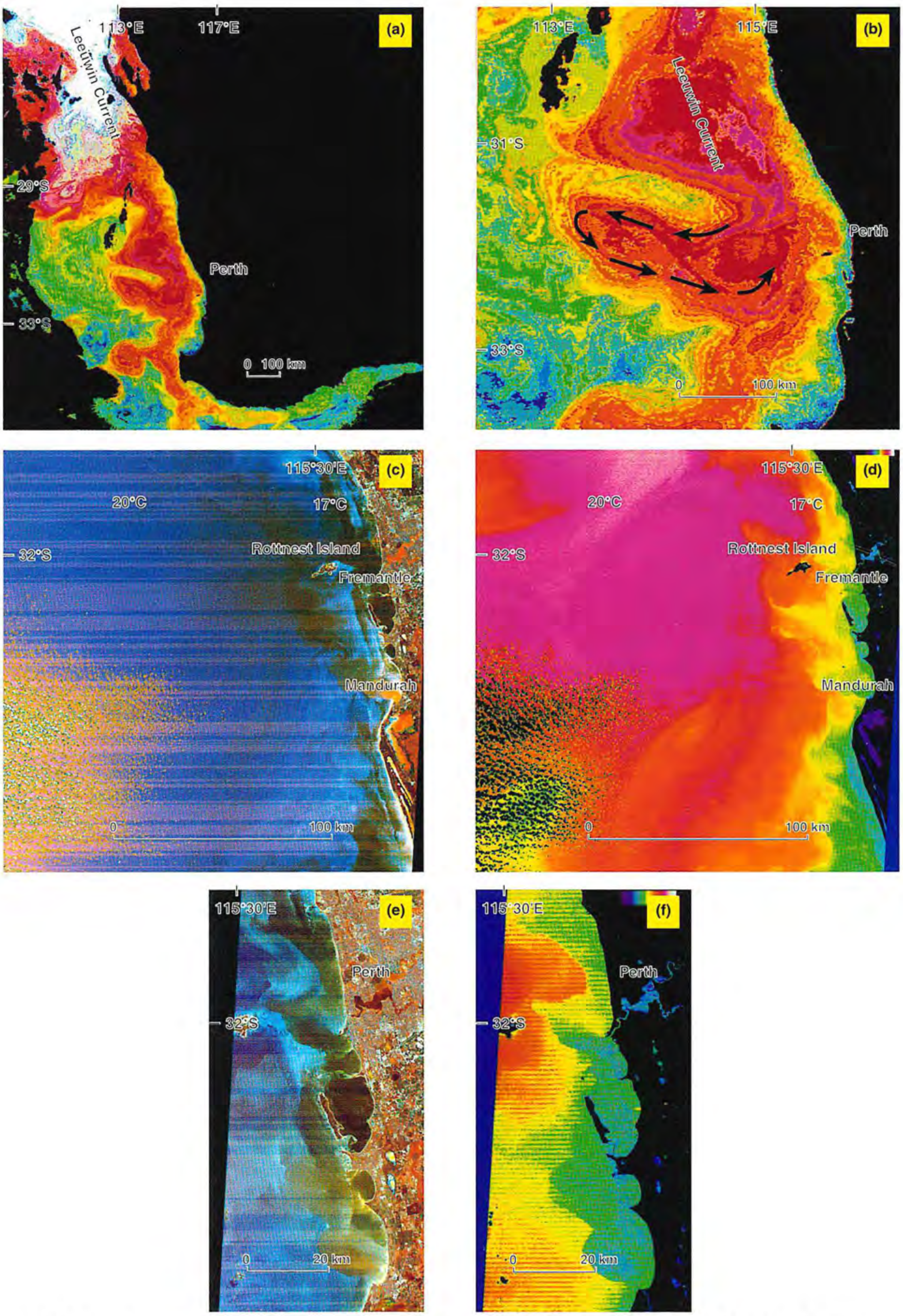


Plate 5.1-1. Satellite imagery of sea-surface temperature (SST) and water colour from NOAA-AVHRR and Landsat TM data collected in 1991. SST images of (a) Leeuwin Current and (b) Leeuwin Current meander west of Rottnest Island on 15 August. Images of the southwest Australian coastal zone showing (c) water colour and (d) SST on 14 August; (e) water colour and (f) SST on 23 August. (Plates a to d reproduced from Wylie et al. 1992).

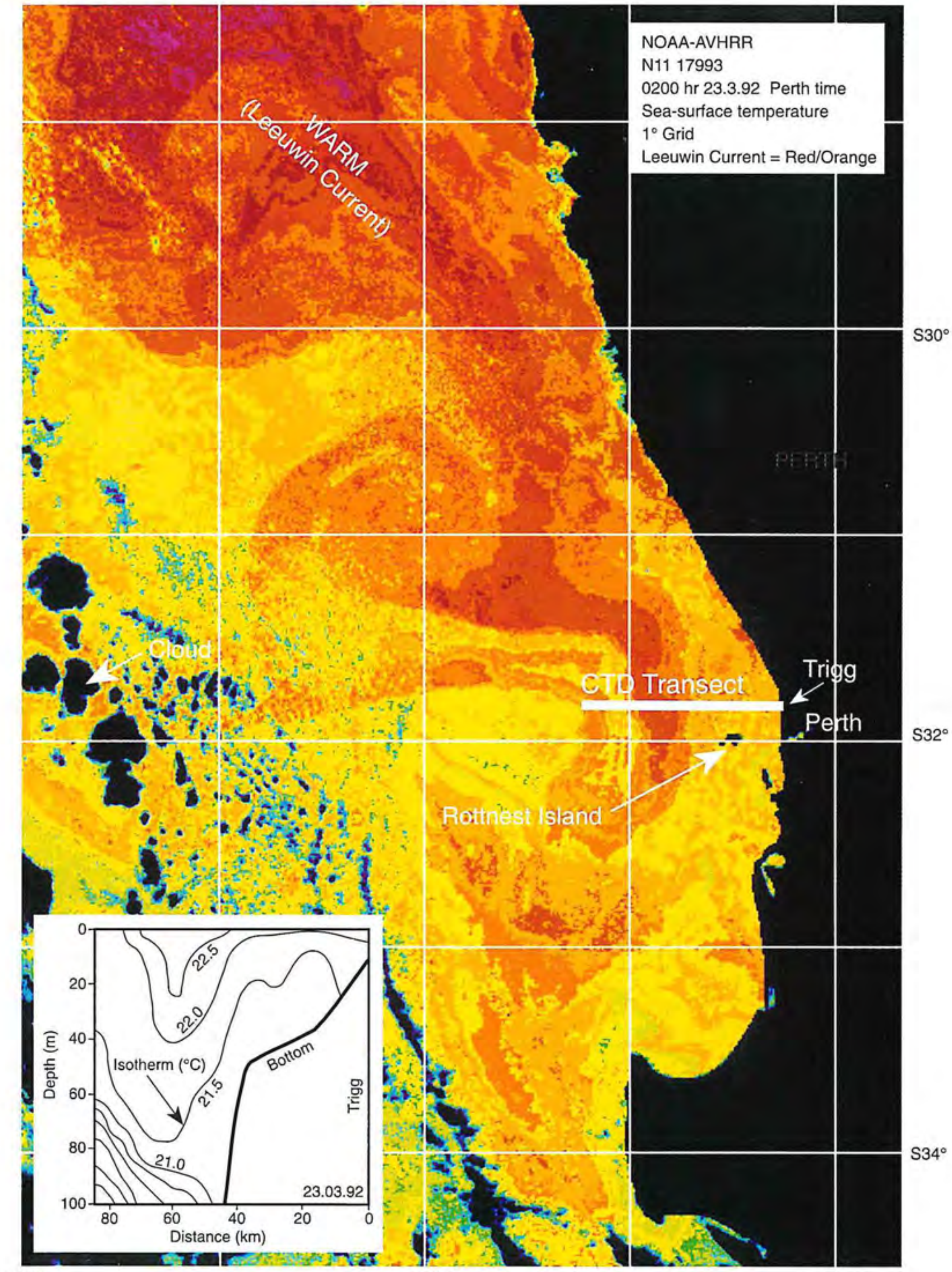


Plate 5.1-2. NOAA-AVHRR sea-surface temperature image off the southwest Australian coast taken at 0200 hrs 23 March 1992. Red indicates the relatively warm Leeuwin Current core. The inset shows a complementary cross-shelf temperature section (23 March 1991) through the shelf zone and warm current core off Trigg.

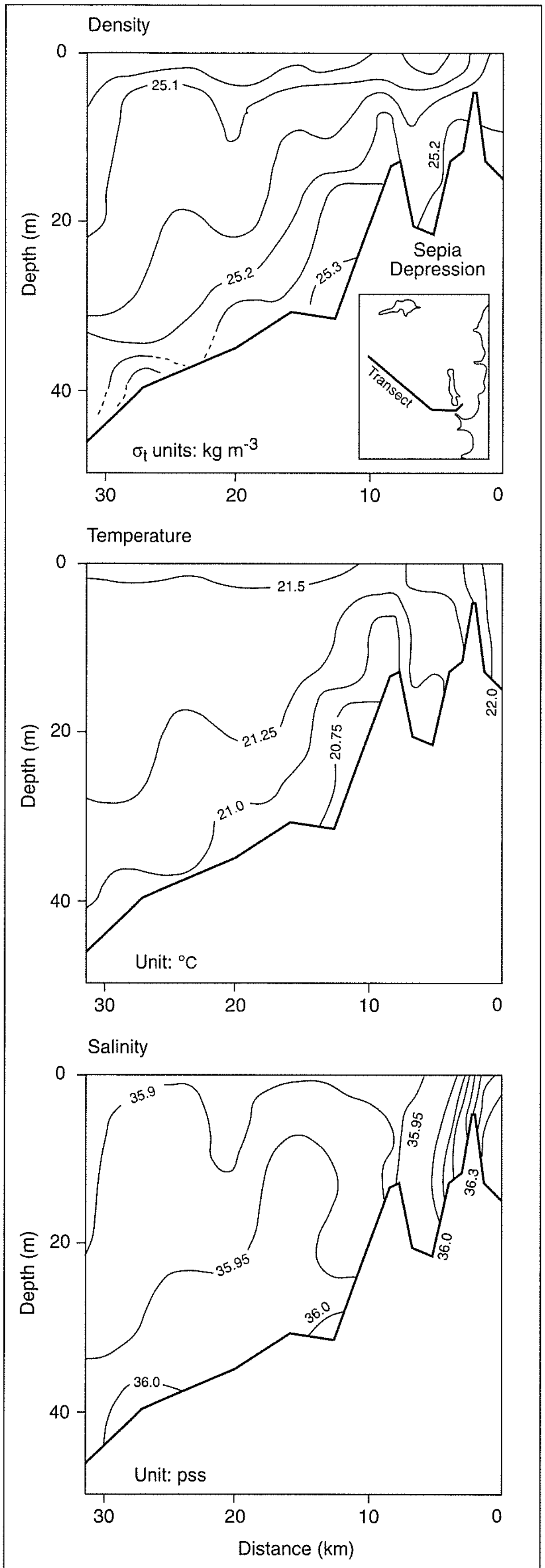


Figure 5.1-8. Cross-shelf vertical sections off Garden Island in March 1992 showing density, temperature and salinity structure indicative of upwelling of shelf waters into Sepia Depression. Inset shows transect location.

Because of the regularly vertically well-mixed nature of the region, barotropic circulation (i.e. not influenced by density effects) predominates. However, the identification of the potential for strong stratification to form during weaker wind periods (about 5 m s^{-1} or less) suggests that density effects could play a role during these periods. As summer progresses evaporation raises the salinity of the coastal zone with respect to offshore waters. Cross-shelf CTD data between Mullalloo Point, Fremantle and the 30 m contour showed that, on occasions during summer, under weak wind conditions, relatively high salinity water resided above low salinity water that was colder and therefore more dense. One possible explanation for this is the advection of saline coastal water offshore as a buoyant layer.

Conclusions

Within the coastal zone (i.e. shoreward of the major reef lines and islands) buoyancy fluxes due to estuarine discharges, solar heating and atmospheric cooling are important in establishing vertical and horizontal density gradients. In a period termed the 'winter-spring' regime (August to October) Cockburn Sound and adjacent waters are less dense than mid-shelf waters, due mainly to positive coastal buoyancy inputs from rivers and solar heating. This is followed by a 'summer' regime (December to March) when river flows are minor and nearshore heating and evaporation have opposing effects on nearshore density and, as a result, cross-shelf density differences diminish. Finally, in the 'autumn' regime (April-July) continued evaporation during autumn and nearshore cooling during early winter results in the nearshore waters being denser than the mid-shelf waters.

These density gradients influence the vertical and horizontal distribution of contaminants, and also the flushing of contaminants from the semi-enclosed basins and lagoons near the coast. The presence of sills, islands, reefs and banks restricts the exchange of contaminants from these basins. Wind-stress is the most important agent that causes water movement and mixing in the nearshore zone, however density gradients (vertical and horizontal) and the earth's rotation strongly modify the nature of wind-driven currents and flushing from the basins.

Once nearshore waters, and contaminants contained in them, have been driven out of these nearshore protected areas, their transport is strongly influenced by shelf-scale and meso-scale processes.

In autumn, winter and into spring the relatively buoyant Leeuwin Current extends over the shelf and a frontal zone between it and coastal water is present. The distance from the shore to this frontal zone varies. Under winds from the southwest to southeast the frontal zone migrates further offshore, inclines to near vertical, and marks the seaward boundary of the northward moving denser nearshore waters.

Under winds from the northeast to northwest the influence of Coriolis force leads to a surface Ekman drift of buoyant water shoreward and a corresponding shoreward migration of the frontal zone, associated with cross-shelf escape of relatively dense (cold) nearshore water as an undercurrent. This describes an effective flushing mechanism for nearshore waters and any contained contaminants. Variations to these dynamical sequences occur as winds swing around the compass during synoptic weather systems.

In summer, by far the most important transport mechanism is that of wind-driven northward coastal flow under the recurrent pattern of moderate to strong south-southwesterly sea-breezes. The nearshore zone was found to be characteristically of higher salinity than mid-shelf waters during summer and into autumn, due mainly to the differential effects of evaporation. In addition, hypersaline water can flow into the coastal zone from the Peel-Harvey Estuary and one such pulse was tracked as a northward coastal flow, entering Comet Bay and Warnbro Sound, driven by sea-breeze winds. Vertical mixing to depths of at least 20-30 m occurs on a regular basis within the nearshore and shelf areas due to a combination of wind-stress and penetrative convection. The Leeuwin Current is either absent or weak over the shelf in summer due to the opposing influence of northward wind-stress.

Hence, the effective management of the coastal waters off Perth, and in particular of the protected nearshore zone in the southern metropolitan coastal waters, requires an appreciation of a wide range of spatial scales of transport, from basin scale, to shelf scale.

5.1.2 Local-scale oceanography

The broad objectives of the local-scale oceanographic studies were to identify the dominant characteristics of the circulation and mixing of the nearshore basins and to develop an understanding of the principal exchange mechanisms between these waters and adjacent mid-shelf waters. The principal methods employed to achieve this were, first, the review of historical oceanographic data, and second, the measurement of currents and density stratification over diurnal (< 1 d), synoptic (< 10 d) and annual (< 1 y) time scales. Particular attention was paid to identifying the key hydrodynamic processes during periods of strong vertical and horizontal density stratification and relatively weak winds (< 7.5 m s^{-1}) as these conditions had not been addressed adequately in other previous studies. A further objective was to provide characteristic information on the oceanography of the study area in order to facilitate the choice, development, and implementation of local-scale hydrodynamic and transport models (see section 6.2).

The specific objectives were: to detail the seasonal variations of physical properties in the coastal waters in response to the major forcings of wind, atmospheric heating and cooling, evaporation, river discharge and regional currents; to determine the major hydrodynamic regimes in relation to vertical and horizontal density gradients, and to determine the major mechanisms of exchange and mixing within and between the semi-enclosed basins for each of the identified hydrodynamic regimes. In addition, the objectives in relation to the modelling component of the SMCWS (see chapter 6) were to provide: an indication of the periods of the year when density effects were likely to be of importance; calibration and validation data for the development and tuning of the models, and forcing data for the model simulations (e.g. salinity-temperature-density structure, meteorological forcings and sea-level gradients).

The local-scale oceanographic studies were focussed primarily on Cockburn Sound in recognition of the central focus of this water body to the SMCWS. In addition, reviews of past studies (Hearn, 1991; D'Adamo, 1992) indicated that the hydrodynamic characteristics of the sound were likely to be of equivalent or greater complexity than for the other 'core areas'. It was therefore reasoned that a model capable of simulating the mixing and transport of water in Cockburn Sound would also be applicable to other areas of interest. Less-intensive studies were undertaken in Warnbro Sound on the basis that the fundamental hydrodynamic processes of the two basins were likely to be similar. Additional studies tracked the brackish plume emanating from the Peel-Harvey Estuary in winter and a coastal flow of high salinity water in summer; in both cases, the entry of these waters into the embayments was documented.

Intensive studies were undertaken during typical winter and summer conditions in Cockburn Sound as these periods had been previously identified as the two distinct and contrasting hydrodynamic 'seasons' on the basis of wind patterns and density structure (Hearn, 1991; D'Adamo, 1992). Historical oceanographic data sets indicated that, during winter, plumes of low salinity Swan River water were periodically driven into Owen Anchorage and Cockburn Sound by southward winds, thereby influencing the stratification of these basins. In summer, the historical data indicated that strong vertical mixing due to sea-breeze winds led to regular vertical destratification to the bottom of Cockburn Sound. Annual cycles were identified in the cross-shelf salinity and temperature differences between the nearshore basins (Owen Anchorage, Cockburn Sound and Warnbro Sound) and the adjacent waters of Sepia Depression and the mid-shelf. For example, the nearshore zone was found to be of relatively low salinity and temperature in winter and of relatively high salinity and temperature in summer, with the transitions occurring some time in spring and autumn (D'Adamo, 1992). The importance of buoyancy fluxes and density effects to the dynamics of Cockburn Sound was generally acknowledged (e.g. Steedman and Craig, 1979); however, there was little detailed understanding of how these factors modified the water circulation, flushing and exchange characteristics of the sound. Further investigations of the hydrodynamic influence of density effects were therefore carried out as part of the SMCWS.

The 'intensive' field surveys were complemented by a series of approximately monthly conductivity-temperature-depth (CTD) surveys over a 20-month period to investigate the seasonal characteristics of the vertical salinity, temperature and density structure in the nearshore and mid-shelf zones, and also the cross-shelf differences in these parameters. The results of the routine surveys, when considered in conjunction with the historical oceanographic data sets, indicated that there was also a distinct 'autumn' regime during which the nearshore embayments were both more persistently vertically stratified in salinity, temperature and density, and also significantly saltier (or cooler by late autumn/early winter) and therefore denser than offshore waters. As a result, 'mini-intensive' autumn surveys were undertaken in March 1994 in Warnbro Sound and in May 1994 in Cockburn Sound to resolve more clearly the characteristics of the hydrodynamics of these nearshore basins during typical autumn conditions.

In addition to comprehensive CTD profiling, data were obtained from satellite imagery, current meters, drogue tracking, acoustic doppler current profilers, sea level recorders, wave recorders and weather stations deployed throughout the study region during various periods over the duration of the oceanographic field programme, from August 1991 to May 1994.

Reviews of past oceanographic studies were conducted by Hearn (1991) and D'Adamo (1992) and summarised in Simpson *et al.* (1993). Further technical details of the oceanographic and related studies conducted since 1991 can be found in Simpson *et al.* (1993), D'Adamo *et al.* (1995 a, b), D'Adamo and Mills (1995 a, b, c), Kruh (1994), Coleman (1995), Barry (1995).

Findings

5.1.2.1 *Forcings*

The wind is a major driving influence on water circulation patterns in the southern coastal waters of Perth. The wind has diurnal, synoptic and annual periodicities, and the hydrodynamic behaviour of the coastal waters responds to these periodicities. The bathymetry steers wind-driven currents and the nearshore zone is surrounded by banks, reefs and islands (Figures 1.1-1, 1.1-2 and Plate 4.1-1), which impose important controls on the exchange between the basins and offshore waters. For example, the presence of the Parmelia Bank restricts water exchange through the northern opening of Cockburn Sound, as does the causeway at the southern opening. These findings support the conclusions of earlier studies (Maritime Works Branch, 1977a, b; Steedman and Craig, 1983; Hearn, 1991).

This study has confirmed that dynamically significant density stratification occurs within the nearshore basins, and between the basins and their surrounding shelf areas, throughout much of the year. In this respect, the role of buoyancy fluxes associated with discharge from the Swan-Canning and the Peel-Harvey estuaries, solar heating and evaporation, and the offshore presence of the Leeuwin Current has been discussed in section 5.1.1. The field data have shown that the density difference between embayment and shelf waters is an important factor in determining the circulation patterns and efficiency of exchange, particularly in relation to the deep basins. Analyses of the field survey data have led to a clearer understanding of the conditions of wind-stress and night-time surface cooling necessary to induce complete vertical mixing of the water column.

Wind speeds greater than about 10 m s^{-1} must occur for at least 5-8 hours for wind-stirring to totally break down 'typical' vertical density stratification in Cockburn Sound and mix the entire water column to the bottom at about 21 m depth. These conditions are only common between November and March, when strong sea-breezes occur on average every 1-2 days, and during winter storms, which occur on average every 7-10 days from June to September. Thus, compared to the surface mixed layer (which is generally less than about 15 m deep), the vertical mixing of bottom waters from Cockburn Sound and Warnbro Sound is restricted for a significant portion of the year. This is most pronounced in autumn, when water densities are relatively high compared to the adjacent oceanic waters and wind speeds are low.

Horizontal density stratification generates density-driven currents and these currents make a significant contribution to the circulation within the basins particularly during autumn to mid-spring when stratification is strong and wind speeds are relatively light and variable in direction.

As discussed in section 2.2.7, in the typically density-stratified Cockburn Sound the circulation patterns are significantly influenced by the earth's rotation (Coriolis force) because current speeds are relatively weak (less than about 0.2 m s^{-1}) (Mills *et al.* 1995) and the dimensions of the basin are sufficiently large compared to the baroclinic radius of deformation (Csanady, 1982).

The numerical modelling component of the SMCWS (see chapter 6) utilises the results of these process studies to simulate the exchange characteristics of the sub-regions within the southern metropolitan coastal waters.

5.1.2.2 *Winter-spring regime*

Cockburn Sound and adjacent waters

The exchange between Cockburn Sound and adjacent offshore waters during the 'winter-spring' regime is influenced markedly by the cyclical passage (7-10 days) of clockwise low pressure systems and the outflow of low salinity Swan River water, which occurs most strongly from about June to September (D'Adamo *et al.* 1995b). During this period groundwater flux into Cockburn Sound is weak compared to river flow. A typical cycle during winter and early spring begins with the southward advection of low salinity water from the Swan River through Owen Anchorage and into the sound by northeasterly to northwesterly winds that precede these frontal systems. The river water is mixed with Cockburn Sound water by the storm winds (> 10 m s^{-1}) which normally accompany low pressure systems as they cross the coast. This buoyancy flux and mixing results in a horizontal density difference between basin waters and the adjacent offshore waters (Figure 5.1-2) and, as moderating southerly winds advect buoyant surface waters out of the Sound, replacement inflows of denser shelf water enter Cockburn Sound via the openings, particularly through the northern opening, and sink to the bottom under gravity. Figures 5.1-9 and 5.1-10 demonstrate this dynamical sequence for the post-storm period (20-23 August) of the intensive winter field survey of 1991. The plunging inflow from the north tends to be deflected to the eastern side of the central basin in response to the earth's rotation. Typically, after 2-3 days, the entire deep basin is influenced by dense inflow. Most of the bottom water in Cockburn Sound is displaced vertically by these inflows and enters the diurnal mixed layer (0-15 m depth) where it is mixed during diurnal wind-mixing and penetrative convection cycles. Water in the mixed layer exits the sound through the northern opening via wind-driven advection caused by strong southerly winds which occur after the 'front' has crossed the coast (Figures 5.1-9 and 5.1-10).

On the basis of these results of the intensive winter oceanographic survey, in conjunction with the statistics of the frequency of occurrence of storms, it was concluded that this dynamical sequence is repeated on average every synoptic cycle (7-10 days) throughout the 'winter-spring' regime. The exact duration of this particular regime will depend on the timing of the onset and duration of strong Swan River flows. Throughout the synoptic cycle the water column in Cockburn Sound is subjected to the diurnal forcings of heating and cooling, and also to occasional weak afternoon sea-breezes. As a result, the water column goes through repeated cycles of vertical density stratification and mixing above the level of the main pycnocline, which is the vertical zone across which the most intense density stratification occurs, due to salinity stratification. This is exemplified by Figure 5.1-9, which shows a typical time series of vertical stratification in central Cockburn Sound during the winter intensive field survey of 13-23 August 1991. This figure also shows the complete vertical mixing of the water column as a result of storm winds, and the reformation of the main pycnocline in Cockburn Sound due to the inflow of dense (more saline and warmer) shelf water, following the storm.

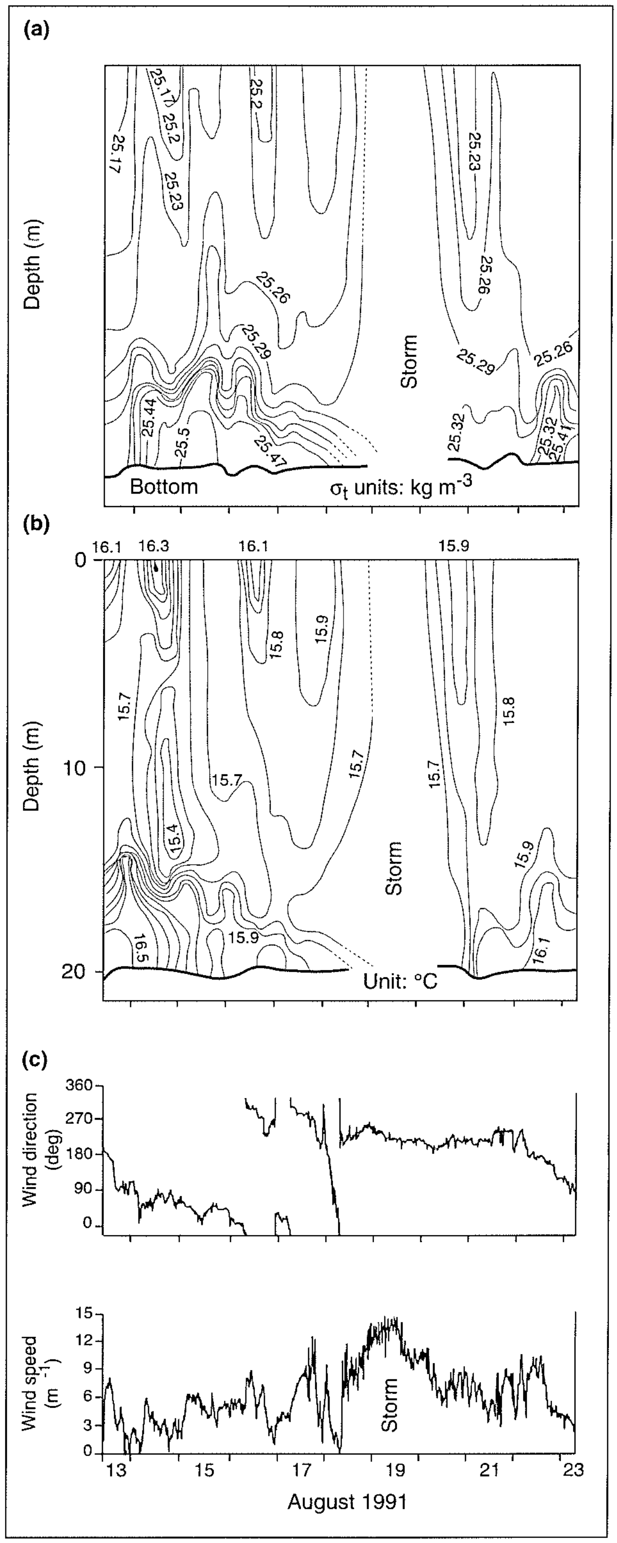


Figure 5.1-9. Time series of vertical (a) density and (b) temperature structure in central Cockburn Sound and (c) wind velocity, showing the combined influences of diurnal heating and cooling, storm mixing and deep-water renewal of high salinity water in winter.

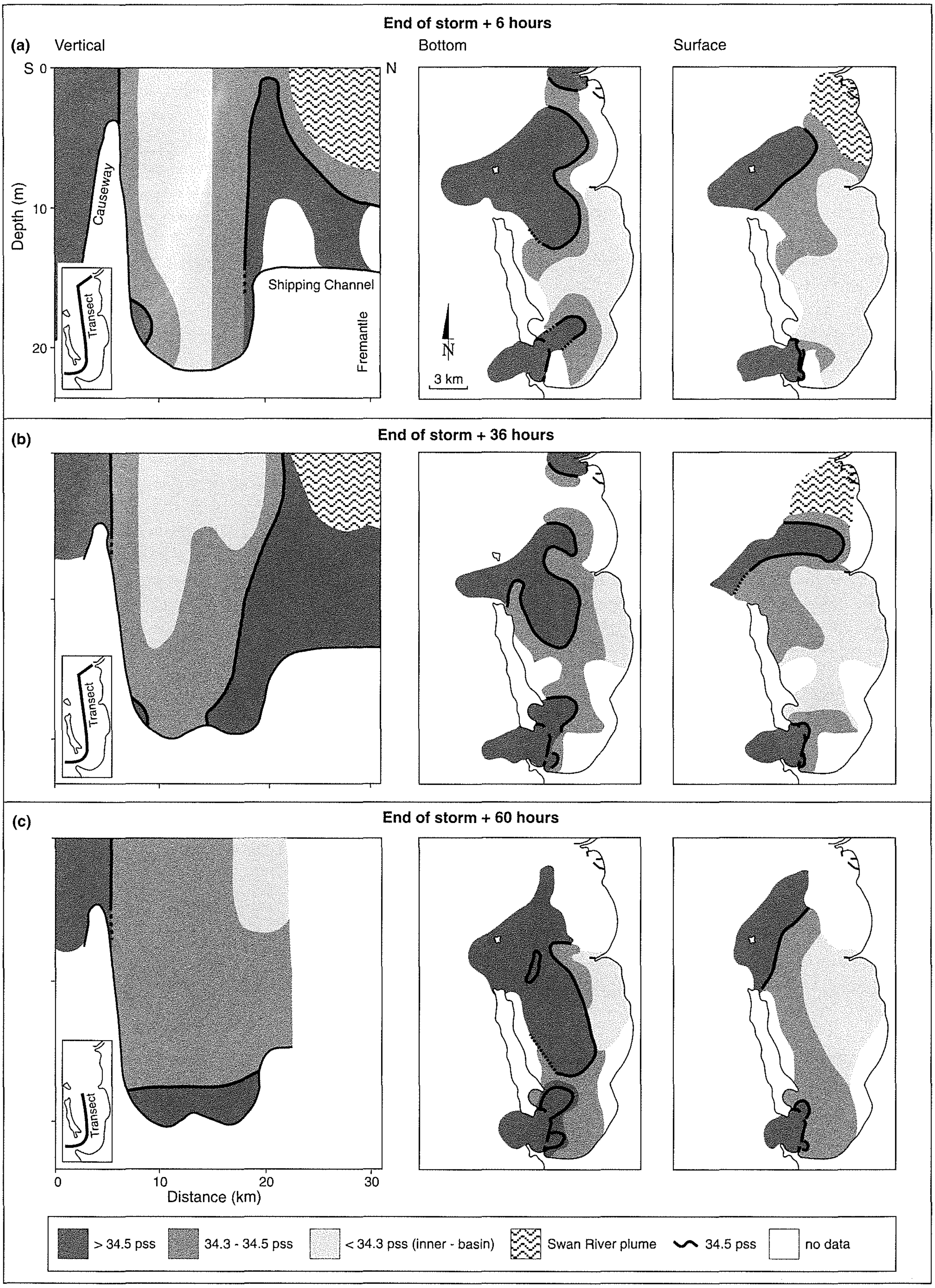


Figure 5.1-10. Time series of vertical and plan (bottom layer and surface waters) salinity structure in Cockburn Sound showing deep water renewal by high salinity shelf waters and surface water exit of low salinity waters after a winter storm in August 1991. Insets show vertical transect paths.

The density structure in Cockburn Sound also undergoes upwelling and downwelling in response to wind forcing (D'Adamo *et al.* 1995b) as is typical of stratified basins subjected to surface wind stress (Imberger, 1994). Both the field measurements (D'Adamo *et al.* 1995b) and the modelling results (see section 6.2) suggest that the response of the density structure includes a component transverse to the direction of the wind, as would be expected in a basin of this size, given the presence of the earth's rotation (e.g. Csanady, 1982).

Sepia Depression and Warnbro Sound

The salinity-temperature data from the surveys of the SMCWS, supported by the modelling results (see chapter 6), indicate that the hydrodynamic influence of the Swan River outflows, as a source of buoyant water, extends to Sepia Depression west of Garden Island. Under typical winds the time scale for this to occur is in the order of 1-2 days. In addition, salinity tracking during the winter intensive survey indicated that low salinity water in Cockburn Sound, derived in part from the Swan River outflow, can be driven from southern Cockburn Sound into Warnbro Sound in 1-2 days via the Shoalwater Bay and Sepia Depression region under east-northeasterly to north-northwesterly winds (D'Adamo *et al.* 1995a). Because only relatively small amounts of freshwater enter Warnbro Sound via groundwater (Appleyard, 1990) the hydrodynamic transport of low salinity waters from Cockburn Sound and Sepia Depression into Warnbro Sound can be significant.

These findings highlight the important role that the Swan River outflow has on the circulation and flushing of Cockburn Sound and adjacent waters in winter and early spring, and emphasise the importance of including density effects in the hydrodynamic modelling of these waters during this period.

Peel-Harvey Estuary outflow and Warnbro Sound

Low salinity outflows from the Peel-Harvey Estuary occur mainly from June to September and considerable loadings of nitrogen and phosphorus are exported to the nearshore marine waters during this period, particularly during average or above average rainfall years (Deeley *et al.* in preparation). A study was undertaken to examine the long-shore transport of these outflows by using salinity and temperature data to track the low salinity plumes during a typical 10-day winter wind cycle in August 1991 (D'Adamo *et al.* 1995a). The results show that, under south-southwesterly winds of 5-10 m s^{-1}, the plume traversed the 25 km distance from Mandurah to Warnbro Sound in 1-2 days (Figure 5.1-11). Contemporaneous current meter data, satellite imagery and nutrient data provided additional support for the conclusion of direct transport between these two areas. These field results are also consistent with the results of model simulations of the transport of Peel-Harvey outflow plumes under similar hydrological and meteorological conditions (see chapter 6). An analysis of long-term wind records indicated that the required south-southwesterly wind events (5-10 m s^{-1} lasting for about one day) occur, on average, about 3-4 times per month between June and September. These data, coupled with the high export of nutrients from the estuary during these months, suggest that the outflow from the Peel-Harvey Estuary has the potential to influence the ecology of Warnbro Sound during winter and spring.

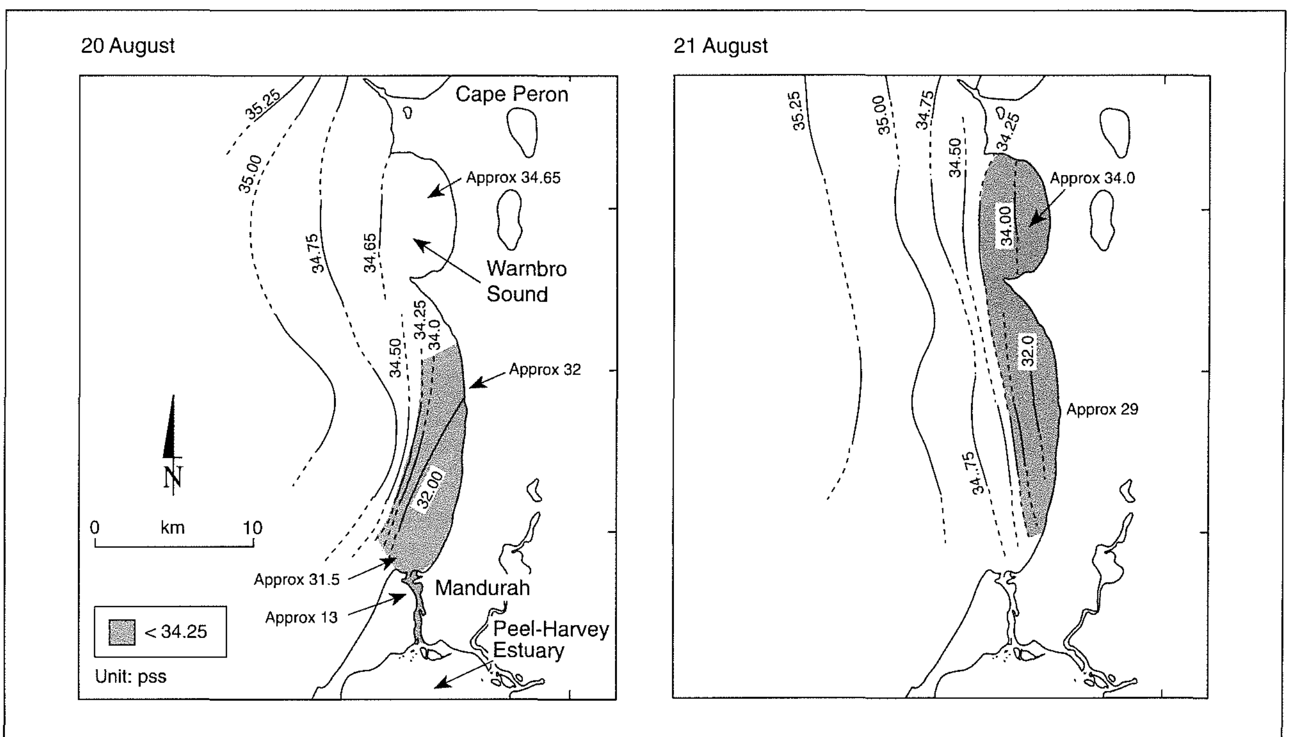


Figure 5.1-11. Surface salinity structure between Mandurah and Cape Peron on consecutive days in August 1991, showing the northward transport of buoyant Peel-Harvey outflow water under south-southwesterly winds.

5.1.2.3 *Summer regime*

Cockburn Sound and adjacent waters

Persistent strong south-southwesterly winds (during afternoon sea-breezes) occur in the coastal regions of Perth during the 'summer' regime from about December to March. As a consequence, transport is primarily wind-driven and the predominant flow direction is northwards within the mid-shelf to nearshore zone (D'Adamo and Mills, 1995a). Cockburn Sound undergoes a regular diurnal cycle of day-time stratification, due mainly to solar radiation, and vertical mixing due mainly to afternoon sea-breezes and penetrative convection (caused by nightly heat losses), as exemplified by the time series of vertical density, temperature and salinity stratification in central Cockburn Sound during the summer intensive surveys of March 1992 (Figure 5.1-12). During 9-13 March 1992 a northward coastal flow was found to enter Cockburn Sound via its northern and southern openings. The inflow was both more saline and colder than resident sound water. Because of this the density of the inflow was greater than the resident basin water and therefore tended to sink and spread along the bottom of the basin, thereby strengthening the diurnally evolving vertical density stratification. The relatively high bottom salinities shown in Figure 5.1-12 were due to this inflow. Although vertical stratification was established on a daily basis its duration was typically short-lived due to regular diurnal mixing by day-time sea-breeze winds and night-time penetrative convection.

Under the typical 'summer' conditions of the intensive field surveys of March 1992 vertical mixing of the stratified waters of Cockburn Sound due to penetrative convection led to upper mixed-layer depths of between about 5 and 15 m, with a mean depth of about 10 m. If the mixing due to penetrative convection was preceded by winds of greater than about 7-10 m s^{-1} for 5-8 hours or more, the water column was fully mixed by dawn. Winds, greater than about 10 m s^{-1} for about five hours, were able to fully mix the water column alone. Analyses of long-term wind records from Fremantle indicate that diurnal forcings should lead to full-depth mixing in central Cockburn Sound about 10-15 times per month between December to February. In general, the results of the summer analyses suggest that, during summer, the combined actions of wind-mixing by afternoon sea-breezes and night-time penetrative convection are likely to fully mix the water column in most of Cockburn Sound and its adjacent waters on average every 1-2 days.

Circulation and transport in Cockburn Sound during 'summer' over synoptic time scales are, therefore, primarily wind-driven and influenced by the bottom topography; density effects are effectively of lesser importance due to the regular mixing that occurs throughout the period of a synoptic cycle.

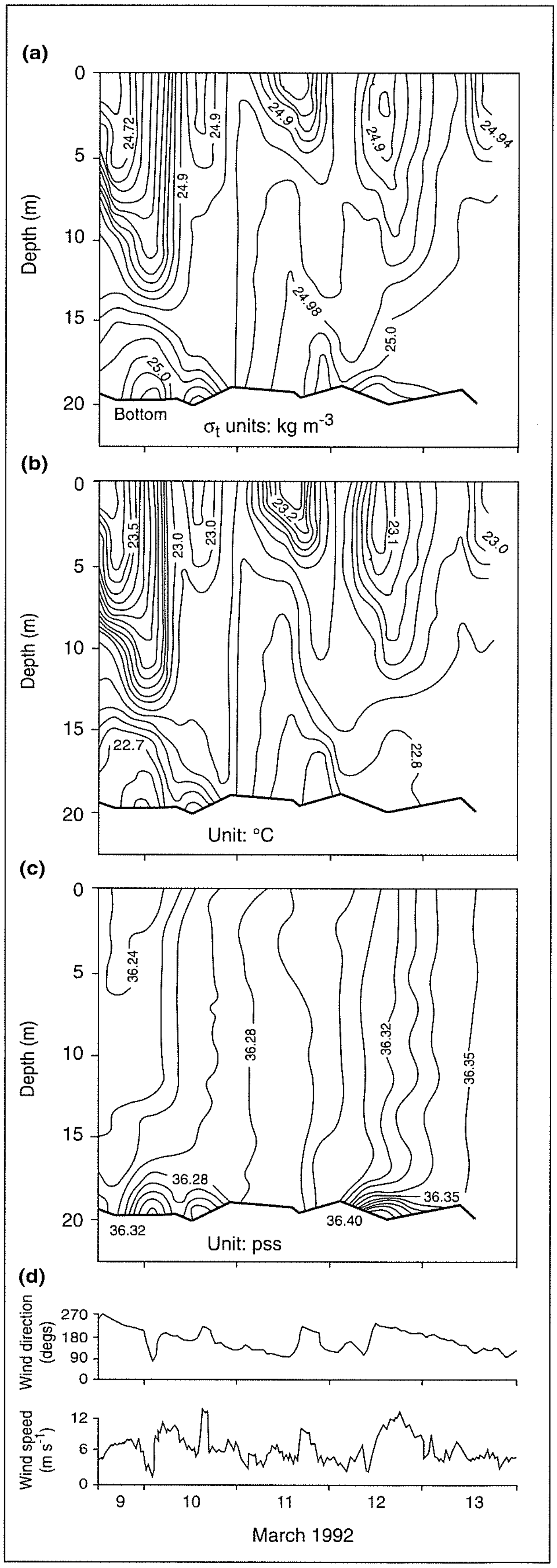


Figure 5.1-12. Time series of vertical (a) density, (b) temperature and (c) salinity structure in central Cockburn Sound and (d) wind speed and direction, showing diurnal heating, cooling and mixing during late summer.

Due to the regularity of full-depth mixing, it was concluded that three-dimensional barotropic hydrodynamic models forced by wind stress were, in the first instance, appropriate for numerical simulations of the main circulation features of this coastal zone (including the nearshore semi-enclosed basins) for typical 'summer' conditions. However, because of the observed diurnal evolution of the stratification by solar radiation and/or advection of high salinity water, baroclinic modelling may be a useful adjunct to account for baroclinic pressure gradients which may be comparable with other forcings, even under vertically well-mixed conditions. Some caution should be exercised in regard to these general conclusions in the more wind-sheltered areas of Cockburn Sound, particularly near Mangles Bay, where the occurrence of full-depth vertical mixing may not be as regular as in the remainder of Cockburn Sound.

Northward transport from the Peel-Harvey Estuary region to Warnbro Sound and its adjacent waters

During the summer field survey period of 9-13 March 1992 (D'Adamo and Mills, 1995a) the circulation of coastal waters off Perth was dominated by longshore wind-driven northward advection due to south-southwesterly sea-breezes. As mentioned above, this water intruded into the series of nearshore semi-enclosed basins along the southern metropolitan coastline. A long-shore salinity gradient was identified, with highest values to the south. This gradient in salinity was augmented by high salinity flow from the Peel-Harvey Estuary. One such pulse of hypersaline Peel-Harvey Estuary outflow was found to flow northward through Comet Bay and then into Warnbro Sound within 1-2 days. These inflows were intermittently traced entering Cockburn Sound as a relatively dense layer propagating along the bottom, but this layer was quickly mixed to the surface by typical strength sea-breezes and penetrative convection acting in concert over a diurnal cycle. Typical northward transport rates of about 10 km per day led to the transport of relatively high salinity water from the vicinity of Mandurah to Cockburn Sound within about four days, driven by south-southwesterly winds. These findings demonstrate that during south-southwesterly winds a hydraulic link also exists in summer between the Peel-Harvey outflow and the nearshore basins north of the estuary. The occurrence of the blue-green estuarine alga *Nodularia spumagina*, which has been commonly found in the Peel-Harvey Estuary in late spring (when south-southwesterly sea-breezes begin to establish themselves as part of the regular diurnal wind cycle), was recorded in Warnbro Sound phytoplankton populations in November 1992 and this further supports these conclusions.

In terms of the ability of diurnal mixing to regularly mix the water column of Warnbro Sound in 'summer', the results for Cockburn Sound have been extended to Warnbro Sound to arrive at similar conclusions. Warnbro Sound is more exposed than Cockburn Sound and receives only minor inputs freshwater via groundwater. The summer intensive survey (9-27 March 1992) provided an indication of the regular afternoon structure in Warnbro Sound and it was found that full-depth vertical mixing during these typical 'summer' conditions was common. Regular sea-breezes and night-time penetrative convection therefore provide enough mixing potential in Warnbro Sound to overcome day-time vertical density gradients on a regular basis, that is, at least as regularly as in Cockburn Sound, which is every 1-2 days.

5.1.2.4 Autumn regime

Cockburn Sound and adjacent waters

During the 'autumn' regime the waters of Cockburn Sound are more saline, and as a result generally more dense, than adjacent coastal and offshore waters (see section 5.1.1). The onshore-offshore density difference is primarily the result of the intensive evaporation that occurs over the summer and autumn months, combined with the differential flushing rates of the basin and the coastal and offshore waters, and secondarily due to the emerging influence of low salinity, relatively warm water introduced to the shelf by the Leeuwin Current during this period (Figure 5.1-2). Into early winter the temperatures of the nearshore zone can drop sufficiently with respect to mid-shelf waters to maintain this cross-shelf density difference (with the nearshore zone being relatively dense), as highlighted by Figure 5.1-2.

The blocking effect of the sills and banks at the openings of the sound tends to confine relatively saline (and therefore dense) basin water to the interior of the basin. Inflows of shelf water are typically less dense compared to the resident high salinity waters in the sound. Hence, during this regime, because vertical mixing is relatively weak, horizontal transport between Cockburn Sound and the adjacent coastal waters occurs largely in the upper half of the water column (0-10 m). A typical vertical salinity section through Cockburn Sound and adjacent shelf waters from 3 May 1994 (D'Adamo and Mills, 1995c) highlights this trait (Figure 5.1-13). During autumn strong south-southwesterly winds, characteristic of summer, are less frequent and night-time heat loss is less due to smaller air-water temperature differences. These conditions, acting in concert, inhibit vertical mixing. Diurnal mixing by penetrative convection was estimated to vary between 3-10 m during 'typical' conditions in autumn. For conditions when the night-time mixing by penetrative convection is preceded by southerly winds of greater than about 7-10 m s^{-1} for 5-8 hours or more, it is predicted that the water column would be fully mixed by dawn. Winds greater than about 10 m s^{-1} for about five hours are predicted to be able to fully mix the water column by wind-mixing alone. Analysis of long-term wind records from Fremantle indicate the former conditions occur, on average, about five times per month and the latter about three times per month during April and May. It is concluded that during the 'autumn' regime the denser topographically constrained waters of Cockburn Sound, particularly the bottom waters, are less subjected to vertical mixing through the water

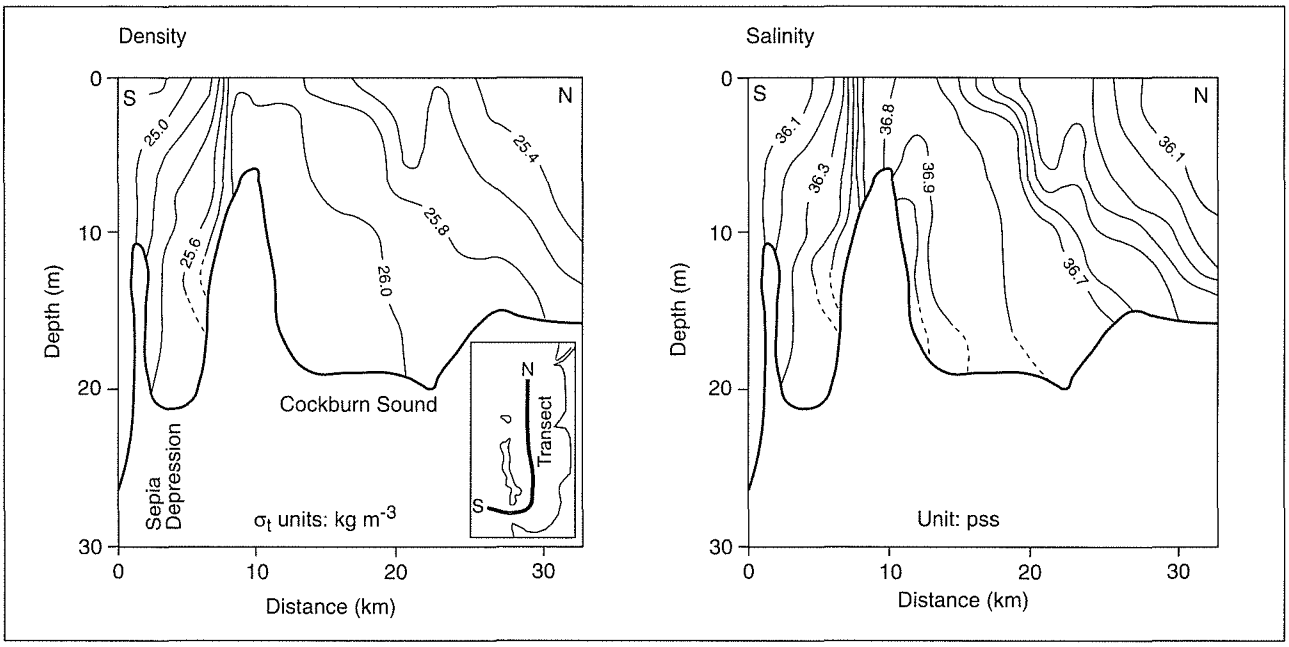


Figure 5.1-13. Vertical density and salinity structure in Cockburn Sound and adjacent waters in May 1994 showing 'typical' autumn stratification.

column than during 'summer'. Thus, exchange of the waters of Cockburn Sound in 'autumn' is potentially lower compared to other times of the year and, as a result, the mean residence time of Cockburn Sound water, particularly bottom waters, can be increased during autumn.

The significance of the vertical and horizontal density structure, characteristic of these waters in autumn, coupled with the relatively weak winds during this period, highlight the importance of considering density related effects in the hydrodynamic modelling of Cockburn Sound in autumn.

Warnbro Sound and adjacent waters

Similar results were returned for Warnbro Sound by the intensive field survey of 21-22 March 1994 (Kruh, 1994) in which the diurnal energetics of the vertical and horizontal structure within the basin was monitored over a typical 'autumn' diurnal period. As for Cockburn Sound, it appears that Warnbro Sound maintains a higher relative salinity than the adjacent waters in late summer to autumn, due to nearshore evaporation and restricted flushing. Below the level of the major sill opening (i.e. below about 5-10 m depth) the water is less frequently subjected to wind-mixing than surface waters due to the vertical stability of the density structure induced by both the presence of higher salinity waters at depth and also due to the day-time surface heating by solar radiation which creates a warm buoyant upper layer. In addition, the presence of relatively buoyant water outside of the sound means that exchange flows can introduce this water into the sound from Sepia Depression and the shelf, thereby complementing the vertically stratifying influences of solar radiation and evaporation within the basin. The results of the field survey suggest that Warnbro Sound's exchange during 'autumn' is significantly influenced by the vertical density structure, the presence of the physical obstruction to exchange by the reef line between its northern and southern points (Mersey and Becher points, respectively), and on-shore wind-stress which acts primarily to drive shelf water into the sound as a surface flow. As for Cockburn Sound, it is predicted that if winds blow at sufficient strengths during the day (of order 10 m s^{-1} for sustained periods of 5-10 hours or more) then it can be expected that full-depth vertical mixing will occur leading to more efficient exchange of the sound's water mass.

Conclusions

In the 'winter-spring' regime estuarine discharges are driven into the nearshore basins by winds, and hence substances derived from the estuaries are transported to various parts of the receiving marine waters. Exchange between the relatively buoyant waters in the basins (such as Cockburn Sound) and the adjacent oceanic waters appears to be more rapid than previously thought due to the tendency for dense inflows to subside to the bottom of the basins, thereby mixing with and displacing resident basin water upwards into the surface mixed layer, where wind stress then drives flow outwards through the openings as wind-driven advection. However, in the 'autumn' regime, the basin water is relatively dense due to evaporation and relatively less dense waters enter the basin as buoyant surface flows, further strengthening the vertical stratification. Hence, vertical mixing, particularly of deep basin waters, is restricted, with near-bottom waters less frequently mixed to the surface during 'autumn' compared to other times of the year. During the 'summer' regime the basins are relatively well-mixed vertically, density gradients are weak horizontally, and mixing and exchange is dominated by wind-driven processes.

The field surveys have shown that a significant degree of hydrodynamic communication exists within the nearshore zone between Fremantle and Mandurah. Buoyant plumes in 'winter-spring' emanate from the estuary mouths and are driven both southwards and northwards, into the nearshore basins, under wind forcing during repeated 7-10 day synoptic cycles. Time scales of the order of 1-2 days were associated with the transport of buoyant plumes in winter from the Peel-Harvey and Swan-Canning estuaries to the basins in the core of the study region (i.e. Cockburn Sound, Sepia Depression, Warnbro Sound and Comet Bay). In 'summer', due to prevailing south-southwesterly winds, the coastal flow is predominantly northward in open shelf or shallow waters, with typical speeds of about 10 km per day. Localised circulation patterns, governed mainly by wind direction and bathymetric distribution, occur within the semi-enclosed embayments and lagoons. In 'autumn', because the waters within the nearshore basins are relatively dense compared to oceanic waters, exchange flows occur predominantly via the upper half of the water column. The resulting vertical salinity stratification, coupled with thermal heating, stably stratifies the water column in density. As a result of the prevalence of weak winds in autumn, vertical mixing to the bottom via surface processes is less frequent compared to other periods of the year. Deep basin waters are therefore believed to reside near the bottom for longer times than during other periods of the year.

The hydrodynamic communication that was identified between the semi-enclosed embayments of the study area, and between the estuarine outflows and the central 'core' regions between Fremantle and Mandurah, suggests that the management of waste inputs to these waters should consider the general issue of inter-connectedness and, in particular, the transport of contaminants in estuarine plumes through the semi-enclosed embayments of the study region.

The findings of this section indicate that, for most of the year, a baroclinic model is required to simulate the key hydrodynamic characteristics of the semi-enclosed waters of this coastal region, particularly during the 'winter-spring' and 'autumn' regimes. The characteristic shelf versus basin density difference is a key factor determining the nature of circulation and exchange within the nearshore zone of semi-enclosed embayments. Wind stress is the principal transport and mixing agent, and penetrative convection also plays an important role in vertical mixing. In the 'summer' regime the basins and adjacent waters display only weak to negligible stratification (in the vertical and horizontal) due to the absence of significant river discharges and due to the occurrence of strong sea-breeze winds, and hence barotropic models can be employed.

5.2 Photosynthesis-irradiance relationships of seagrass

Plants are commonly termed 'primary producers' because they synthesise their own 'food' (carbohydrates) from the fundamental building blocks of carbon dioxide and water through the process of photosynthesis. Photosynthesis requires 'sun-light' as an energy source, so plants can only survive where the intensity and duration of light reaching their leaves are sufficient to ensure that photosynthetic production exceeds respiratory losses. Seagrasses are important primary producers in the coastal waters off Perth and generally occur in relatively shallow, depositional environments where water circulation and flushing are restricted. Wastes discharged to these protected areas tend to be retained for longer periods than in better-flushed environments, and hence the biota that inhabit these areas are exposed to these contaminants for longer periods than in the more energetic areas. Elevated nutrient loadings to these waters are of particular concern as they provide conditions that favour the formation of algal blooms which, in turn, reduce the amount of light available to benthic plants for photosynthesis (see section 5.3). This combination of factors make seagrasses particularly vulnerable to the effects of nutrient enrichment of coastal waters.

The scenario outlined above occurred in Cockburn Sound in the 1960s and 1970s and lead to the widespread loss of seagrass meadows, particularly on the eastern margin. Cambridge *et al.* (1986) concluded that shading by higher than normal standing crops of phytoplankton and epiphytes was responsible for the seagrass losses and was a direct consequence of excessive nutrient inputs.

The relationships between seagrass photosynthesis and light duration/intensity can be described mathematically and used to construct computer-based models to simulate seagrass growth under different conditions. These models can be used as environmental management tools to assist in determining critical light thresholds and for deriving nutrient-related environmental quality criteria. These criteria can be used by managers to help prevent a repetition of the nutrient-related impacts on seagrass communities that occurred in Cockburn Sound, or the Albany Harbours, for example.

To construct a realistic growth model of a particular seagrass species based on photosynthesis:irradiance relationships, other factors such as the effects of temperature, season and water depth on photosynthetic responses must also be understood and incorporated. If the model is to be generally applicable (i.e. non site-specific), any variability in response to these factors would also need to be determined for populations at the extremes of the geographic range of the species and incorporated if necessary (see section 6.3).

The principal aim of this study, undertaken in collaboration with the CSIRO, was to determine the effects of temperature

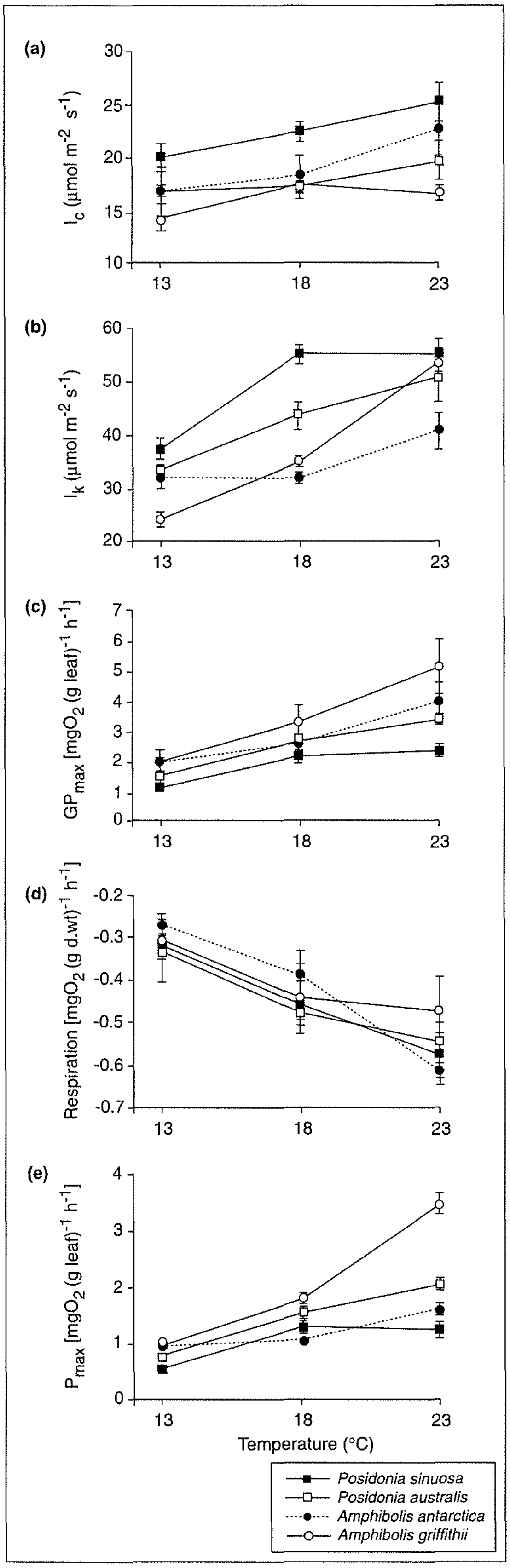


Figure 5.2-1. The effects of temperature on aspects of the photosynthesis and metabolism of the seagrasses Posidonia sinuosa, Posidonia australis, Amphibolis antarctica and Amphibolis griffithii: (a) compensating light intensity, I_c (b) onset of light saturation, I_k (c) maximum gross photosynthetic rate, GP_{max} (d) respiration rate, and (e) maximum net photosynthetic rate, P_{max}.

on the irradiance requirements and metabolic performance of *Posidonia sinuosa, Posidonia australis, Amphibolis griffithii* and *Amphibolis antarctica*, the dominant meadow-forming seagrass species in southwestern and southern coastal waters of Western Australia. In addition, preliminary experiments were conducted to compare the seasonal, depth and latitudinal effects on irradiance requirements and metabolic performance of the most prevalent meadow-forming seagrass in the coastal water of Perth, *Posidonia sinuosa*.

Further technical details can be found in Masini *et al.* (1990), Simpson *et al.* (1993), Masini *et al.* (1995b) and Masini and Manning (1995a, b and in press).

Findings

Maximum gross photosynthetic rates (GP_{max}), net photosynthetic rates (P_{max}), respiration rates and the amount of light required to allow photosynthetic oxygen production to balance respiratory consumption (I_c), and to saturate photosynthesis (I_k), generally increased over the temperature range 13 °C to 23 °C (Figure 5.2-1). Temperature optima for photosynthesis were at 23 °C or above for *P. australis*, *A. griffithii* and *A. antarctica*, and between 18 and 23 °C for *P. sinuosa*. The lowest metabolic rates and light requirements tended to occur at 13 °C. In general, the genus *Amphibolis* had higher photosynthetic rates, and these were reached at lower light intensities, than the genus *Posidonia*. Maximum achievable growth rates derived from the laboratory-derived photosynthetic data for these genera were similar to growth rates measured in the field (Masini and Manning, 1995b).

The photosynthetic response of *P. sinuosa* at a range of water temperatures and irradiances was not different for plants collected during summer or winter, indicating that the seasonal growth pattern of *P. sinuosa* is related directly to prevailing environmental conditions, such as water temperature, light intensity and daylength. The results of laboratory experiments suggest that biomass-mediated controls on photosynthetic rates occur in the field. To balance photosynthesis and respiration, a higher irradiance (measured at the top of the canopy) is required by plants in dense meadows compared to thin meadows, due to self-shading effects. However, higher photosynthetic rates can be achieved in dense meadows, as the inhibitory effects of very high light intensities on photosynthesis are reduced (Masini *et al.* 1995b). The critical light intensities and photosynthetic rates of *P. sinuosa* in summer were consistently higher at the southern end than at the midpoint of this species' geographic range in Western Australia (Masini and Manning, 1995b).

Seagrass plants that had been growing for a prolonged period under low-light conditions showed no measurable signs of 'adaptation' to those conditions (Masini *et al.* 1995b). However, *P. sinuosa* plants collected from near their natural maximum depth (=light) limits in Perth's coastal waters displayed morphological and physiological evidence of adaptation to the conditions found at these depths; the leaves were shorter, narrower and thinner, and the plants had a higher photosynthetic efficiency, and lower light requirements to saturate photosynthesis than the shallow water plants. These data indicate that *P. sinuosa* plants collected from the bottom of natural depth gradients have different photosynthetic responses compared to plants growing in shallow water and that these differences do not reflect 'rapid' responses to low light (phenotypic adaptations); rather, they are genetically determined (Masini and Manning, 1995b). These data suggest that there are genetically-distinct populations of *P. sinuosa* plants in Perth's coastal waters. This suggestion is supported by the findings of Waycott (1995) who identified at least 15 genotypes of a closely related species, *Posidonia australis*, within an area of 320 m^2 indicating a high level of genetic variation in this species in the Perth region.

Conclusions

Comparisons between *in-situ* and metabolically derived growth rates suggest that simulation models based on laboratory measurements of metabolism can be used to predict the growth of seagrasses in the natural environment. Temperature- and biomass-dependence are required to accurately simulate the growth of *P. sinuosa*. There is no seasonal rhythm in the photosynthetic response to light and water temperature in *P. sinuosa*. Photoadaptation in *P. sinuosa* does not occur as a short-term (e.g. months) response to imposed low light, rather the timescales for the apparent 'photoadaptation' observed in naturally occurring 'deep-water' plants are longer and may reflect selection processes operating during the development of these meadows over the last few thousand years (Davies, 1970; Semeniuk and Searle, 1986). On this basis it would appear that the measured differences in photosynthetic response of shallow- and deep-water plants are genetically determined. The potential for genetic variability in *Posidonia* is reflected in the high level of speciation that has occurred in this genus in southern Australia (see section 2.4).

5.3 Light attenuation

The survival of plants is related to the amount of light reaching their leaves and growth occurs when light levels ensure photosynthetic production exceeds respiratory losses (Masini *et al.* 1995b). Light availability to seagrasses is determined by biological and physical processes attenuating light through the water column and at the leaf surface (Figure 5.3-1). Nutrient enrichment of coastal waters can lead to excessive growth of phytoplankton in the water column and/or epiphytic algae on the leaf surface, with the result that less light reaches benthic plant communities (Dennison *et al.* 1993). Furthermore, in relatively shallow coastal waters, the resuspension of sediments into the water column by waves and the discharge of suspended particulates and/or dissolved organics (e.g. tannins) from estuaries are additional processes which influence the attenuation of light through the water column (Kirk, 1994). Thus nutrient enrichment and increases in suspended material, separately or in concert, can lead to a decline in benthic plant communities due to light starvation. An understanding of the biological and physical processes that influence light reaching benthic plant communities is, therefore, of critical importance to their conservation.

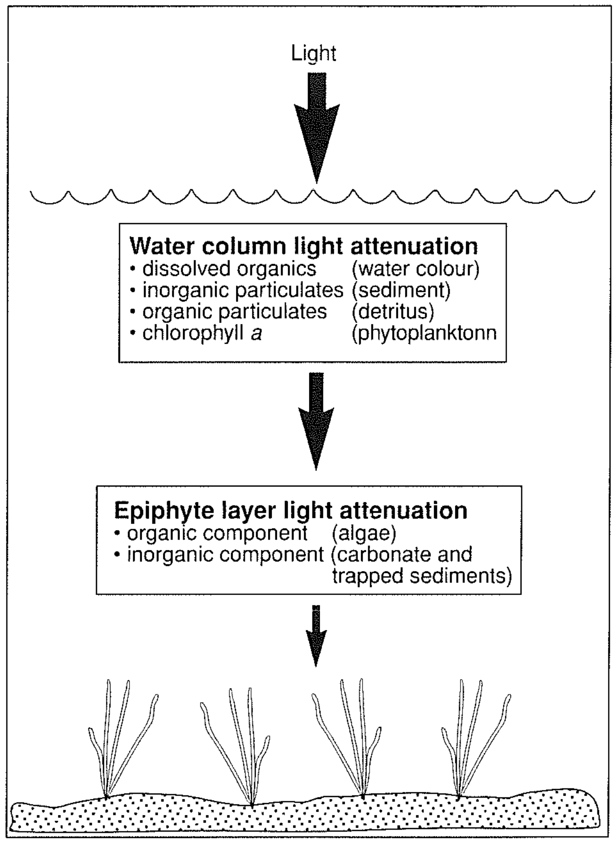


Figure 5.3-1. Schematic of physical and biological processes that influence the amount of light reaching the leaves of seagrasses.

5.3.1 Light attenuation through the water column

Water column light attenuation is influenced by the optical properties of the water itself, the concentration and species composition of phytoplankton, the amount of inorganic and organic particulate material suspended in the water and the concentration of dissolved organic molecules, all of which reflect, refract, absorb and scatter incident radiation (Kirk, 1994). The relative contribution of these factors to the attenuation of light through the water column is determined by physical and biological processes and, therefore, can vary greatly for different water bodies, depending on their physical setting and trophic status. Coastal waters are generally shallow, biologically productive and often influenced by rivers and waves (i.e. water colour and suspended

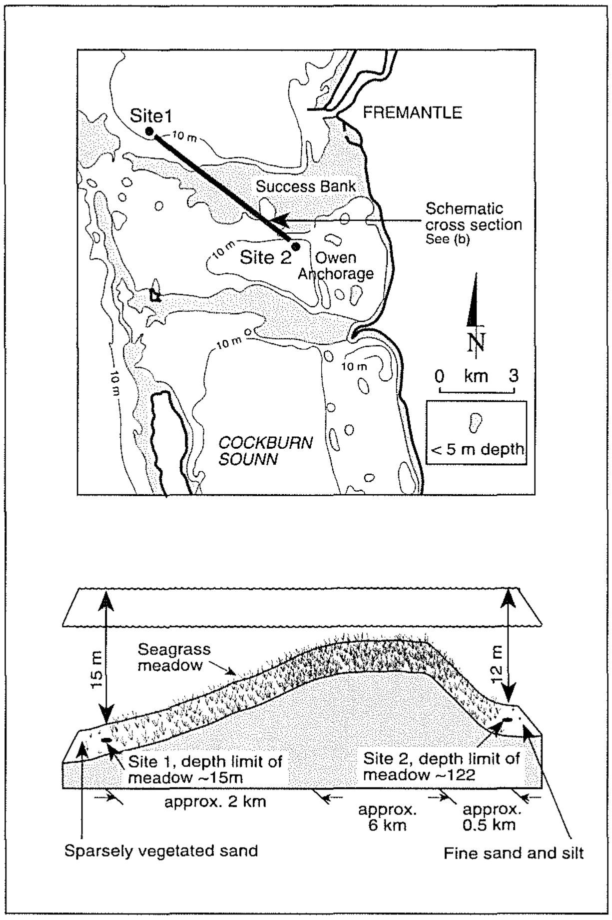


Figure 5.3-2. Location map and schematic of Success Bank showing seagrass distribution, water depth, sediment type and site locations for light attenuation and seagrass minimum light requirement studies.

particulates), so that the relative importance of the factors attenuating light is likely to change considerably with both space and time (season), due to changes in key physical (e.g. wave climate) and biological (e.g. algal production) processes.

The major factors contributing to the attenuation of light through the water column, and the physical and biological processes influencing these factors, were studied at two sites considered to be broadly representative of the offshore *semi-exposed* zone and inshore *protected* basin zone in the southern coastal waters of Perth. The locations of the sites, their water depths and proximities to major bathymetric features are illustrated in Figure 5.3-2. Day-time water column light attenuation was calculated at 20-minute intervals for periods of about six weeks during summer, autumn, winter and spring, between March 1992 and June 1993. Seasonal and site differences in the relationships of light attenuation with wind and swell were examined using contemporaneous data. Instantaneous light profiles and measurements of chlorophyll *a* and suspended particulates were collected and analysed routinely to examine and quantify the relative contribution of these factors to water column light attenuation, for input to the ecological model (see section 6.4).

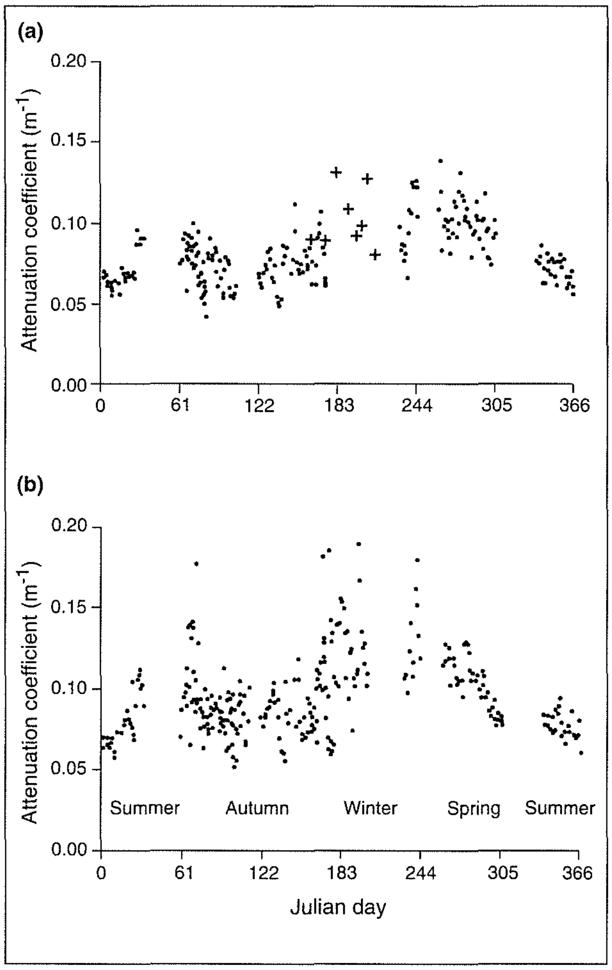


Figure 5.3-3. Composite annual cycle of mean daytime vertical light attenuation coefficients at (a) site 1 and (b) site 2. Crosses represent instantaneous light attenuation values.

Further technical details can be found in Burt *et al.* (1993a) and Burt *et al.* (1995b).

Findings

Water column light attenuation varied seasonally at both sites and was highest and most variable in winter, and lowest and least variable during early summer and autumn (Figure 5.3-3). A secondary maximum occurred in late summer at both sites. Light attenuation was generally higher at the inshore site with mean annual values of 0.094 m^{-1}, compared to 0.079 m^{-1} at the offshore site. Mean day-time light attenuation ranged between approximately 0.05 to 0.19 m^{-1} and 0.04 to 0.14 m^{-1} at the inshore and offshore sites, respectively.

Swell height was significantly correlated with light attenuation at the offshore and inshore sites during each of the seasonal study periods; the correlation was strongest during winter (Figure 5.3-4). Wind speed (generating wind waves) was correlated with light attenuation at the two sites during winter, however this was probably due to a strong cross correlation between wind speed and swell height. By contrast, during summer, a relatively low energy period with

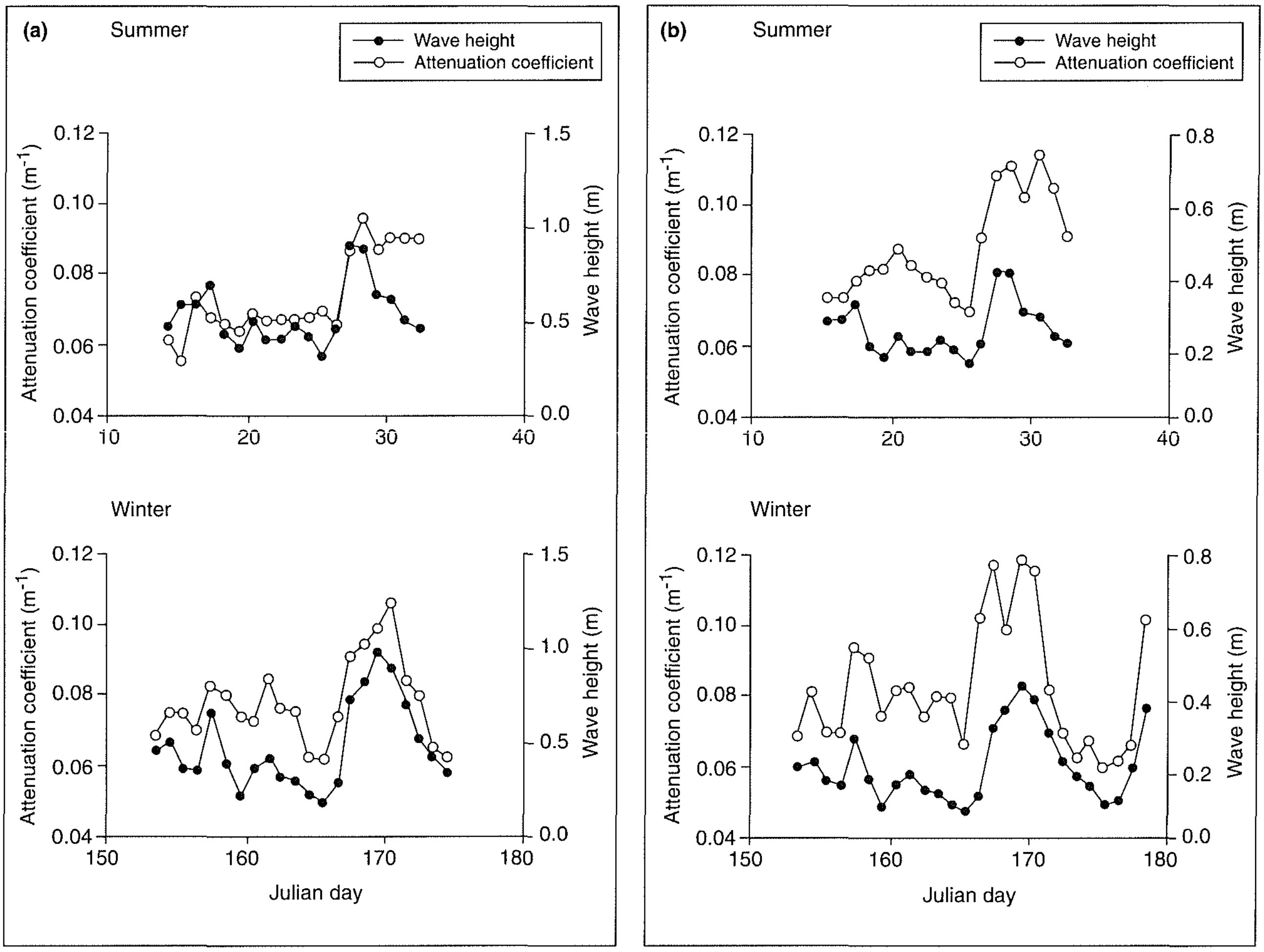


Figure 5.3-4. Representative timeseries of mean significant wave height and attenuation coefficient at (a) site 1 and (b) site 2 during summer and winter.

regard to swell, there was a significant correlation between wind speed and light attenuation at the inshore site only. These data suggest that the resuspension of sediments by long period swell waves is an important process influencing light attenuation in Perth's coastal waters, particularly during winter, and that locally generated wind waves during summer also have an important influence on light attenuation in the shallower inshore waters.

The differences in the sediment characteristics between the offshore and inshore waters appear to contribute to the differences in water column light attenuation. The offshore sediments consist mostly of coarse carbonate sands whereas the inshore sediments are finer sands and silt (Burt *et al.* 1995d; Figure 4.5-1). The organic fractions in both zones are similar (Figure 4.5-2). Wind wave heights are generally similar in both zones however swell height more than halves between offshore and inshore waters (Burt *et al.* 1995b). The composition of the material suspended in the water column, and caught in the sediment traps, supports the conclusion that the finer sediments inshore are resuspended by both swell and wind waves, in contrast to the offshore sediments, which are generally too coarse to be resuspended by wind waves alone.

Comparisons between the rates of deposition in sediment traps in the upper and lower layers of the water column can be used to differentiate between remote and local sources of suspended material, providing a further understanding of the processes influencing water column light attenuation. The deposition rates in the top and bottom sediment traps were correlated at the offshore and inshore sites during winter and the offshore site during the non-winter period, that is, when the respective sites were generally down-wind of Success Bank. This suggests that material resuspended in the relatively shallow waters on the bank by swell and wind waves is then transported toward and across the sites by wind-driven currents. The absence of a correlation between the top and bottom sediment traps at the sites during other times of the year suggests that sediment deposition is the result of local resuspension.

Sediment resuspension by waves is a function of wave energy, sediment grain size and density (de Jonge and van Beusekom, 1992). The relationship between swell height and sediment resuspension (measured as the deposition rate using sediment traps) is non-linear, with a distinct swell height threshold required to initiate sediment resuspension after which sediment resuspension increases rapidly (Figure 5.3-5).

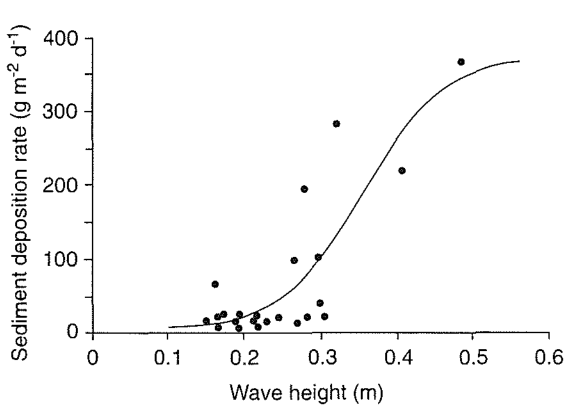


Figure 5.3-5. Relationship between mean sediment deposition rate and mean significant wave height in the inshore zone.

This relationship indicates that a relatively small increase in swell height across this critical threshold can cause a relatively large increase in sediment resuspension and, consequently, water column light attenuation. Periods of relatively high wave energy, when mean sediment deposition rates frequently exceeded the resuspension threshold, occurred throughout most of the year (May-November) at the offshore site but were confined to the winter period (June-August) at the inshore site. The terms 'wave' and 'non-wave' periods are used here to differentiate between these periods and the rest of the year.

The relationships between suspended particulates, including phytoplankton (measured as chlorophyll *a*) and instantaneous light profiles were examined to determine the relative contribution of these factors to water column light attenuation in the study area. During the non-wave period the biomass of phytoplankton was the primary factor influencing light attenuation at the two sites. By contrast, during the wave period, light attenuation in the offshore zone was primarily related to the concentration of phytoplankton and inorganic particulates (carbonate and silica sands), while, in the less exposed inshore zone, the concentration of organic particulates was the key factor. These results support the view that resuspension of sediments by swell waves is an important process influencing water column light attenuation in the inshore and offshore zones of Perth's coastal waters, especially during periods of moderate to heavy swell and the biomass of phytoplankton is an additional factor contributing to light attenuation in both zones during periods of low swell.

On occasions during the study period, especially in winter and early spring, high light attenuation values were occasionally recorded at the offshore site in the upper layer of the water column, coinciding with observations of the Swan River plume over the site. The Swan River flows strongest each year from approximately June to September, producing a characteristic plume of buoyant (low salinity) water (D'Adamo *et al.* 1995b). The location and extent of the plume is variable, determined largely by prevailing wind conditions, and the plume is sometimes visible (by water colour), as illustrated by satellite imagery (Plate 5.1-1c and e). The offshore site is approximately 6 km from the mouth of the Swan River, and data from this site supports the conclusion that plumes of stained estuarine water can influence water column light attenuation on occasions in the local receiving waters.

Conclusions

These findings indicate that anthropogenic activities that modify the wave climate and/or lead to increases in phytoplankton standing crop have the potential, individually or together, to significantly alter the light reaching benthic plant communities. In the study area, prolonged increases in water column light attenuation will ultimately change the vertical distribution and biomass of the underlying benthic plant communities. The extent of this effect will depend on the magnitude of the change in light attenuation and the depth distribution of these communities.

5.3.2 Light attenuation through the epiphyte layer

Changes in species composition and biomass of algae growing on the leaves of seagrasses (epiphytes) are typical responses to the combined or separate effects of changing nutrient status and wave energy of waters over seagrass meadows (May *et al.* 1978; Cambridge, 1979; Harlin and Thorne-Miller, 1981; Cambridge *et al.* 1986; Hillman *et al.* 1991). Epiphytic algae reduce light reaching the surface of the seagrass leaf, and the degree of light reduction depends on the biomass and species composition of the epiphyte assemblage (Silberstein *et al.* 1986). Therefore, the relationship between epiphyte assemblage and light reduction through this layer is important in understanding the effects of nutrient enrichment on seagrass communities.

Algae growing on artificial seagrass leaves (strips of transparent plastic) were used to examine this relationship between epiphyte biomass and light reduction. Seasonal and site influences on the relationship were examined over an annual cycle at two sites considered to be broadly representative of a relatively high energy 'oligotrophic' zone (offshore) and a relatively low energy 'eutrophic' zone (inshore). These relationships were quantified for input into the ecological model.

Further technical details can be found in Burt (1994) and Burt *et al.* (1995a).

Findings

The composition of the epiphyte assemblage at the inshore site was predominantly filamentous red and brown algae during summer and winter with similar species richness and composition. In contrast, at the offshore site, the epiphyte assemblage changed from predominantly filamentous species with relatively high species richness in summer to predominantly coralline species with relatively high species richness during winter. The biomass of the epiphyte assemblages at the two sites was similar for most of the year (non-winter period) but during winter there was a marked reduction in biomass at the inshore site and an increase at the offshore site (Figure 5.3-6).

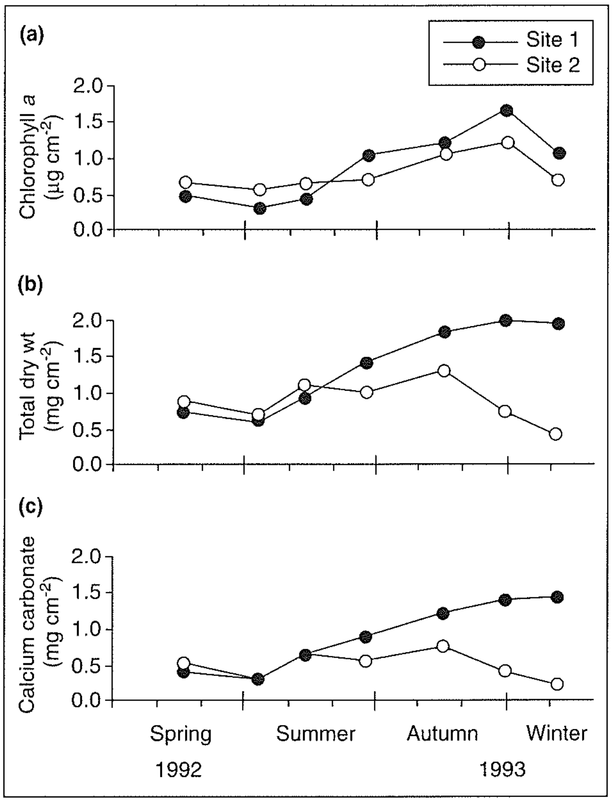


Figure 5.3-6. Seasonal variation in epiphyte biomass at sites 1 and 2 expressed as (a) chlorophyll a, (b) total dry weight and (c) calcium carbonate per unit area of seagrass leaf.

Epiphytic assemblages dominated by filamentous red and brown algae are usually characteristic of eutrophic waters (Cambridge *et al.* 1986; Hillman *et al.* 1991). The mean concentration of total inorganic nitrogen was generally higher throughout the year in inshore than in offshore waters, particularly during winter. Furthermore, the concentrations of chlorophyll *a* (an index of nutrient enrichment), were usually higher inshore than in offshore waters. Together, these findings suggest that the composition of the inshore epiphyte assemblage was primarily nutrient induced.

In offshore waters there were large seasonal changes in the species composition of the epiphyte assemblage with a shift from non-coralline red algae during summer to coralline red algae during winter. Previous studies have reported differential erosion of the various epiphyte forms during periods of high wave energy, with higher erosion rates for fine filamentous and erect foliose algae compared to 'low profile' encrusting coralline algae (Cambridge, 1979). In high energy waters coralline algae can outgrow filamentous algae and become the dominant algal type (Kendrick, 1991). Swell height is significantly higher in the study area during winter and is approximately 50% higher at the offshore site (see section 5.3.1). This seasonal difference in wave height between inshore and offshore waters supports the conclusion that the seasonal changes in species composition of the offshore epiphyte assemblage were due to the loss, through erosion and abrasion, of 'loosely' attached non-coralline algae during winter storms providing the coralline species with a competitive advantage.

During summer, species diversity of the epiphyte assemblage and the relative occurrence of coralline algae were substantially higher offshore than in the inshore waters. Nutrient enrichment has been associated with changes in the species diversity and composition of epiphyte assemblages, with a higher diversity and higher proportion of coralline species in oligotrophic waters compared to eutrophic waters, where the assemblages are generally dominated by filamentous species (May *et al.* 1978; Harlin and Thorne-Miller, 1981, Hillman *et al.* 1991). On the basis that, in summer, wave heights in the two zones are similar, and the inshore zone is relatively eutrophic, the differences between the zones in the composition of the epiphyte assemblages are probably nutrient induced.

By comparison, during winter, species richness of the offshore epiphyte assemblage was lower than the inshore assemblage, again reflecting the relative exposure of these two zones to swell waves, with the loss of non-coralline species from the relatively exposed offshore waters.

Significant relationships between periphyton biomass and light attenuation through this layer were evident at both sites, with differences between sites and seasons reflecting changes in species assemblage, particularly the relative abundance of coralline algae. These results suggest that the different organic and carbonate contents of different species assemblages can result in epiphyte assemblages having different optical properties that can significantly change the relationship between periphyton biomass and light attenuation.

The relationship with periphyton biomass, expressed as total dry weight, provides the only generic relationship with no significant site or seasonal differences (Figure 5.3-7). The functional form and detail of this relationship is very similar to relationships determined by Silberstein *et al.* (1986) and

Manning (1994) from a wide range of environmental conditions. On this basis it would be reasonable, in the first instance, to apply this general relationship to epiphyte assemblages growing on *P. sinuosa* throughout the temperate coastal waters off Western Australia.

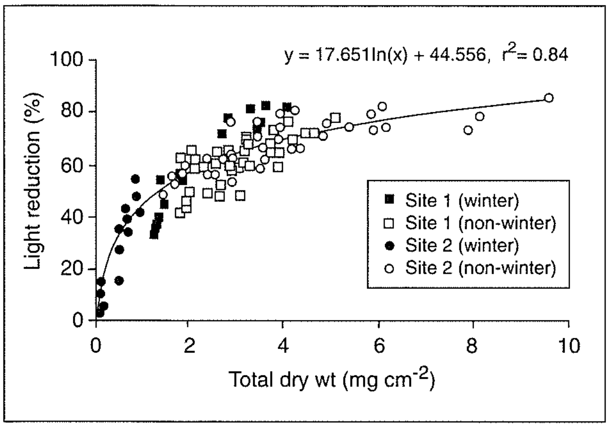


Figure 5.3-7. Relationship between periphyton biomass on the leaves of Posidonia sinuosa seagrass and light reduction through the periphyton layer.

The logarithmic nature of the light reduction relationship means that the relative effect of a given increase in biomass on percent light reduction is dependent on the initial biomass. At the inshore and offshore sites, the seasonal ranges in epiphyte biomass occur at relatively low biomass values where the slope of the curve is relatively steep, resulting in large reductions in percent light for relatively small increases in biomass. In this study periphyton biomass, expressed as total dry weight, ranged between 0.5 and 2.0 mg cm^{-2}. An increase of 1.0 mg cm^{-2} in epiphyte biomass at the lower end of the range (i.e. 0.5 to 1.5 mg cm^{-2}) would cause the percent light reduction through the epiphyte layer to change from 32 to 52%. An equivalent 1.0 mg cm^{-2} increase at the higher end of the range (i.e. 1.5 to 2.5 mg cm^{-2}) would result in light reduction changing from 52 to 61%, a much smaller relative change.

Conclusions

The relationship between epiphyte biomass and light attenuation indicates that a small increase in biomass can significantly reduce the light reaching the surface of the leaf. Changes in the species composition of epiphyte assemblages can also have a significant effect on the light reduction relationship. Therefore the amount of light reaching the surface of seagrass leaves is largely influenced by the combined effects of changing nutrient status and wave energy of waters over seagrass meadows. In the temperate coastal waters off southwestern Australia, seagrass meadows typically grow within semi-protected embayments and lagoons, in clear oligotrophic waters with relatively low standing crops of epiphytes. In these conditions, seagrasses growing at their depth limits are extremely vulnerable to factors which stimulate the growth of epiphytic macroalgae (e.g. nutrient enrichment) or reduce wave energy (e.g. coastal engineering structures).

5.4 Minimum light requirements of Posidonia sinuosa

Light availability is generally regarded as the dominant factor controlling seagrass depth distribution (Duarte, 1991). If a seagrass meadow is to survive, sufficient light is required to ensure that photosynthetic production is greater than, or equal to, the growth and respiratory requirements of the plant over an annual cycle (Masini *et al.* 1995b). At a given water depth, light availability at the epidermis of the seagrass leaf is dependant on the clarity of the water and the amount of epiphytic material coating the leaves. For a given water clarity and biomass of epiphytes, the amount of light available for seagrass photosynthesis and growth diminishes with depth. Therefore seagrasses will grow at greater depths in clear, oligotrophic waters and with low epiphyte biomass compared to turbid or eutrophic waters and with high epiphytic loads. Once *Posidonia* seagrass meadows are lost they do not re-establish in the order of decades and the most common indirect cause of seagrass loss is through light starvation resulting from shading by a nutrient-induced proliferation of phytoplankton and epiphytic algae (Clarke and Kirkman, 1989). Relationships between the vertical light attenuation coefficient and the concentration of phytoplankton and suspended organic and inorganic matter in the water column, and between epiphyte biomass and light reduction, have been derived in parallel studies described in Section 5.3. Information on the minimum light requirements of seagrasses could provide the basis of an 'early warning' system to detect potentially deleterious changes in aquatic ecosystem health and allow pre-emptive management action to be taken before serious or irreversible damage occurs.

The objective of the present study was to determine the minimum *in-situ* light requirements of *Posidonia sinuosa* seagrass meadows in Perth's coastal waters. The study involved measuring the light regime at the natural depth limit of two *P. sinuosa* meadows (12 m and 15 m) over an annual cycle which, after taking into account the shading of light by the epiphytes, allowed the light regime at the epidermis to be quantified over the annual cycle.

Further technical details can be found in Masini *et al.* (1995a).

Findings

The average amount of PAR (Photosynthetically Active Radiation) reaching the level of the seagrass canopy at a depth of 12 m at the inshore site in Owen Anchorage (see Figure 5.3-1) was 9.0% of the PAR measured immediately below the water surface (surface PAR) over the annual cycle and was highest in summer and lowest in spring. At the offshore site in Gage Roads (15 m depth) the meadow received 6.7% of surface PAR over the annual cycle with a maximum in summer and minimum in spring. The annual mean epiphytic biomass at the offshore site was higher than the inshore site and reduced light by an average of 49% compared with 42% at the inshore site. The main difference between sites occurred during winter when the epiphyte biomass reduced PAR by 56% at the offshore site compared to 35% at the inshore site which was dominated by filamentous and foliose red and brown algae at the time (section 5.3.2). When shading by epiphytes is taken into account, the PAR reaching the epidermis of the seagrass leaf is lower than that reaching the canopy and averaged 5.1% and 3.4% of surface PAR at the inshore and offshore sites respectively. The seasonal pattern of PAR reaching the epidermis was similar to that at the canopy.

Daylength and mid-day PAR intensity vary with season and at the latitude of Perth (32 °S) range from about 10 hours and 500 μmol m^{-2} s^{-1} during winter to 14 hours and 2000 μmol m^{-2} s^{-1} during summer. Global radiation for the 1992/93 study period was only about 2% higher than the 18-year average and, on that basis, can be considered representative of a 'typical' year. Seagrasses such as *Posidonia*, which have low photosynthetic production to respiration ratios, have morphological characteristics such as underground storage structures which allow a buffering of short-term energy deficits, but nonetheless they must achieve a positive energy balance over an annual cycle for long-term survival (Masini *et al.* 1995b). Consequently, these plants require substantial periods where PAR is sufficient to effectively saturate photosynthesis in order to produce surplus energy that is stored in the form of sugars and starch that can be utilised to meet the respiratory requirements of the plant during periods of low PAR. The number of hours of each day that PAR is sufficient to saturate photosynthesis varies with season, due to the seasonal differences in daylength and PAR intensity, but it is not directly related to these factors alone, as photosynthesis is temperature-dependent (see section 5.2) and less PAR is required at low temperatures than at high temperatures. Seasonal changes in epiphytic loading must also be taken into account.

At the inshore site, PAR was sufficient to allow the onset of light saturation (H_k) for 5.1 hours per day on average over the annual cycle and ranged from 9.3 hours during summer to 2.0 hours during winter. At the offshore site, the annual average was 3.5 hours per day and ranged from 0.4 hours during winter to 8.2 hours during summer. There was insufficient PAR on average to actually saturate photosynthesis (i.e. H_{sat}=0) during winter at either site. At the offshore site, photosynthesis was only saturated during summer (2.5 hours per day), and at the inshore site H_{sat} was $\leq$ 1 hour during spring and autumn and 4.8 hours during summer. The data for both sites were averaged (Table 5.4.-1) and, on the basis of the H_{sat} periods, show that photosynthetic production is highest during summer and lowest during winter. Simulation modelling of seagrass growth supports this view and indicates that the plants are in a positive carbon balance during summer and a negative carbon balance during winter (see section 6.3). These findings suggest that significant reductions in PAR availability during summer, such as by phytoplankton or epiphytic algal blooms, will upset the annual energy balance and jeopardise long-term survival.

Table 5.4-1. Mean minimum photosynthetically active radiation (PAR) expressed as a percentage of PAR immediately below the water surface, and number of hours per day that PAR was above the point of onset of PAR saturation of photosynthesis (H_k), and above the level required to actually saturate photosynthesis (H_{sat}), for Posidonia sinuosa at Success Bank.

Season	PAR at canopy	PAR at epidermis	H_k	H_{sat}
	(% of surface PAR)		(hours day $^{-1}$)	
Winter	6.2	3.5	1.2	0
Spring	5.2	3.2	4.0	0.5
Summer	11.4	6.2	8.8	3.7
Autumn	8.5	4.1	3.4	0.1
Annual mean	7.8	4.3	4.4	1.1

The minimum light requirements of *P. sinuosa* were 7.8% and 4.3% of surface PAR at the canopy and at the epidermis, respectively, based on an average of the two study sites (Table 5.4.-1). The sites were located at the local depth limits of meadow formation and the dry weight above-ground biomass of the meadows ranged from about 12 g m^{-2} during winter to 25 g m^{-2} during summer, which represents approximately 5% of the biomass of a typical meadow growing under optimal conditions. From these data, and giving consideration to the low biomass at the study sites, the minimum light requirement of *P. sinuosa* is $\geq$ 5% of surface irradiance measured at the epidermis, and therefore with moderate epiphytic loadings (1 mg dry weight cm^{-2} seagrass leaf; 45% PAR reduction) the minimum measured at the canopy would be $\geq$ 10% of surface irradiance. The minimum irradiance at the canopy derived in this study is reasonably consistent with the value of 16% of surface PAR calculated using a relationship generated from analysis of the depth distributions of 31 marine angiosperm species world-wide (Duarte, 1991).

With a moderate epiphytic load, the *P. sinuosa* depth limit can be approximated as $D = {}^{1}/_{AC}$ where D is the maximum depth limit in metres and AC is the light attenuation coefficient expressed on a $\log_{10}$ basis. With low epiphytic loads (0.4 mg cm^{-2}; 30% PAR reduction) the relationship becomes $D = {}^{1.105}/_{AC}$ and under high loads (2.5 mg cm^{-2}; 60% PAR reduction) it is $D = {}^{0.862}/_{AC}$. These relationships are presented graphically in Figure 5.4-1 and highlight the dramatic effect of small increases in attenuation coefficient on the depth of seagrass survival, especially in clear waters. These relationships, derived from *in-situ* measurements, are very similar to relationships derived from computer simulation modelling using laboratory-derived metabolic data (see section 6.3.5). The changes in seagrass depth distribution that have occurred in Cockburn Sound, and the present depth distributions in Owen Anchorage/Gage Roads (where the *in-situ* relationships were derived) and Warnbro Sound provide support for these relationships. The light attenuation coefficient in Cockburn Sound during the late 1970s was 0.13 m^{-1} (Cary *et al.* 1995a) and these conditions are thought to have existed when the seagrass banks on the 9 m sill on the eastern margin of Cockburn Sound were lost during the early 1970s (Cambridge and McComb, 1984). From Figure 5.4-1, a light attenuation coefficient of 0.13 m^{-1} would only allow *P. sinuosa* to survive to a depth of 7.7 m under the moderate epiphytic loading category. There is evidence to indicate that epiphytic loads were substantially higher than this and, at times, probably exceeded the high load category described here (Silberstein *et al.* 1986). On the basis of this relationship, seagrass with high epiphytic loads could have only survived to a maximum depth of 6.6 m. The current annual average light attenuation coefficient of Cockburn Sound is approximately 0.11 m^{-1} (section 4.7.2) which, applying the moderate epiphytic load category would only allow seagrass survival to a depth of 9.1 m. The light attenuation coefficient of Warnbro Sound was about 0.09 m^{-1} in the late 1970s and has remained relatively stable over the past 15-20 years according to available data (Cary *et al.* 1995a). With moderate epiphyte loads, this light attenuation coefficient would allow *P. sinuosa* to survive to a depth of 11.1 m, which is similar to its actual depth distribution in the Sound. The light attenuation coefficient of Warnbro Sound is thought to be similar to that of Cockburn Sound prior to industrialisation, and accordingly,

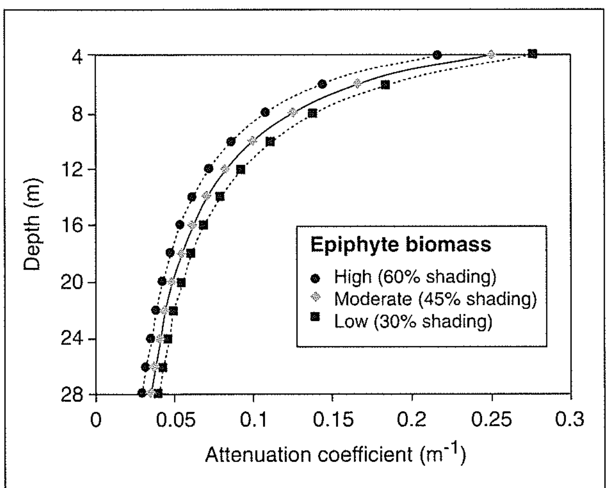


Figure 5.4-1. The inter-relationship between depth of seagrass survival, water column light attenuation and epiphyte shading.

P. sinuosa would have theoretically existed at a depth of about 11 m, which is consistent with historical records for Cockburn Sound (Department of Conservation and Environment, 1979).

In Princess Royal Harbour on the southwest coast of Western Australia, the maximum depth of *P. sinuosa* survival regressed to about 5 m when the light attenuation coefficient was about 0.16 m^{-1} and epiphyte loads were high as a result of eutrophication (Simpson and Masini, 1990); under these conditions the maximum depth of survival is predicted to be 5.4 m.

Posidonia species have been recorded at a depth of 27 m in the clear, offshore waters of Geographe Bay situated about 250 km south of Perth (Cambridge, 1980). According to the relationships in Figure 5.4-1, the light attenuation coefficient of these waters would need to be approximately 0.041 m^{-1} to support *P. sinuosa* with low epiphytic loads. The offshore waters in the mid-shelf region off Perth have a mean attenuation coefficient of 0.039 m^{-1} (Cary *et al.* 1995b) and are likely to be of similar clarity to offshore waters of Geographe Bay, providing further support for the relationship outlined previously.

These findings suggest that the minimum light requirements of *P. sinuosa* described by the relationships shown in Figure 5.4-1 are generally applicable across the depth and geographic range of this species.

Conclusions

Posidonia sinuosa meadows with moderate epiphyte loadings require an annual average of approximately 10% of PAR immediately below the water surface to reach the canopy level to survive in the long-term. This equates to approximately 5% at the epidermis of the leaf. The relationship between maximum depth of seagrass survival, epiphyte loading and water column vertical light attenuation coefficient is consistent with past and present seagrass depth distributions in south-west Western Australia and is an important interpretative tool for evaluating the results of water quality monitoring programmes. The results of this study highlight the effect of even small increases in light attenuation coefficient on the depth distribution of the ecologically important meadow-forming seagrasses such as *P. sinuosa*.

5.5 Phytoplankton-zooplankton interactions

A classic response of coastal ecosystems to nutrient enrichment is the stimulation of algal productivity. This occurred in Cockburn Sound during the 1970s when excessive nutrient inputs resulted in frequent and intense phytoplankton 'blooms' which, in turn, caused decreased light penetration through the water column, resulting in reduced productivity and contributing to the catastrophic decline of seagrass communities (Department of Conservation and Environment, 1979).

Phytoplankton abundance in the wider coastal waters off metropolitan Perth is naturally low and this is generally attributed to nutrient limitation (Chiffings, 1987). However, in the absence of evidence to the contrary, an alternative explanation for this low abundance may be high zooplankton grazing pressure. Conversely, the relatively high phytoplankton biomass in the nutrient-enriched waters of Cockburn Sound may be due to low zooplankton grazing pressure.

It is generally accepted that phytoplankton and zooplankton populations interact, and this interaction can be both positive and negative (e.g. Frost, 1980). For example, heavy grazing pressure can limit phytoplankton standing crops, or conversely, excretion of inorganic nutrients into the water column by zooplankton can stimulate phytoplankton growth. Some phytoplankton species or cell size classes are selectively grazed in preference to others, and this will advantage or disadvantage certain phytoplankton species, depending on whether they are preferred or not.

Phytoplankton are a major food source for zooplankton; therefore high standing crops and fast growth rates of phytoplankton can support higher zooplankton abundances by allowing heavier grazing pressure to be maintained, whereas low phytoplankton abundance and growth rates can only sustain low zooplankton abundances.

The primary productivity of Cockburn Sound is still phytoplankton dominated and changes in relative phytoplankton abundance, measured as chlorophyll *a* concentrations, remain the principal ecosystem response to nutrient inputs (Simpson *et al.* 1993). Thus, to understand the relationship between nutrient inputs and phytoplankton biomass, it is also necessary to understand the interaction between phytoplankton populations and zooplankton grazers.

Sections 4.8 and 4.9 described the results of two-year intensive studies of the phytoplankton and zooplankton populations in Cockburn and Warnbro sounds. In this section the two data sets are integrated to identify relationships between these two groups and to facilitate a better understanding of the trophodynamics of phytoplankton populations in these waters.

Further technical details can be found in Simpson *et al.* (1993); Helleren and John (1994); Helleren and John (1995).

Findings

The abundances of phytoplankton and zooplankton (Figure 5.5-1) in the two embayments were positively correlated when data were averaged over the study period (Figure 5.5-2). This correlation was strong ($r^2 = 0.672$, $P < 0.05$) suggesting that either one group is controlling the standing crop of the other or that common site specific factors are influencing both populations. The composition of the phytoplankton and zooplankton assemblages in the two embayments were similar suggesting that the types of trophic interactions between the zooplankton and phytoplankton would be common to both embayments.

Grazing potential

Of the three major groups of zooplankton found in Cockburn and Warnbro sounds (section 4.9), the copepods are generally regarded as the most significant grazers of phytoplankton (Ikeda, 1977; Anderson, 1994). Copepods feed by filtration and/or predation and it is generally thought that the adult copepods in the small size classes (e.g.< 0.75 mm) are predominantly filter feeders grazing on phytoplankton, whereas the larger copepods are predatory and may feed on other zooplankton. The copepods in the embayments are dominated (~80%; see section 4.9) by individuals in the small size range and are likely to be exerting a substantial grazing pressure on the phytoplankton assemblages.

The cladocerans were the second most abundant zooplankton group and were dominated by *Penilia avirostris*. This species feeds almost exclusively on phytoplankton, and specifically phytoplankton less than about 30 µm in diameter, which includes most phytoplankton species encountered during this study. *Penilia* is widely distributed in the world's temperate oceans where small phytoplankton species dominate and it can grow and reproduce even when phytoplankton concentrations are relatively low (Paffenhoffer and Orcutt, 1986). Cladocerans, in particular *Penilia avirostris*, were found to graze only a small proportion of the phytoplankton standing stock in Tolo Harbour, Hong Kong (Wong *et al.* 1992) but the relatively high abundances of this group compared with other zooplankton groups found in the present study suggests that cladocerans are likely to be significant grazers of phytoplankton in Perth's coastal waters.

Although the ecology of the third major group of zooplankton in these waters, the protozoans (radiolarians) is poorly described, the skeleton bearing species such as those found in the present study are known to be able to utilise their supporting skeletal structures to capture relatively large prey, including adult and larval copepods, trochophore larvae, diatoms and silicoflagellates (Anderson, 1993).

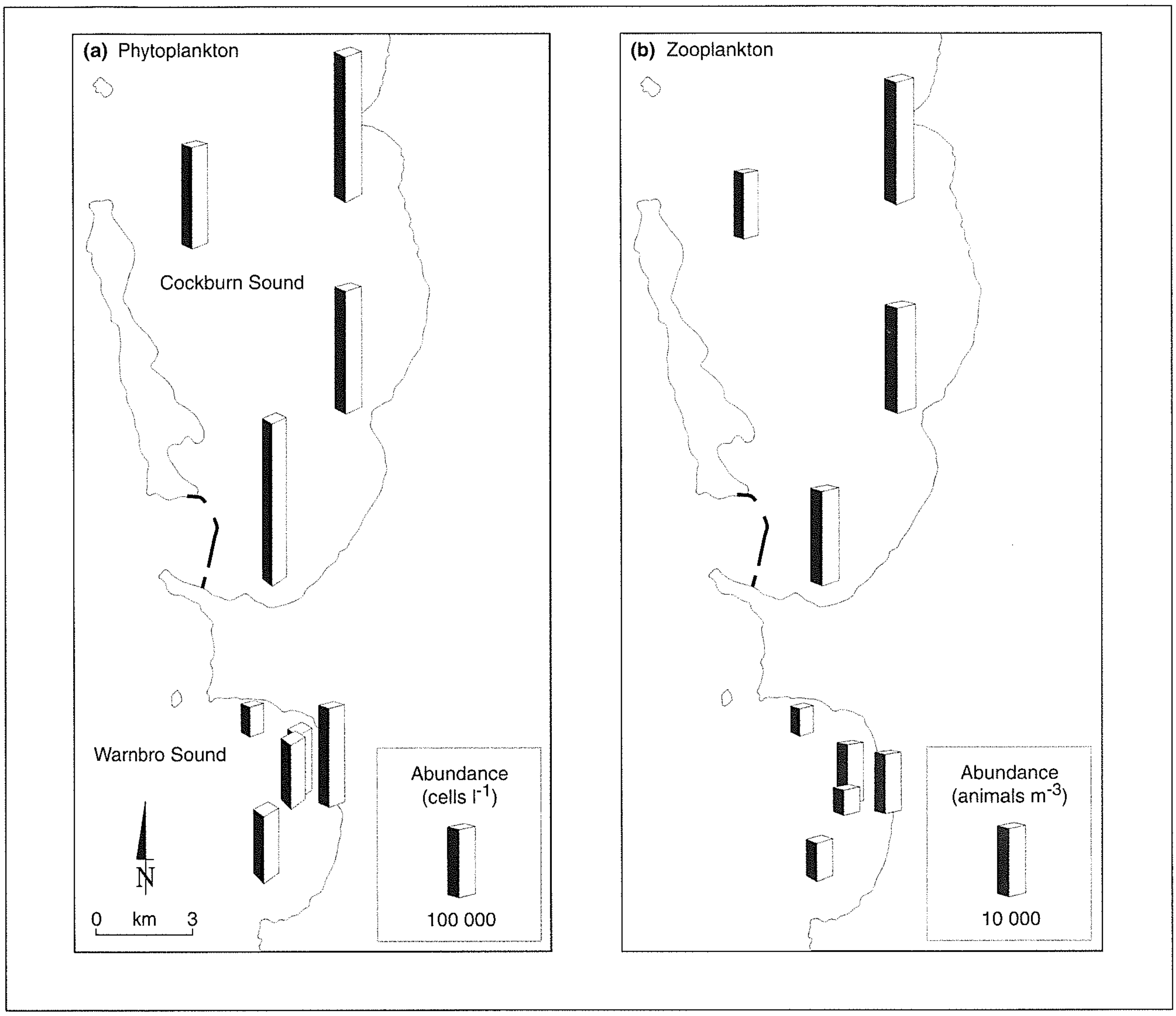


Figure 5.5-1. Abundance of (a) phytoplankton and (b) zooplankton in Cockburn Sound and Warnbro Sound. Values are vertically integrated averages over the period August 1992 to August 1994.

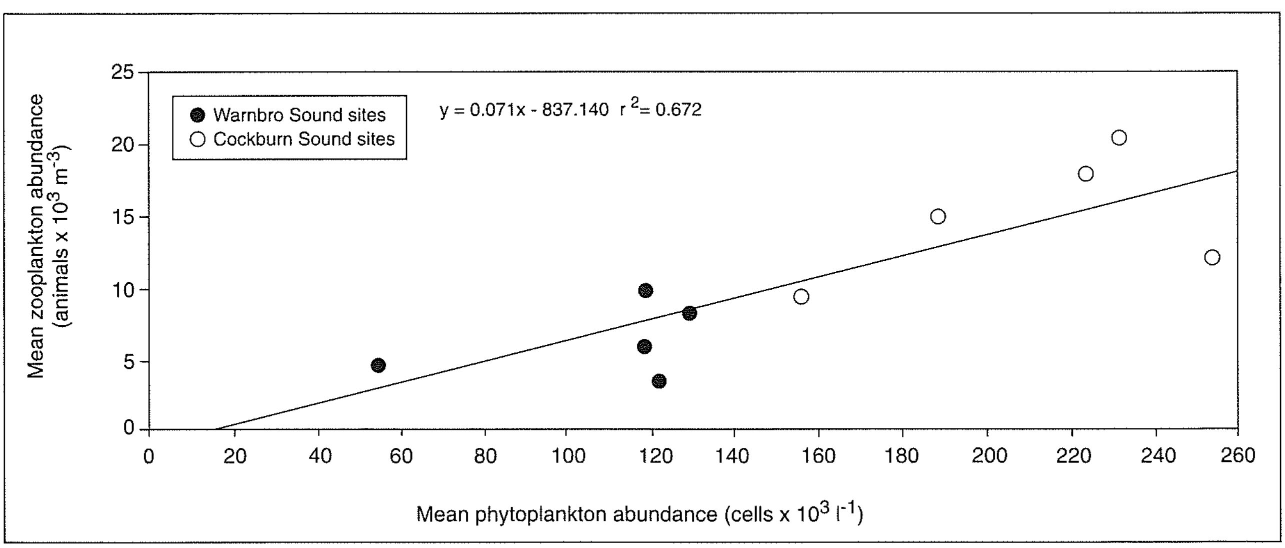


Figure 5.5-2. Relationship between the mean abundance of phytoplankton and zooplankton over the period August 1992 to August 1994 at sites within Cockburn Sound and Warnbro Sound.

On this basis, and given the relatively high abundances of radiolarians found during late winter, this group has the potential to exert a significant grazing pressure on the phytoplankton communities of the embayments.

Of the minor groups of zooplankton present, polychaete and mollusc larvae, tunicates and tintinnids also graze phytoplankton. However, the low abundances of these groups suggest that their impact on phytoplankton populations in the study area would be relatively low, compared to the three major groups.

Of the major phytoplankton assemblages identified during this study (see section 4.8) the diatoms were most abundant and would definitely be the most preferred as a food source for the planktonic grazing community. Dinoflagellates have high cellular concentrations of protein, lipids and carbohydrates (Nielsen, 1991) and are the second most preferred group, but were never found in high abundances. Cyanobacteria are generally considered to be the least favoured of the major phytoplankton groups present. Little is known of the relative preferences of the zooplankton grazers for silicoflagellates, however the abundant silicoflagellate population in both embayments during winter is potentially a large food source to the zooplankton able to access it.

Although the potential for interaction between various groups of zooplankton and phytoplankton has been identified in this and other similar studies conducted elsewhere, seasonal changes in abundances of each of these groups are not solely determined by predator-prey interactions. Other confounding influences such as predator-herbivore interactions are also very important and will tend to mask all but the strongest and most simplistic predator-prey interactions.

Phytoplankton-zooplankton inter-relationships

Visual assessment of the time series plots of phytoplankton and zooplankton abundances (Figures 4.8-3 and 4.9-3) show evidence of some general inter-relationships, primarily between the copepods/cladocerans and diatoms, and also between the radiolarians and silicoflagellates. Peaks in copepod abundance tended to occur during spring and summer, the time of the year when diatom blooms occur (see section 4.6.2). Cladocerans displayed a similar seasonal pattern of occurrence, suggesting that both copepods and cladocerans may be preferentially grazing on diatoms. This is supported by the observed increase in copepod and cladoceran abundances in Cockburn Sound during late winter 1994, which coincided with a rapid decline in a monospecific bloom of the diatom *Skeletonema costatum*; no corresponding diatom or associated crustacean blooms occurred in Warnbro Sound at that time. There were no apparent associations between the abundance of dinoflagellates and any of the major zooplankton groups, and the dinoflagellates remained consistently at low abundance throughout the study period. Given the high nutritional value of dinoflagellates, the possibility that the abundance remains low due to selective feeding by the grazing community rather than because of unfavourable environmental conditions, cannot be ruled out. Increases in radiolarian abundance occurred in late winter in both embayments and coincided with the decline in numbers of silicoflagellates which dominate the phytoplankton assemblage throughout the region during winter (see section 4.8). Radiolarian abundance peaked in early spring and declined to low numbers by the end of spring, a pattern that would be expected if the radiolarians were grazing the silicoflagellates. Additionally, the magnitude of the radiolarian peak during late winter was related to the magnitude of the preceding silicoflagellate bloom (Figure 5.5-3) which is also indicative of a significant relationship between the two groups (Figure 5.5-4).

Although the general relationships outlined above were apparent, simple statistical analyses of time series plots of spatially averaged and of temporally averaged phytoplankton and zooplankton data for Cockburn Sound and Warnbro Sound showed no consistent significant correlations between any of the phytoplankton and zooplankton groups or between chlorophyll *a* and zooplankton, even when the data were offset (by one, two and three months) to account for possible lags in zooplankton response times.

Trophic interactions

The phytoplankton and zooplankton abundance data were analysed to provide an insight into the overall productivity of the two embayments. As is the case with any abundance-based estimates of productivity, several assumptions were made and these are as follows:

- Phytoplankton standing crop (expressed as chlorophyll *a*) was the annual average for both sounds.
- Zooplankton standing crop was taken to be the annual average adult copepod abundance in each sound (expressed as animals l^{-1}) and was used to represent the grazing pressure of the entire zooplankton community.
- Zooplankton grazing rate = 0.10 μgchl *a* animal^{-1} day^{-1}.
- Settling/advective loss = 5% of phytoplankton biomass day^{-1}.
- Phytoplankton Carbon:Chlorophyll *a* ratio = 50:1.

The doubling time of phytoplankton populations can be estimated by dividing the phytoplankton standing crop by the combined daily losses due to grazing and settling/advection, if it is assumed that the phytoplankton standing crop remains relatively constant on a day-to-day basis. Calculated in this way the doubling times for the phytoplankton was about 2.3 days on average for each embayment (Table 5.5-1). Although the doubling times for the phytoplankton populations were the same in each embayment the standing crop of phytoplankton was about 2.5 times higher in Cockburn Sound than in Warnbro Sound and therefore the production per cubic metre was also about

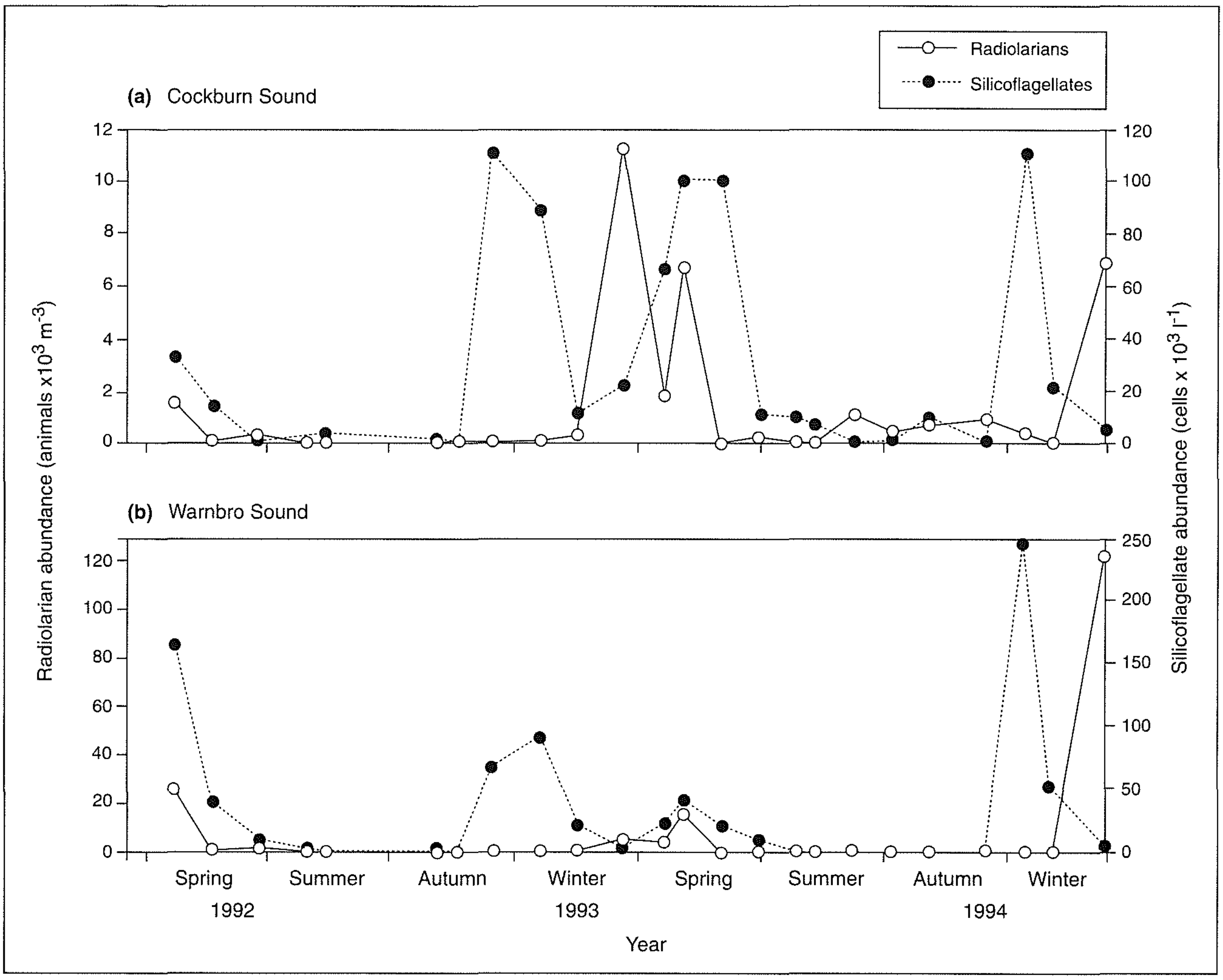


Figure 5.5-3. Seasonal changes in mean basin-scale silicoflagellate and radiolarian abundance in (a) Cockburn Sound and (b) Warnbro Sound.

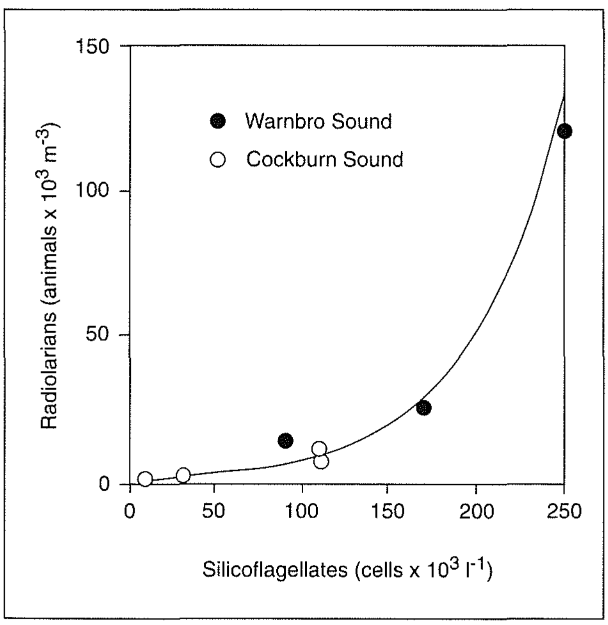


Figure 5.5-4. Relationship between peak basin-scale silicoflagellate abundance and subsequent peak radiolarian abundance in Cockburn Sound and Warnbro Sound.

2.5 times higher in Cockburn Sound than in Warnbro Sound. Cockburn Sound is approximately five times larger than Warnbro Sound by volume and therefore the primary production of Cockburn Sound attributable to phytoplankton (22 009 tC y^{-1}) is about 12.4 times greater than of Warnbro Sound (1 771 tC y^{-1}) (Table 5.5-2). Assuming an average depth of 15 m the pelagic primary productivity of Cockburn Sound is relatively low for nearshore coastal waters by world standards and that of Warnbro Sound is comparable with values for offshore oceanic waters (Table 5.5-3).

Nitrogen budgets

The productivity of phytoplankton, and most other primary producers for that matter, is principally determined by light availability, water temperature and macro-nutrient availability. Other factors, such as micro-nutrient availability, are considered to be secondary factors and generally do not over-ride the effects of the primary limiting factors mentioned above. Nitrogen is the macro-nutrient most limiting plant growth in these waters (section 4.7).

Table 5.5-1. Comparison of phytoplankton standing crop, zooplankton grazing pressure, and phytoplankton doubling times in Cockburn Sound and Warnbro Sound.

	Cockburn Sound	Warnbro Sound
Average depth (m)	15	15
Volume (l)	$1.5x10^{12}$	$0.3x10^{12}$
Chlorophyll *a* standing crop (µg Chl *a* l^{-1})	1.85	0.76
Settling/Advective loss (µg Chl *a* l^{-1} d^{-1})	0.093	0.038
Copepod standing crop (animals l^{-1})	7.1	2.8
Grazing loss (µg Chl *a* l^{-1} d^{-1})	0.710	0.285
Total loss (µg Chl *a* l^{-1} d^{-1})	0.803	0.323
Phytoplankton doubling time (days)	2.30	2.35

Apart from the size difference of the two embayments, they have similar bathymetry, sediment types, habitat types and have similar seasonal patterns and ranges in water temperature and global radiation. The major difference between them is the magnitude of the nutrient loads they receive from anthropogenic sources; Cockburn Sound is currently estimated to receive a daily load during summer of between 1 300 and 3 200 kg of nitrogen compared with Warnbro Sound which receives about 50 kgN d^{-1}. Given the

Table 5.5-2. Comparative productivity of phytoplankton in Cockburn Sound and Warnbro Sound.

Productivity	Cockburn Sound	Warnbro Sound
Daily rates		
µgChl *a* l^{-1} d^{-1}	0.80	0.32
mgC m^{-3} d^{-1}	40.20	16.20
mgC m^{-2} d^{-1}	603.00	243.00
Annual rates		
gC m^{-2} yr^{-1}	220.10	88.50
tonnes C yr^{-1}	22 009.00	1771.00

similarity in the physical and biological characteristics of the embayments it would appear that the differences in their respective nutrient loadings is the most likely cause of the different phytoplankton productions.

The carbon to nitrogen to phosphorus ratio of phytoplankton is generally considered to be about 106:16:1 on a molar basis (Redfield, 1958) but data collected by Bastyan and Hillman (unpublished; cited in Bastyan and Paling, 1995) suggest that the C:N:P ratio of the phytoplankton of Cockburn Sound is more like 106:10.4:1 and supports the general assumption that the growth of phytoplankton is nitrogen limited. The nitrogen requirement to meet the measured production in the two embayments can be calculated by applying these C:N ratios (Table 5.5-4). The zooplankton data can also be used to estimate a nitrogen requirement for each of the two embayments. The zooplankton of Cockburn Sound and Warnbro Sound are dominated by copepods (see section 4.9), and although many zooplankton species are omnivores and graze other zooplankton, for the purposes of these

Table 5.5-3. Comparison of phytoplankton standing crop and daily productivity in Cockburn and Warnbro sounds with other regions of the world. Annual means in parentheses. (indicates mean of summer months).*

Location	Standing crop (µgchl *a* l^{-1})	Productivity (gC m^{-2} d^{-1})	Source
Gulf of Carpentaria*	0.1 - 1.6	3.19	Jeffrey and Hallegraeff (1990)
Southern Indian Ocean	0.28	3.12	Jeffrey and Hallegraeff (1990)
Upwelling regions	-	0.5 - 3.15 (1.2)	Hatcher (1994)
Tasmanian waters	0.3 - 2.4	0.336 - 2.88	Jeffrey and Hallegraeff (1990)
Tasman Sea*	-	1.176	Jeffrey and Hallegraeff (1990)
Arabian Sea	-	0.3 - 1.0	Jochem *et al.* (1993)
Indian Ocean (110°E)	0.17	0.888	Jeffrey and Hallegraeff (1990)
Cockburn Sound	**1.85**	**0.603**	**This study**
Shelf seas & neritic waters	-	0.35 - 1.0 (0.6)	Hatcher (1994)
Coral Sea*	0.11 - 0.24	0.264 - 0.600	Jeffrey and Hallegraeff (1990)
Great Australian Bight*	0.1 - 0.4	0.437	Jeffrey and Hallegraeff (1990)
Warnbro Sound	**0.76**	**0.243**	**This study**
Coastal Australia (200 nm zone)	<0.05	<0.250	Hatcher (1994)
Open ocean	-	0.05 - 0.5 (0.2)	Hatcher (1994)
Temperate and southern oceans	1.5	-	Jeffrey and Hallegraeff (1990)
Tropical and sub-tropical oceans	0.05 - 0.5	-	Jeffrey and Hallegraeff (1990)

calculations the abundance of adult copepods was used as a measure of potential grazing pressure of the total zooplankton population. It was also assumed that the nitrogen content was 10-20 μgN animal^{-1} (Jorgenson *et al.* 1991), the population had a three week (21 day) generation time and therefore required 0.48 - 0.95 μgN animal^{-1} d^{-1}. The nitrogen requirements estimated from the zooplankton and from the phytoplankton data are very similar (Table 5.5-4) and suggest that the nitrogen requirements to maintain the current levels of planktonic production in Cockburn Sound and Warnbro Sound are approximately 3200 tN y^{-1} and 260 tN y^{-1} respectively or 8.75 tN d^{-1} and 0.70 tN d^{-1}. The estimated nitrogen requirement for maintaining the current level of primary production in Cockburn Sound is similar to the range 5.9 - 9.1 tN d^{-1} calculated from Bastyan and Paling (1995) after correcting an error in the C:N:P ratio used in their calculations.

Table 5.5-4. Annual nitrogen requirements to support phytoplankton and zooplankton production in Cockburn and Warnbro sounds.

	Cockburn Sound	Warnbro Sound
Phytoplankton		
chl *a* (μg l^{-1})	1.85	0.76
annual production (tonnes C yr^{-1})	22 010	1770
nitrogen requirement (tonnes N yr^{-1})	2522 - 3865	203 - 311
nitrogen requirement (tonnes N d^{-1})	6.91 - 10.59	0.56 - 0.85
Zooplankton		
copepods (animals l^{-1})	7.1	2.8
nitrogen requirement (tonnes N yr^{-1})	1846 - 3693	146 - 291
nitrogen requirement (tonnes N d^{-1})	5.06 - 10.12	0.40 - 0.80

These values are far in excess of anthropogenic loadings and indicate that nutrient transformations and recycling are important in maintaining planktonic primary productivity. Bastyan and Paling (1995) estimated that inorganic nitrogen released from the sediments in Cockburn Sound could account for between about 970-1950 tN y^{-1} (2.65 and 5.34 tN d^{-1}) and for Warnbro Sound the equivalent range was between about 68-110 tN y^{-1} (0.19 and 0.30 tN d^{-1}). The most likely source of this nitrogen is detrital rain from the water column (e.g. dead phytoplankton and zooplankton and zooplankton faecal pellets) which is subsequently released as dissolved inorganic nitrogen after it is remineralised through aerobic and anaerobic bacterial decomposition pathways. Zooplankton excrete nitrogen, and assuming a 70% assimilation efficiency derived from the review by Holloway and Jenkins (1993), this is estimated to be equivalent to approximately 43% of their daily nitrogen requirement. This excreted nitrogen is primarily in the form of ammonium (NH_4^+) which is the preferred form of nitrogen for phytoplankton uptake (e.g. Harrison *et al.* 1983) and growth, and therefore will be efficiently recycled.

Although the methods used to determine the nitrogen requirements of the plankton are simplistic, they correspond reasonably well to the estimates of internal and external loadings to these embayments (Table 5.5-5). The balance obtained between the estimated nitrogen loads and estimated nitrogen requirements suggests that the major pathways of nitrogen cycling for the planktonic component of the embayment ecosystems are described.

Table 5.5-5. Comparison between total nitrogen inputs from external and internal sources, and nitrogen required to maintain current levels of planktonic primary production in Cockburn and Warnbro sounds.

	Cockburn Sound	Warnbro Sound
INPUTS (kgN d^{-1})		
anthropogenic	1 300** - 3200*	50**
sediment efflux	2650 - 5340	190 - 300
zooplankton excretion	3250	260
TOTAL (kgN d^{-1})	7200 - 11 790	500-610
REQUIREMENTS (kgN d^{-1})	6910 - 10 590	560 - 850

* derived from loading relationship (Cary *et al.* 1995)
** derived from contaminant inputs inventory (Muriale and Cary, 1995)

Conclusions

Several zooplankton-phytoplankton interactions were apparent. At the coarsest level, the correlation between average phytoplankton and zooplankton abundances over the study period is indicative of predator-prey interaction or common controlling factors. At a finer level of resolution two different associations were evident. The first was between copepods/cladocerans and diatoms. Copepods and cladocerans show similar seasonal trends in abundance and were both most abundant at the time when diatoms dominated the phytoplankton assemblages. These similarities suggest that all three groups are responding to the same stimuli or there are predator-prey interactions operating between each of the two zooplankton groups and the diatoms. Although both zooplankton groups appear to be grazing on diatoms it is possible that they are not in direct competition as *Penilia avirostris* has a preference for small particles (< 30 μm) which would allow these two groups to coexist if the dominant copepods were exploiting a larger size class of diatoms. The second association was evident between the silicoflagellates and the radiolarians. Although only one documented instance of radiolarians grazing on silicoflagellates could be found in the literature, the late winter initiation of radiolarian blooms and the positive relationship between the magnitude of silicoflagellate and radiolarians blooms is indicative of grazing interaction. It is possible that these radiolarian grazers may play an important role in limiting the magnitude and duration of silicoflagellate blooms. The apparent increase in silicoflagellate numbers in the two embayments since the late 1970s may be due to insufficient grazing pressure. This could result from an inability of grazers to multiply fast enough to keep up with increased silicoflagellate production or it is possible that the growth rates of silicoflagellates have remained relatively constant but there has been a decline in the numbers of silicoflagellate grazers and hence grazing pressure. In either case, the interaction between silicoflagellates and radiolarians appears important but is poorly understood world-wide and should be investigated further.

The suggestion that dinoflagellate densities may be kept low by grazing pressure rather than unsuitable environmental conditions or a competitive disadvantage imposed by another phytoplankton group also requires further investigation. If this is the case, the likelihood of an increase in the abundance of toxic or potentially toxic dinoflagellate species is reduced while the grazing pressure is maintained. Any disruption of the grazing food chain such as through the introduction of toxicants (e.g. pesticides) could therefore have a dramatic effect on the density and/or composition of the dinoflagellate assemblage.

The difference in nutrient loadings to Cockburn and Warnbro sounds is the most likely cause of the different phytoplankton productions. The close agreement between the estimated nitrogen loads and estimated nitrogen requirements of the plankton in Cockburn and Warnbro sounds suggests that the major pathways of nitrogen cycling for the planktonic component of the embayment ecosystems are described.

5.6 Sediment oxygen flux

There are anecdotal reports of 'fish and crab kills' occurring in the deep basin in the southern end of Cockburn Sound during February/March 1992. During late summer/early autumn, air temperatures are high and the sea-breeze pattern can sometimes give way to weak easterly winds that last for several days. As a result, water temperatures are high and vertical mixing and flushing of this area may be reduced (D'Adamo and Mills, 1995c). In addition, the water column can exhibit strong vertical density stratification at this time of year which further restricts the flushing of bottom waters (see section 5.1.2.4). These conditions are conducive to oxygen depletion of bottom waters, particularly in sedimentary basins, such as Cockburn Sound, where sediments contain relatively high amounts of organic carbon. Sediments are also potentially a significant source of nutrients to the water column and sediment nutrient release rates in Cockburn Sound are higher under anoxic conditions (Bastyan and Paling, 1995). Thus, an understanding of the oxygen flux between sediments and overlying waters and of the frequency and duration of physical conditions that restrict the circulation of bottom waters is important in assessing the potential for deoxygenation events from both faunal stress and nutrient enrichment perspectives.

A laboratory study was undertaken to examine the oxygen flux characteristics of intact sediment cores representing the range of sediment types found in the coastal waters of Perth. The objective of this programme was to provide comparative data on sediment oxygen flux to help determine the relative potential for deoxygenation events to occur at the sediment sampling sites. This study was also part of a collaborative research project, with the WAWA, undertaken to assess the relative contribution of sediments to water column nutrient loading at these sites. Dissolved oxygen data loggers were deployed for about 20 days at two sites in Cockburn Sound during April/May 1994. Vertical profiles of dissolved oxygen through the water column were also measured weekly at sites in Cockburn Sound and Warnbro Sound from December 1993 to March 1994.

Further technical details and relevant data can be found in Masini (1993), Cary and D'Adamo (1995) and Masini (1995).

Findings

The oxygen consumption rates in the dark (respiration), gross and net oxygen production rates in the light, and some physical and biological characteristics of the sediments of the study areas are shown in Table 5.6-1. Dark respiration rates of sediments in the five sub-regions of the study area were relatively uniform; mean rates ranged from -8.3 to -21.6 $mgO_2\ m^{-2}\ h^{-1}$ and were not significantly related to any of the measured physical and biological characteristics of the sediments. For example, the sediments of the deep basins of Cockburn Sound and Warnbro Sound are fine-textured and have high organic and water contents, but have similar respiration rates to the coarser and less organically-rich sediments of the Marmion/Whitfords lagoon and Swanbourne areas. The main metabolic difference between the areas was found in the gross oxygen production rates measured under conditions of high light intensity (i.e. sufficient to achieve maximum photosynthetic rates), with an order of magnitude higher gross production rate in the Marmion area compared to Cockburn Sound (Table 5.6-1). This production of oxygen is attributable to benthic microalgal photosynthesis (Masini, 1990). Net oxygen fluxes (rate of consumption plus rate of production) were positive under high light intensities in all regions except the deep basin sites of Cockburn Sound. The chlorophyll *a* content of a sediment provides an index of benthic microalgal biomass but this measure also includes dead phytoplankton cells that have settled on the seabed or have been deposited as zooplankton faecal pellets. The low ratio of chlorophyll *a* to phaeophytin (a chlorophyll degradation product) provides some indication that the majority of chlorophyll *a* in the sediments of the Cockburn Sound deep basin, and to a lesser extent the Warnbro Sound deep basin, is of phytoplankton origin.

Simulation modelling suggests that the sediment oxygen demand in Cockburn Sound is sufficient to deplete the oxygen content of the bottom 1 m of the water column by 70% in 10 days assuming an initial dissolved oxygen concentration of 7 $mgO_2\ l^{-1}$ and that the sediment is the only source or sink of oxygen. At all other sites, benthic production of oxygen at high light intensities is sufficient to maintain oxygen levels at around saturation in the bottom 1 m indefinitely (Masini, 1993).

Vertical profiles of dissolved oxygen concentrations taken weekly between December 1993 and March 1994 (Cary and D'Adamo, 1995) indicate that reduced oxygen concentrations in bottom waters were more common and pronounced in the Cockburn Sound deep basin than in the Warnbro Sound deep basin and were most frequent during the autumn period, which is characterised by light and variable prevailing winds, weak sea-breezes, vertical density stratification and maximum water temperatures. In Warnbro Sound, depressions in dissolved oxygen concentrations of bottom waters of the deep basin were generally minor and dissolved oxygen remained within 5-10% of saturating concentrations. At the shallow sites (2-5 m) on the northern and southern banks of Warnbro Sound, dissolved oxygen concentrations were often higher near the sea bed than the surface and this is attributed to oxygen evolution during photosynthesis by benthic microalgae on the sediments (Masini, 1990) and by seagrass in adjacent meadows. There was no evidence of increased oxygen concentrations in the bottom waters over the Cockburn Sound eastern bank (~ 9 m), where the seagrass meadows were lost in the early 1970s. Within the Cockburn Sound deep basin, low oxygen concentrations were common in the southern end, particularly in the Mangles Bay area. The lowest measured dissolved oxygen concentration was 2.8 $mgO_2\ l^{-1}$ (40% saturation) and this occurred near the bottom of the water column in Mangles Bay on 24 March 1994. This sampling date was characterised by light easterly winds.

An *in-situ* dissolved oxygen data logger was deployed 0.2 m above the seabed in the Mangles Bay area for the period 28 April to 17 May 1994; data were averaged and recorded every 15-minutes. The data indicated that dissolved oxygen concentrations ranged from about 1.9 to 3.5 $mgO_2\ l^{-1}$ on 29 April and rose rapidly to about 5 $mgO_2\ l^{-1}$ by the early hours of 1 May (Masini, 1995). The period of low oxygen concentration (29-30 April) coincided with high water temperatures (> 21 °C) and light winds (< approximately 5 $m\ s^{-1}$) with a dominant easterly component. The subsequent and rapid rise in dissolved oxygen concentrations coincided with the onset of a sea-breeze pattern on the afternoons of 29 and 30 April and a gradual reduction in water temperature. For the remainder of the deployment period, in the presence of cooler water (~ 20 °C), dissolved oxygen concentrations generally fluctuated in the range 4.5 to 7.0 $mgO_2\ l^{-1}$, falling briefly to a value of 3.3 $mgO_2\ l^{-1}$ (50% saturation) during a weak, easterly wind event.

Dissolved oxygen measurements between 15 and 27 April 1994 (immediately preceding the period of low dissolved oxygen concentrations in Mangles Bay discussed above) at a site in the deep basin midway along Garden Island did not fall below about 4 $mgO_2\ l^{-1}$ (60% saturation) and were much less variable than in Mangles Bay, even though weak winds occurred for some of this period. This suggests that the Mangles Bay end of the Cockburn Sound deep basin is not as well mixed and contains discrete bodies of water that are more isolated from sources of oxygen replenishment than the more northerly parts of the Cockburn Sound deep basin.

Numerical simulation modelling of the Cockburn Sound hydrodynamics shows that flow through the causeway openings is into the sound under southwesterly winds and out of the sound under prolonged easterlies (Mills and D'Adamo, 1995c). This suggests that the southern end of Cockburn Sound is better flushed with oceanic waters under southwesterly winds compared with prevailing easterly winds

Table 5.6-1. Dissolved oxygen flux rates and characteristics of sediments from five locations in Perth's coastal waters.

Location	Dissolved oxygen flux rates			Sediment characteristics					
	Gross production	Consumption	Net production	Organic content		Moisture content	Chlorophyll *a*	Phaeophytin	Chlorophyll *a*/ phaeophytin
		($mgO_2\ m^{-2}\ h^{-1}$)		(mg per g dry weight)	(mg per g wet weight)	(%)	($mg\ m^{-2}$)	($mg\ m^{-2}$)	
Cockburn Sound	7.0	-17.2	-10.2	88.5	42.6	50.4	8.3	28.9	0.29
Marmion Lagoon	69.3	-18.2	51.1	14.0	10.2	24.5	17.1	10.9	1.57
Warnbro Sound	55.8	-12.7	43.1	74.0	38.9	47.5	18.2	26.2	0.69
Sepia Depression	42.2	-8.6	33.7	34.0	24.7	27.5	10.5	7.2	1.46
Swanbourne	83.0	-20.2	62.8	8.0	6.4	19.8	15.0	8.5	1.76

(see section 6.2). Hydrodynamic field measurements have shown that the waters of the Cockburn Sound deep basin are typically warmer (and more saline) than adjacent offshore waters during autumn (D'Adamo and Mills, 1995b). Periodic depressions in the water temperature time series from Mangles Bay generally coincide with southwesterly sea-breezes and are indicative of intrusions of water from the shallow margins or offshore.

The higher frequency and greater magnitude of oxygen depletion in bottom waters of Cockburn Sound, compared with Warnbro Sound, appear to be related more to the very low gross oxygen production rates of Cockburn Sound sediments than to their consumption rates, which are comparable to most other sediments analysed (Table 5.6-1). The spatial patterns of dissolved oxygen in the bottom waters of Cockburn Sound suggest that the waters of Mangles Bay have the highest likelihood of suffering deoxygenation events. The findings of the field and laboratory studies provide a plausible explanation for anecdotal reports of 'fish and crab kills' in the southern portion of the deep basin of Cockburn Sound during late summer to autumn, when the likelihood of extended periods of vertical stability is greatest (D'Adamo and Mills, 1995c; Coleman, 1995).

Conclusions

Laboratory experiments and simulation modelling suggest that the oxygen demand of the sediments over the majority of the Perth's coastal waters can be balanced by oxygen evolved during benthic microalgal photosynthesis, given sufficient light levels at the sea bed. In the deep basin of Warnbro Sound, depressed oxygen levels in bottom waters are likely to be infrequent and relatively minor. In Cockburn Sound, the net sediment oxygen demand of the deep basin sediments and restricted flushing, especially in the southeastern corner, increases the likelihood of the development of low dissolved oxygen concentrations and, under certain conditions, the sediments may become anoxic. Any change in the hydrodynamic regime which restricts oxygen replenishment of deep basin waters (e.g. increased vertical stratification) will increase the likelihood of sediment anoxia events. Sediment anoxia inhibits denitrification (biologically-mediated conversion of nitrate into nitrogen gas, which is biologically inert) because of reduced nitrification (production of nitrate from organic matter) which results in a lower availability of nitrate. This often leads to elevated inorganic nitrogen release in the form of ammonium, which is highly available and taken up by most algal species in preference to nitrate (Harrison *et al.* 1983; Bastyan and Paling, 1995). The reducing environment in anoxic sediments also favours the accumulation of toxic compounds such as heavy metals (Chegwidden, 1979). Prolonged anoxia of bottom waters can cause the death of sessile and slow moving benthic fauna and significantly alter community structure.

Low dissolved oxygen concentrations recorded in deep basin waters of Cockburn Sound were associated with periods of weak, predominantly easterly winds and high water temperatures suggesting that late summer/autumn is the period of the year when deoxygenation is most likely to occur. Laboratory-based metabolic studies indicate that benthic primary producers can elevate dissolved oxygen concentrations in surrounding waters and this finding is supported by field measurements in shallow (< 5 m) waters near seagrass meadows. It seems likely that significant losses of benthic primary producers such as seagrasses can alter the dissolved oxygen balance of water bodies.

5.7 Remote sensing as a water quality monitoring tool

The widespread death of seagrasses in Cockburn Sound, during the early 1970s, and Princess Royal Harbour near Albany, during the 1980s, were the result of nutrient enrichment. This environmental degradation may have been prevented or reduced if pro-active, long-term monitoring programmes had been in place. *In-situ* monitoring methods are expensive and time consuming; these are the principal reasons that routine surveillance monitoring of coastal waters is not carried out in Western Australia.

However, recent advances in the remote sensing of marine waters have demonstrated that these technologies can be useful and cost-effective monitoring tools in mesotrophic and eutrophic coastal and estuarine waters (Lavery *et al.* 1990; Pattiaratchi *et al.* 1991). To assess the usefulness of these tools in the coastal waters off southwest Western Australia, a study was undertaken in collaboration with the University of Western Australia, the Department of Land Administration, and the Commonwealth and Scientific Industrial Research Organisation. Multi-temporal, empirical algorithms, applicable to Cockburn Sound, were developed for phytoplankton biomass (expressed as chlorophyll *a*), secchi depth and sea surface temperature. These algorithms were obtained from multiple regression analyses performed between digital data from the Landsat Thematic Mapper (TM) satellite and ground-truth data collected concurrently with the satellite over-passes.

Further technical details can be found in Lavery *et al.* (1990) and Pattiaratchi *et al.* (1991).

Findings

The spatial variability in chlorophyll *a* concentrations within Cockburn Sound during the high algal growth period from summer to autumn can be between about 1 and 10 $\mu g\ l^{-1}$, but with typical variations around the sound of less than about 3 $\mu g\ l^{-1}$ (Cary *et al.* 1991). Pattiaratchi *et al.* (1991) employed multi-temporal, step-wise regression techniques to develop predictive relationships between contemporaneous TM data and chlorophyll *a* measurements in Cockburn Sound. Pattiaratchi *et al.* (1991) found, on the basis of these relationships, that the Landsat TM was able to sense chlorophyll *a* levels in Cockburn Sound with a regression standard error corresponding to an accuracy of ± 0.3 $\mu g\ l^{-1}$ (Figure 5.7-1). Hence, for situations when the chlorophyll *a* variation around the basin is significantly greater than this, as is often the case during summer and autumn, the use of Landsat TM could be useful, especially in conjunction with selected ground-truthing data.

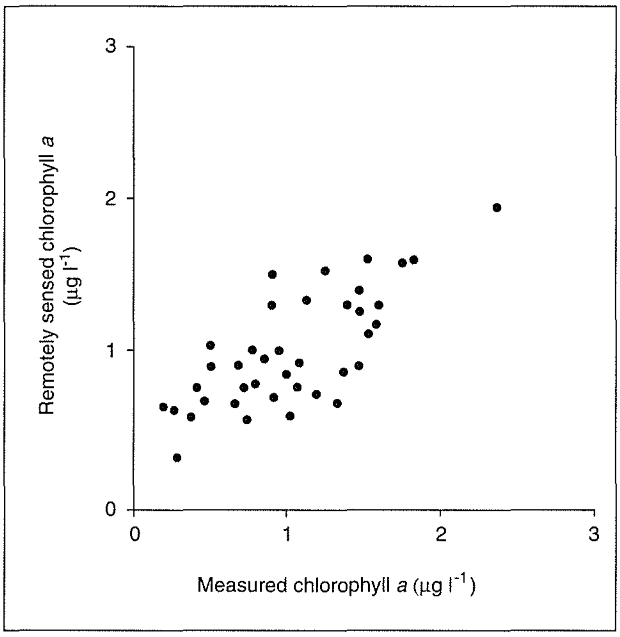


Figure 5.7-1. Relationship between remotely sensed and measured chlorophyll a concentrations in Cockburn Sound.

Problems with the use of such satellites for this purpose are (i) the spatial and temporal variability of cloud cover or other atmospheric particulates such as smoke and haze, (ii) the contamination of the signal from water-borne chlorophyll *a* by the spectral response from benthic plants in clear waters and, (iii) the infrequency of satellite passes. However, the results have provided sufficient confidence to suggest that this technique should be pursued as the ability to define narrower wavelength bands improves with emerging remote sensing technology. This improvement is presently occurring with the continuing development of air-borne multi-spectral scanners. Such scanners can be deployed from aircraft and thus have the advantage of being able to avoid cloud cover if mounted in low-flying mode, and also to be deployed at the most suitable times. In addition, the cost of flights compares favourably with the cost of acquisition of data from satellites.

The regression results for secchi disk depths were also reasonable in terms of comparing measured and sensed data, with the regression standard error for the data corresponding to an accuracy of ± 1.2 m. Secchi disk depths of over 5 m are common in the coastal waters and hence satellite remote sensing could be a useful technique for measurement of this parameter.

Sea-surface temperature, obtained from the thermal band 6 of the Landsat TM, was compared to field data and found to be accurate to within ± 0.5 degrees Celsius. Typically, the natural spatial variation of surface temperature within Cockburn Sound is of this order, but the variation between nearshore and offshore waters can exceed this. In addition warm water outfalls can result in elevated temperature plumes with temperature excesses of greater than 0.5 degrees Celsius above ambient. Hence, the use of satellite remote sensing can have applicability for the measurement of sea-surface temperature in some cases.

Conclusions

The *in-situ* monitoring of water quality over large areas of the marine environment is generally labour intensive, logistically complex and therefore expensive. However, there are continuing advances in the resolution of remote sensing instruments for this purpose. There are planned launches over the next five years of several satellites, such as the SEASTAR satellite with its SeaWIFS sensor (Atlantic Centre for Remote Sensing of the Oceans, 1994). The SeaWIFS will be able to sense chlorophyll *a* in the range 0.05 - 50 $\mu g\ l^{-1}$, with a certainty of about ± 35 per cent (Hooker and Esaias, 1993). Imminent advances in remote sensing technology mean that remote sensing can be considered as an increasingly more viable monitoring tool for parameters such as chlorophyll *a*, light penetration and sea-surface temperature over ranges that commonly occur in Perth's coastal waters. Their use should prove to be cost-effective for the long-term monitoring of water quality off Perth and in providing baseline water quality information for other coastal regions.

5.8 Other studies

As previously mentioned, the studies outlined above are the process studies undertaken as part of the Southern Metropolitan Coastal Waters Study and, together with a series of other studies, conducted as part of WAWA's Perth Coastal Waters Study (PCWS), provide the basis for the development of the COASEC nutrient-effects model (van Senden, 1994). Independent studies have also been undertaken that provide valuable insights into some of the key processes regarding the impacts of nutrient inputs to Perth's southern metropolitan coastal waters. A brief summary of other process studies of relevance to the SMCWS is given below.

5.8.1 Physical oceanography

The physical data sets of the PCWS were collected to identify and characterise the dominant physical processes which influence circulation patterns within Perth's coastal waters and to provide data sets required for the forcing and validation of a hydrodynamic model. The study domain was set at two spatial scales; firstly, from Yanchep to Mandurah and offshore to the continental slope, to investigate shelf scale processes, and secondly, about the Beenyup and Cape Peron domestic wastewater outfalls, to investigate the local scale hydrodynamics pertinent to mixing and dispersion of the wastewater plumes. Details of the oceanographic studies undertaken for the PCWS are presented in Pattiaratchi *et al.* (1995).

Seasonal CTD surveys were conducted from April 1993 to February 1994 and comprised monthly transects from the nearshore zone to the 140 m contour some 50 km offshore. Intensive CTD surveys were conducted in April 1992 (Beenyup outfall) and February 1993 (Cape Peron outfall).

Currents were monitored at six different locations along a cross-shelf transect from the Whitfords Lagoon for various periods throughout 1993. Two deep water current meter strings were located on the outer continental shelf, with currents measured at four depths throughout the water column. Four other sites were monitored closer to the Whitfords Lagoon, and meters were positioned near-surface and near-bottom at these sites.

Satellite imagery was acquired by the Department of Land Administration and processed for the PCWS. The objective was to identify the spatial and temporal characteristics of regional currents, such as the Leeuwin Current and the "Capes Current". This resulted in a series of NOAA-AVHRR sea-surface temperature (SST) images of the Western Australian coastal zone at approximately weekly intervals for more than a year. In addition, a smaller number of Landsat Thematic Mapper images were acquired and processed to provide more spatially resolute SST images and also sea-colour data.

A suite of meteorological data was collected at coastal sites for the PCWS (e.g. solar radiation, air temperature, relative humidity, and wind). Wave and sea-level data were drawn from Department of Transport data sets.

Three-dimensional hydrodynamic modelling studies, carried out as part of the PCWS, are reported by Pattiaratchi and Knock (1995). The model was applied at shelf-scale to examine aspects of the broader circulation patterns off Perth, and over two smaller domains, to provide advection fields in the vicinity of the Ocean Reef (Beenyup), Swanbourne and Cape Peron domestic wastewater outfalls.

Speedy and Hearn (submitted) have recently modelled the change in the barotropic flow regime of Cockburn Sound due to the presence of the Garden Island causeway. A two-dimensional, depth-averaged barotropic model was applied. The results are consistent with earlier depth-averaged modelling results (Maritime Works Branch, 1977a, b; Hearn, 1991) which suggest that the presence of the causeway has reduced the barotropic exchange rate of the sound (i.e. the combined volume exchange rate via the north and south entrances) by a factor of approximately two. In addition, Dr C J Hearn and Dr J R Hunter (CSIRO, Division of Oceanography) have performed several localised modelling investigations of circulation and effluent dispersion in Cockburn Sound, in relation to the flushing efficiencies of proposed marinas/boat harbours, and the mixing and spreading of effluent plumes originating from the Cockburn Sound shoreline near the Kwinana industrial zone (e.g. Hearn, 1991; Hearn, 1992).

Mr Grant Ryan (Department of Transport) has applied a numerical model to simulate the dynamics of the Dawesville and Mandurah channel outflows, and also the influence of the Dawesville Channel on water levels in the Peel-Harvey Estuary (Department of Marine and Harbours, 1993). This information is of relevance to understanding the efflux of nutrients in Peel-Harvey Estuary outflows to the coastal zone.

Stephens and Imberger (submitted) have conducted field and analytical investigations of the dynamics of the stratified Swan River estuary, including an analysis of the exchange dynamics at the estuary mouth. The estuary contributes fluxes of buoyancy (in brackish water), nutrients and other contaminants to Perth's southern coastal waters, and hence this investigation is of relevance to the SMCWS.

5.8.2 Ecology

Other ecological process studies have recently been conducted in Perth's coastal waters and some are relevant to the SMCWS. These studies have primarily been conducted at universities and the CSIRO and are briefly described below.

Lavery (1993) reviewed the literature on nutrient-related macroalgal processes of possible relevance to Perth's coastal waters. This review included information and data on the effects of nutrient enrichment on macroalgal production, nutrient 'stripping' of the water column by macroalgae, the influence of macroalgal production on water column light attenuation through detrital re-suspension, the extent of nitrogen recycling by macroalgal wrack, and nutrient-induced changes in macroalgal assemblages.

Hillman *et al.* (1994) conducted a study on behalf of the WAWA to quantify the seasonal and spatial changes in the biomass of the major benthic primary producers (seagrasses and macroalgae), epiphytic plant assemblages on seagrasses, and secondary benthic producers (invertebrates >1 mm) using a combination of field measurements and information derived from the scientific literature. This study also identified the major assemblages of invertebrate grazers on seagrass epiphytes and examined the seasonal variation in the biomass of these assemblages. Environmental factors influencing the biomass of these assemblages were investigated, particularly nutrient enrichment.

The effects of temperature on the relationship between light and productivity of seagrasses (*Posidonia sinuosa* and *Amphibolis griffithii*) and assemblages of seagrass epiphytes was examined using metabolic chambers (Walker *et al.* 1994b). As an alternative to this metabolic technique, experiments were also undertaken using large aquaria (mesocosoms) to examine the effects of light, temperature and nutrients on the growth of seagrasses (*P. sinuosa* and *A. antarctica*) and seasonal assemblages of epiphytes.

Jernakoff and Nielson (1994) conducted a series of laboratory and field experiments to examine the interaction between invertebrate grazers of seagrass epiphytes and the epiphyte assemblage. These studies, and values obtained from a previous review of the scientific literature (Jernakoff *et al.* 1993), provided input parameters, such as grazing demands and mortality rates, for the grazing sub-model of COASEC.

Lavery *et al.* (1993) reviewed sediment nutrient processes in Perth's coastal waters, especially the relative influence of sediment functions on nutrient cycling. Rosich *et al.* (1994) conducted a pilot study of sediment nutrient release rates at ten sites throughout the coastal waters of Perth, including sites in the deep basins of Cockburn and Warnbro sounds, and in Sepia Depression. Nitrogen and phosphorus fluxes from the sediments under aerobic and anaerobic conditions were measured using intact sediment cores. More recently, a comparative study of sediment nutrient release rates between sites in Cockburn and Warnbro sounds was undertaken by Bastyan and Paling (1995).

6. MODELLING

6.1 Introduction

This chapter summarises the hydrodynamic and ecological modelling studies that were conducted as part of the SMCWS.

Hydrodynamic modelling

The hydrodynamic modelling was carried out in conjunction with the water quality and oceanographic field studies (sections 4.7 and 5.1) to investigate water circulation and mixing patterns of the study area in response to a range of meteorological, oceanographical and hydrological conditions. In the context of the SMCWS, the aims of the hydrodynamic modelling were to determine the major hydrodynamic transport pathways of water-borne substances, to identify existing and potential links between contaminant source locations and areas of environmental sensitivity, and to determine the flushing characteristics of the coastal embayments.

The use of numerical hydrodynamic modelling in studies of this nature has increased steadily and in the past few years three-dimensional baroclinic models have been applied to refine the understanding of the hydrodynamics of some systems. Confidence in the ability of a model to provide realistic simulations has first to be established by obtaining satisfactory agreement between model results and oceanographic field data sets collected under known forcing conditions in a given area. Once this has been achieved the model simulations are able to complement the understanding derived from field surveys by providing results at a much finer spatial and temporal resolution than would be economically feasible with field survey techniques. The simulations can also be designed to examine alternative scenarios, not encountered in the field surveys, involving changed bathymetry (for example, due to proposed construction works), different contaminant input rates and locations, and other combinations of forcing conditions.

The field and modelling studies have highlighted major hydrodynamic processes which contribute to the circulation, mixing and flushing of these waters. These include baroclinic and barotropic (non-baroclinic) processes. *Baroclinic* processes involve horizontal differences in water density which modify the fluid pressure distribution and thereby contribute to the horizontal momentum of the water. In addition, vertical density stratification influences the efficiency of vertical mixing processes. Density gradients arise from heterogeneities in the temperature and salinity of the seawater. The oceanographic field data for the study area indicate that stratification, in the form of horizontal and/or vertical gradients of temperature and salinity, is almost invariably present. The potential importance of density effects, including baroclinic processes, to circulation, mixing and flushing in the study area has been acknowledged in previous studies (e.g. Steedman and Craig, 1979, 1983; Hearn, 1991) but not studied in detail. Both the field measurement and modelling components of the SMCWS were designed to provide further understanding of their role and importance.

The major focus of the hydrodynamic modelling effort described here has been on the series of embayments, basins and channels which comprise the SMCWS nearshore area. 'Local-scale' model domains extending to about 20 km offshore have been established to encompass these water bodies and to include outflows from the Swan-Canning and the Peel-Harvey estuaries. However, to gain some appreciation of the relative influence of the wind and long-shelf sea-level slope in forcing circulation at a broader scale, several 'shelf-scale' model simulations have been conducted over a larger area, extending along the entire Perth metropolitan coastline from Yanchep to Tim's Thicket and across the width of the continental shelf. Modelling of Perth's coastal waters at 'shelf-scale' has also been recently undertaken by Pattiaratchi and Backhaus (1992) and Pattiaratchi and Knock (1995).

The model used for the SMCWS is described in section 6.2.2. Some shelf-scale simulation results are presented in section 6.2.3. The main hydrodynamic modelling section is 6.2.4, which focusses on the local-scale circulation and dispersion characteristics throughout the nearshore embayments, basins and channels. Section 6.2.4 presents comparisons between current meter data and simulated currents to demonstrate the simulation ability of the model; it summarises results from the modelling of wind driven buoyant plumes discharged from the Swan-Canning and the Peel-Harvey estuaries; it presents the simulation of three distinct seasonal hydrodynamic regimes for Cockburn Sound, the nature of each being determined primarily by the horizontal density differences between the sound and the surrounding shelf waters, and by wind forcing; it briefly examines the effects of construction of the Garden Island Causeway on the flushing of Cockburn Sound; and finally, it presents the modelled circulation and mixing response of Warnbro Sound to wind forcing, given the initial presence of vertical density stratification. Section 6.2.5 summarises model results of the basin-scale transport and dispersion of materials released from a point source in Cockburn Sound.

Ecological modelling

The ecological modelling was carried out in conjunction with the ecological base-line and process studies (Chapters 4 and 5) to investigate the consequences of nutrient enrichment on key components of coastal ecosystems in Perth's southern metropolitan coastal waters.

The modelling focussed on formulating, calibrating and testing algorithms representing the processes by which nutrients affect the structure and function of local marine ecosystems so as to develop relationships between nutrient load and ecological response. This provided a technical basis upon which to assess the short- and long-term consequences of nutrient load management strategies for the key embayments of the study area. It also assisted in developing nutrient-related environmental quality criteria (Chapter 3).

The usefulness of ecological models for environmental management has been the subject of debate over the past 2-3 decades. An emerging theme in this debate is that complex 'ecosystem' models which attempt to represent all pathways, interactions and forcing functions are still of limited usefulness for environmental management and decision making. However, it is now recognised that specific process modelling can be extremely valuable (GESAMP, 1993). In this latter approach, a hypothesis (or set of hypotheses) concerning particular linkages between stress and response is formulated as a model and tested against data.

In this study, the relationship between nutrient load and ecological response was described by a suite of simple, empirical and deterministic stress-response sub-models. Much of the algorithm development was undertaken in conjunction with the PCWS, especially the development of the integrated Coastal Ocean Ecology model COASEC (van Senden, 1994). The stress-response pathway which forms the basic structure of this model was initially outlined by Masini *et al.* (1991) in a conceptual model of nutrient-effects on local marine communities. The seagrass *Posidonia sinuosa* was chosen as the key ecological health indicator because it was known to be susceptible to eutrophication and because of the inherent ecological significance of this seagrass species in the nearshore coastal waters of southern Western Australia (Simpson *et al.* 1993).

The key ecological sub-models are briefly described in section 6.3.2. Progress made in the development and testing of the ecological sub-models is described in section 6.3.3. The results of the simulations and their relevance to management are described in section 6.3.4.

The modelling tools developed during the course of this study can be used to investigate other scenarios which may arise for the study area. The models have been structured in such a way that they can be further developed for application to other ecosystems. A discussion of the potential for further application and development of these models in the context of the need for environmental management and decision-making is given in section 6.4.

6.2 Hydrodynamic modelling

6.2.1 Model requirements

The hydrodynamic modelling component of the SMCWS was based on reviews of the local and regional oceanography (Hearn, 1991; D'Adamo, 1992), the findings of the SMCWS oceanography characterisation phase (see section 5.1) and on a review of available models. The reviews and characterisation confirmed the importance of wind as a primary driving force for the currents and highlighted the fine balance existing in the study region between natural stabilising processes that cause lighter water to move over or form above heavier water (e.g. freshwater runoff and surface heating) and destabilising processes, such as turbulent mixing by wind stress or penetrative convection due to surface heat loss. The characterisation also suggested the significance of the earth's rotation at basin-scale and shelf-scale, and the essentially three-dimensional nature and time-dependence of the water circulation and mixing regimes.

The hydrodynamic model was therefore required to be three-dimensional, time-dependent, and to be able to simulate responses to a wide range of forcings including imposed (meso-scale or regional) horizontal pressure gradients, tidal forcings, wind stress, horizontal buoyancy fluxes (from estuaries), vertical buoyancy flux (from air/sea transfers of heat and water), local pressure gradients (due to water surface slopes and horizontal density gradients set up within the study area), bottom frictional stresses and the effects of the earth's rotation. The model needed to take account of the relatively complex bathymetry of the study area and to be able to simulate a range of barotropic and baroclinic mechanisms, with realistic vertical and horizontal structure in water movement and water density, and realistic vertical mixing.

Three-dimensional baroclinic modelling techniques are now sufficiently advanced (Spaulding *et al.* 1992) that these models may be used in conjunction with appropriate field measurements to investigate the water circulation and hydrodynamic transport of substances in coastal water bodies under a broad range of meteorological and oceanographical conditions.

6.2.2 Description of the model

The *Princeton Ocean Model*, generically known as the 'Blumberg-Mellor Model' (Blumberg and Mellor, 1980, 1987; Mellor, 1993) was chosen for application to the SMCWS. Over recent years this model has undergone development and successful application to diverse regions such as Chesapeake Bay, the Mid-Atlantic Bight, the Gulf of Mexico, the Great Lakes, and the east and south coasts of Australia.

This time-dependent, fully three-dimensional model numerically solves the non-linear primitive equations for the conservation of mass, momentum, salt and heat. The density distribution is derived from the modelled seawater temperature, salinity and pressure fields using an equation of state. Horizontal density gradients give rise to baroclinic forcings which feed back into the momentum balance. The vertical density gradients affect the water column stability which modifies vertical mixing in the model, as determined by a turbulence-closure sub-model (Mellor and Yamada, 1982). The model utilises a sigma coordinates system (Phillips, 1957) in which the vertical coordinate is scaled to water depth. In addition to three-dimensional simulations, the model can also be used in two-dimensional (depth-averaged) form.

Two versions of the Princeton Ocean Model have been used in the SMCWS: a Fortran version, as described in Mellor (1993) and a version modified and rewritten in the C language by Herzfeld (1995) and further developed during the SMCWS (Mills and Essers, 1995). The C language version of the model incorporates a range of open boundary conditions and employs the positive definite advection algorithm of Smolarkiewicz and Clark (1986). The far-field transport of effluent plumes from estuaries and ocean outfalls has been simulated in the model by introducing point sources of volume, heat and salt, following the method of Lazure and Salomon (1991).

6.2.3 Shelf-scale modelling

6.2.3.1 *Aims*

Several shelf-scale simulations were conducted to gain a general appreciation of the broader circulation features off the Perth metropolitan region, the influence of Rottnest Island, which forms a major obstacle to flow along the shelf, and to understand the respective roles of wind stress, steric height gradient and the earth's rotation in driving this circulation. This work was also used to help classify water quality zones across the shelf and to guide more detailed local-scale modelling of the SMCWS area (see section 6.2.4). Further details may be found in Mills and D'Adamo (1995a).

6.2.3.2 *Model domain and grid*

The hydrodynamic model was applied to the continental shelf area shown in Figure 6.2-1 which is bounded by Yanchep and Tim's Thicket, a north-south distance of 131 km, and by the mainland coast to 48 km west of Fremantle (maximum depth 185 m). The model grid consisted of an array of rectangular cells of dimensions 1 km (east-west) by 2 km (north-south). The offshore open boundary condition was clamped and the north and south cross-shelf open boundary conditions were treated as cyclic.

The model was forced by wind stress and an imposed long-shelf pressure gradient. The latter was represented explicitly in the long-shelf momentum balance to facilitate the use of cyclic boundary conditions, following the approach of Pattiaratchi and Backhaus (1992). The model was used in two-dimensional, depth-averaged form to obtain the results described in the following section.

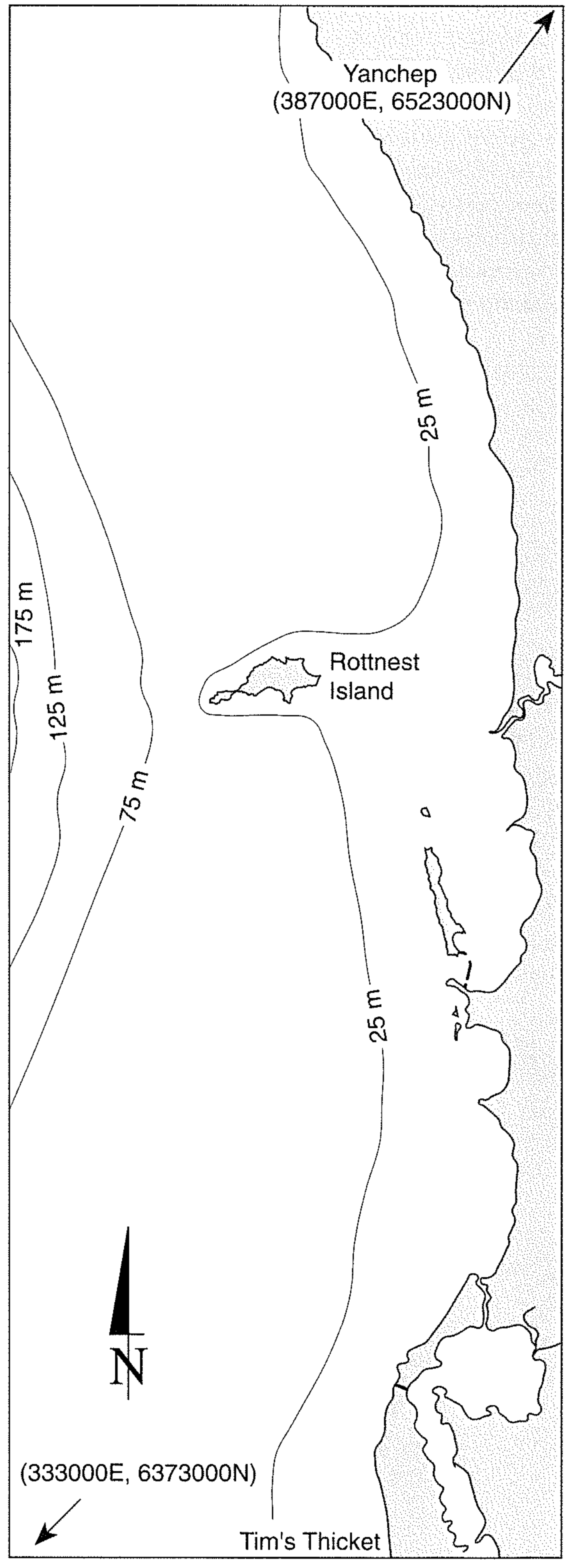


Figure 6.2-1. Domain for the shelf-scale hydrodynamic model simulations.

6.2.3.3 *Roles of wind stress, wind direction, steric height gradient and bathymetry*

Depth-averaged model simulations were conducted to investigate the shelf-scale circulation forced by southerly winds in the presence of an opposing regional, long-shelf pressure gradient. These simulations were allowed to proceed toward steady state, and the resultant current vector fields after five days, for wind speeds of 4.5 and 8 m s^{-1}, are shown in Figure 6.2-2. For each wind speed the model predicted that there was an inshore (shallow water) zone of northward (downwind) flow where the wind stress dominated the long-shelf pressure gradient and an offshore (deep water) zone of southward flow (upwind) where the long-shelf pressure gradient dominated the wind stress. The location of the boundary between northward and southward depth-averaged flow was found to depend on the relative strength of the long-shelf wind stress and the opposing regional pressure gradient. For greater wind speeds, this boundary was located further offshore, and the near-shore, downwind current speed increased. Rottnest Island was found to create a major obstacle to long-shelf flow, causing current strengths to be enhanced about its western and eastern ends, flow separation to occur, and a zone of weak recirculating flow to form on the downstream side of the island. Southerly wind and resulting northward currents in Sepia Depression led to inflow to Cockburn Sound through its southern entrance and northeastward flow through Challenger Passage and across the reef line north of Garden Island. In the semi-enclosed waters of Cockburn Sound and Warnbro Sound, the depth-averaged currents were found to be much weaker than in the adjacent Sepia Depression as a result of wind-induced water level gradients in these embayments which forced water to move predominantly upwind over the deep basins, while the wind forced water downwind over the shallower margins of these embayments.

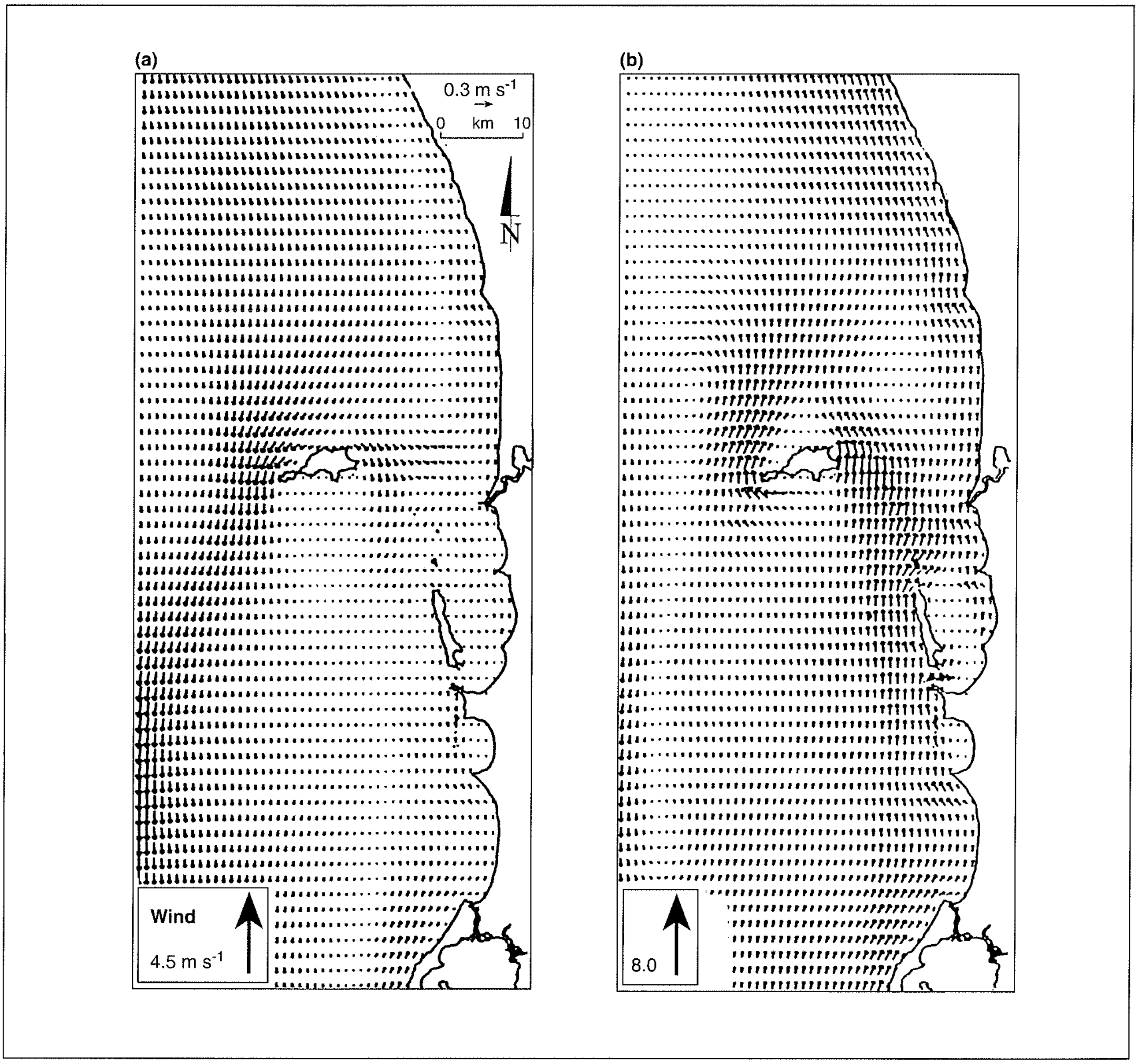


Figure 6.2-2. Modelled shelf-scale depth-averaged velocity fields in response to a north to south sea-level slope and southerly wind of (a) 4.5 m s^{-1} and (b) 8.0 m s^{-1}.

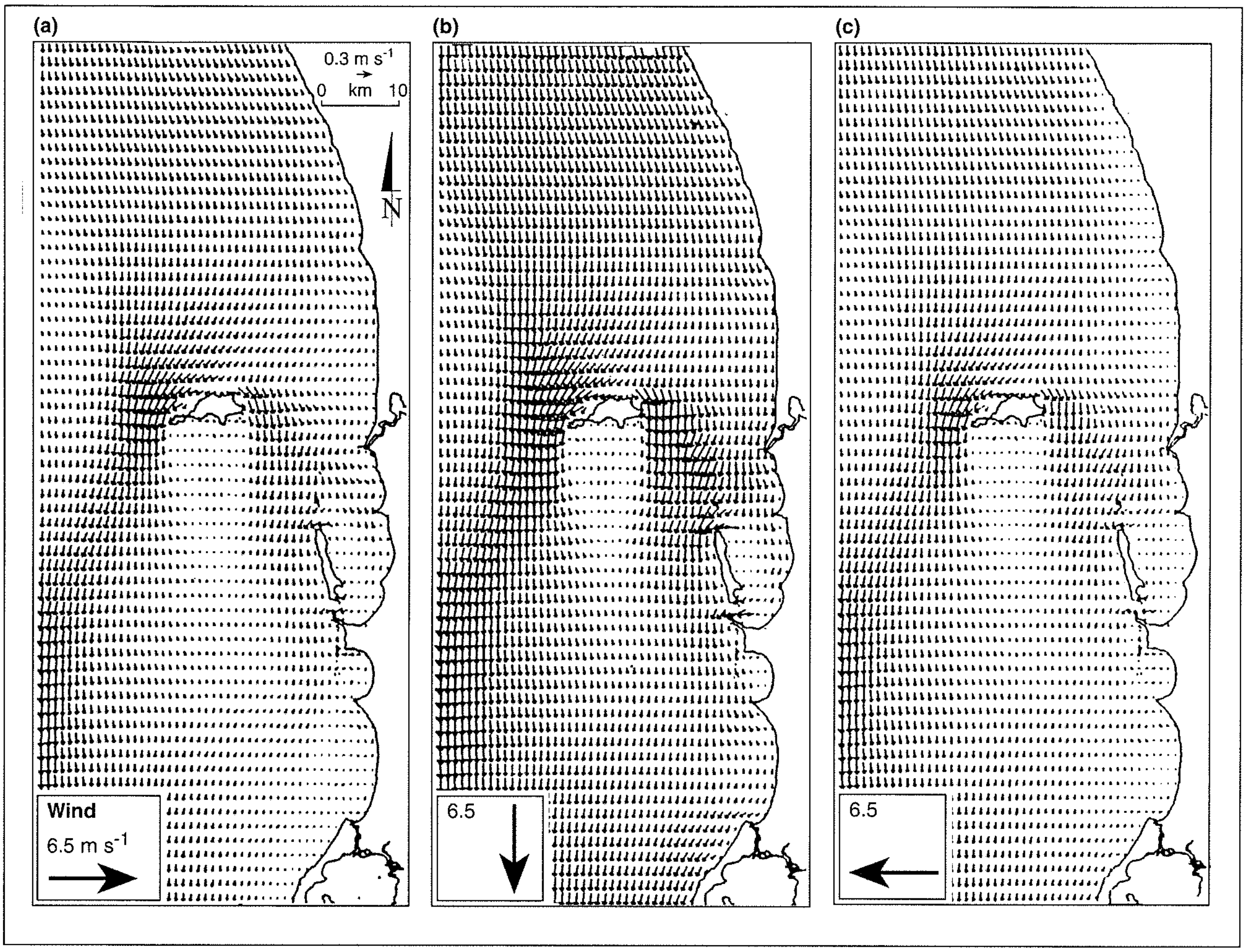


Figure 6.2-3. Modelled shelf-scale depth-averaged velocity fields in response to a north to south sea-level slope and (a) westerly, (b) northerly and (c) easterly wind.

Another series of depth-averaged model simulations was conducted to examine the effects of winds from different directions on the depth-averaged currents in the presence of a long-shelf, poleward sea level pressure gradient force. The current vector fields from the westerly, northerly and easterly wind-driven simulations are shown in Figure 6.2-3a, b and c, respectively. For westerly winds, depth-averaged flow was again to the south across most of the shelf, however this result depended on the relative strength of the wind and the long-shelf pressure gradient. For northerly and easterly winds, depth-averaged flow was to the south across most of the shelf.

The results of the simulations forced by southerly winds, discussed above and represented by the vectors in Figure 6.2-2, showed that southward flow was encountered only offshore, and there was northward flow in Sepia Depression, inflow through the southern entrance of Cockburn Sound and northeastward flow through Challenger Passage. Depth-averaged outflow through the southern entrance and Challenger Passage occurred for northerly and easterly winds, and accompanying southward flow in Sepia Depression. For westerly winds, depth-averaged outflow through the southern entrance and Challenger Passage and accompanying southward flow in Sepia Depression occurred, however this result depended on the relative strength of the wind and the long-shelf pressure gradient.

The model was re-run with several settings of the long-shelf pressure gradient in the presence of a southerly wind. The value of the long-shelf pressure gradient was found to appreciably influence shelf currents and flow through the southern opening of Cockburn Sound, however there was minimal change in the current speed and direction throughout the wider portions of the sound.

Conclusions

The two-dimensional shelf-scale modelling indicates that the coastal water currents are directed predominantly long-shore under a wide range of wind forcing conditions. Dissolved and suspended substances discharged to the coastal zone are therefore transported by the currents in a predominantly long-shore direction. Hence these substances tend to remain in the coastal zone, which is biologically productive and is used for multiple purposes by the community. For this reason managers need to carefully consider the potential cumulative impacts (ecological and social) of discharges to the coastal zone, taking into account the temporal and spatial scales of the transport processes.

The modelling shows that, in summer, when winds from the southerly quadrants prevail, the nearshore waters are predominantly transported to the north. Hence, under these conditions, point and diffuse contaminant discharges to these waters mainly affect the water quality to the north of the discharge. This was exemplified in the summer intensive field measurements by tracking a coastal band of high salinity water northward from the vicinity of Mandurah over a distance of about 30 km in three days (D'Adamo and Mills, 1995a).

In summer the mean prevailing southerly wind is stronger relative to the opposing regional sea level gradient than at other times of the year. Under these conditions, following the trend of the shelf-scale modelling results, the boundary between northward (coastal) flow and southward (offshore) flow should be located far from the coast in summer. This conclusion is consistent with findings from observational data (e.g. Cresswell and Golding, 1980; Cresswell, 1991) which confirm that in summer a wind-driven high salinity northward flow occupies most of the width of the continental shelf, with southward flow occurring further offshore. Hence the shelf waters which flush the nearshore coastal zone are predominantly drawn from the south during summer.

Contrary to the summer situation, in winter, the mean long-shelf component of the wind is much weaker and the opposing sea-level gradient exerts a greater influence on currents over the shelf. The model results suggest that under these conditions the boundary between northward (nearshore) flow and southward (offshore) flow should on average be located closer to the coast. As noted by Cresswell (1991) and Mills *et al.* (1996), in winter, the southward moving Leeuwin Current is found to extend shoreward across the continental shelf and therefore the shelf waters which flush the coastal zone are predominantly drawn from the north.

In winter, wind reversals occur every 3-4 days. The model results give depth-averaged coastal currents of order 0.1 m s^{-1}, hence it is to be expected that, in unconfined waters, successive north-south excursions of water occur over distances of about 30 km. Since the length scale of these excursions is smaller than the length of the Perth metropolitan coastline (order 100 km) long-shelf transport is not necessarily an efficient materials flushing mechanism in winter. This highlights the issue of cross-shelf exchange and its potential importance to flushing cumulative waste discharges in winter, when the long-shore transport can be limited by major periodic wind shifts. A more detailed understanding of cross-shelf transport mechanisms and their contribution to the export of waste materials from Perth's coastal waters is required for the environmental management of these waters.

The shelf-scale modelling indicates that the semi-enclosed embayments exhibit basin scale circulation and have lower advection speeds than the inner shelf areas (e.g. Sepia Depression). Hence particular management attention is required in these areas which are likely to be more vulnerable to stress.

These conclusions provide the motivation for more detailed modelling at the local scale.

6.2.4 Local-scale modelling

6.2.4.1 *Aims*

Perth's southern metropolitan nearshore coastal waters comprise a series of semi-enclosed basins and channels, including Owen Anchorage, Cockburn Sound, Warnbro Sound and the Sepia Depression (Figures 1.1-1, 1.1-2 and Plate 4.1-1). These water bodies are typically up to 15-20 m deep and range from about 3-10 km in width. Their geomorphology and bathymetry have been described in sections 2.3 and 4.1.1, respectively. The aim of the 'local-scale' hydrodynamic modelling was to determine the spatial and temporal scales of the transport of water-borne materials throughout this area. Key issues investigated by the modelling were: the trajectories and mixing of plumes from the Swan-Canning Estuary and the Peel-Harvey Estuary; the exchange fluxes between semi-enclosed embayments and their surrounding waters; the flushing times of the embayments; the internal circulation of individual basins and embayments; and the far-field spread of contaminant plumes from municipal, industrial and groundwater discharges.

6.2.4.2 *Characteristics of the local-scale circulation*

The design of a numerical model application must take into account the goals of the modelling (see above) and the primary bathymetric and hydrodynamic characteristics of the area to be modelled.

Wind is a major driving force on the waters of the study area (Steedman and Craig, 1979, 1983). Wind-driven current speeds are of order 0.1 m s^{-1}. Major wind shifts occur about once every 3-4 days (Breckling, 1989), and in the intervening periods the advective length scale of wind-driven currents over the inner continental shelf is of order 30 km (Mills *et al.* 1996). In semi-enclosed embayments the wind drives water towards or away from coastlines resulting in sea level slopes and resultant horizontal gradients in the pressure field which, in conjunction with the wind stress, lead to local circulation patterns. In a depth-averaged description of the flow field, these circulation patterns are referred to as topographic gyres (Csanady, 1982).

Current meter data show that the waters of Sepia Depression, off Cockburn Sound and Warnbro Sound, have a net southward flow in autumn-winter and a net northward flow in summer (Steedman and Associates, 1981). Following the method of Scott and Csanady (1976) a linear regression of long-shelf current component against long-shelf wind stress component may be used to infer an effective long-shelf pressure gradient which accounts for a residual current, even under zero wind stress. Steedman and Associates (1981) and Pattiaratchi *et al.* (1995) found for the shelf waters off Perth that the inferred pressure gradients are equivalent to a sea level slope of order 10^{-7} and and varies throughout the year, with greatest poleward gradients in winter. Simple momentum balance calculations and modelling confirm that long-shelf pressure gradients of this magnitude drive currents of order 0.1 m s^{-1} over the mid shelf (depths of about 20-50 m). In addition, the model demonstrates that, within Cockburn Sound, the volume rate of throughflow driven by these pressure gradients is limited by frictional impedance and the small flow area through the causeway openings at the southern entrance of the sound. Hence, the wind-driven currents largely determine the circulation patterns within the sound.

The earth's rotation tends to deflect currents anti-clockwise in the southern hemisphere. Over time periods of order one day or more, and for water current speeds of order 0.1 m s^{-1}, typical of the study area, rotational effects become significant in water bodies which have horizontal dimensions of several kilometres or more. In such cases the Rossby number (Csanady, 1982) is less than one.

D'Adamo (1992) reviewed past data and concluded that density stratification (horizontal and/or vertical) was almost always present in Perth's nearshore coastal waters. Vertical density stratification influences the way in which the wind-induced momentum is mixed through the water column, and therefore influences the circulation. The presence of horizontal density gradients which influence the pressure field, introduces additional forces driving the motion.

If the width of an embayment largely exceeds the baroclinic radius of deformation (Csanady, 1982) corresponding to its vertical density stratification, rotational effects lead to upwelling and downwelling responses transverse to the principal (long-shore) current direction. These dynamical responses are contained within a band having a width that compares with the baroclinic deformation radius along each side, and each side behaves independently of the other. If the width of the embayment scales with the baroclinic deformation radius, then the effects of the lateral boundaries as well as coriolis effects are significant throughout. If the basin is considerably smaller than the baroclinic deformation radius, then the upwelling and downwelling response of the stratification will occur mainly in the direction of the wind.

For typical ranges of vertical density stratification (i.e. a density difference of 0.1 to 0.5 kg m^{-3} in 20 m water depth (D'Adamo and Mills, 1995b)) the baroclinic radius of deformation in Perth's coastal embayments, channels and inner-shelf is in the range 1-3 km. Cockburn Sound is about 15 km long and up to 10 km wide. The basins of Warnbro Sound, Mangles Bay and Owen Anchorage, although smaller than Cockburn Sound, nonetheless have horizontal dimensions which scale with the baroclinic radius of deformation.

Horizontal gradients of density may result from buoyancy fluxes, non-uniform vertical mixing, upwelling of the density structure and other causes. If the wind weakens appreciably, then baroclinic adjustment flows (e.g. Gill, 1982) may occur, as the water body tends toward a state of gravitationally stable equilibrium. Where rotation is important and horizontal density fronts are present, this leads to currents in a direction approximately parallel to the fronts, and tends to maintain these structures. The important influence of rotation on the buoyancy-driven circulation of the sound is discussed later in this chapter (see section 6.2.4.8).

Scaling analysis (Steedman and Craig, 1979, 1983) shows that dynamical effects due to the advection of momentum can be important where currents undergo spatial variations of order 0.1 m s^{-1} within horizontal distances of 1 km. These effects alter the distribution of momentum and modify flow patterns locally, for example where recirculating eddies form in the lee of headlands or islands.

The tides of the SMCWS area are mainly diurnal with a maximum spring range at Warnbro Sound of about 0.9 m and an annual mean range of about 0.5 m (Hearn, 1991). Scaling analysis (Steedman and Craig, 1983) indicates that tidal current speeds are generally very weak, of order 0.01 m s^{-1} in open shelf waters and broad coastal embayments. However they may be stronger near narrow channels or shallow banks and reefs. For example, the modelling of Speedy and Hearn (submitted) has shown that tidal currents can be locally significant through the causeway bridge openings at the southern entrance to Cockburn Sound and across the adjacent Southern Flats. Since the tidal currents are generally an order of magnitude smaller than the wind-driven currents, and the advective length scale of tidal flows is only about 500 m, much less than that for wind-driven flows, tidal forcing effects have been excluded from the model simulations to be discussed.

6.2.4.3 *Model domain and grids*

Two model domains were used for the local-scale modelling of the southern metropolitan coastal waters, a northern domain and a southern domain. The northern model domain (Figure 6.2-4) extends 60 km long-shore, from City Beach in the north to mid Comet Bay in the south, and 23 km offshore, from the mainland coast to Rottnest Island and the vicinity of the 35 m depth contour. The maximum depth in the northern model domain is 39 m.

The southern model domain (Figure 6.2-5) extends 63 km long-shore, from Owen Anchorage in the north to Tim's Thicket in the south (about 8 km south of Dawesville) and 29 km offshore of the mainland coast to a maximum depth of 44 m.

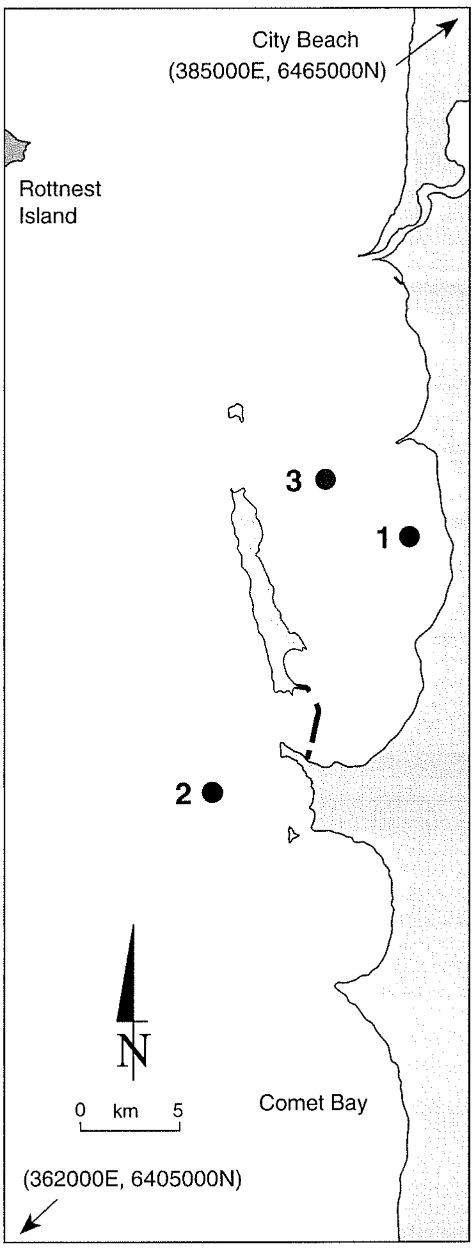


Figure 6.2-4. Domain for the local-scale (northern) hydrodynamic model simulations showing locations of current meter sites.

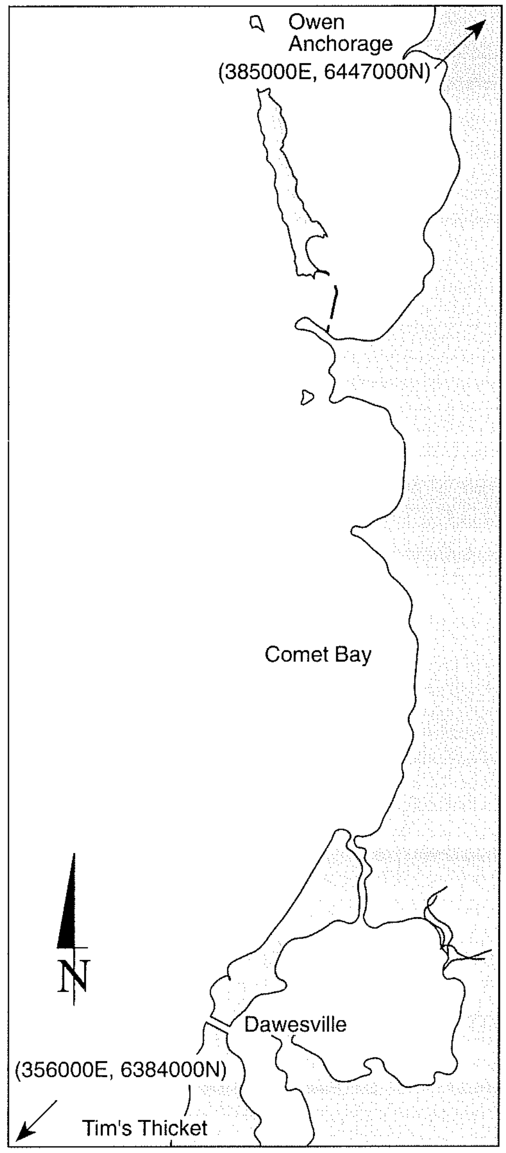


Figure 6.2-5. Domain for the local-scale (southern) hydrodynamic model simulations.

In each case the model is applied to a grid system which divides these domain areas into 500 m square cells. This grid is sufficient to spatially resolve important hydrodynamic boundary features, such as upwelling/downwelling zones and coastally-attached buoyant plumes, which have widths that scale with the baroclinic radius of deformation (typically about 1-3 km for these waters). The coastline and bathymetry of the area are also satisfactorily represented in the main, with a few exceptions. In particular, it should be noted that the shipping channel through Parmelia and Success banks is not resolved. The cross-sectional flow areas of the two bridge openings in the causeway at the southern entrance to Cockburn Sound have been correctly specified within the grid by slightly adjusting the water depth of these openings. Finer grid resolution would have been desirable, however it would have led to impracticably large computing times for each simulation, given the computing resources available. It was deemed equally important for the model to encompass the major bathymetric features in the areas surrounding the coastal embayments, including the islands, reefs and banks, and to remove open boundary effects away from the areas of principal interest. Hence the above balance was struck between spatial resolution and model domain area.

The vertical discretisation used for the 'local-scale' simulations consisted of 12 sigma levels, forming 11 layers, two thinner ones near the water surface, and near the sea-bed, and the remaining seven layers of equal thickness. Implemented in this way, the model was used to investigate the three-dimensional barotropic and baroclinic response of the region to a range of external forcings, including surface wind stress, regional sea level gradient, buoyancy inputs from rivers and estuaries, and the mean seasonal density differences between shelf and embayment waters.

6.2.4.4 *Model validation*

Two approaches have been used to validate the model. For barotropic simulations the model performance has been assessed on the basis of comparisons between time-series of measured currents from meters deployed in Cockburn Sound and in Sepia Depression (Figure 6.2-4) and time-series of simulated currents from the corresponding model grid cells. While such point-wise comparisons can be readily made and quantified in statistical terms, it is economically feasible to deploy current meters at only a few points throughout the modelled area. Baroclinic simulations calculate the spatial distributions of seawater properties such as salinity, and the changes in these distributions which arise from the movement and mixing of the nearshore coastal waters. The broad characteristics and temporal changes in the measured and simulated spatial salinity fields have been compared in these cases as a means of assessing the simulation ability of the model. Such comparisons have been made wherever possible, and are documented throughout this chapter.

The model validation exercise described in this section was carried out on the northern 'local-scale' (500 m grid) model domain which extended long-shore from City Beach in the north to mid Comet Bay in the south, and offshore to the approximate location of the 35 m depth contour. The three-dimensional barotropic circulation of Cockburn Sound and surrounding waters was modelled for the period 7-29 March, 1992, using real wind data and an estimate of the mean long-shelf pressure gradient, derived from analysis of wind and current meter records.

D'Adamo and Mills (1995a) reported that during much of the simulation period the horizontal density differential between Cockburn Sound waters and adjacent shelf waters was less than 0.1 kg m^{-3} and that the vertical density difference in Cockburn Sound was less than about 0.1 kg m^{-3} (in 20 m) with regular vertical mixing. On the basis of these data they concluded that the application of a three-dimensional barotropic model would provide a reasonable approximation of the hydrodynamic behaviour for this period.

Time-series and progressive vector runs of wind data from the DEP Naval Base station (on the mainland coast of Cockburn Sound) and from the Bureau of Meteorology station on Rottnest Island are shown in Figure 6.2-6 for the simulation period. The progressive vector runs for these time-series show periods of southerly, easterly and northwesterly winds, and highlight the presence of cross-shore horizontal shear in the wind field, particularly in the north-south wind component.

Using inshore wind forcing, it was found that currents in eastern Cockburn Sound could be satisfactorily simulated, however currents in Sepia Depression were less well simulated. With only offshore wind forcing, the currents in Cockburn Sound were over-estimated by the model. Consequently, the wind stress field used to validate the model was derived from a linear interpolation in the cross-shore direction of the computed hourly wind stresses from these two wind stations.

A linear regression analysis was performed between long-shelf wind stress and long-shelf current time-series (collected at site 2 in Sepia Depression, shown in Figure 6.2-4) for the period 11 March to 26 April 1992. Following Scott and Csanady (1976) and Steedman and Associates (1981), the results of this analysis were used to infer a long-shelf sea level slope of about -1×10^{-7} for this period and this value was used to guide the specification of an imposed pressure gradient forcing in the validation run of the hydrodynamic model.

Modelled current data from grid cells corresponding to the current meter locations (Figure 6.2-4) were extracted to form hourly time-series. The measured and modelled current time-series data for the simulation period are shown in Figure 6.2-7 for the sites in eastern Cockburn Sound and Sepia Depression. Table 6.2-1 shows the mean error and the root mean square error for the simulated northward and eastward current components, respectively.

The model satisfactorily hindcast the measured currents from the mooring sites in eastern Cockburn Sound and in Sepia Depression. Agreement between model results and current meter measurements for the northern Cockburn Sound current meter site near the Parmelia Bank and the shipping channel was less favourable. This is probably due to the inability of the model grid to resolve the shipping channel, and to the occasional slumping of a northward moving coastal plume of slightly elevated salinity into the northern end of the Cockburn Sound basin (D'Adamo and Mills, 1995a).

As mentioned above, the model was also used in baroclinic mode to simulate the transport of estuarine plumes and the

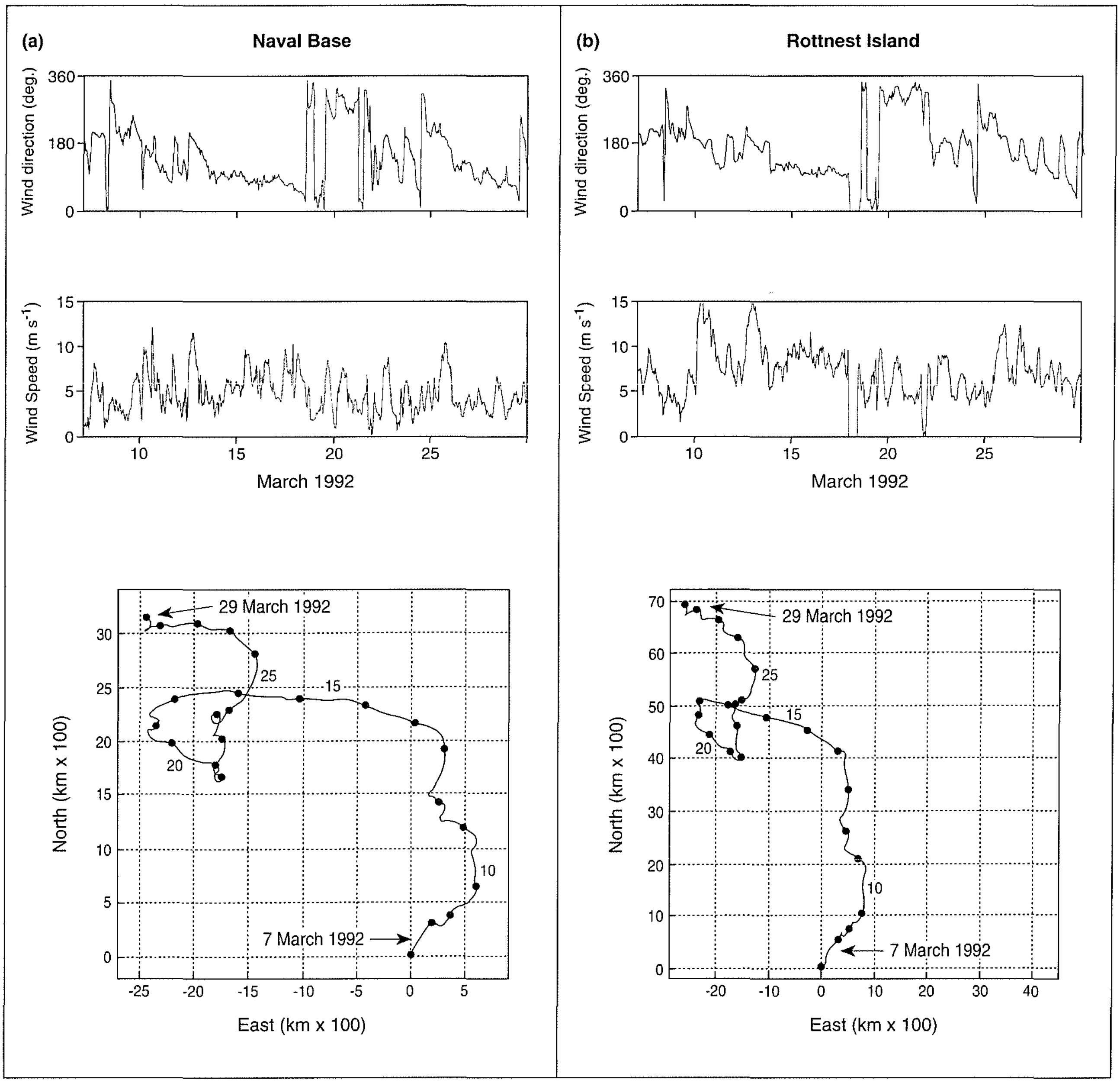


Figure 6.2-6. Wind data input from (a) Naval Base and (b) Rottnest Island for the barotropic model validation. The wind data are presented as speed and direction time series and progressive vector runs.

flushing of stratified embayments in winter and autumn and, wherever possible, comparisons are made between simulated and observed water salinity and density structures to confirm that the model is reproducing the main features of the measured hydrodynamic response of these waters.

6.2.4.5 *Investigation of the Swan-Canning Estuary outflow into marine waters*

From about June to September each year strong riverine flows enter the Swan-Canning Estuary and low salinity water is discharged from the estuary to the coastal waters, forming buoyant surface plumes whose movement is strongly influenced by the wind (D'Adamo *et al.* 1995b).
It is important to understand the direction, rate and extent of transport of these plumes, since they carry loads of nutrients, fine particulates and other catchment-derived contaminants into the coastal waters and influence the hydrodynamics of the coastal zone.

A series of three-dimensional, baroclinic model simulations was conducted to investigate wind-driven, far-field transport of the buoyant Swan-Canning Estuary plume. Different wind conditions were used for each simulation, but the estuarine discharge rate, based on a freshwater flow of 60 $m^3 s^{-1}$, was held constant across all simulations. The method of Lazure and Salomon (1991) was used to model the discharge of estuarine water. The simulations were initialised with no stratification in the receiving marine waters, however a stratification field developed in the course of the simulations as a result of the influx of low salinity water. The model results for each wind condition are presented in Figure 6.2-8 and

Table 6.2-1. Mean error and root mean square (Rms) error for the simulated north-south and east-west components of the currents for the three-dimensional barotropic model validation.

Station	North-south component		East-west component	
	Mean error (m s^{-1})	Rms error (m s^{-1})	Mean error (m s^{-1})	Rms error (m s^{-1})
1 (4.8/8.6)	0.03	0.04	- 0.01	0.02
1 (2.3/8.6)	0.01	0.02	0.00	0.02
2 (15.0/22)	ND	ND	ND	ND
2 (3.0/22)	0.06	0.11	0.04	0.05

(4.8/8.6) - current meter located 4.8 m above sea bed in a total water depth of 8.6 m
ND - Intermittent current meter data only

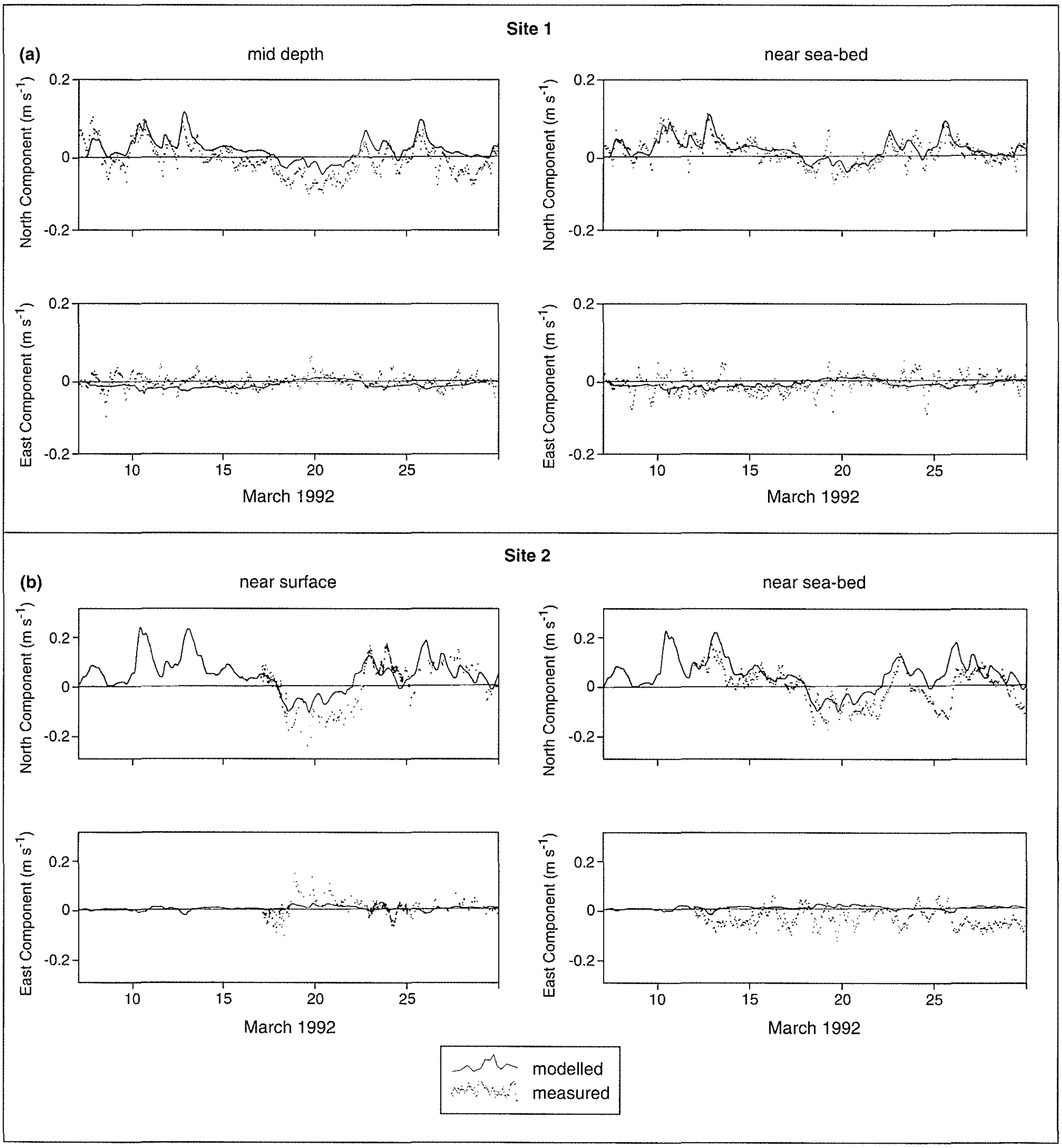


Figure 6.2-7. Comparison at (a) site 1 and (b) site 2 between measured and modelled currents from the barotropic model validation, driven by winds in Figure 6.2-6.

discussed in the following sections. Further details may be found in Mills and D'Adamo (1995b).

Past field observations (reviewed in Hearn (1991) and D'Adamo (1992)) indicate that this plume could extend into Owen Anchorage, Cockburn Sound, Sepia Depression and Warnbro Sound. The simulation results provide a systematic understanding of the influence of wind on plume transport and show that the model is able to reproduce major transport pathways of the plume.

Transport of Swan-Canning Estuary water into Cockburn Sound

The modelling predicts that northeasterly, northerly and northwesterly winds transport Swan-Canning estuarine water into Cockburn Sound as a surface buoyant plume, and that this occurs typically within 1.5 days. Under northeasterly wind, low salinity water enters Cockburn Sound across the whole width of its northern entrance, and within Cockburn Sound the buoyant plume tends to the eastern side of the deep basin (Figure 6.2-8a). A vertical salinity section across the sound (Figure 6.2-9a) confirms that the southward-moving low-salinity plume is deepest over the eastern side of the basin. The northeasterly wind also forces the buoyant plume through Challenger Passage into Sepia Depression, where it is driven southward next to the west coast of Garden Island. The earth's rotation plays a significant role in these dynamics, deviating water to the left of the wind direction.

Under northerly and northwesterly winds, the discharged estuarine water is transported southward as a deep narrow plume attached to the mainland coast (Figures 6.2-8b and c, respectively). Once within Cockburn Sound the plume continues to extend southward and its low salinity water inundates the seabed of the eastern margin of the sound, as shown by the vertical salinity structure (Figure 6.2-9b).

Under westerly wind, the estuarine plume is confined to within 3-5 km of the coastline (Figure 6.2-8d). Although the plume extends both to the north and south of the point of discharge, low salinity water accumulates preferentially to the south, in the embayment between Fremantle and Woodman Point. The plume reaches the northeast corner of Cockburn Sound in about 1.5 days. The westerly wind forces downwelling of buoyant plume water against the mainland coast and induces a steeply-inclined front between the plume and adjacent ambient water.

Northward transport of Swan-Canning Estuary water

Under southwesterly winds the plume is driven northward in contact with the mainland coast (Figure 6.2-8e). The plume has a strong frontal boundary to the south and the west. Cross-sections of salinity show strong vertical gradients near the base of the plume at depths of less than 10 m.

Under southerly winds, the plume is driven to the north-northwest (Figure 6.2-8f) which is to the left of the wind stress direction due to deviation by the Coriolis force. The plume broadens with distance from the source and is essentially detached from the seabed in water depths greater than about 8 m.

Offshore transport of Swan-Canning Estuary water

A southeasterly wind can drive buoyant plume water directly offshore, in a westerly direction (Figure 6.2-8g). The plume has a strong surface front on the southern side. A vertical section of salinity along the axis of the plume (Figure 6.2-9c) indicates that it is a buoyant surface feature, mainly detached from the seabed. However the plume undergoes significant local deepening on the upstream side of banks and reefs, exposing them to lower salinity water to depths of about 15 m. The hydraulic conditions for deepening of buoyant plumes in the presence of banks and sills has been further investigated by Chao and Paluszkiewicz (1991).

Transport of Swan-Canning Estuary water to Sepia Depression and thence southward

Under an easterly wind the estuarine plume extends southwest and reaches Carnac Island and the northern end of Garden Island within one day (Figure 6.2-8h). There is limited entry of buoyant water into northwest Cockburn Sound. In the main, the plume moves into Sepia Depression and is transported southward, with the plume confined to the east of Five Fathom Bank. Cross-shelf vertical sections of salinity show that the southward moving plume is attached to the west coast of Garden Island (Figure 6.2-9d). The vertical salinity structure of the plume comprises tightly spaced isohalines which slope downward from west to east as they approach Garden Island. The deepening of the salinity structure from west to east suggests a coastal downwelling circulation within the plume, and is consistent with a southward driven buoyant coastal flow influenced by rotation (e.g. Chao, 1988).

As discussed above, northeasterly wind drives low salinity water past the northern end of Garden Island and thence southward along Sepia Depression, as well as directly into Cockburn Sound (Figure 6.2-8a).

Transport of Swan-Canning Estuary water under calm conditions

Under calm conditions, the low salinity plume initially spreads radially due to its buoyancy. After about one day, when the plume has developed a radius comparable to the baroclinic radius of deformation (1-3 km), the plume water tends to circulate anticlockwise about the source under the influence of the earth's rotation. To the south of Fremantle, this circulation is blocked by the mainland coast, and the plume water is forced to flow southward as a boundary current (Figure 6.2-8i). It takes 1.5-2 days for plume water to enter the northeast corner of Cockburn Sound, and thereafter it continues to propagate southward along the eastern margin of the sound.

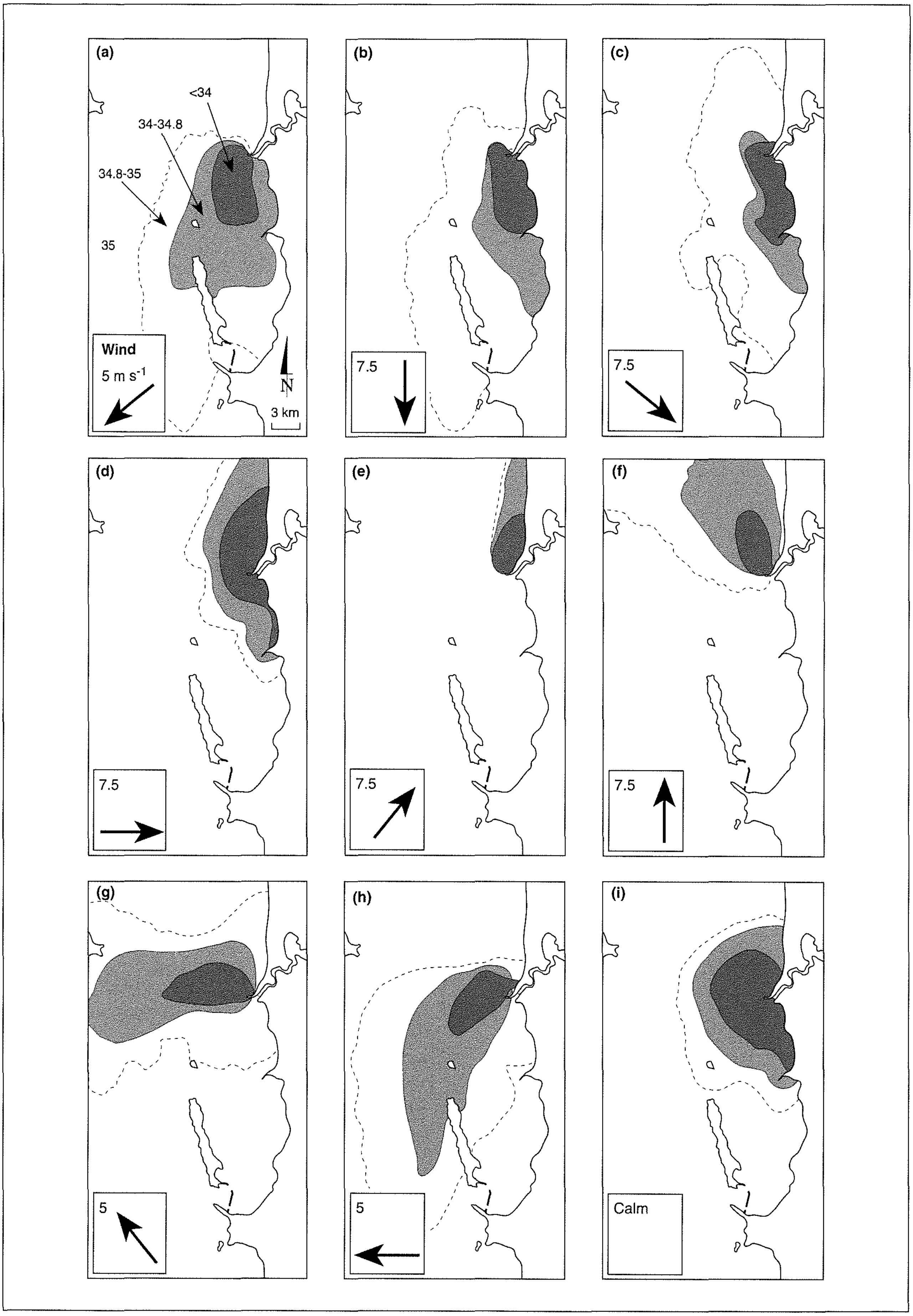


Figure 6.2-8. Baroclinically modelled surface salinity fields representing the transport of Swan-Canning estuarine plumes after 1.5 days under (a to h) eight constant wind conditions and (i) calm conditions.

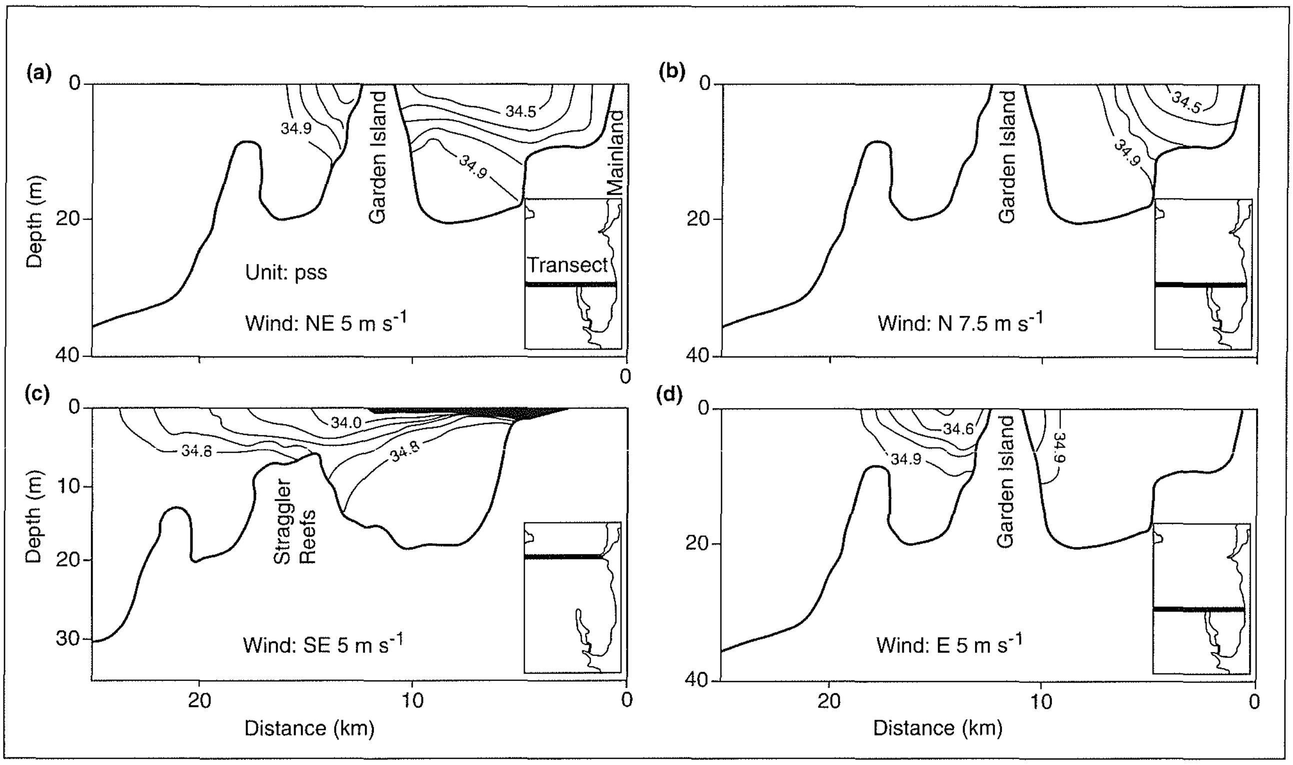


Figure 6.2-9. Baroclinically modelled cross-shelf vertical salinity structures representing the transport of Swan-Canning estuarine plumes after 1.5 days under four (a to d) constant wind conditions.

Transport of Swan-Canning Estuary water under variable wind conditions

The transport of the buoyant Swan-Canning estuary plume was simulated using local wind data for the period 9-18 August 1991 (Figure 6.2-10a and b) during which time the wind veered anticlockwise from southeasterly through to westerly, with strong northwesterly winds from 16-18 August. The simulation proceeded from an initial, specified salinity field (based on field data) with lower salinity water in Cockburn Sound relative to surrounding shelf water. Output from this simulation is presented for comparison with field data collected on 16 August.

By midday of 16 August 1991, that is 7.5 days after start of the simulation, the modelled plume extended west to southwest past the northern tip of Garden Island, and southward along Sepia Depression as far as the entrance to Warnbro Sound (Figure 6.2-11a). This followed a period of easterly to northerly wind during the preceding 30 hours.

Comparisons between model results and field data

Selected comparisons are provided here to illustrate the broad agreement between the modelled and measured transport of buoyant Swan-Canning Estuary outflows.

Water quality measurements conducted during the August 1991 winter survey (see section 4.8.1) found that the diatom *Skeletonema costatum* emanated from the Swan-Canning Estuary and was present in the coastal waters. On 16 August 1991, after preceding easterly to northerly winds, this diatom was found in Cockburn Sound and also extended southward along the Sepia Depression (Figure 6.2-11b). The model predicted a similar behaviour of the buoyant Swan-Canning estuarine discharge under these wind conditions (Figure 6.2-11a).

Further support for these model results is drawn from the early work of Environmental Resources of Australia (1970), where pulses of buoyant Swan-Canning estuarine water (coloured brown by the planktonic riverine diatom *Melosira*) were photographically tracked during winter 1970. Under easterly to northeasterly winds (see Figure 6.2-12a) the algae were observed to enter Cockburn Sound across the full width of its northern entrance, and also to round northern Garden Island and form a thin plume moving southward along Sepia Depression, as predicted by the model (Figures 6.2-8a and h). Under northerly to northwesterly winds (Figure 6.2-12b) the algae were found to move southward around Woodman Point and to proceed along the eastern side of the sound next to the mainland coast, as was also predicted by the model (Figures 6.2-8b and c).

The northward propagation of buoyant estuarine discharge plumes as coastally attached features under southwesterly winds was measured in the field and captured by satellite imagery (Plate 5.1-1e and f, from Wylie *et al.* (1992)) during the SMCWS August 1991 winter intensive survey. This plume behaviour under such winds was as predicted by the model (Figure 6.2-8e).

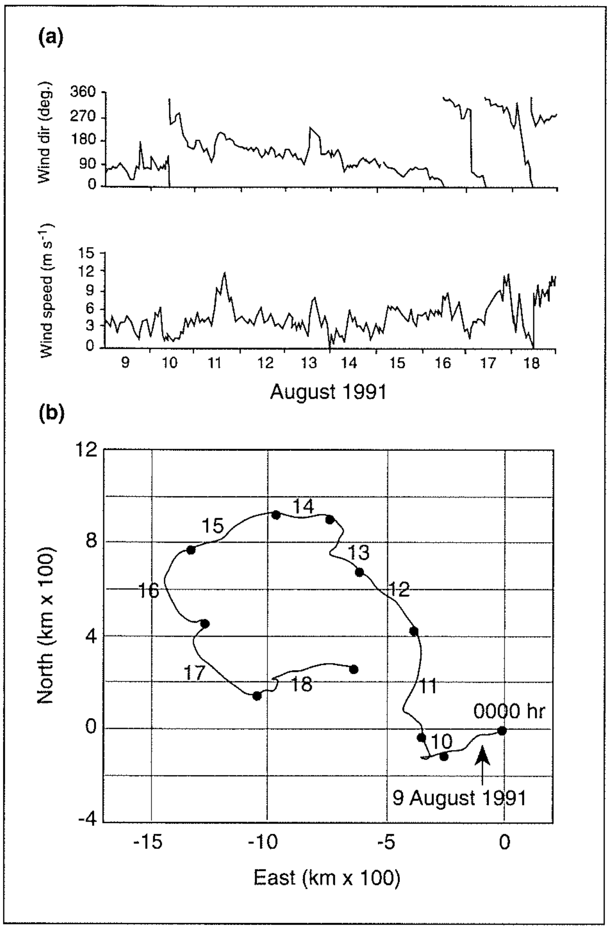


Figure 6.2-10. Wind data input for the baroclinic modelling of the transport of the Swan-Canning estuarine plume under typical pre-storm winter conditions: (a) time series and (b) progressive vector run.

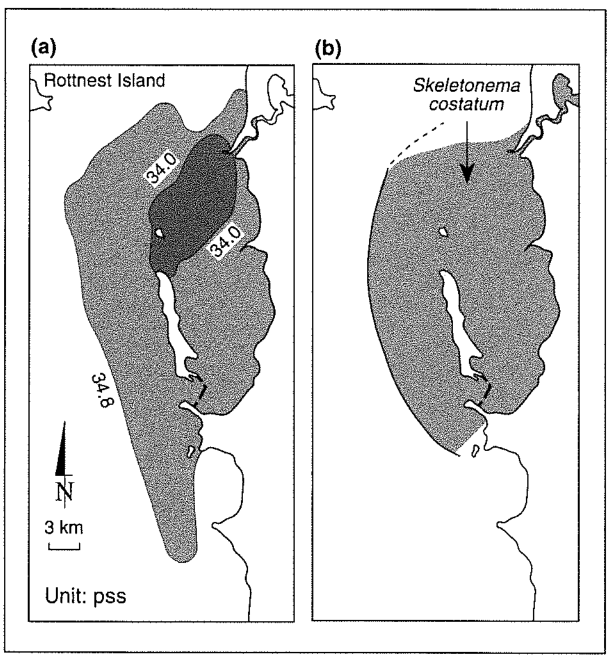


Figure 6.2-11. Comparison between (a) simulated surface salinity and (b) measured surface distribution of the estuarine diatom Skeletonema costatum associated with the Swan-Canning estuarine plume.

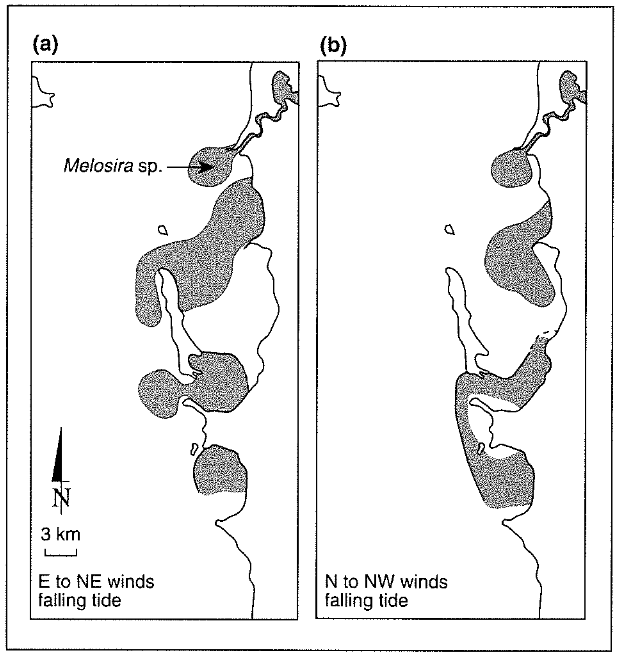


Figure 6.2-12. Observed positions of successive tidal pulses of Swan-Canning estuarine water (coloured brown by the estuarine diatom, Melosira) under (a) east-northeasterly winds and (b) north-northwesterly winds (from Environmental Resources of Australia, 1970).

Conclusions

Buoyant plumes discharged from the Swan-Canning Estuary are driven under most wind conditions toward nearshore areas which are generally biologically productive and sought after for multiple human uses. These areas are therefore exposed to loads of nutrients, phytoplankton and various catchment-derived contaminants contained in the discharges and this may result in environmental changes such as reduced productivity of benthic plants through deterioration of the light climate (see section 5.3.1) and toxic contamination with consequent physiological impairment of biota (see section 4.11). Nitrogen loads from the catchments to the Swan-Canning Estuary have increased by more than 350% since the 1960s (Deeley, in preparation). Maintenance of the environmental quality of Perth's coastal waters is therefore inextricably linked to the management and limitation of contaminant losses from the catchments and waterways of the Swan-Canning Estuary.

The wind drives buoyant surface plumes more rapidly than surrounding surface coastal waters. Within 1-2 days, under favourable wind conditions, water from the Swan-Canning Estuary can reach much of the SMCWS area, including Gage Roads, Owen Anchorage, Cockburn Sound, Sepia Depression, Warnbro Sound and the area about Rottnest Island. This is consistent with the findings from the shelf-scale and local-scale water quality studies (sections 4.7 and 4.8) that, in winter, estuarine outflows are a key factor influencing water quality. The spatial and temporal scales of transport of the Swan-Canning discharge should be taken into account in future environmental monitoring and management plans for Perth's coastal waters.

The plume modelling suggests that the eastern side of Success Bank and the northeastern margin of Owen Anchorage is exposed to nutrients discharged from the Swan-Canning Estuary under a wide range of wind conditions. This high nutrient exposure helps to explain the observed high biomass of epiphytes on seagrass leaves and the decline in the seagrass meadows of these areas (see section 4.2).

The modelling confirms that, under west to southwest wind conditions, the plume extends north of Fremantle next to the shore. The high incidence of the physiological deformity *imposex* in *Thais orbita* (see section 4.11) collected from the Cottesloe reefs, an area where boating activity is low, the link between tributyltin contamination and this deformity, and the direction of the plume under west to southwest winds, together suggest that the tributyltin may be transported to these reefs from the mouth of the Swan-Canning Estuary and the inner port area.

Buoyant plumes tend to deepen and broaden as they approach submarine banks and reefs. The marine biological communities associated with these banks and reefs are therefore exposed to contaminants contained in the plume waters to depths even greater than the normal thickness of the plume.

Episodic winds from the northerly quadrants in winter drive low salinity water discharged from the Swan-Canning Estuary southward. As a result of these incursions of the estuarine plume into Cockburn Sound the salinity of the sound is lowered. This leads to a density difference between Cockburn Sound and the surrounding shelf waters which (as shown by the field data in section 5.1.2 and model results to follow in section 6.2.4.9) is one of the key determinants of the circulation and flushing regimes of the sound in winter. Further investigations of the winter flushing and exchange between Cockburn Sound and its surrounding waters need to take account of the influence of buoyancy inputs to the sound from the Swan-Canning Estuary.

6.2.4.6 *Investigation of the Peel-Harvey Estuary outflow into marine waters*

The salinity of the Peel-Harvey estuary undergoes a marked annual cycle, ranging from low values (< 10 pss) in winter to high values (35-45 pss) in summer (McComb *et al.* 1981; Hodgkin *et al.* 1985; D'Adamo and Lukatelich, 1985). Hence, this estuary discharges buoyant water in winter and dense water in late summer, relative to the receiving coastal waters. The behaviour of both buoyant and dense plumes has therefore been modelled. As pointed out previously, the modelling has thus far been confined to the situation before the construction of the Dawesville Channel, because it was under this situation that the SMCWS field data were collected. Further details may be found in Mills and D'Adamo (1995b).

Transport of the buoyant plume from Mandurah

The three-dimensional baroclinic model was used to simulate the transport of the buoyant plume from the Peel-Harvey estuary under several wind conditions and with an estuarine discharge based on a freshwater flow of 60 $m^3 s^{-1}$. Field studies (D'Adamo *et al.* 1995a, Mills *et al.* 1996) have shown that this plume can extend into Comet Bay, Warnbro Sound, Sepia Depression and offshore, under winds from the southerly quadrants.

For southwesterly winds the model predicted that the plume is confined against the coast and driven northward through Comet Bay, as shown by the simulated surface salinity fields at 0.5, 1 and 1.5 days (Figure 6.2-13). The leading front of the plume extended 12 km into Comet Bay in 0.5 days. It then crossed the reefs and shallows off Becher Point and moved into Sepia Depression and Warnbro Sound. After 1.5 days the plume had reached Penguin Island and plume water extended across much of the surface of Warnbro Sound. In Comet Bay the model predicted that the plume had an offshore width of 4-5 km and a depth of 8-10 m. The model results are in good agreement with observations of the plume under similar winds during the period 20-22 August, 1991, shown in Figure 5.1-11 and discussed further in D'Adamo *et al.* (1995a).

For southerly winds, the model predicted that the plume develops to the north and north-northwest, and separates from the shore line within 4 km of Mandurah (Figure 6.2-14a). The plume deepens as it approaches the shallows and reefs about Becher Point, to the north of Comet Bay, and is forced into Sepia Depression.

Under southeasterly winds the model predicted offshore transport of the buoyant plume (Figure 6.2-14b), as a surface feature, mainly detached from the seabed, but with deepening occurring just upstream of the reef-lines. Satellite imagery (Plate 5.1-1c and d) confirms the offshore extension of the buoyant Mandurah plume under southeasterly wind conditions.

Transport of the dense plume from Mandurah

The three-dimensional, baroclinic model was used to simulate the transport of a negatively buoyant plume originating from discharge of high salinity (40 pss) estuary water at Mandurah. Under summer conditions, the mean daily tidal exchange volume through the Mandurah Channel (before the construction of the Dawesville Channel) was estimated to be 5.5 x 10^6 m^3 (Hodgkin *et al.* 1985) which is equivalent to an average discharge rate of about 60 $m^3 s^{-1}$.

Under constant southeasterly wind (5 m s^{-1}) the model predicted that the hypersaline discharge mixes downwards and spreads across the sea bed as a negatively buoyant plume. The dense plume spreads about 3 km to the north and northwest, where it is constrained by the bathymetry of southern Comet Bay and the Murray Reefs.

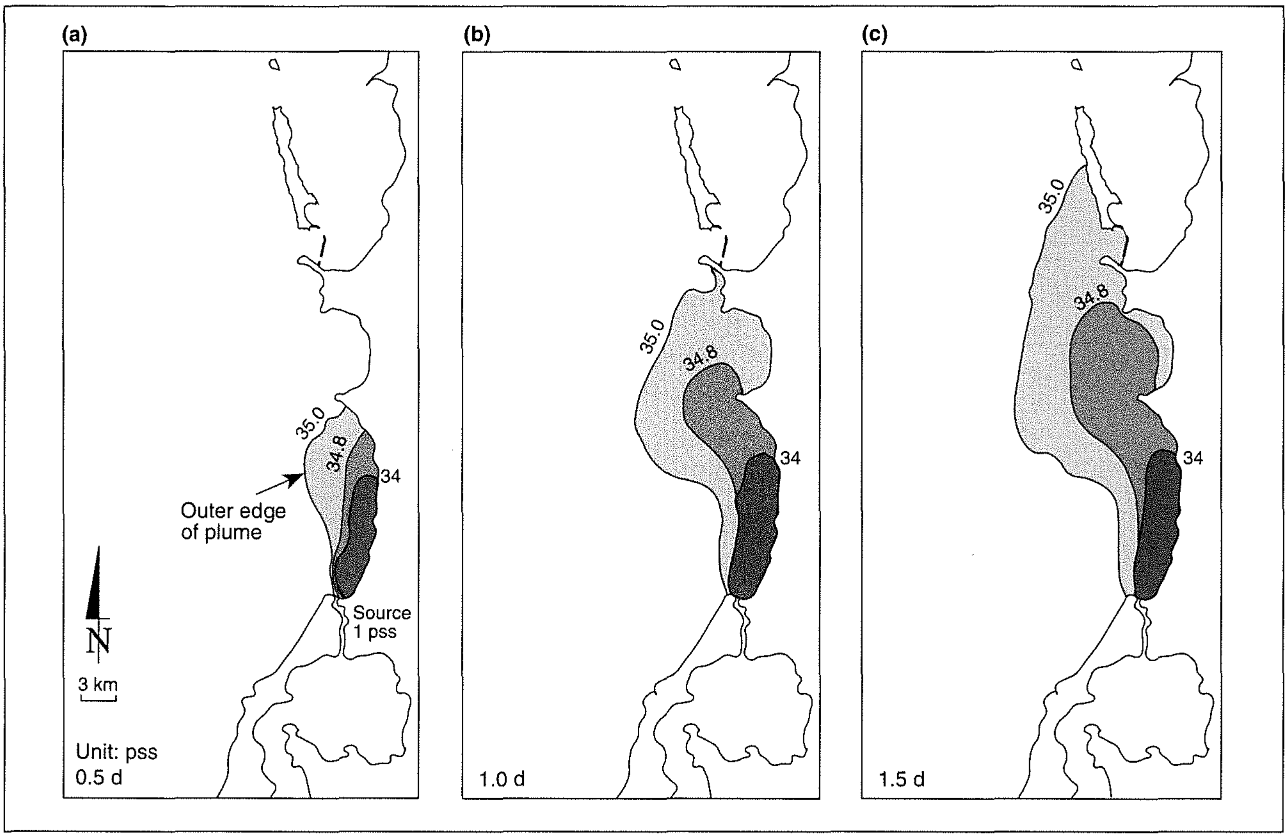


Figure 6.2-13. Baroclinically modelled surface salinity fields representing the transport of a buoyant Peel-Harvey estuarine plume under a southwesterly wind of 7.5 m s^{-1} after (a) 0.5 days (b) 1.0 day and (c) 1.5 days.

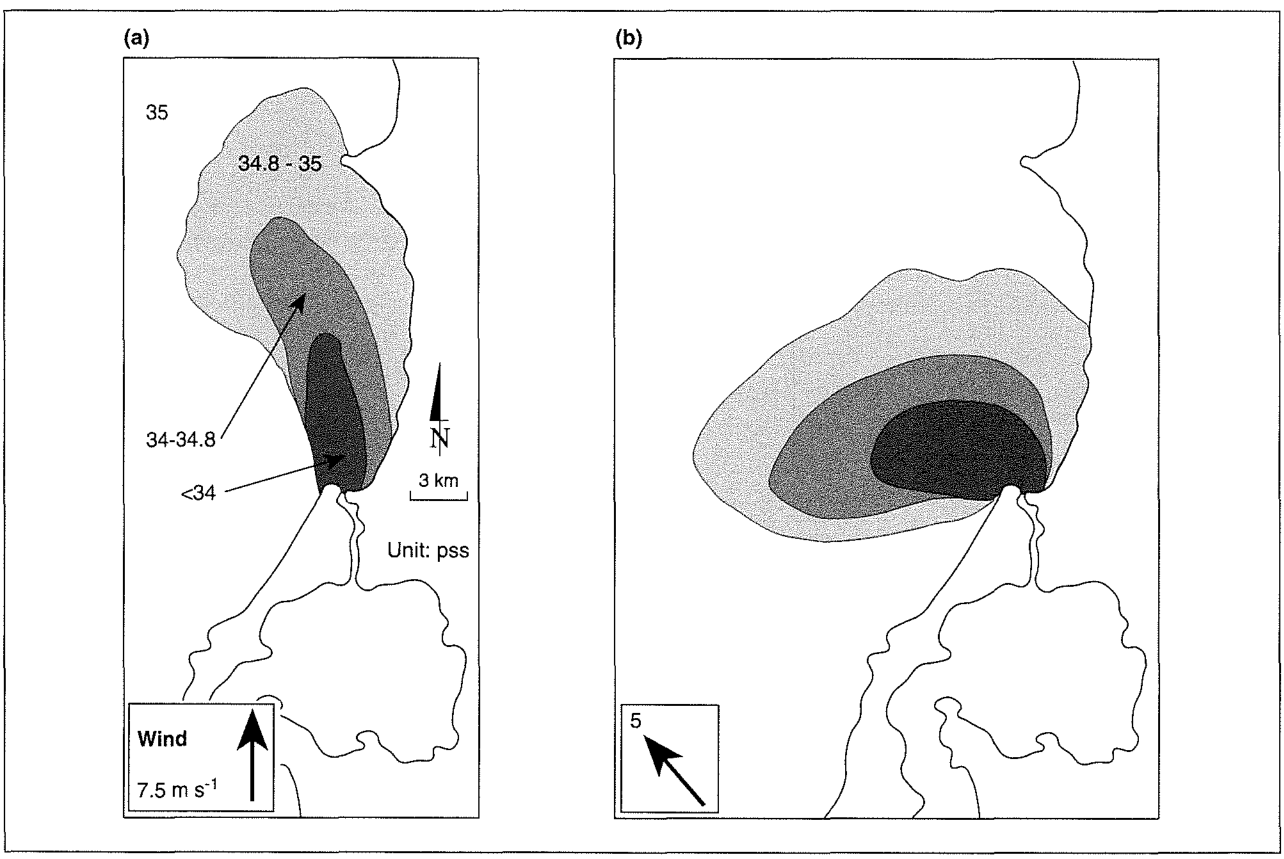


Figure 6.2-14. Baroclinically modelled surface salinity fields representing the transport of buoyant Peel-Harvey estuarine plumes after 0.75 days under (a) a southerly wind of 7.5 m s^{-1} and (b) a southeasterly wind of 5 m s^{-1}.

The plume then veers anticlockwise until it is obstructed by the coastline and is forced to flow southwestward. The dense plume was predicted to spread to the southwest as a coastally-attached plume, and to extend about 12 km southwest of Mandurah after 3.5 days (Figure 6.2-15). By contrast with the behaviour of the buoyant plume which tended to move well offshore under southeasterly winds, the dense plume remained nearshore when forced by the same wind.

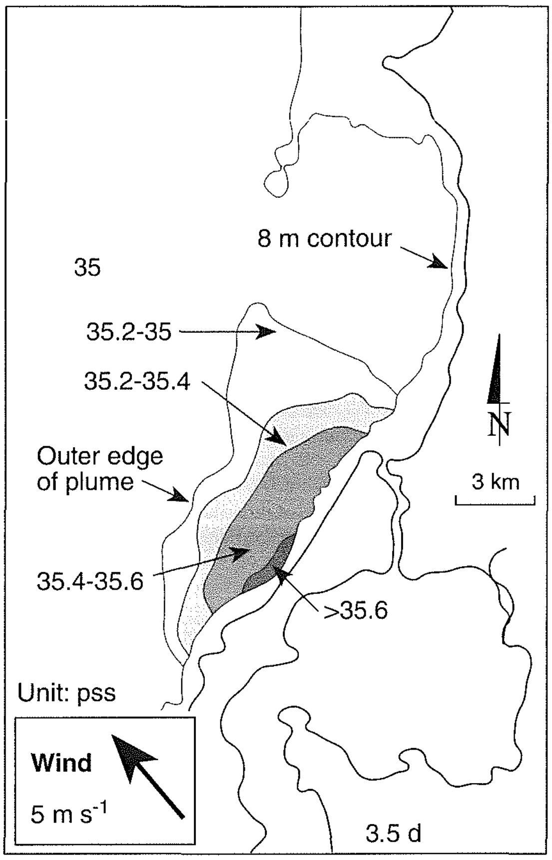


Figure 6.2-15. Baroclinically modelled surface salinity fields at 8 m depth representing the transport of a dense Peel-Harvey estuarine plume after 3.5 days under a southeasterly wind of 5 m s^{-1}. The salinity of the source water was set at 40 pss.

Under constant southwesterly wind (7.5 m s^{-1}) the model predicted similar behaviour for the dense plume, with mixing down to the sea floor, spreading northward into Comet Bay for several kilometres, and then veering in an anticlockwise sense until the presence of the coast deflected the plume toward the southwest.

There is some anecdotal evidence (from before the opening of the Dawesville Channel in 1994) to support this modelled behaviour. There have been observations from Falcon Bay, approximately 7 km southwest of the Mandurah Channel, of a southwestward moving plume of brown coloured water from Mandurah, hugging the coast. These observations were made on 2-3 occasions during mid-late summer under sustained light easterly wind conditions (M J Pannell, personal communication). Although there are no known field studies which can confirm the simulated behaviour of the dense plume, the anecdotal reports and the modelling results can guide future monitoring exercises.

Conclusions

The numerical simulations supported by field data show that buoyant water discharged from the Peel-Harvey Estuary can be driven into Comet Bay, Warnbro Sound and Sepia Depression within 1-2 days under favourable wind conditions. The plume can also be driven off-shore. Discharge from the Peel-Harvey Estuary contains significant loads of nutrients, phytoplankton and catchment-derived contaminants. Since the 1960s, nitrogen loads from rivers flowing into the Peel-Harvey Estuary have increased by 450%, based on the estimates of Deeley *et al.* (in preparation). Shelf-scale water quality investigations identified outflow from the Peel-Harvey Estuary as a major determinant of water quality in winter (see sections 4.7 and 4.8) with widespread increases in nutrient levels up to 5-10 times background values being monitored. Hence catchment management plans designed to limit losses of nutrients and contaminants to the waterways should be motivated not only by the need to rehabilitate the estuary but also by the need to protect the environmental quality of Perth's southern metropolitan coastal waters.

The Shoalwater Islands Marine Park has been established in recognition of the high environmental and social value of this area. It has been shown in this section and in section 5.1.2 that the Peel-Harvey Estuary plume can readily enter the marine park under southwesterly wind conditions which occur frequently throughout the year. Hence management of the environmental quality of the marine park must take into account the role of the Peel-Harvey Estuary plume (and other plumes from external sources) in contributing to the total loads of nutrients and other contaminants entering the marine park.

The behaviour of the Peel-Harvey Estuary plume depends on the density of the discharged water relative to oceanic water. During periods of strong river flow the plume occurs as a low salinity buoyant surface feature. In late summer/autumn the discharge becomes strongly hypersaline and the plume mixes down and spreads across the seabed, being strongly guided by the bathymetry. At intermediate times of the year the plume may have a similar density to surrounding coastal waters. The dispersion patterns of nutrients and other contaminants discharged from the Dawesville Channel require further investigation. The changes in plume behaviour arising from the strong annual cycle of salinity and density in the Peel-Harvey Estuary need to be recognised in future monitoring of the plume and its effects on the marine environment.

6.2.4.7 *Transport of water leaving the southern entrance of Cockburn Sound*

During the winter runoff period the waters of Cockburn Sound are typically less saline and less dense than the surrounding shelf waters, due mainly to the incursion of low salinity plumes from the Swan-Canning Estuary under winds from the northerly quadrants (see sections 5.1 and 6.2.4.5) and their mixing with the resident waters of the sound. Model results for this period indicate that easterly, northeasterly, northerly and northwesterly winds drive Cockburn Sound water out through the causeway bridge openings. Under these wind conditions, water exiting the southern entrance of Cockburn Sound in winter is driven southward as a buoyant plume past Shoalwater Bay and to the entrance of Warnbro Sound and Comet Bay. Figure 6.2-16 (from D'Adamo *et al.* 1995a) shows the results of a model simulation forced by southeasterly to northeasterly winds recorded on 13-15 August, 1991. After about 1.5 days, buoyant water from the southern entrance of Cockburn Sound had been transported to the Warnbro Sound entrance, and after 2.5 days, the plume had entered Comet Bay. This is consistent with the conclusion from temperature-salinity survey data collected during this period (D'Adamo *et al.* 1995a) that low salinity water was transported from southern Cockburn Sound to Warnbro Sound under these conditions.

The model was also run under calm conditions (no wind) starting from an initial density distribution, typical of winter, with Cockburn Sound water being less saline and more buoyant relative to the surrounding waters. This initial

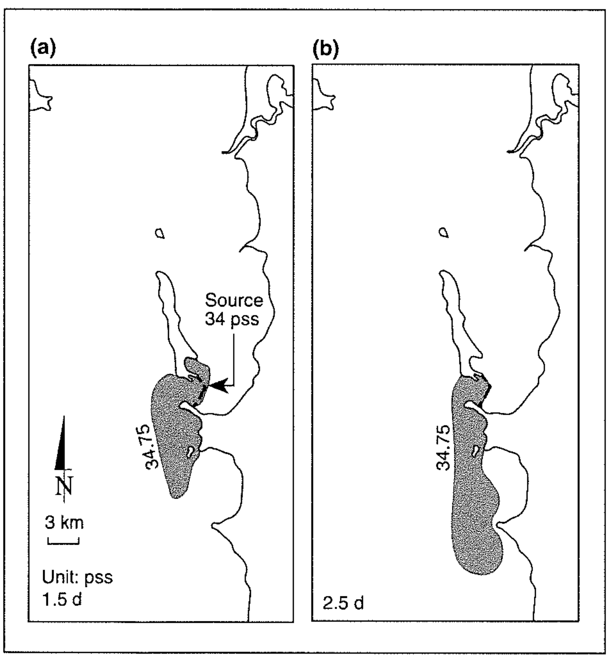


Figure 6.2-16. Baroclinically modelled surface salinity fields representing the southward transport of buoyant water out of the southern opening of Cockburn Sound under southeast to northeast winds (starting at 0000 hrs 13 August 1991, Figure 6.2-10) after (a) 1.5 days and (b) 2.5 days.

density structure is gravitationally unstable and, as shown by the simulated surface salinity contours (Figure 6.2-17), relatively buoyant Cockburn Sound water flows through the southern entrance into Sepia Depression and is then transported southward. Part of the buoyant plume subsequently enters Warnbro Sound.

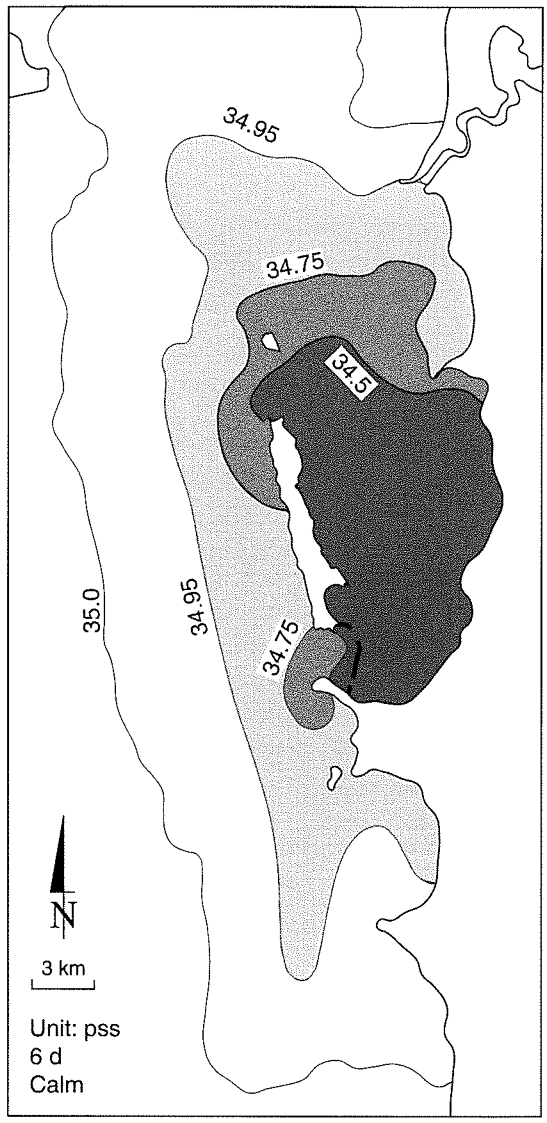


Figure 6.2-17. Baroclinically modelled surface salinity field representing buoyant outflow from Cockburn Sound after 6 days during calm 'winter' conditions.

In autumn, the waters of Cockburn Sound are typically more dense than the surrounding shelf waters (see section 5.1). Model results indicate that easterly, northeasterly, northerly and northwesterly winds tend to drive Cockburn Sound water out through the causeway bridge openings. Water exiting the southern entrance of Cockburn Sound tends to mix down to the sea bed of Sepia Depression and to be driven southward as a dense plume under the above wind conditions.

Conclusions

Under favourable wind conditions, buoyant water can be driven out of the southern entrance of Cockburn Sound and, within 1-2 days, transported to Warnbro Sound and Comet Bay. This result further highlights the interconnectedness between the nearshore embayments. In particular, discharges from the southern entrance of Cockburn Sound are likely to periodically enter the Shoalwater Islands Marine Park. Hence the ongoing environmental monitoring of Cockburn Sound, the marine park, and other interconnected areas should be coordinated.

6.2.4.8 *Investigation of the seasonal hydrodynamic regimes of Cockburn Sound*

The oceanographic characterisation of the SMCWS area identified distinct annual patterns in the water salinity, temperature and density of Cockburn Sound and of the surrounding shelf waters (see section 5.1.1, and D'Adamo and Mills (1995b)). It showed that there is typically a difference in density between these waters, and that this density difference reverses in sign twice as each year progresses (refer to Figure 5.1-2). The field observations also suggested that the density difference between Cockburn Sound and shelf waters is a key factor in determining the nature of the circulation and flushing regimes of the sound. Three major hydrodynamic regimes were described: the 'winter-spring' regime, the 'autumn' regime and the 'summer' regime (see section 5.1.1).

The purpose of this section is to present the results of modelling each of these 'seasonal' hydrodynamic regimes. Further details may be found in Mills and D'Adamo (1995c). The model has been able to reproduce major features of the advection field and the temporal development of the density distribution, as characterised by intensive field data sets. It has therefore been used to gain further understanding of the circulation and exchange patterns, and to estimate the flushing rates of the sound. The modelling has also served to determine the relative importance of various forcings (induced by wind, seasonal density differences between the sound and shelf waters, and the earth's rotation) to the essential nature of these regimes. For reasons of computing economy, initial density differences were introduced to the model (via the equation of state) solely through the prescription of an initial salinity distribution, and water temperature was held constant. These simulations concentrate on understanding the influence on embayment flushing of the spatially-variable seasonal climatology of these waters. At present the simulations do not incorporate the diurnal cycle of surface heat and evaporative fluxes, however these should be included in future work, as field data have shown them to influence stratification and vertical mixing of the sound.

The model simulations were initialised with waters of different salinities outside and within Cockburn Sound. *Flushing rates* were calculated from the simulated change of salt content in pre-defined control volumes of the model. Complete (100%) flushing would have occurred at a time when the salinity of a control volume within the sound became equal to the initial salinity of the external waters. Defined in this way, the flushing generally approaches 100% in an asymptotic manner.

Water exchange fluxes across the boundaries between Cockburn Sound and the surrounding waters were computed for each of the model simulations. The volume inflows and outflows through the northern and through the southern openings of the sound were each separately calculated for each model time step, taking into account the transverse and vertical variations in the water currents across the openings. These volumes were then cumulated over successive time steps for the duration of the simulation. The *cumulative volume flux time* for Cockburn Sound was defined as the time required for the combined gross inflows through both the northern and southern openings to cumulate to a volume equal to the capacity of Cockburn Sound.

The flushing times for Cockburn Sound (e.g. the times required to reduce the initial salt excess/deficit of the sound by a factor of e^{-1}, equal to 0.37) are generally longer than the cumulative volume flux times, because the change in salt content is a result of net salt flux across the entrances and depends on the details of advection and mixing that take place throughout the water body. However, the cumulative volumetric flux times are calculated solely on the basis of gross inflows across the boundaries of the water body, and do not account for the internal circulation and mixing.

Modelling the 'winter-spring' hydrodynamic regime of Cockburn Sound

The waters of Cockburn Sound become less dense than shelf waters during winter, due to wind-driven southward incursions of the low salinity Swan-Canning estuary plume into the sound (as modelled in section 6.2.4.5; see Figure 6.2-8a to c) and subsequent vertical mixing. Warming of the sound waters in spring prolongs this density difference. The period 18-23 August 1991 was chosen for simulation of the 'winter-spring' hydrodynamic regime for two reasons. Firstly, the model results could be compared with an intensive field data set (D'Adamo *et al.* 1995b) which had been collected during this period. Secondly, complete vertical mixing of the sound by storm winds on 18-19 August 1991 (Figure 6.2-18) justified the use of an initial model salinity distribution which was vertically uniform, with less saline water in the sound and more saline water external to the sound (Figure 6.2-19). This salinity distribution gave rise to corresponding density differences in the model which affected the dynamics of circulation, exchange and vertical mixing. Furthermore, salinity was used in the model to determine the rates of flushing (as defined above) between shelf and sound waters, and to trace the re-establishment of vertical stratification in the sound. The model was accordingly initialised and was forced by local wind data for this period and by a low salinity (buoyant) source at the mouth of the Swan-Canning Estuary.

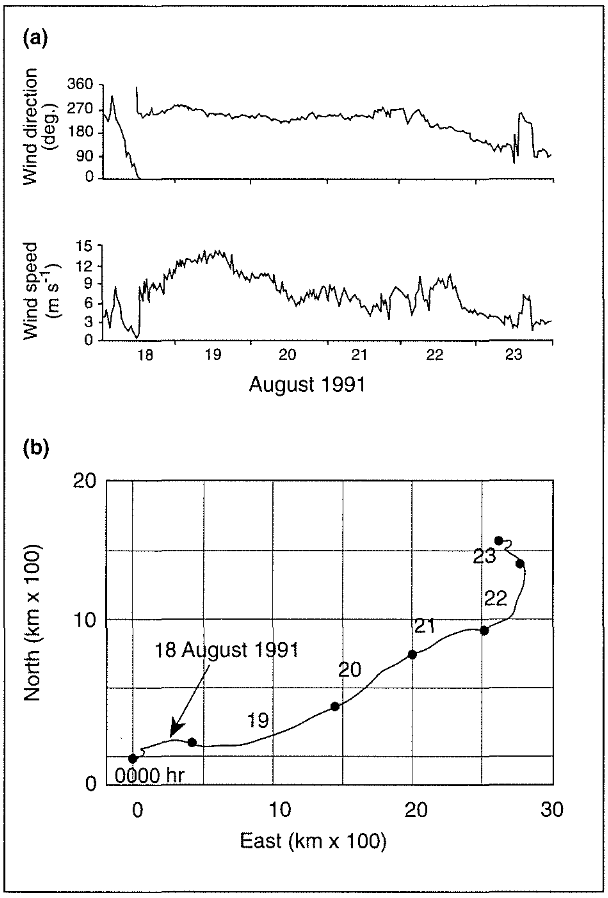


Figure 6.2-18. Wind data used for the baroclinic model simulation of wind-driven flushing and deep-water renewal in Cockburn Sound under post-storm 'winter' conditions: (a) time series and (b) progressive vector run.

As simulated by the model, the sound remained vertically well-mixed during the storm (west-southwesterly winds of 10 m s^{-1} or more) with strong horizontal salinity gradients to the north and south (Figure 6.2-20a). Vertical stratification was re-established (Figure 6.2-20b) during the following 36 hours of southwesterly winds (5-10 m s^{-1}) by the entry of relatively dense, high salinity water into Cockburn Sound through its northern and southern openings, the subsidence of this water and its horizontal transport across the deep basin. Incoming high salinity water from the north penetrated as a bottom layer to within a few kilometres of the Southern Flats before encountering high salinity inflow from the southern causeway entrances. The high salinity water both uplifted and partially mixed with the sound's lower salinity waters, a portion of which was forced out of the sound as wind-driven surface flows. Under winds (5-11 m s^{-1}) swinging to the south during the following 36 hours, the simulation (Figure 6.2-20c) predicted further dense water inflow and the renewal of deep basin water, as indicated by the presence of bottom water with salinity greater than 34.5 pss salinity throughout the length of the basin.

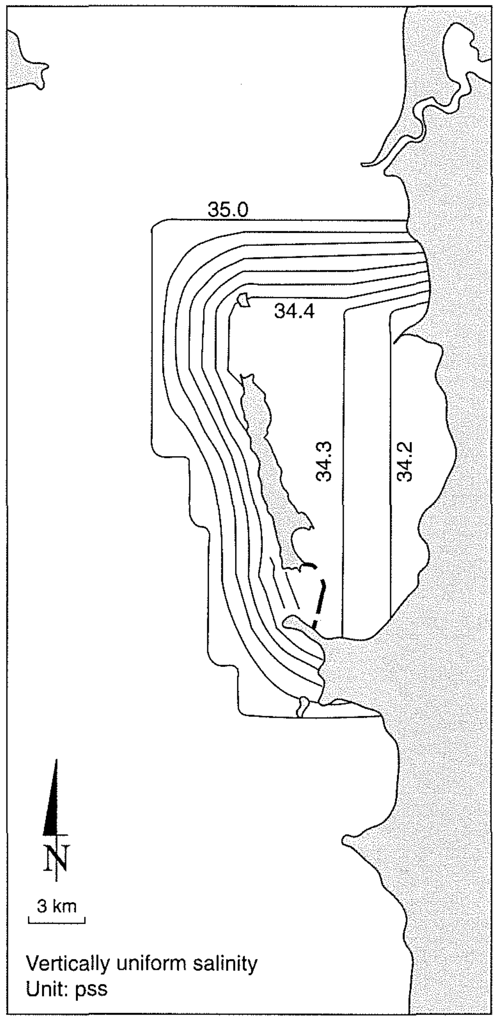


Figure 6.2-19. Starting horizontal salinity field for the baroclinic model simulations of the 'winter' hydrodynamics of Cockburn Sound.

As shown in Figure 6.2-21 the deep basin zone (> 15 m below sea surface) of Cockburn Sound was flushed more rapidly than the near-surface zone (0-5 m below sea surface) of the sound due to the subsidence of dense incoming water and its transport across the deep basin. The deep basin zone was 50% flushed and the near-surface zone was 35% flushed within six days, under the wind forcing conditions and the initial salinity/density difference conditions applied to this simulation (Figures 6.2-18 and 6.2-19, respectively).

During the period of the simulation (westerly to southerly winds) essentially all of the volume output was across the northern opening and, of the inputs, 66% was across the northern opening and 34% across the southern opening. However the magnitude of southward flux across the east-west section from James Point to Garden Island was less than 40% of that across the northern entrance of the sound.

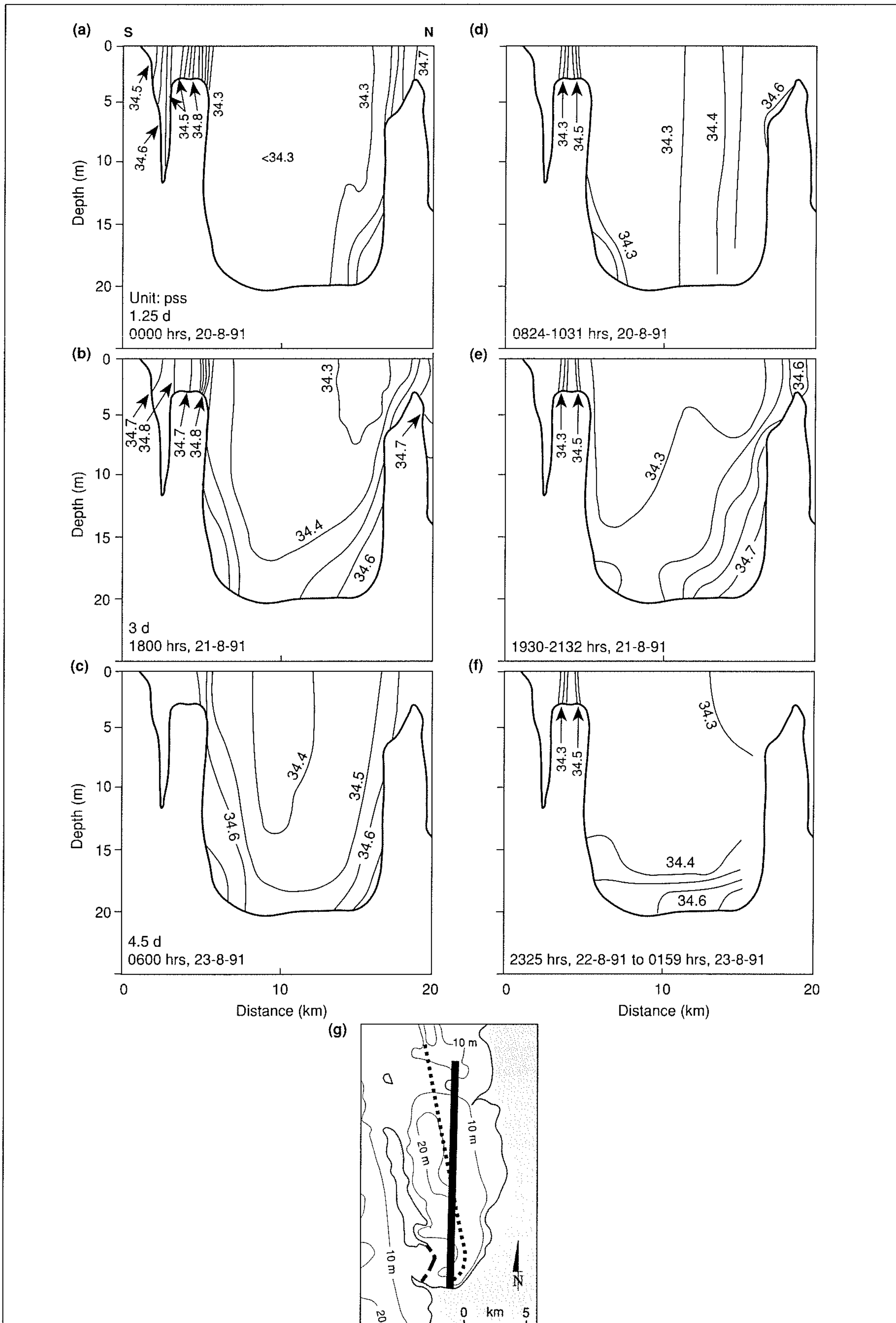


Figure 6.2-20. South-north vertical salinity sections showing deep-water renewal in Cockburn Sound during the post-storm period of 20-23 August 1991 from (a-c) baroclinic model simulations and (d-f) field measurements (wind data as in Figure 6.2-18). Measured salinity was projected onto the model transect path (g).

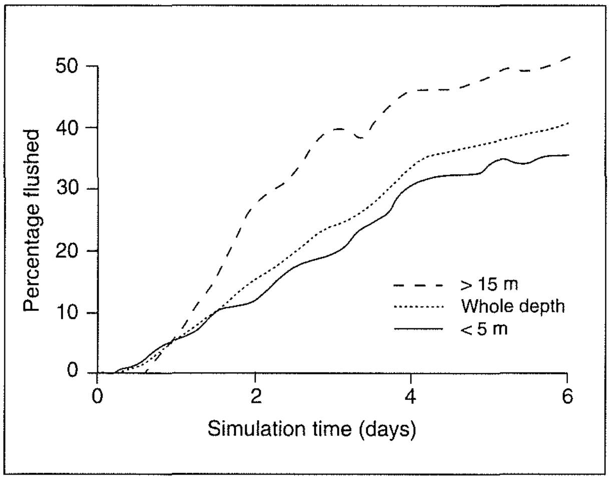


Figure 6.2-21. Baroclinically modelled flushing rates for three depth zones (whole depth, < 5 m and > 15 m) of Cockburn Sound from a 'winter' (18-23 August 1991) simulation under wind conditions shown in Figure 6.2-18.

A schematic of these volume fluxes is given in Figure 6.2-22, which suggests that, in the southern portion of Cockburn Sound, the volume throughflow via the southern opening and the two-way volume exchange across the section west of James Point were of comparable magnitude. In the northern portion of Cockburn Sound the two-way volume exchange across the northern opening was significantly greater than either throughflow due to the flux through the southern opening of the sound, or the two-way volume exchange across the James Point section. The two-way exchange across the northern opening was about twice the magnitude of the throughflow from the southern opening under the conditions of this simulation.

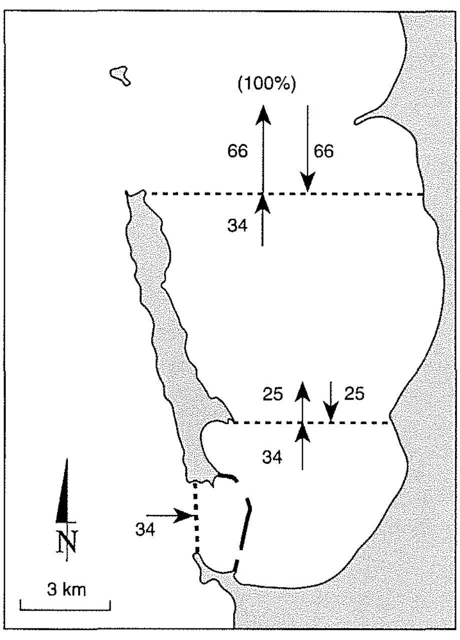


Figure 6.2-22. Schematic of modelled volume fluxes across the openings and an inner west-east section of Cockburn Sound from a 'winter' (18-23 August 1991) baroclinic simulation under winds shown in Figure 6.2-18. Volume fluxes are expressed as a percentage of total outflow.

The initial 'winter' density structure is gravitationally unstable, because buoyant basin water is located beside denser shelf water. Modelling (Mills and D'Adamo, 1995c) has shown that, in the absence of other forcings, this density structure undergoes baroclinic adjustment, influenced by the earth's rotation, with denser shelf water slumping into the sound and more buoyant water overflowing out of the sound (Figure 6.2-23). The buoyant water forms coastally-attached surface plumes which extend southward along the western coasts of Garden Island and Shoalwater Bay, with some buoyant water entering Warnbro Sound (Figure 6.2-23a). The denser shelf water first slumps into the northwest corner of the sound's deep basin, moves to the east and then spreads southward (Figure 6.2-23b and c). It took about six days for the dense inflow to travel the length of the Cockburn Sound basin (Figure 6.2-23c). The cumulative volume flux time for Cockburn Sound for this purely baroclinic (with rotation) simulation was about 12 days, which is about 50% longer than for the corresponding simulation without rotational effects.

The cumulative volume flux time for pure baroclinic adjustment (with rotation) is much smaller than that due to tidal exchange (about 30 days) which was cited by Maritime Works Branch (1977b) as the base level of exchange for the sound. The simulations have shown that the gravitational adjustment process involves the transport of external water throughout the length of the sound in about six days, so that basin-scale mixing is more efficient than for the tidal mechanism, which has a typical advective length scale of about 500 m, much less than the length of the sound.

The cumulative volume flux time for pure baroclinic adjustment (with rotation) in Cockburn Sound was 12 days, about three times longer than for the simulation with wind forcing also included. This suggests that exchange due to baroclinic adjustment alone is significant, but considerably less than exchange due to the combined effects of density and wind forcing for wind speeds in excess of about 5 m s^{-1}. However, as has been shown in Figure 6.2-20 and 6.2-21, the presence of the density difference between Cockburn Sound and the surrounding waters can strongly influence the nature of the wind-forced circulation of the sound, resulting in differential flushing rates with depth.

Modelling the 'autumn' hydrodynamic regime of Cockburn Sound

The model was used to investigate exchange processes between Cockburn Sound and surrounding shelf waters during the 'autumn' regime (Mills and D'Adamo, 1995c). The period 29 April to 6 May 1994 was simulated in view of the availability of detailed water structure data for 3 May 1994 and to complement the understanding of circulation, mixing and exchange processes drawn from autumn field data (D'Adamo and Mills, 1995c).

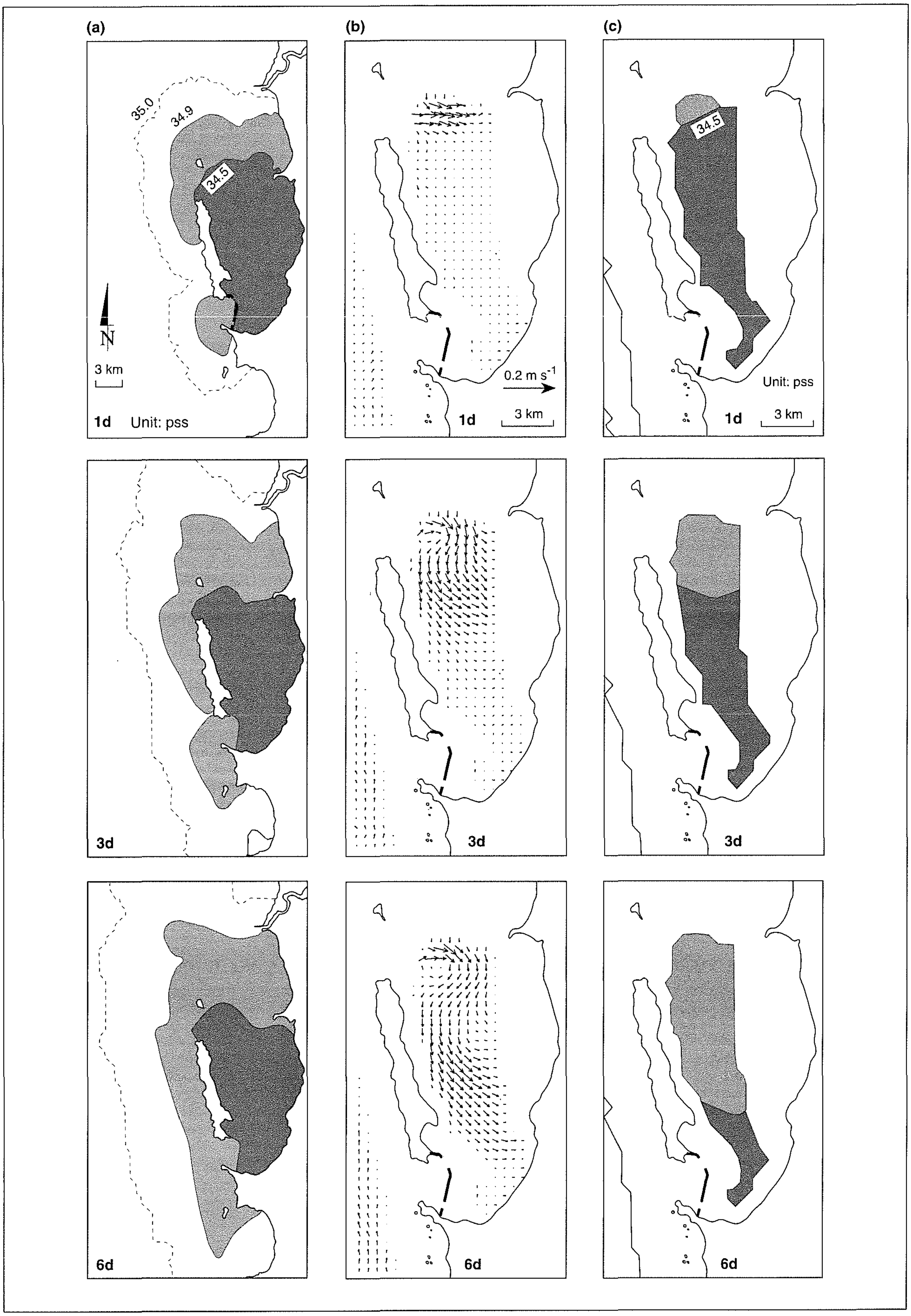


Figure 6.2-23. Baroclinic model simulation of density-induced exchange (with no wind forcing) between buoyant Cockburn Sound water (set at 34.4 pss) and dense shelf water (set at 35.0 pss) after one, three and six days, showing (a) surface salinity, (b) bottom velocity and (c) bottom salinity.

The initial salinity field supplied to the model consisted of two vertically uniform water masses, one with 36 pss salinity, occupying Cockburn Sound, and the other with 35 pss, occupying the remainder of the model domain (Figure 6.2-24). This gave an initial density difference of about 0.75 kg m^{-3} between the sound and shelf waters. While this is a typical value for the 'autumn' density difference between the sound and the shelf (D'Adamo and Mills, 1995c) there were no field data available during the week preceding 3 May 1994 to provide further guidance in the setting of the initial density field and, in particular, its vertical structure. Hence the model and field results could be compared only in the most general terms. The model was forced by the density field and with local wind data (Figure 6.2-25) for the period.

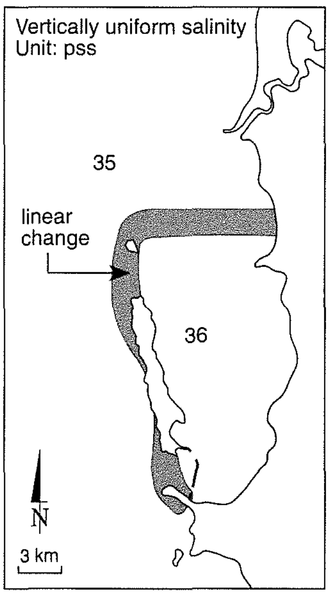


Figure 6.2-24. Starting horizontal salinity field for the baroclinic model simulations of the 'autumn' hydrodynamics of Cockburn Sound.

Figures 6.2-26a, b and c, and 6.2-27a, b and c show near-surface distributions of salinity and currents, respectively, at selected times during the simulation. Figures 6.2-28a, b and c present the corresponding vertical salinity structures along a south to north transect line through Cockburn Sound.

The simulation indicates that relatively low salinity shelf waters were introduced into Cockburn Sound and partially mixed, forming buoyant stratified layers (5-10 m thick) which tended to over-ride resident deep basin water (Figure 6.2-28), shelter it from wind mixing and isolate it from the major openings of the sound. Wind-driven advection of the buoyant layers was significant (Figure 6.2-27) and the model results suggest that it took about 3-4 days for the central surface waters of the basin to be influenced by shelf water (Figure 6.2-26). The modelled salinity field varied much more rapidly near the surface than in the deep basin of the sound.

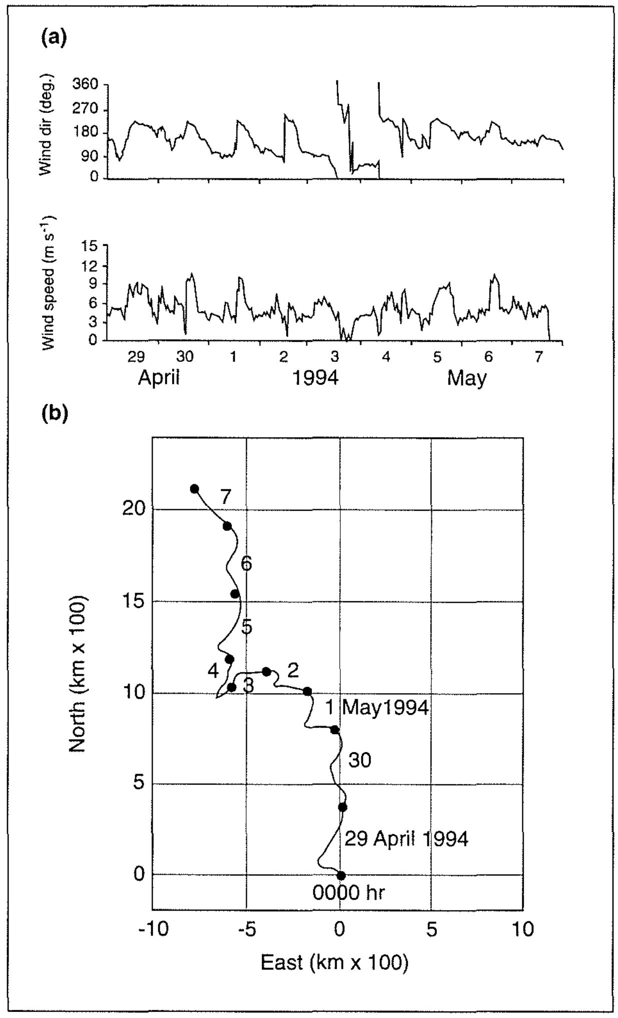


Figure 6.2-25. Wind data used for the baroclinic model simulation of exchange between Cockburn Sound and shelf waters under 'autumn' conditions: (a) time series and (b) progressive vector run.

A 30 hour period of predominantly easterly winds preceded the salinity-temperature-density field measurements of 3 May 1994. The model results, 4.5 days into the simulation, indicate that these winds drove surface inflow to the sound via the northern opening and outflow under the causeway bridges (Figure 6.2-27a) and that there was also sub-surface outflow through Challenger Passage. Relatively low salinity water was transported southward to central Cockburn Sound (Figure 6.2-26a) as a surface layer which was separated from deep basin waters by a mid-depth zone of vertical stratification (Figure 6.2-28a). In contrast, the salinity contours at the southern end of the basin downwelled and the vertical stratification was reduced (Figure 6.2-28a). Measurements of vertical salinity structure along a south-north transect of Cockburn Sound were made on 3 May 1994. The measured salinity structure (Figure 5.1-13) shows an inclined upper layer of relatively low salinity water in the northern half of the Cockburn Sound basin which does suggest that there had been recent entry of relatively low salinity, buoyant water across the northern opening of the sound under predominantly easterly winds, as predicted by the model.

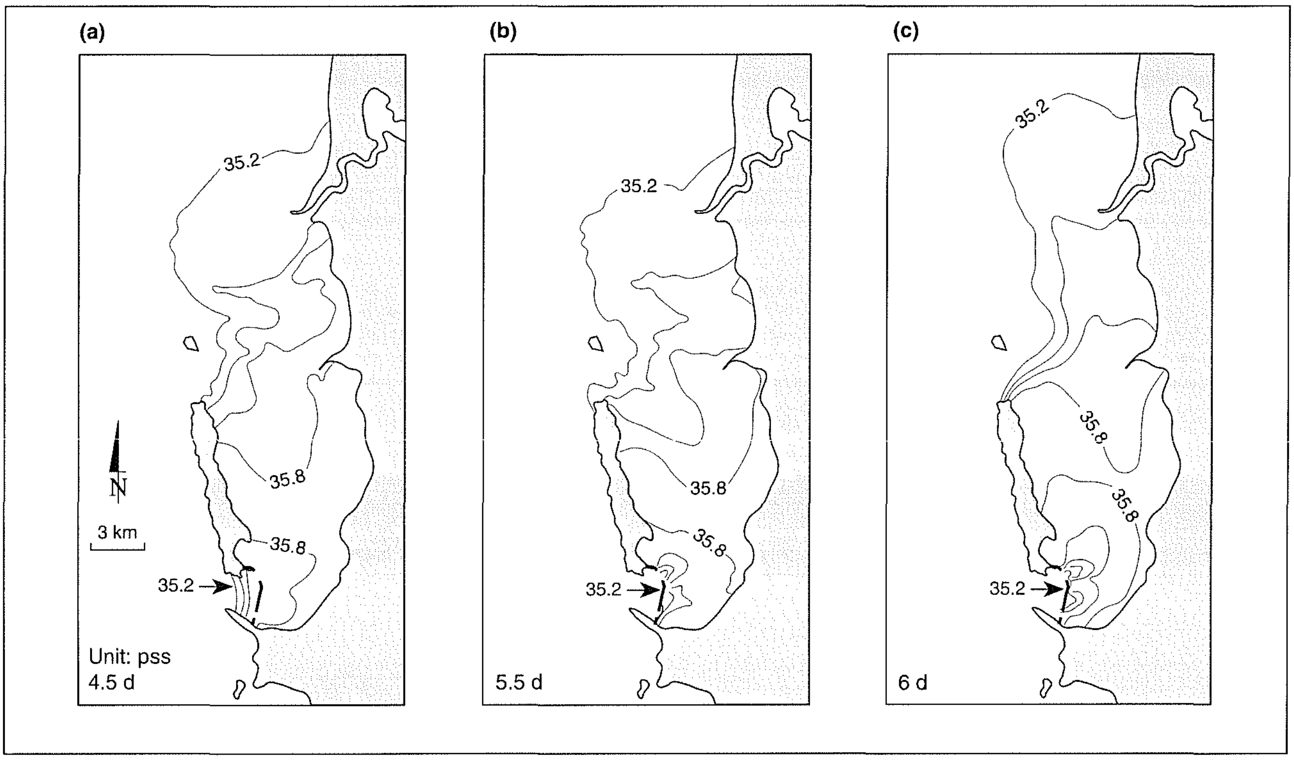


Figure 6.2-26. Baroclinically modelled surface salinity fields representing 'autumn' exchange between dense Cockburn Sound water and buoyant shelf water forced by winds (from Figure 6.2-25) after (a) 4.5 days (b) 5.5 days and (c) 6 days.

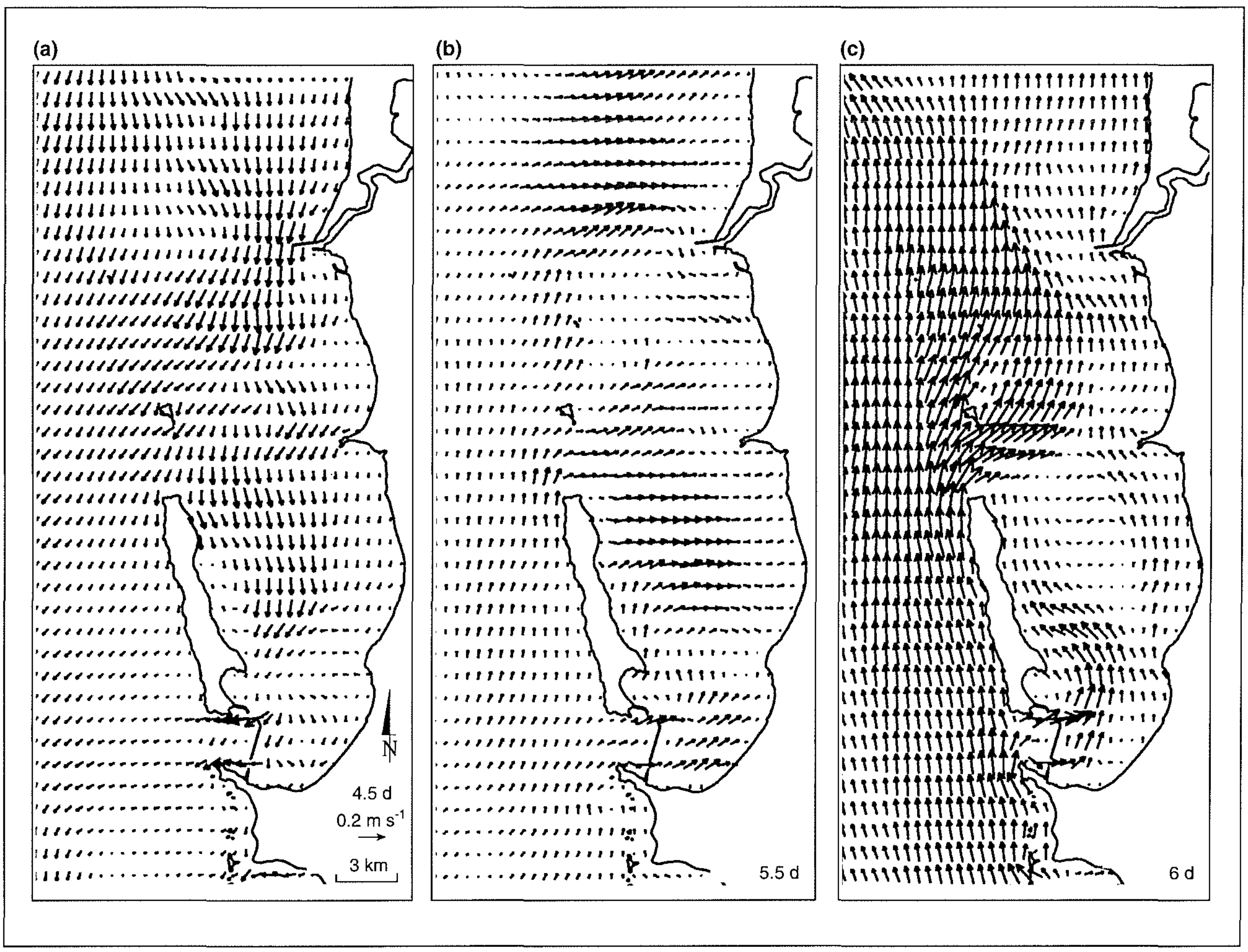


Figure 6.2-27. Baroclinically modelled surface current fields representing the 'autumn' exchange between dense Cockburn Sound water and buoyant shelf water forced by winds (from Figure 6.2-25) after (a) 4.5 days (b) 5.5 days and (c) 6 days.

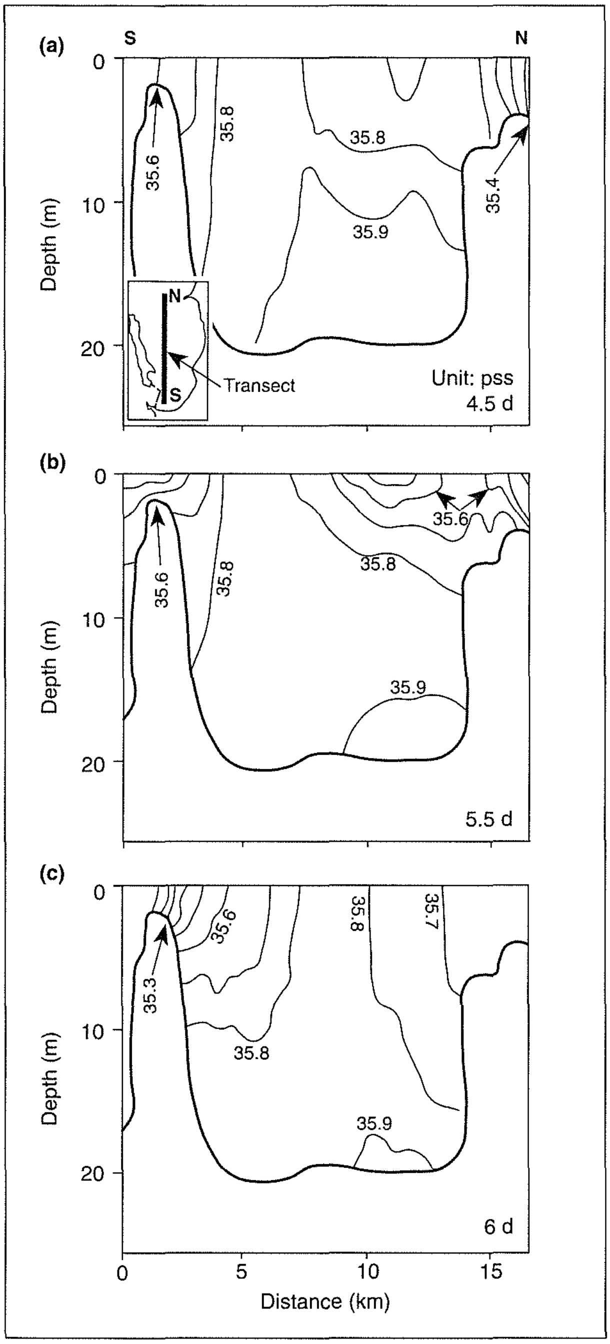


Figure 6.2-28. Baroclinically modelled south-north vertical salinity structure representing the 'autumn' exchange between dense Cockburn Sound water and buoyant shelf water forced by winds (from Figure 6.2-25) at (a) 4.5 days (b) 5.5 days and (c) 6 days.

Further model results are presented from this simulation for midday on 4 May 1994. The winds had remained easterly to northeasterly since the previous day, except for the last hour or two, when winds strengthened from the southwest. The surface currents (Figure 6.2-27b) responded rapidly to this recent wind shift. However, apart from in the close vicinity of the causeway bridge openings, the salinity distribution reflected the influence of the earlier northeasterly winds and showed the resultant southward migration of considerably lower salinity (e.g. 35.6 pss) surface waters to the centre of the sound (Figure 6.2-26b) and the accompanying intensification of the near-surface vertical stratification (Figure 6.2-28b).

The model predicts that the ensuing southwesterly sea-breezes of 5 and 6 May 1994 drove low salinity water into the sound through the causeway bridge openings and then transported it northward (Figures 6.2-26c and 6.2-27c). This led to a strengthening of the near-surface stratification in southern Cockburn Sound to a depth of about 10 m (Fig 6.2-28c). Buoyant water which had previously entered via the northern opening was downwelled against central Parmelia Bank (as shown in Fig 6.2-28c by the steepness of the salinity contours) reducing the vertical stratification and making this area more susceptible to wind mixing.

As shown by the model results in Figure 6.2-29 the near-surface zone (0-5 m below sea surface) of Cockburn Sound was flushed at a greater rate than the deep basin zone (> 15 m below sea surface) of the sound for this 'autumn' simulation. After nine days the near-surface zone was 34% flushed and the deep basin zone was 25% flushed. This result is consistent with the introduction of buoyant water into the sound as an upper layer which inhibited the vertical mixing and flushing of denser deep basin waters.

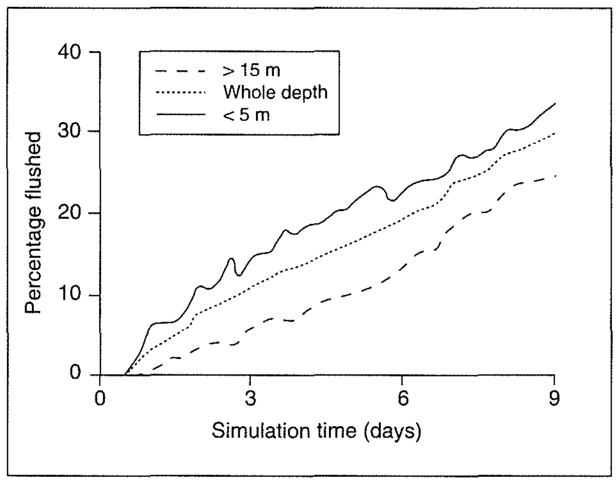


Figure 6.2-29. Baroclinically modelled flushing rates for three depth zones (whole depth, < 5 m and > 15 m) of Cockburn Sound from an 'autumn' simulation (wind data as in Figure 6.2-25).

During the period of the simulation (mainly southerly to easterly winds) 97% of the gross volume output was across the northern opening and, of the inputs, 78% was across the northern opening and 22% across the southern opening. A schematic of these volume fluxes is given in Figure 6.2-30. The two-way exchange across the northern opening is about 3.5 times the magnitude of the throughflow exchange under the conditions of this simulation.

Other simulations (Mills and D'Adamo, 1995b) indicate that easterly, northeasterly, northerly and northwesterly winds are favourable to the migration of relatively buoyant, external

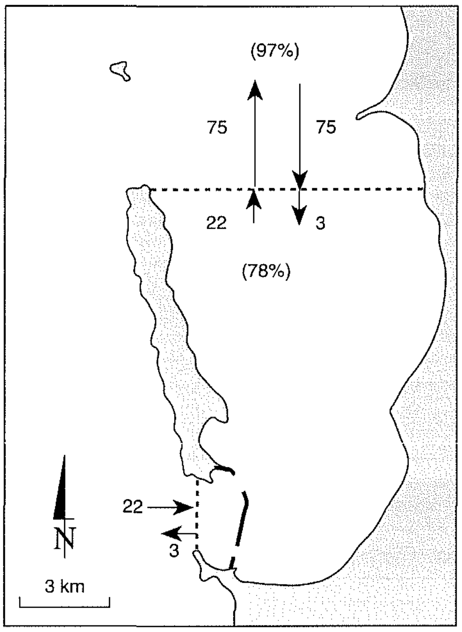


Figure 6.2-30. Schematic of modelled volume fluxes across the openings of Cockburn Sound from a baroclinic 'autumn' simulation (wind data as in Figure 6.2-25). Volume fluxes are expressed as a percentage of total outflow.

surface waters across the northern entrance into Cockburn Sound. Southwesterly and southerly winds favour the entry of external waters into Cockburn Sound through the causeway openings.

In autumn, prevailing easterly winds last typically for 2-3 days, but on occasions 5-6 days of easterlies are experienced. Figure 6.2-31 presents the modelled vertical salinity structure along a south-north transect after five days of 5 m s^{-1} easterly winds. The vertical stratification extends over the entire length of the main Cockburn Sound basin. Under prevailing easterly winds the deep basin water flushing (salt depletion) rates were lower for the southern section of Cockburn Sound than for the entire sound (Figure 6.2-32).

The initial 'autumn' density structure is gravitationally unstable and modelling (Mills and D'Adamo, 1995c) shows

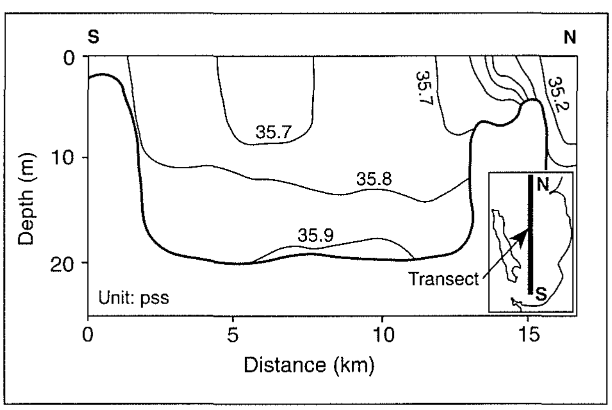


Figure 6.2-31. Baroclinically modelled south-north vertical salinity structure of Cockburn Sound under 'autumn' conditions after 5 days of constant easterly winds at 5 m s^{-1}.

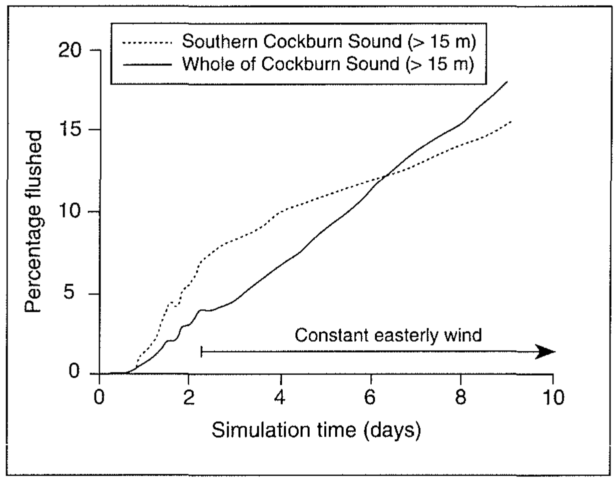


Figure 6.2-32. Baroclinically modelled flushing rates of deep Cockburn Sound water (> 15 m depth) for the entire basin and for the southern part of the basin (between James Point and the causeway) from an 'autumn' simulation. The model was forced with real winds as shown in Figure 6.2-25 for 0-2.3 days, and thereafter by a constant easterly wind of 5 m s^{-1}.

that, in the absence of other forcings, it undergoes baroclinic adjustment subject to rotational effects, resulting in the entry to the sound of buoyant shelf water. Inflow to Cockburn Sound from the north occurs as a buoyant plume attached to the eastern coast, while inflow from the south through the causeway openings propagates northward as a plume attached to the Garden Island coast (Figure 6.2-33) reaching the northern end of the island within four days. The widths of these plumes are typically several kilometres, which scales with the baroclinic radius of deformation (1-2 km) for the sound. The cumulative volumetric flux time for the baroclinic adjustment simulation is about 11 days, which is twice that for the 'autumn' simulation with both wind and density forcing. The combined action of wind and density forcing results in higher exchange rates than for the case of density forcing alone. However, in the presence of wind forcing, it is the characteristic density difference between the sound and external waters in 'autumn' that enhances circulation and flushing of near-surface waters relative to deep basin waters.

The cumulative volume flux time for pure baroclinic adjustment in 'autumn' (about 11 days) is smaller than that due to tidal exchange (about 30 days) which was cited by Maritime Works Branch (1977b) as the base level of exchange for the sound. This suggests that in fact the underlying mechanism of gravitational relaxation provides the base level of exchange in 'autumn'.

Modelling the 'summer' hydrodynamic regime of Cockburn Sound

The transition from the 'winter-spring' hydrodynamic regime to the 'autumn' regime is a complex one (D'Adamo and Mills (1995a). However, on the basis of detailed salinity-temperature-density field data, D'Adamo and Mills (1995a)

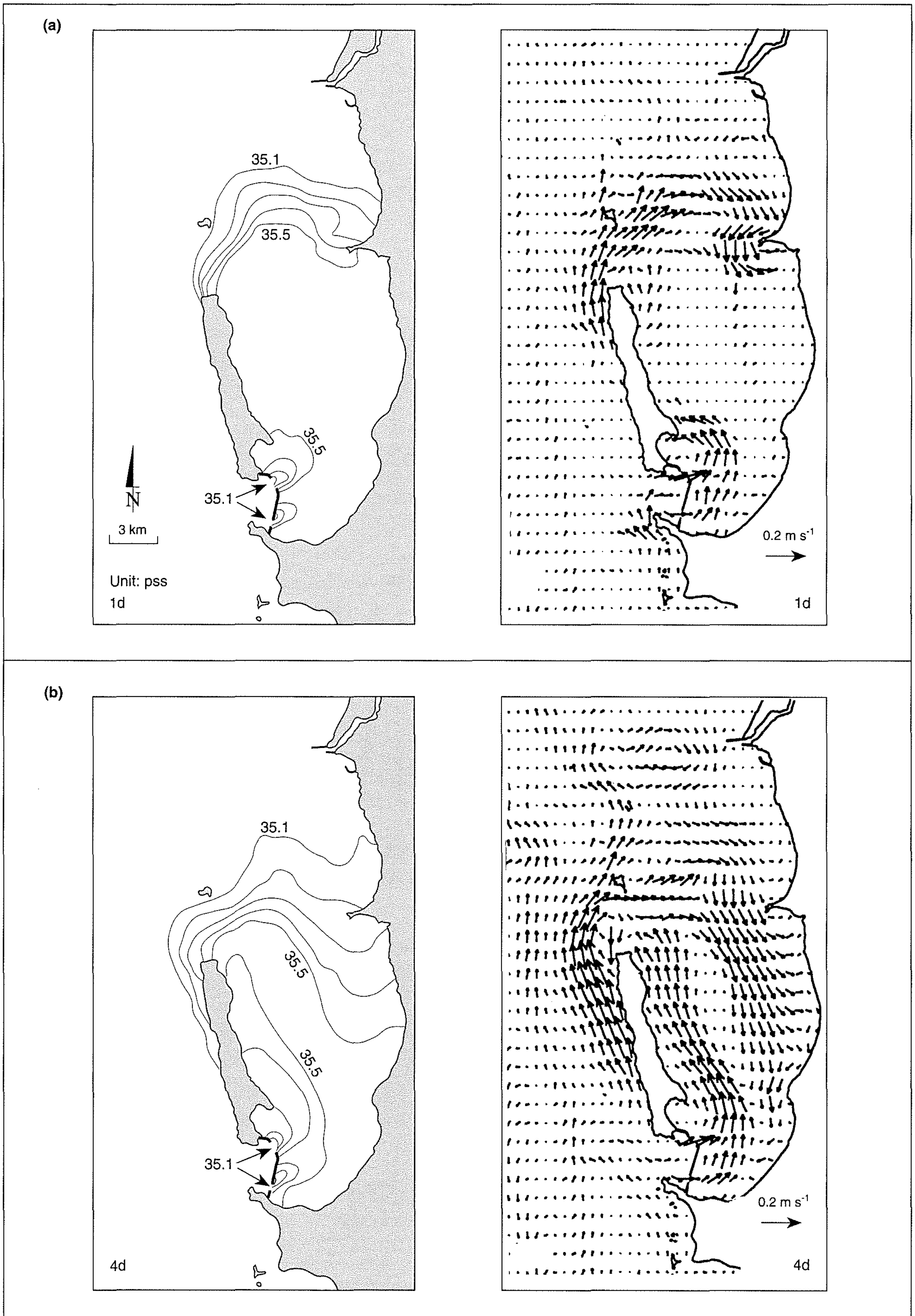


Figure 6.2-33. Baroclinically modelled surface salinity and velocity fields representing density-induced exchange between dense Cockburn Sound and buoyant shelf waters during calm 'autumn' conditions after (a) 1 day and (b) 4 days.

suggested that it would be reasonable to apply a three-dimensional barotropic model during periods (i.e. the 'summer' regime) when the water densities of the shelf and Cockburn Sound were approximately equal and recurrent full depth mixing due to sea-breeze events was occurring in the sound. During the intensive field exercise of March 1992, for example, the horizontal density differential between Cockburn Sound waters and adjacent shelf waters was less than 0.1 kg m^{-3} and the vertical density stratification in Cockburn Sound was generally less than about 0.1 kg m^{-3} (in 20 m) with regular vertical mixing (D'Adamo and Mills, 1995a).

Previous oceanographic studies of Cockburn Sound (Maritime Works Branch, 1977a; Steedman and Craig, 1979, 1983; Speedy, 1994; Speedy and Hearn, submitted) applied depth-averaged barotropic models, forced by wind, long-shelf pressure gradients and/or shelf currents, which indicated that the water circulation of the sound is comprised of two components: a direct throughflow, and a system of interacting topographic gyres (Csanady, 1982) mainly controlled by wind and bathymetry. The system of gyres generally dominates the modelled current speeds and directions within the sound. The depth-averaged currents are approximately in the direction of the wind over shallow margins of the sound, with return flows in the deeper central basin. For example, when southwesterly winds predominate, the flow tends northward along the eastern and western margins and southward in the central basin. The depth-averaged topographic gyres were virtually closed circulation features with severely limited volume exchange rates across the wide northern opening of the sound under wind conditions which occur for most of the time. These rates of volume exchange were found to be of similar magnitude to the rates of exchange due to throughflow in the presence of the causeway (Maritime Works Branch, 1977a; Speedy, 1994; Speedy and Hearn, submitted).

The period 7 March to 4 April 1992 was chosen for the three-dimensional barotropic simulation for two reasons: firstly, current meter data were available to enable comparisons between modelled results and field data; secondly, the analyses of intensive salinity-temperature-density field data collected during the period 9-27 March 1992 provided justification for using a barotropic version of the model. The water density variable in the model was set to a constant in space and time (to provide a barotropic simulation) and the salinity variable in the model was used to represent a conservative, dynamically-passive tracer. The simulation was initialised with low tracer concentration in Cockburn Sound and high tracer concentration in the rest of the model domain (Figure 6.2-34), in order to trace the exchange between internal and external waters. The model was forced with locally recorded wind data for the simulation period (Figure 6.2-35).

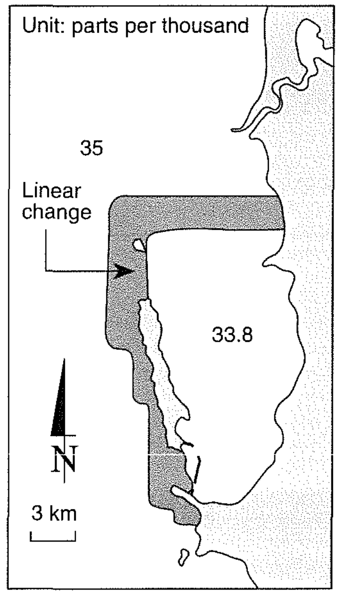


Figure 6.2-34. Starting horizontal distribution of tracer concentration for the barotropic model simulations of the 'summer' hydrodynamics of Cockburn Sound. Vertical distribution of tracer is uniform.

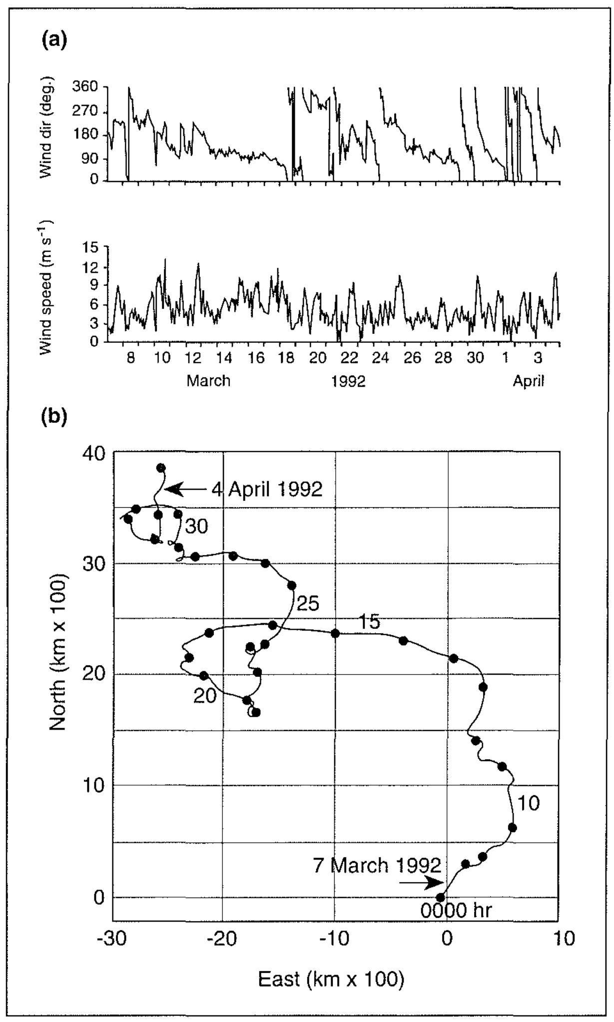


Figure 6.2-35. Wind data used for the barotropic model simulation of the hydrodynamics of Cockburn Sound under typical 'summer' conditions: (a) time series and (b) progressive vector run.

The model predicted three-dimensional barotropic circulation patterns generally consistent with the topographic gyres of previous depth-averaged modelling studies (Maritime Works Branch, 1977a; Steedman and Craig, 1983; Speedy, 1994; Speedy and Hearn, submitted). Figure 6.2-36 shows the simulated horizontal current vector fields at three depths (1 m, 7 m and 12 m) below the water surface for south-southwesterly winds. The flow is northward along the shallow eastern and western coastal margins of the sound and southward in the central basin. The surface current field shows the flow of water from Cockburn Sound across Parmelia Bank and to the eastern margin of Owen Anchorage. The vertical profiles of current speed and direction over the eastern margin of Cockburn Sound (Figure 6.2-37) shows that flow is approximately downwind throughout the water column and that there is a wind-driven surface boundary layer and a frictionally-impeded boundary layer near the sea bed. The surface current speed is approximately 50% greater than the depth-averaged current. The non-uniform depth profile of the currents could result in significant differential transport between particles which are positively and negatively buoyant (e.g. phytoplankton).

For east-northeasterly winds (perpendicular to the principal axis of the sound) the modelled circulation consists of south to southwest currents outside of the sound with west to southwest currents over Parmelia Bank and through Challenger Passage. Water movements are much weaker within the sound, except over the Southern Flats and through the causeway bridge openings where strong outflows occur (Figure 6.2-38).

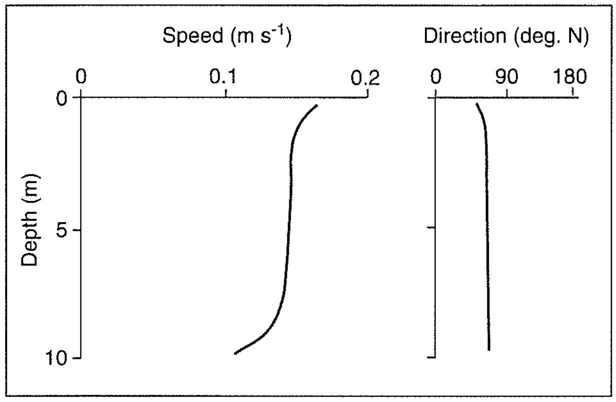


Figure 6.2-37. Barotropically modelled vertical profile of current speed and direction for Cockburn Sound (eastern margin) from the 'summer' simulation of the sound and shelf waters after 6 days (wind data as in Figure 6.2-35).

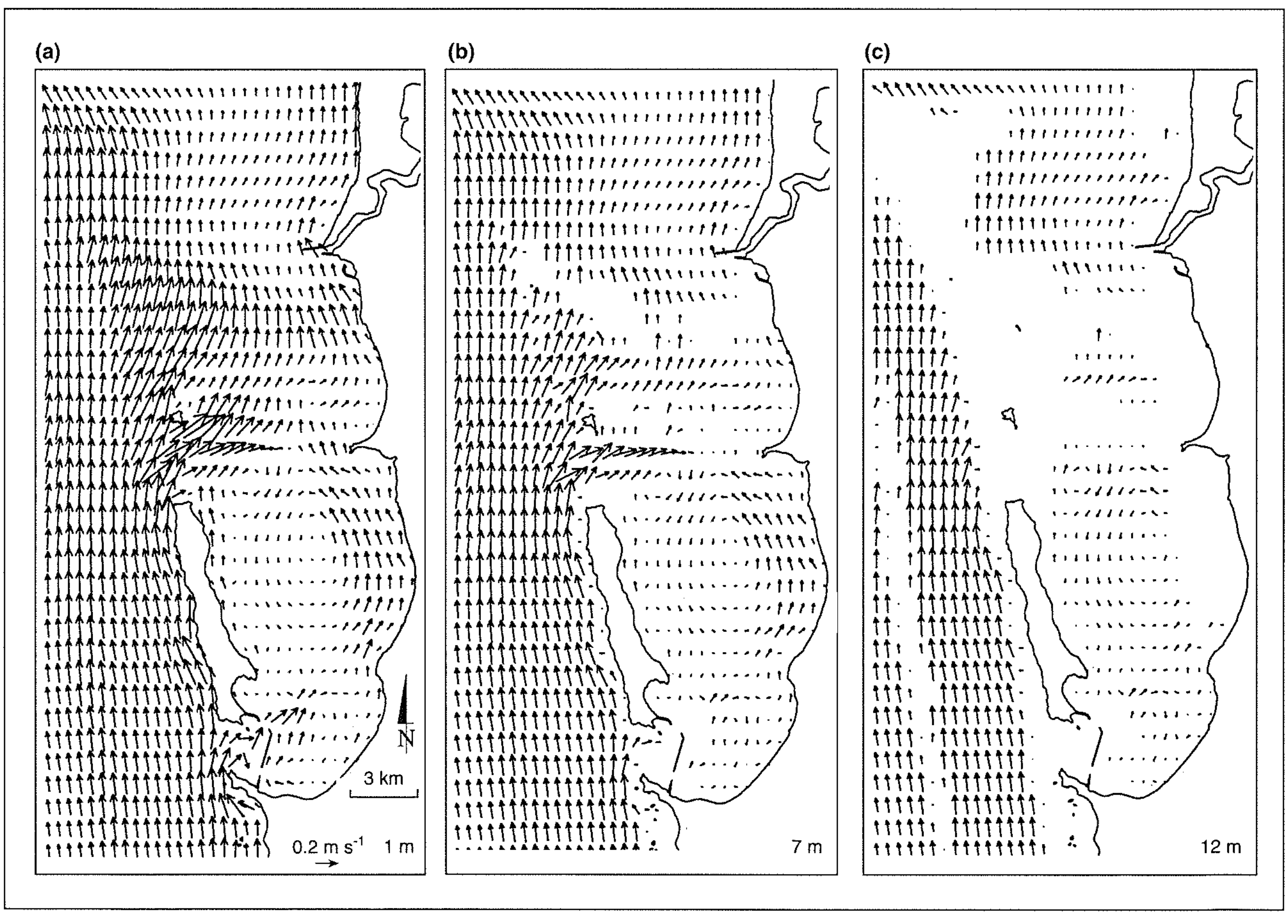


Figure 6.2-36. Barotropically modelled horizontal velocity fields from the 'summer' simulation of Cockburn Sound and shelf waters responding to south-southwesterly winds after 6 days at (a) 1 m depth (b) 7 m depth and (c) 12 m depth (wind data as in Figure 6.2-35).

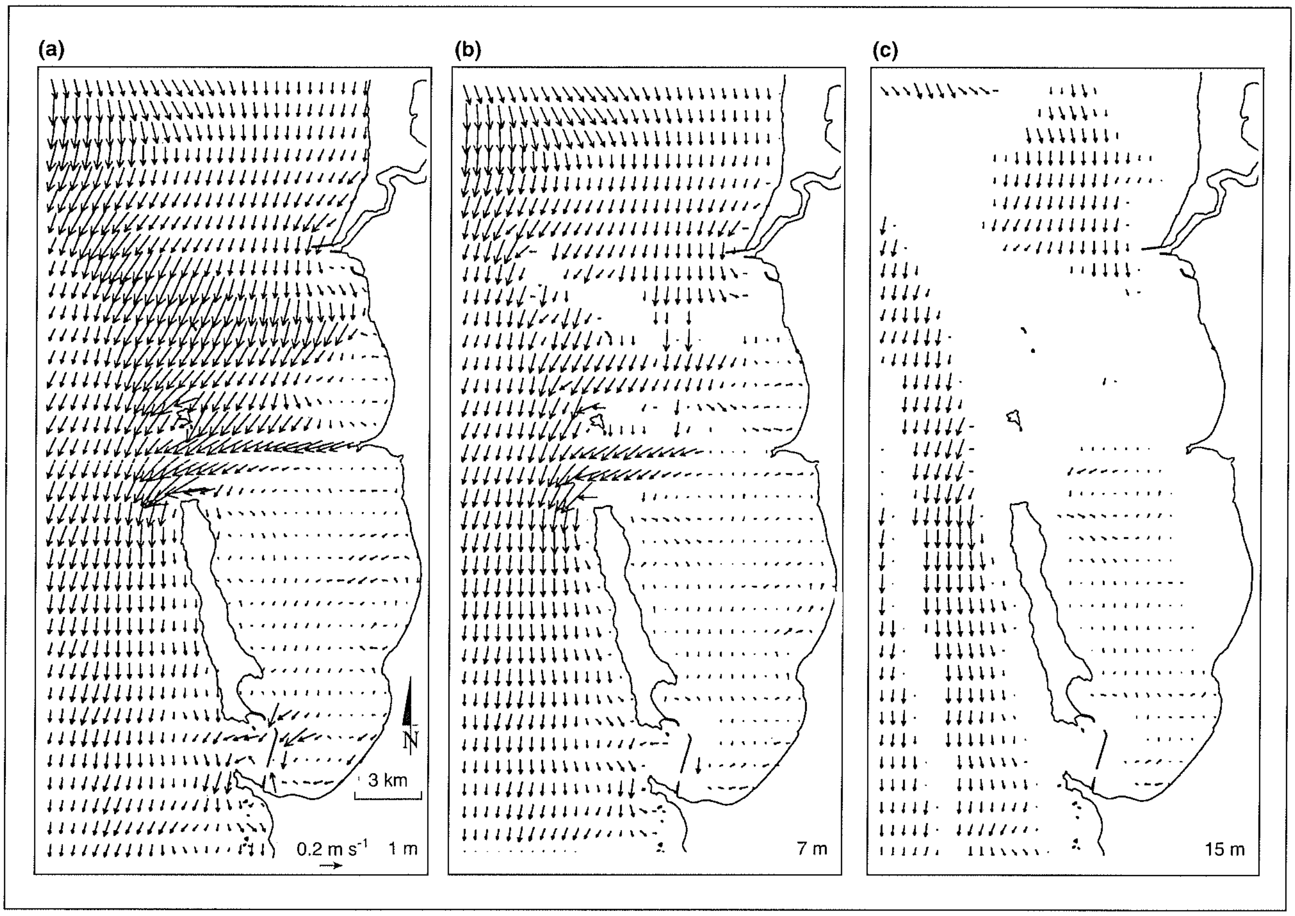


Figure 6.2-38. Barotropically modelled horizontal velocity fields from the 'summer' simulation of Cockburn Sound and shelf waters responding to east-northeasterly winds after 11 days at (a) 1 m depth (b) 7 m depth and (c) 115 m depth (wind data as in Figure 6.2-35).

In general, the time-scale for vertical mixing to the bottom of the sound is comparable to the advection time (time to advect across one grid cell) and hence the salinity (tracer) is vertically well-mixed throughout the model, although horizontal concentration gradients are still present.

With the benefit of a longer (28 day) simulation it can be seen (Figure 6.2-39) that the flushing of Cockburn Sound progresses approximately in the form of an exponential decay. The model predicts that 45% flushing has been achieved in 28 days. There is minimal difference in the flushing rates of the near-surface (0-5 m below sea surface) zone and the deep basin (> 15 m below sea surface) zone, which is to be expected for a barotropic simulation, since full-depth mixing is not impeded by density stratification and occurs on time-scales much shorter than the salt flushing times. Figure 6.2-40 shows the successive cumulative volume flux times for Cockburn Sound calculated from the results of the 'summer' regime (barotropic) simulation. These were the successive time intervals required for the gross volume inflows to cumulate to the volume capacity of Cockburn Sound. The cumulative volume flux times were in the range 6.5-10 days for this simulation.

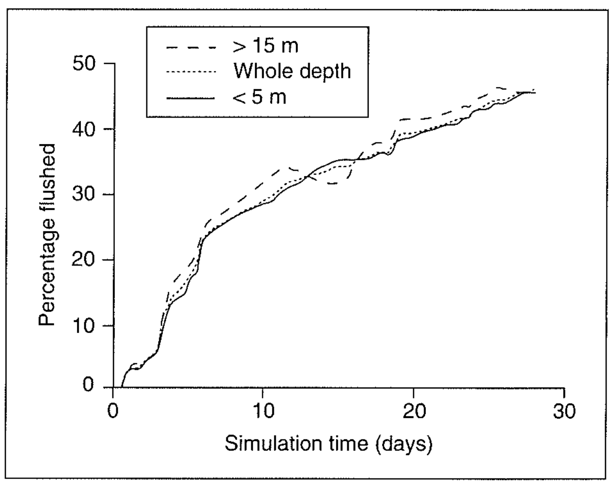


Figure 6.2-39. Barotropically modelled flushing rates for three depth zones (whole depth, < 5 m and > 15 m) of Cockburn Sound from the 'summer' simulation (wind data as in Figure 6.2-35).

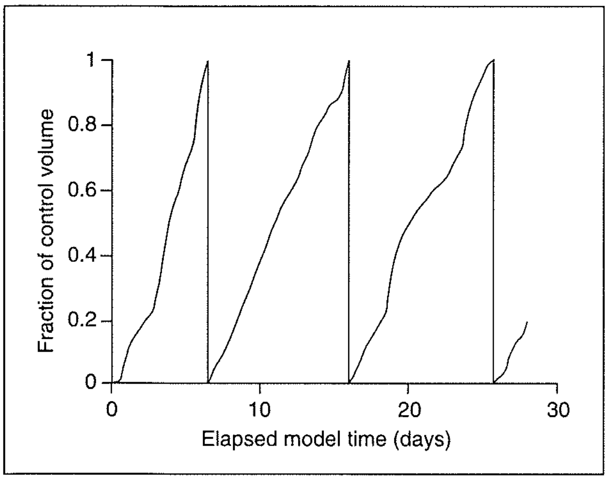


Figure 6.2-40. Successive time intervals required for cumulative volume inflows to equal the volume of Cockburn Sound from the 'summer' barotropic simulation (wind data as in Figure 6.2-35).

Conclusions

Previous studies of the Cockburn Sound hydrodynamics employed two-dimensional, depth-averaged barotropic models which calculated water circulation patterns composed of wind-driven topographic gyres and throughflow. These model studies did not account for the dynamical role of density gradients. However field measurements have shown that vertical and/or horizontal density gradients are almost always present and this has emphasised the need to understand the role of density gradients in mediating the wind-driven circulation of the sound.

The three-dimensional, baroclinic simulations described in this section have shown that the presence of an underlying density difference between the sound and external shelf waters (and its change from season to season) plays a key role in determining the nature of the wind-driven exchange, circulation and flushing of Cockburn Sound. Field data show that the shelf-sound density difference undergoes an annual cycle, with Cockburn Sound water being relatively dense in autumn/early winter, and relatively buoyant in winter/spring. This annual cycle and its relationship to the seasonal hydrodynamic regimes of the sound is further discussed in D'Adamo and Mills (1995b) and in section 5.1. Three broad 'seasonal' regimes have been identified from field data and modelled. In 'winter', buoyant estuarine plume water is forced into Cockburn Sound by northerly winds, and then vertically mixed by storm winds. Under moderating southerly wind conditions, buoyant surface water is transported out of the sound and is replaced by relatively dense shelf water which subsides and is transported across the deep basin of the sound. In 'autumn', relatively buoyant water is driven in across the shallow sills at the entrances of the sound and forms surface stratified layers which confine and isolate deep basin water from exchange. Under well-mixed conditions in 'summer' the three-dimensional circulation characteristics are generally consistent with the wind-driven topographic gyres and throughflow described by previous studies.

To further establish the above findings, three parallel simulations were conducted with the same wind forcing in each case, but different initial salinity/density distributions appropriate to the 'autumn', 'winter' and 'summer' regimes. As shown in Figure 6.2-41, the flushing of the near-surface zone (0-5 m below sea surface) is advantaged by a salinity/density distribution with a relatively dense Cockburn Sound (typical of 'autumn'), the flushing of the deep basin zone (> 15 m below sea surface) is advantaged by a relatively buoyant Cockburn Sound (typical of 'winter'), and flushing is intermediate for the 'summer' (barotropic) simulation. This model result is in substantial agreement with the findings from field data of D'Adamo (1992), D'Adamo and Mills (1995b).

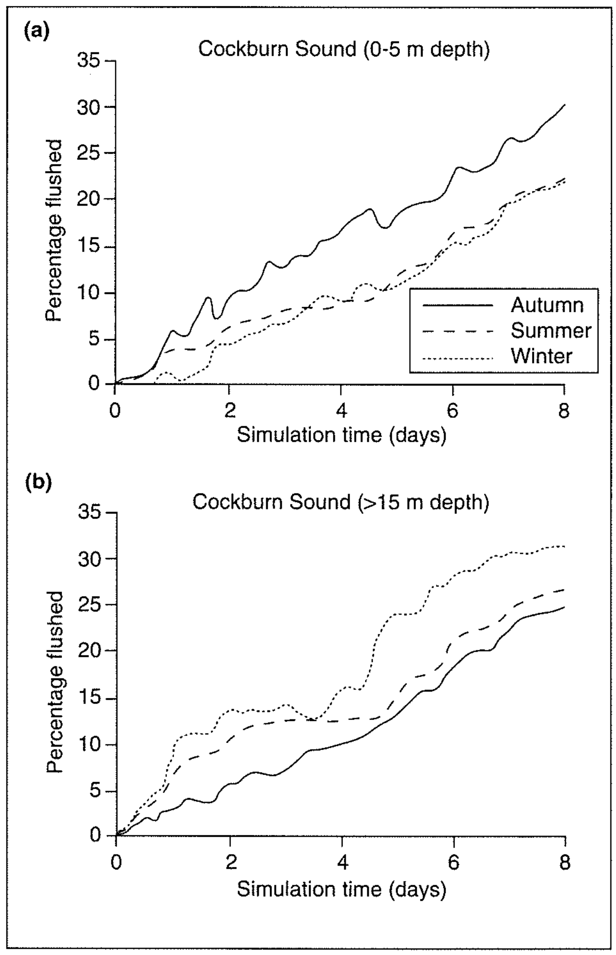


Figure 6.2-41. Comparison of modelled flushing rates for Cockburn Sound from three simulations using the same wind forcing (as in Figure 6.2-25) but with starting salinity distributions for 'autumn', 'summer' and 'winter', respectively, for (a) 0-5 m depth, and (b) > 15 m depth.

For winds in excess of about 5 m s^{-1}, the addition of the wind forcing increases the exchange rates substantially beyond those for gravitational relaxation alone. However, for winds up to 10 m s^{-1}, in the presence of shelf-embayment density differences, exchange can introduce vertical stratification to the basin, enhancing the gravitational stability of the sound. The density stratification changes the way in which the momentum of the wind is distributed throughout the water column, with the wind-driven circulation being mediated by density. The simulations have confirmed that density effects can remain of dynamical significance in association with wind speeds up to 10 m s^{-1}, as concluded by D'Adamo (1992) who reviewed historical density stratification and wind data. The presence of vertical stratification does not necessarily imply a lack of flushing. In 'winter', for example, high vertical stratification in Cockburn Sound results from an efficient flushing process.

In each of the 'autumn', 'summer' and 'winter-spring' wind-driven baroclinic simulations the two-way volume exchange across the northern opening was greater by a factor ranging from 2 to 3.5 compared to the magnitude of flow through the causeway. This result is at variance with the findings of the two-dimensional model studies, which concluded that this factor was close to unity. Since the three-dimensional model is able to resolve vertical current profiles, it could more accurately calculate exchange across the deep northern opening section.

In the absence of forcings such as wind, tide and long-shelf pressure gradients, an embayment-shelf density difference still leads to underlying exchange and flushing because this distribution of water masses is gravitationally unstable. The model has been used to simulate the gravitationally-driven circulation for 'winter' and 'autumn' conditions. It has been shown that, although the influence of the earth's rotation lowers the intrinsic rate of baroclinic exchange, it is still considerably greater than the estimated rate of tidal exchange during 'autumn' and 'winter'. Furthermore the modelling has shown that the density-induced gravitational exchange involves penetration of shelf water along the entire length of the sound within about 10 days, whereas the length scale of tidal advection is about 500 m, so that flushing of Cockburn Sound due to tidal processes alone is likely to occur as a slow diffusive process.

The distinct seasonal hydrodynamic and transport regimes identified in Cockburn Sound should be taken into account in the design of future programmes to investigate the dispersion of effluent and to monitor environmental quality in the sound.

6.2.4.9 *Effects of the Garden Island Causeway on the flushing of Cockburn Sound*

Construction of a rock-fill causeway between Garden Island and the mainland (west of Rockingham) was completed in June 1974, providing vehicular access to the HMAS Stirling naval facility. The causeway traverses the southern entrance to Cockburn Sound, following the shallowest route between the land masses. Two bridge openings were incorporated into the causeway, with widths of 300 m and 625 m and average water depths of 2.8 m and 4.5 m, respectively. The cross-sectional flow area along the causeway alignment was reduced to approximately one third of its original value, before construction.

Past studies

Between 1970 and 1976 a series of field surveys was commissioned by the Commonwealth Government of Australia to assess the causes of water flow between Cockburn Sound and the surrounding waters and to report on the changes in flow which occurred as a result of the causeway construction. A complementary numerical hydrodynamic modelling study was initiated, making use of a two-dimensional, depth-averaged, barotropic modelling system with a nested grid facility, developed at the Danish Hydraulics Institute (Abbott *et al.* 1973). Based on these studies, the Maritime Works Branch (1977a and b) concluded that the causeway had (a) reduced volume flow rates through the southern entrance of the sound to about 40% of their original values, (b) reduced the combined rate of volume exchange across the northern and southern entrances of the sound to about 40-60% of its original value, (c) significantly changed water current speeds within about 1500 m of the causeway, but had not significantly altered the wind-driven current speeds within the broad interior of Cockburn Sound.

Steedman and Craig (1979, 1983) applied a two-dimensional, depth-averaged, barotropic model to study the wind-driven circulation of Cockburn Sound and to estimate the resultant volume exchange across the northern opening. This study concluded that the two-way exchange flux across the northern opening due to wind-driven topographic gyres was low, and generally of the same order of magnitude as estimates from field data of the net flux through the causeway.

In reviewing these works, Hearn (1991) suggested that the causeway may have led to a significant difference in the overall 'ventilation' of the sound under barotropic conditions, even although the mixing and dispersion of contaminant plumes would not have appreciably changed at a more local scale.

More recently, Speedy (1994) and Speedy and Hearn (submitted) have conducted high resolution (100 m cell size) depth-averaged, barotropic modelling of Cockburn Sound and surrounding waters, for the configurations before and after installation of the causeway. The conclusions of this study are broadly consistent with those of the Maritime Works Branch (1977a) model study, which have been listed above. In particular, it was found that the net volume fluxes through the causeway openings are of similar magnitude to the two-way, depth-averaged, barotropic exchange rates across the much larger northern opening due to topographic gyres alone. This study also found that the presence of oscillatory (tidal) flows through the causeway openings had little influence (through frictional damping) on the magnitude of the net throughflows.

These studies all used two-dimensional, depth-averaged, barotropic models which did not account for the effects of density gradients on the dynamics, and were unable to resolve vertical variations in the currents.

Application of the three-dimensional baroclinic model

The importance of density effects on the hydrodynamics and flushing of the sound has been detailed in section 5.1 and modelled in section 6.2.4.8. The three-dimensional baroclinic model was therefore used to investigate the influence of the causeway on flushing in the presence of density stratification. An additional advantage of this approach is that model variables such as water salinity can again be used to trace the extent to which external water penetrates into the sound and mixes with resident water, and this provides a further appreciation of the effects of the causeway on flushing of the sound.

The model grid representation of the two causeway bridge openings gives a combined cross-sectional flow area of 3500 m^2, compared to an area of 11 100 m^2 along the same alignment for model simulations without the causeway. This gives a flow area reduction of 67%, which is close to the reduction of 73% used by Speedy (1994) and Speedy and Hearn (submitted). However, as previously cautioned, the horizontal grid size of the baroclinic model is too coarse to resolve the exact form and orientation of the bridge openings. Finer model resolution was precluded by the area of the total domain and the computing resources available.

Effects of the Garden Island Causeway in 'autumn'

The 'autumn' simulation was re-run with the same wind forcing and initial salinity (density) distribution, but without the causeway, to determine how this affected the flushing rates and circulation patterns of the sound.

Under southerly winds the simulated throughflows via the southern opening were stronger in the absence of the causeway, with a more rapid penetration of relatively low salinity, buoyant shelf water into the sound and then northwards, and next to the entire eastern shoreline of Garden Island (Figures 6.2-42a and b). Another significant difference in the two simulations is that under southerly wind the absence of the causeway resulted in greater penetration of denser water northward along the mainland coast toward Fremantle.

During a subsequent period of predominantly easterly wind, shelf water penetrated into the sound via the northern opening. A comparison of the simulated surface salinity contours for 1200 h 5 May 1994 shows that, without the causeway, most of the surface waters of the sound (which had an original salinity of 36 pss) had been significantly mixed with inflowing low salinity shelf water (Figure 6.2-42d). By contrast, in the simulation with the causeway present (Figure 6.2-42c), a large central portion of Cockburn Sound remained close to its original surface salinity.

Vertical sections of salinity along the sound show that greater inflow via the unobstructed southern opening under predominantly southerly winds lowered the salinity in the southern half of the sound over greater horizontal and vertical distances, and reduced the entry of buoyant water across the northern opening (Figures 6.2-43a and b). At 0000 hrs 5 May 1994, after predominantly easterly winds, in the absence of the causeway, a mound of high salinity (> 35.8 pss) water was restricted to the central and northern end of the deep basin and overlain by less saline surface water, whereas, with the causeway present, the entire bottom of the sound was covered with this water (Figures 6.2-43c and d).

Differences in the rates of flushing of Cockburn Sound during 'autumn' conditions, with and without the causeway, are illustrated in Figures 6.2-44a, b and c for the near-surface zone (0-5 m below sea surface), the deep basin zone (> 15 m below sea surface) and the whole water column, respectively. These flushing rates were derived from changes in calculated salt content during simulations where the initial salinity of Cockburn Sound water was greater than that of the surrounding waters. In each case the flushing rates were greater in the absence of the causeway. Both with and without the causeway the surface zone was most readily flushed and the deep-basin zone was least rapidly flushed.

The baroclinic model predicted that the volumetric flux through the southern opening was reduced to 45% of its former magnitude by the construction of the causeway. This is in good agreement with previous results (Maritime Works Branch, 1977a and b; Speedy, 1994; Speedy and Hearn, submitted) considering the spatial resolution of the model. The 'autumn' simulations further showed that the combined *gross* volumetric influxes via the northern and southern entrances were reduced (in the presence of the causeway) to about 80% of their original values. This result differs considerably from the previous depth-averaged modelling studies which gave a combined (northern and southern openings) flux reduction to about 50% of the pre-causeway value.

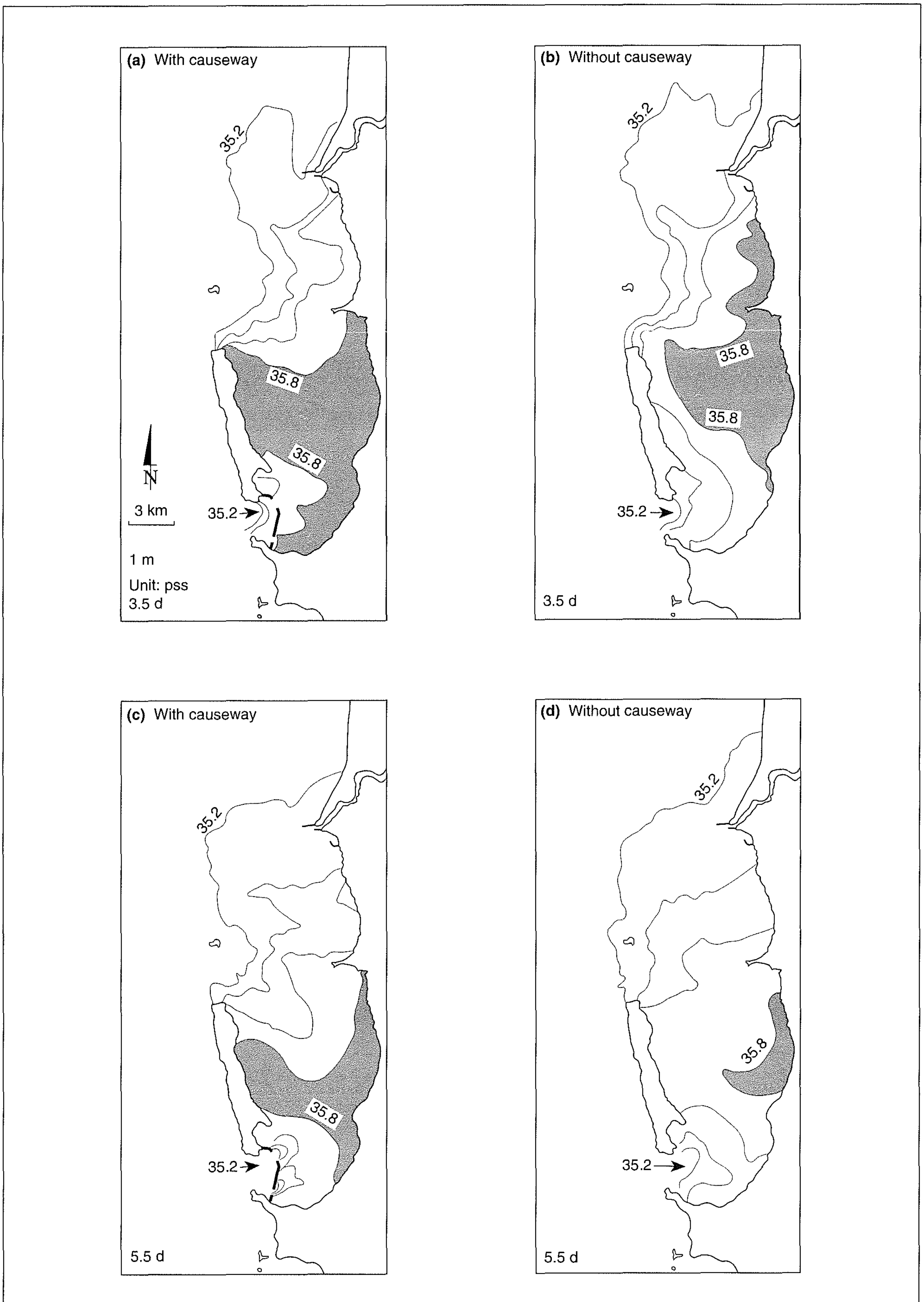


Figure 6.2-42. Baroclinically modelled surface salinity fields from the 'autumn' simulations (a) with the causeway and (b) without the causeway after 3.5 days and (c) with the causeway and (d) without the causeway after 5.5 days. Shading indicates a salinity greater than 35.8 pss.

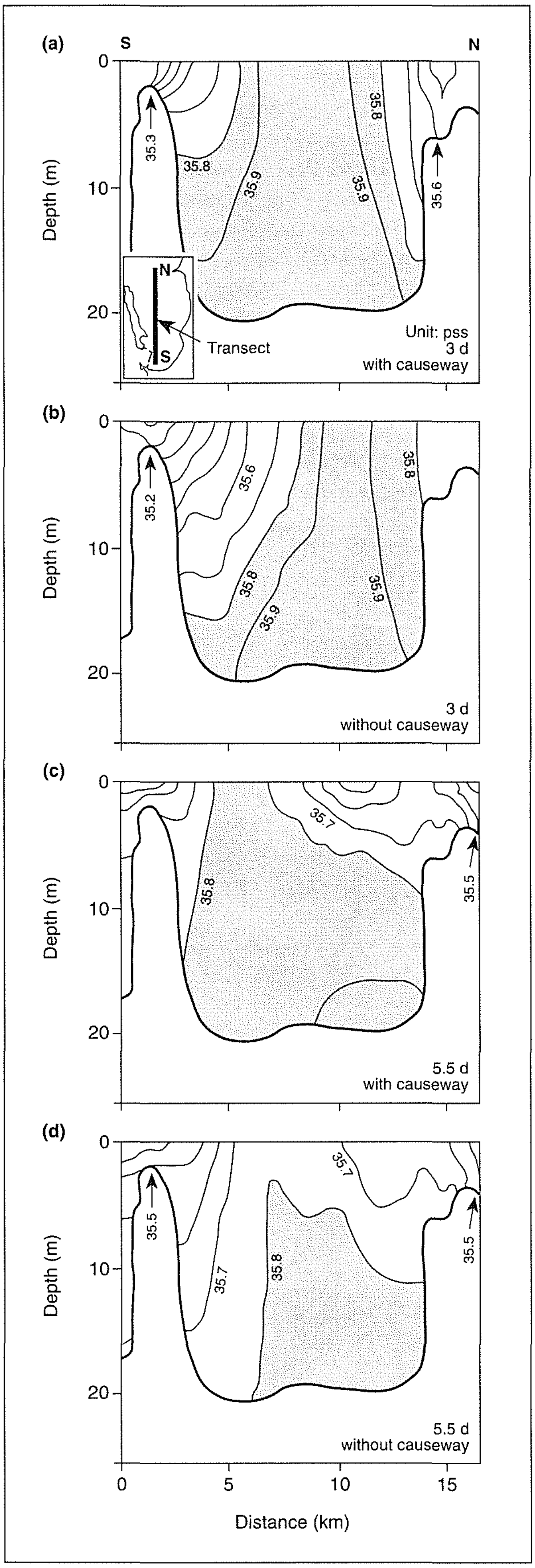


Figure 6.2-43. Baroclinically modelled south-north vertical salinity structures from the 'autumn' simulations (a) with the causeway and (b) without the causeway after three days and (c) with the causeway and (d) without the causeway after 5.5 days. Shading indicates a salinity greater than 35.8 pss.

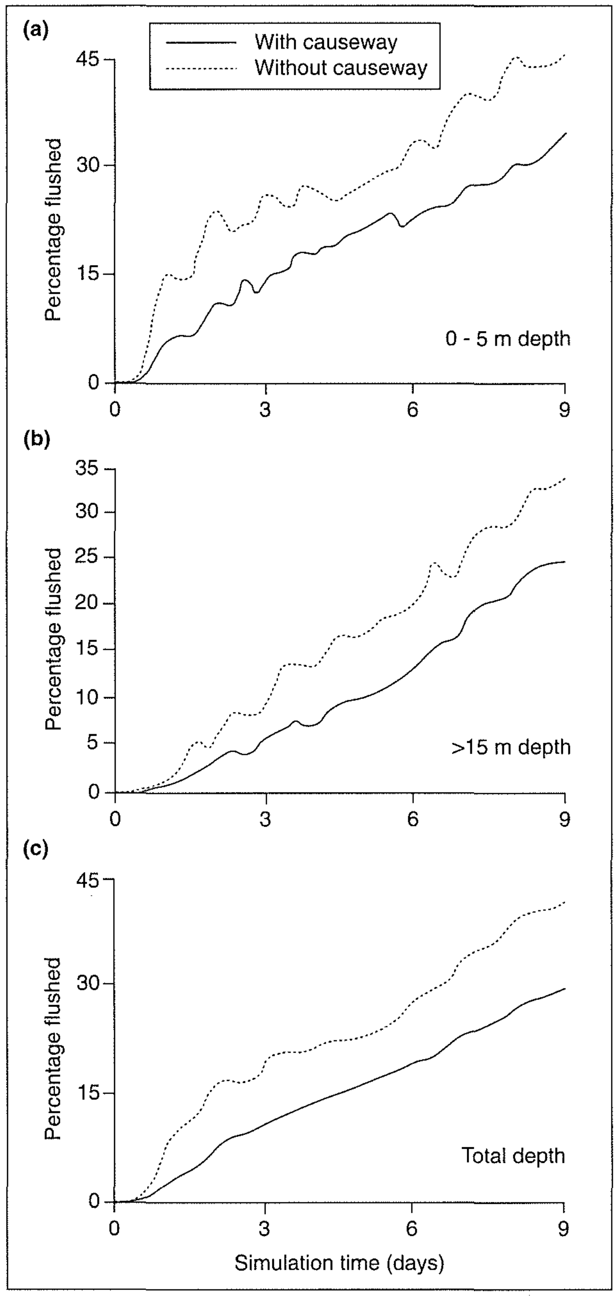


Figure 6.2-44. Baroclinically modelled flushing rates for Cockburn Sound comparing 'autumn' simulations with and without the causeway for three depth zones: (a) 0-5 m, (b) > 15 m and (c) total depth.

A primary reason for this difference is the ability of the three-dimensional model to resolve vertical shear, that is the changes in current speed and direction in the vertical, whereas a depth-averaged model can not do this and therefore can not estimate the component of water exchange that takes place due to vertical shear. Figure 6.2-45 illustrates a depth profile of current speed and direction at a point along the northern opening of the sound, exemplifying the shear that can occur there. These results suggest that volume exchange across the northern entrance of the sound may have been underestimated in the past using depth-averaged models, however the past estimates for the southern entrance are likely to be reliable, because the shallow depths there would rarely sustain flow reversals under wind-driven conditions.

For the 'autumn' simulations both with and without the causeway, the density excess of Cockburn Sound waters relative to shelf waters in 'autumn' was an important factor in determining the nature the circulation. In both simulations the deep basin zone of the sound was less readily flushed in 'autumn' than the near-surface zone.

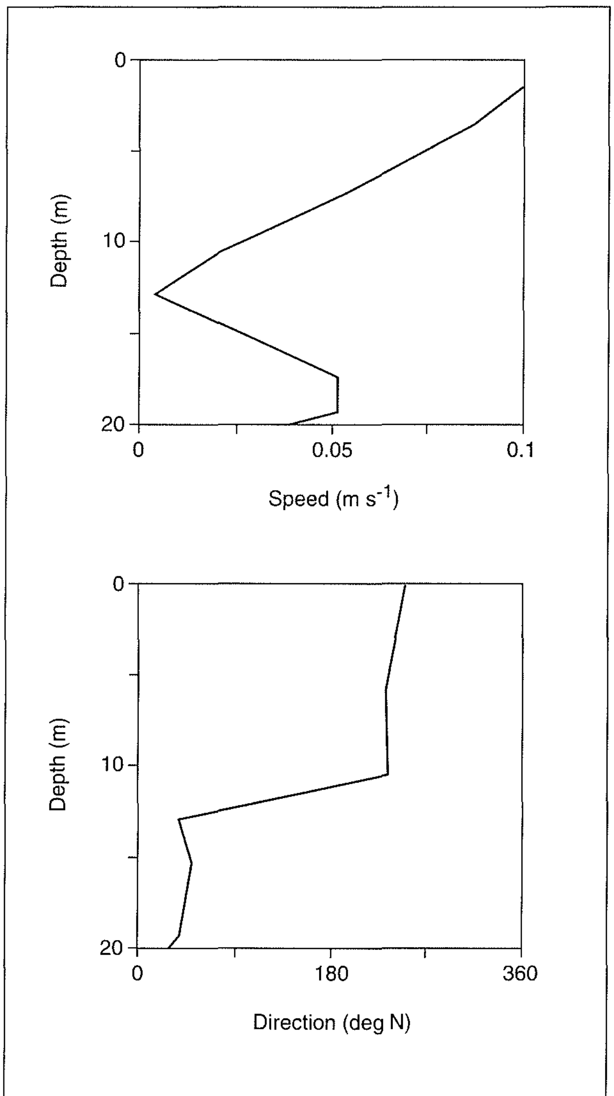


Figure 6.2-45. Baroclinically modelled vertical profile of current speed and direction at the northern opening of Cockburn Sound from the 'autumn' simulation.

Effects of the Garden Island Causeway in 'winter-spring'

The 'winter-spring' hydrodynamic regime of Cockburn Sound was simulated with and without the causeway present, using local wind data for the period 9 August 1991 to 18 August 1991. In the absence of the causeway, winds from the southwest quadrant forced a greater inflow to Cockburn Sound through its southern opening. The relatively dense inflow subsided and moved northward across the deep basin of the sound. This resulted in more rapid mixing with and displacement of buoyant sound water in the southern half of the basin in the absence of the causeway, as shown by comparing south-north vertical salinity sections for the two simulations (Figure 6.2-46). The corresponding near-bottom current vector plots for the two simulations are shown in

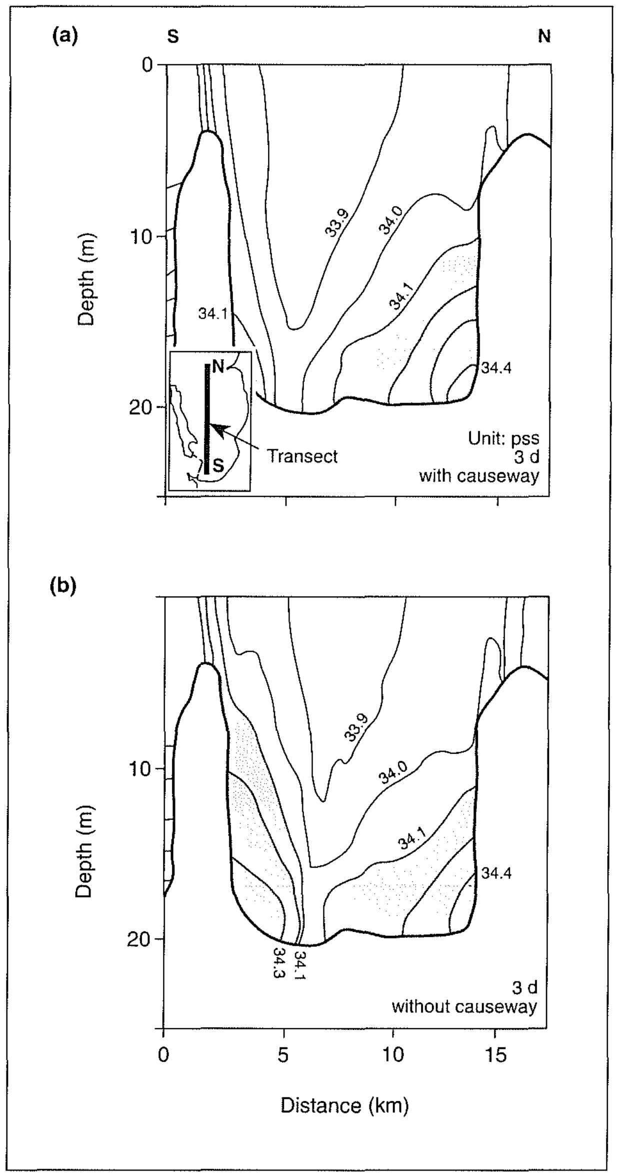


Figure 6.2-46. Baroclinically modelled south-north vertical salinity structure from the 'winter' simulations (a) with and (b) without the causeway after 3 days (winds as in Figure 6.2-10). Shading indicates a salinity greater than 34.1 pss.

Figure 6.2-47 and show the relative increase in flow speeds in the southern portion and the extended spatial influence of the southern inflow to the sound when the causeway is not present. In summary, the 'winter-spring' mechanism of deep basin water renewal was simulated both with the causeway present and absent, however the rate was quicker in the absence of the causeway.

Flushing rates for the 'winter-spring' regime of Cockburn Sound, with and without the causeway, were calculated from the modelled changes in salt content for various control volumes. These results are presented in Figure 6.2-48 and suggest that the presence of the causeway has led to a somewhat decreased flushing rate for the entire sound, including both the near-surface and the deep basin zones.

Conclusions

The results of past two-dimensional modelling studies, which suggest that, on average, the flux through the southern opening of Cockburn Sound has been reduced by construction of the causeway to about 40% of its former value, are supported by this study.

The total (combined) volume exchange rate via both the northern and southern entrances of Cockburn Sound has been reduced as a result of the causeway construction, though the three-dimensional modelling results suggest that the overall reduction is about 30%, rather than the 50% suggested by the previously reported two-dimensional modelling studies.

Former studies reported by Maritime Works Branch (1977a and b) invoked tidal currents as a base flushing mechanism under calm conditions (with an exchange time of about 30 days), and concluded that the construction of the causeway would not have significantly altered the tidal exchange rates. This study has pointed out that, in the presence of seasonal density differences between waters of the shelf and Cockburn Sound, the exchange rates due to baroclinic adjustment alone are generally greater than those due to tide. The gravitational adjustment flows penetrate along the length of Cockburn Sound, whereas the length scales of tidal advection are generally of the order of 500 m.

The onset of distinct circulation regimes ('autumn', 'winter-spring' and 'summer') due to seasonal density differences between shelf and embayment waters was predicted to occur both in the presence and absence of the causeway.

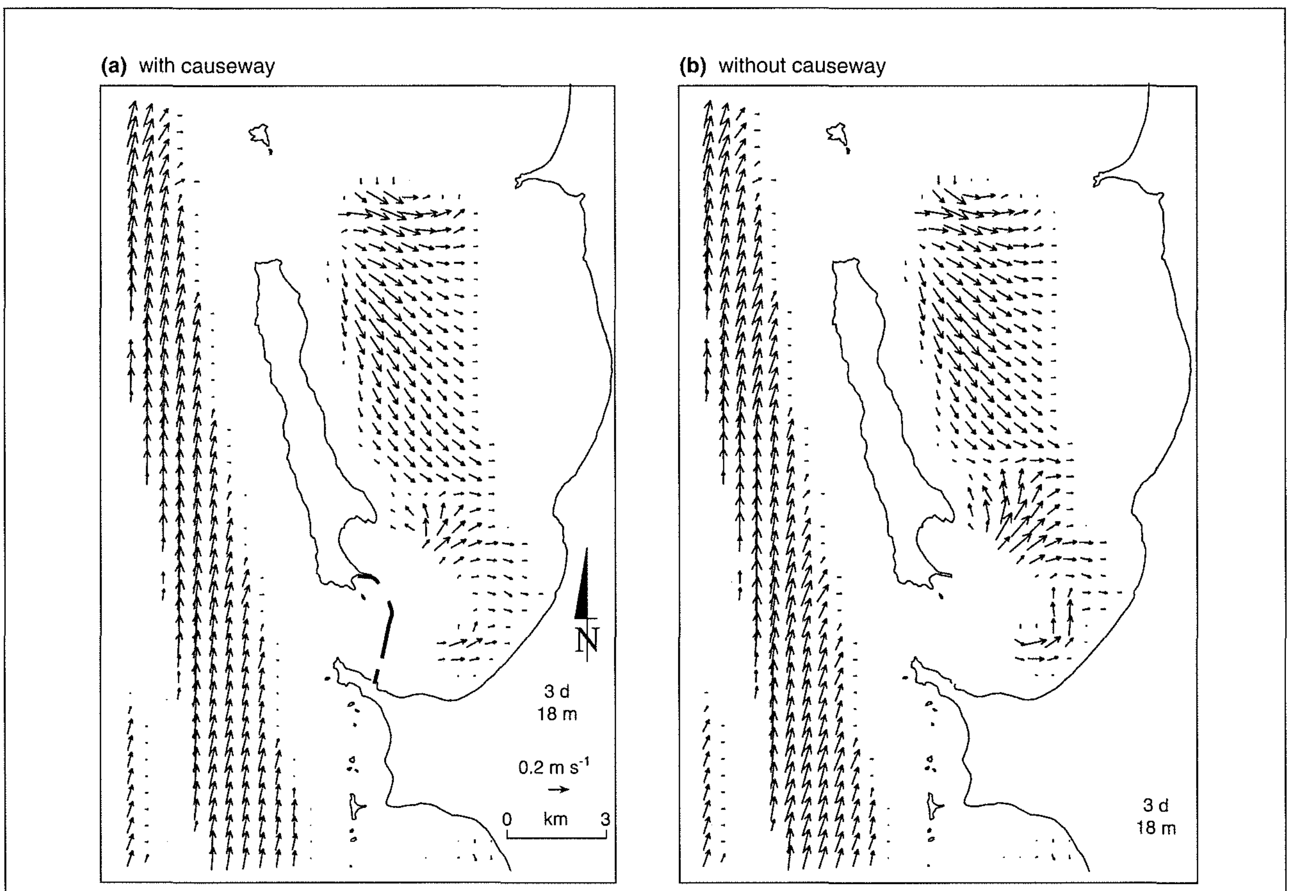


Figure 6.2-47. Baroclinically modelled horizontal velocity fields at a depth of 18 m from the 'winter' simulations (a) with and (b) without the causeway after 3 days (wind data as in Figure 6.2-10).

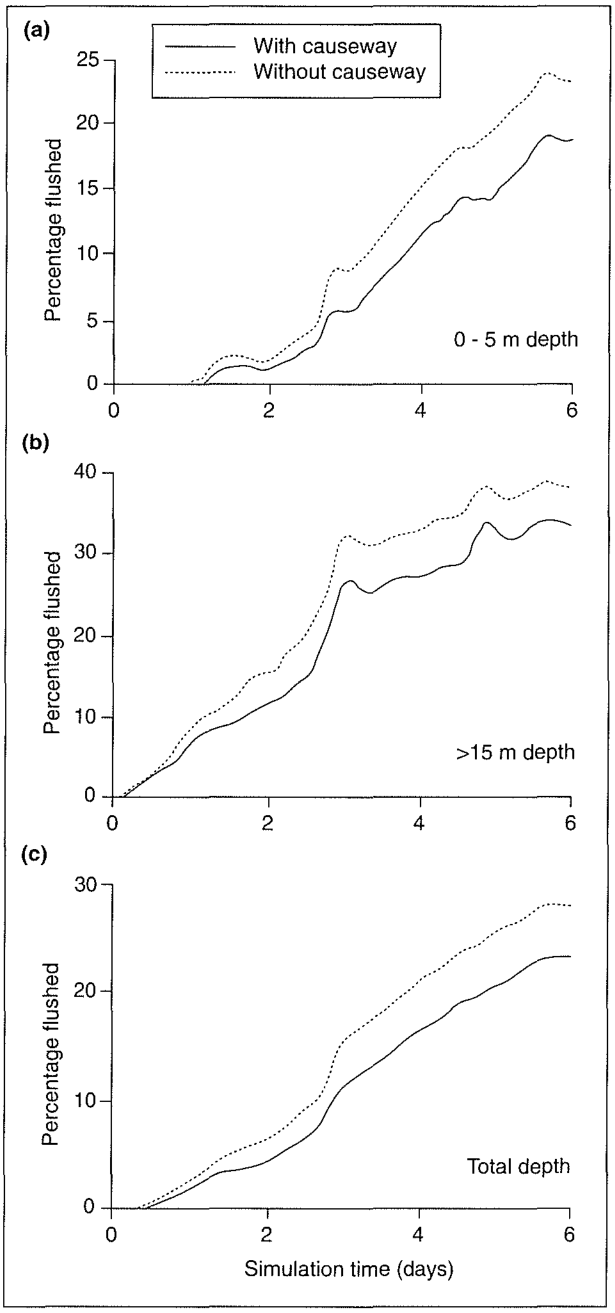


Figure 6.2-48. Baroclinically modelled flushing rates for Cockburn Sound comparing 'winter' simulations with and without the causeway for three depth zones, (a) 0-5 m, (b) > 15 m and (c) total depth.

Comparing the results of the baroclinic simulations, with and without the causeway, it was concluded that the most significant differences in the modelled salinity and advection fields in Cockburn Sound occurred in the southern half of the sound, reflecting the proximity of this region to the inflows via the southern opening.

6.2.4.10 Investigation of the hydrodynamics of Warnbro Sound

Warnbro Sound (see Figure 1.1-1, 1.1-2 and Plate 4.1-1) is a semi-circular embayment south of Cockburn Sound and mid way between the mouths of the Swan-Canning and Peel-Harvey estuaries. The sound is 4.5 km wide and its seaward boundary is about 7 km long. The main basin of Warnbro Sound has depths typically between 15 and 20 m. However sandbanks and a chain of reefs form sill areas which limit the depth of water exchange with the ocean to less than 5 m, except for a few passages (up to 10 m deep) through the centre of the reef chain.

The wind-driven barotropic circulation of Warnbro Sound was modelled by Gersbach (1993). He concluded that winds with significant easterly or westerly components led to greater rates of exchange between the sound and Sepia Depression than northerly or southerly winds of the same speed, because they drove water directly across the western opening of the sound.

Field observations from the SMCWS found vertical and horizontal density stratification in the waters of Warnbro Sound on most occasions throughout the year. For this reason baroclinic modelling was conducted (Mills and D'Adamo, 1995d) to examine the wind-driven circulation of Warnbro Sound in the presence of density stratification, and the response of the density structure to wind forcing.

Response of Warnbro Sound to wind forcing in the presence of stratification

The baroclinic model (northern 'local-scale' domain, as shown in Figure 6.2-4) was initialised with linear vertical salinity stratification of 0.035 pss m^{-1} and forced by constant winds. This salinity gradient corresponds to a density gradient of about 0.025 kg m^{-4} which is near the upper end of the observed range of vertical density stratification in Perth's coastal embayments (D'Adamo and Mills, 1995b). The results presented here are derived from a sub-area of the total

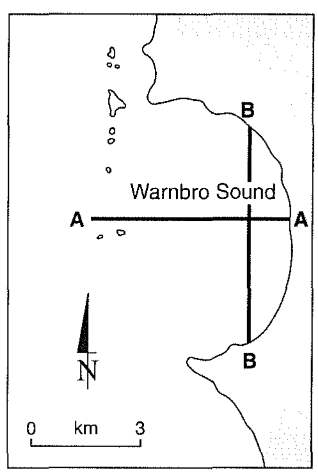


Figure 6.2-49. Locations of vertical salinity sections from the Warnbro Sound baroclinic simulations.

model domain. The locations of simulated salinity sections are shown in Figure 6.2-49.

Figures 6.2-50a, b and c show the simulated currents at three depths after 12 hours of 7.5 m s^{-1} westerly wind. It can be seen that surface water is driven into the sound over the northern and southern sill areas and that through the deeper reef passages there is a corresponding outflow, drawn from sub-surface recirculation in the deep basin. Figures 6.2-51a and b show the contours of the simulated salinity distribution along a west-east and a south-north vertical section (see Figure 6.2-49) of Warnbro Sound. The west-east vertical salinity structure (Figure 6.2-51a) shows wind-induced vertical mixing, together with upwelling on the western side and downwelling on the eastern side of the sound, and suggests that the outflow through the deeper reef passages is partially composed of upwelled high salinity water from the deep basin. The south-north vertical salinity distribution (Figure 6.2-51b) shows two upper regions of low salinity to the north and south of the sound overlying a vertically-stratified, deep basin structure. The low salinity regions resulted from wind-driven advection across the shallow sills into the sound. Intense downwelling of low salinity water on the northern side of the sound led to the southward displacement of stably-stratified resident water in the deep basin. Twenty-four hours of wind forcing removed the vertical stratification in the sound.

For steady 5 m s^{-1} easterly winds and the same initial vertical stratification, the model results (Figures 6.2-52a, b and c) show that surface water is driven out of the sound over the shallows and that there is a compensating inflow to the sound through the deeper reef passages. At 5 m depth this inflow is directed eastward, with horizontal circulation cells to each side, however at greater depths the inflow is directed predominantly toward the northeast corner of the basin.

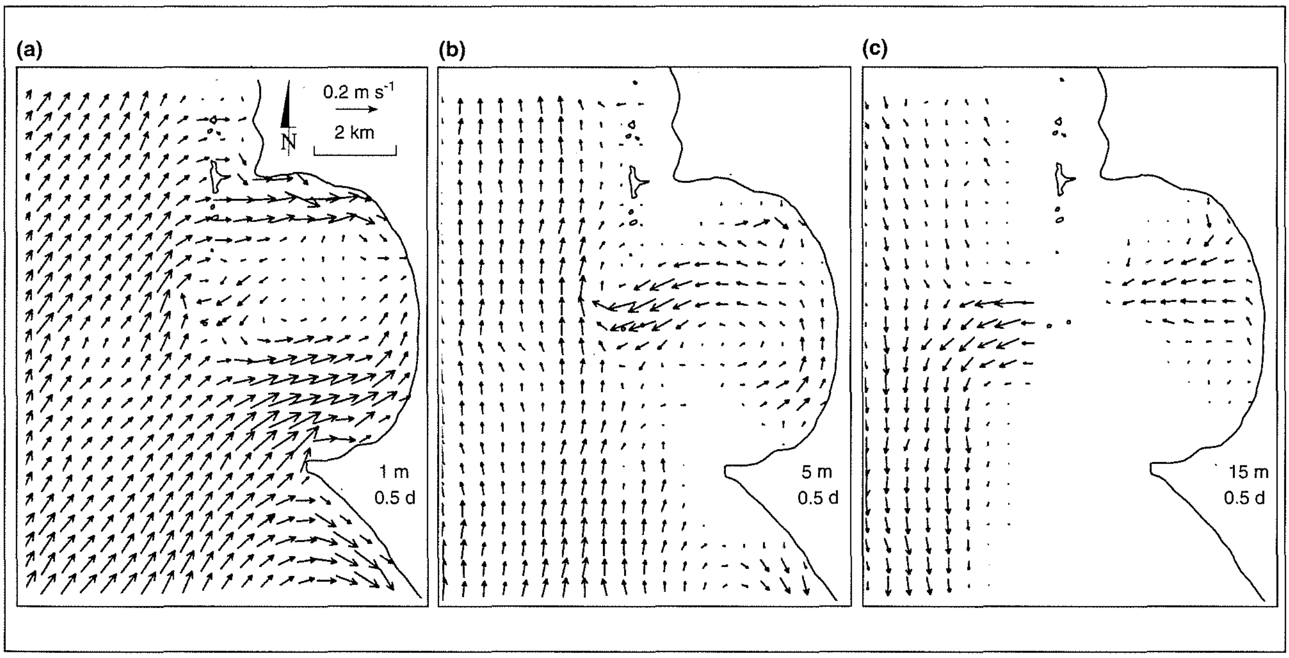


Figure 6.2-50. Baroclinically modelled horizontal velocity fields for a vertically stratified Warnbro Sound after 0.5 days of westerly wind at 7.5 m s^{-1} at depths of (a) 1 m, (b) 5 m and (c) 15 m.

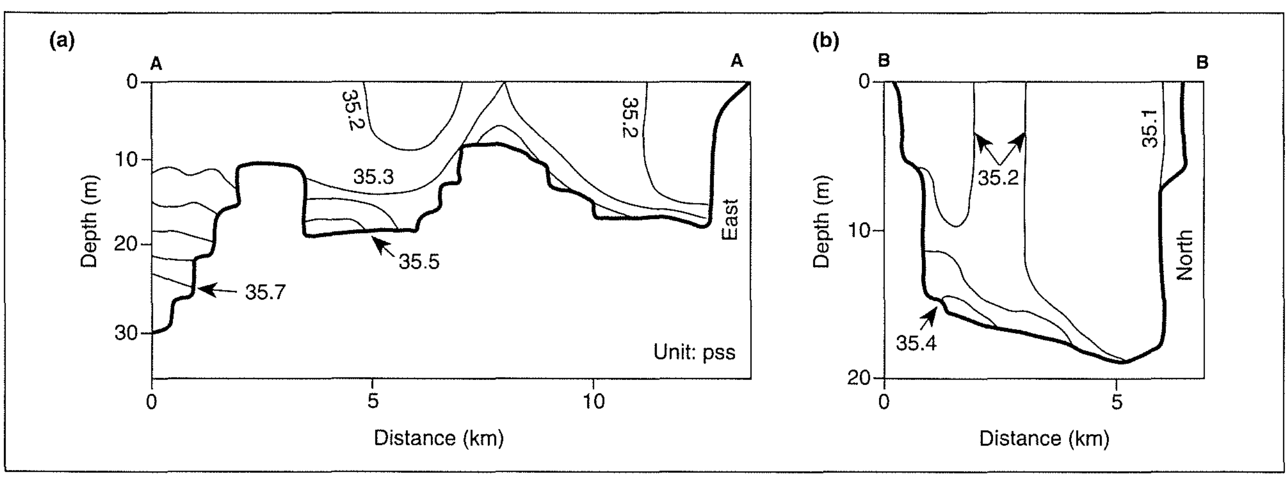


Figure 6.2-51. Baroclinically modelled vertical salinity sections for a vertically stratified Warnbro Sound after 0.5 days of westerly wind at 7.5 m s^{-1} along transects (a) AA, and (b) BB (see Figure 6.2-49).

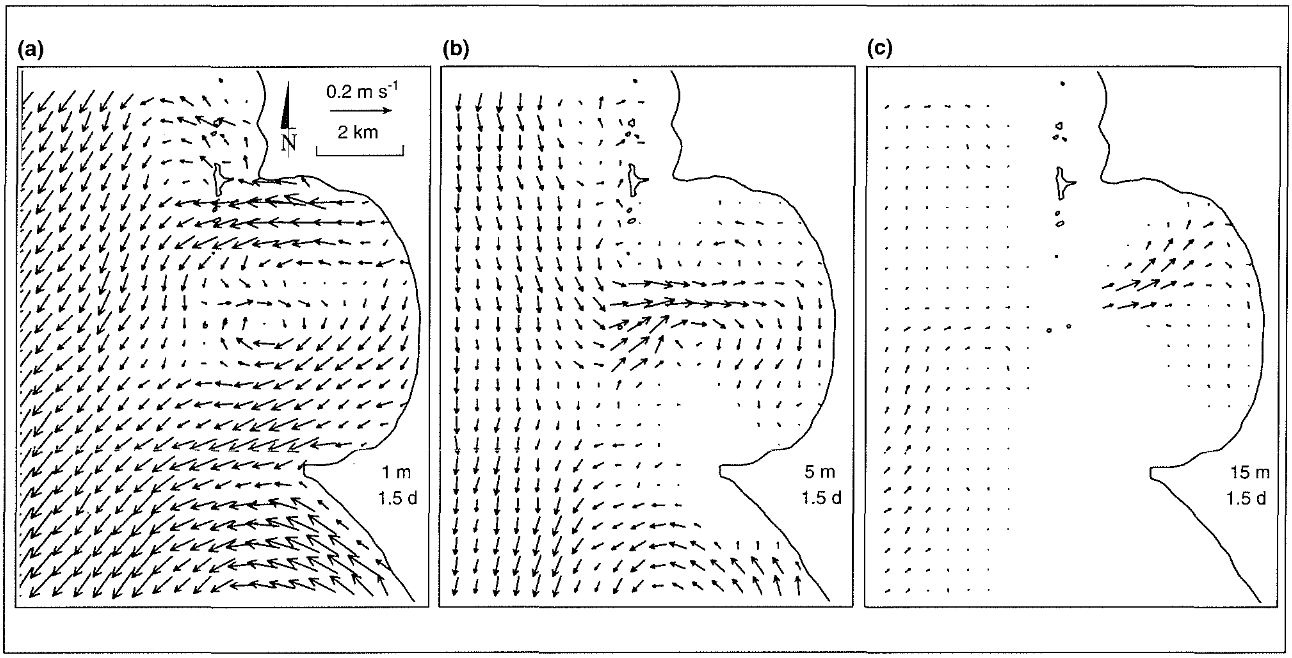


Figure 6.2-52. Baroclinically modelled horizontal velocity fields for a vertically stratified Warnbro Sound after 1.5 days of easterly wind at 5 m s^{-1} at depths of (a) 1 m, (b) 5 m and (c) 15 m.

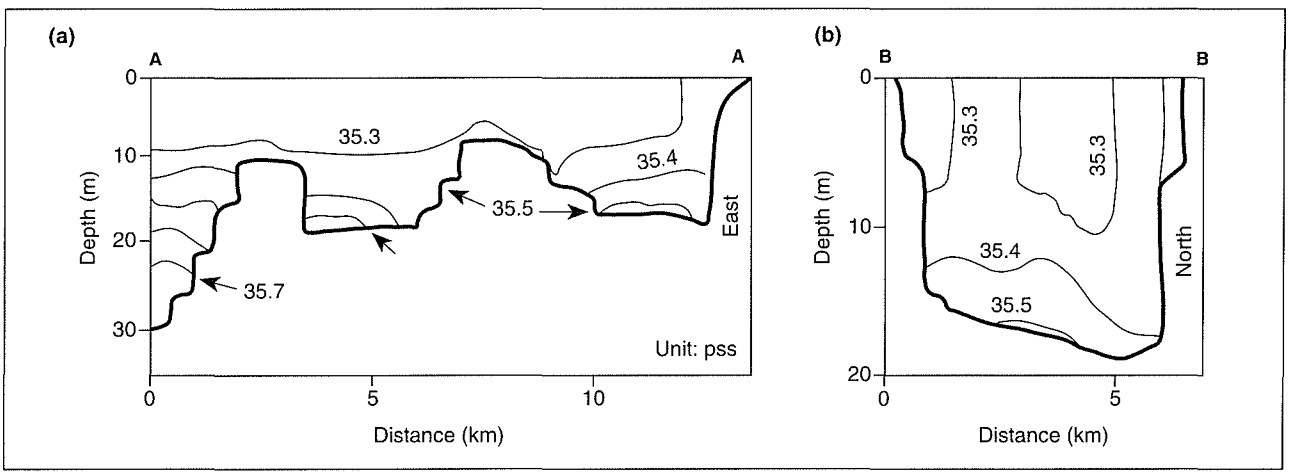


Figure 6.2-53. Baroclinically modelled vertical salinity sections for a vertically stratified Warnbro Sound after 1.5 days of easterly wind at 5 m s^{-1} along transects (a) AA, and (b) BB (see Figure 6.2-49).

Figures 6.2-53a and b show the contours of simulated salinity on the west-east and south-north vertical sections, 36 hours after the wind forcing began. Figure 6.2-53a illustrates downwelling of the salinity structure on the western side of the basin and upwelling to the east under easterly winds, and indicates that the influx through the central reef passages is of low salinity water. The south-north transect (Figure 6.2-53b) shows a buoyant plume-like structure, consistent with this influx, and also indicates downwelling of relatively low salinity water against the northern side of the basin at depths greater than about 10 m, and the displacement of stably-stratified deep basin water to the south. After 36 hours the sound remains vertically stratified at depths greater than about 10 m, indicating a low potential for vertical mixing of the deep basin under these conditions.

Figures 6.2-54a, b and c show circulation patterns at three depths for a steady 7.5 m s^{-1} southerly wind. The flux of near-surface waters between the sound and Sepia Depression is more limited here than for the cases of easterly and westerly winds. The water currents within the main basin of Warnbro Sound are weak; the surface currents are directed downwind and sub-surface recirculation flows upwind. After 12 hours a marked east-west salinity gradient has developed across the entrance to Warnbro Sound (Figure 6.2-55a) as a result of wind-induced upwelling on the eastern side of Sepia Depression and downwelling on the western side of Warnbro Sound. Figure 6.2-55b shows that there is also strong downwelling of the salinity structure to the north of the deep basin and upwelling to the south. Remnant vertical stratification in Warnbro Sound is by this time restricted to a

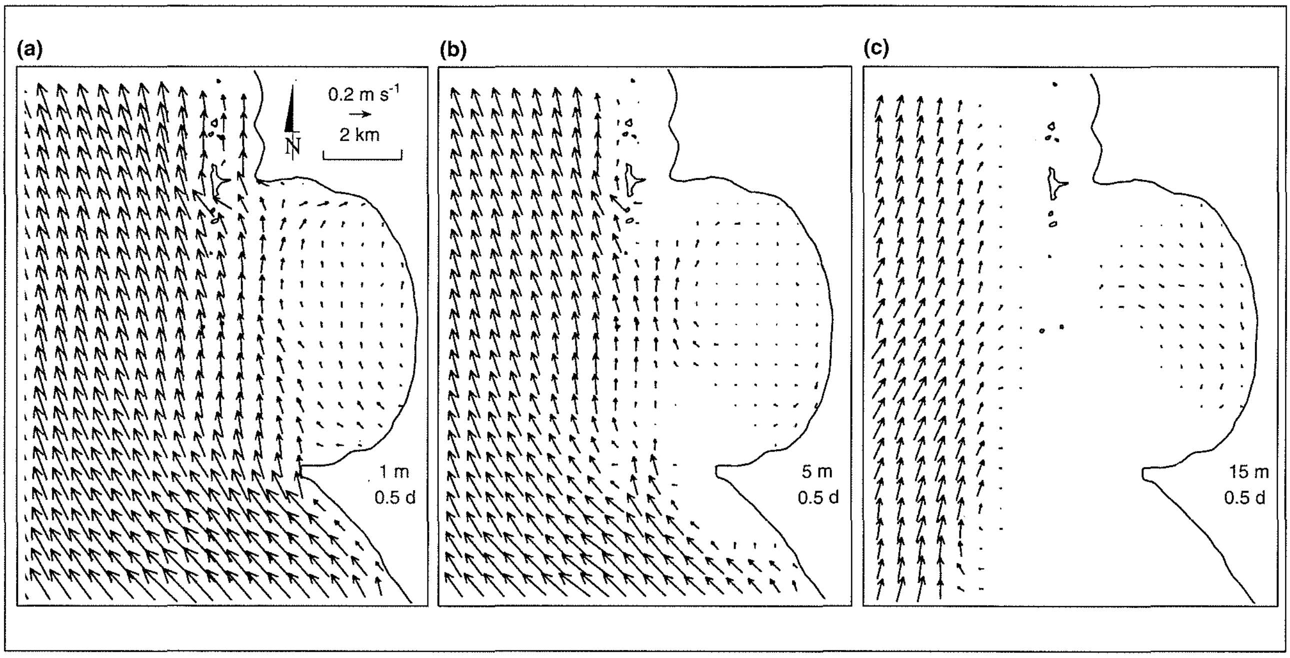


Figure 6.2-54. Baroclinically modelled horizontal velocity fields for a vertically stratified Warnbro Sound after 0.5 days of southerly wind at 7.5 m s^{-1} at depths of (a) 1 m, (b) 5 m and (c) 15 m.

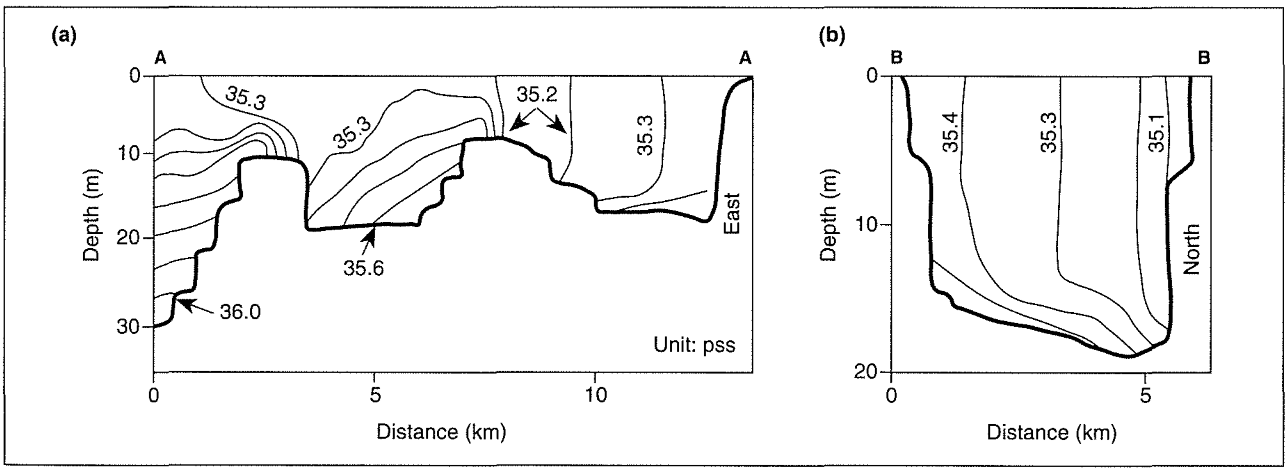


Figure 6.2-55. Baroclinically modelled vertical salinity structures for a vertically stratified Warnbro Sound after 0.5 days of southerly wind at 7.5 m s^{-1} along transects (a) AA, and (b) BB (see Figure 6.2-49).

bottom layer of only a few metres thickness, indicating a rapid rate of vertical mixing under these conditions.

Likewise, for 7.5 m s^{-1} northerly winds the exchange fluxes between the sound and Sepia Depression are limited. After 12 hours the salinity structure shows a marked downwelling to the south and an upwelling to the north of the basin. Upwelling also occurs to the west of Warnbro Sound and Sepia Depression. By this time the initial stratification in the sound has been largely removed except for a layer within a few metres of the seabed.

Influence of buoyant plumes on Warnbro Sound

The baroclinic model was used to simulate the transport of buoyant plumes in winter from the Swan-Canning Estuary, the Peel-Harvey Estuary and Cockburn Sound under variable winds (see sections 6.2.4.5, 6.2.4.6 and 6.2.4.7). Modelling and field studies (see section 5.1.2) have confirmed that these plumes can move long-shore and reach the seaward entrance to Warnbro Sound within 1-2 days, or they may be driven offshore, depending on wind conditions. Hence the entrance to Warnbro Sound is exposed to a rapid succession of water masses with different physical and chemical properties. Modelling also indicated that buoyant plume water entering Warnbro Sound can lower the salinity of near-surface waters and cause major changes in the salinity-density structure of the sound within a day (e.g. see Figure 6.2-13 and 6.2-23). Winter field observations (D'Adamo *et al.* 1995a) confirm that such changes occur on these time scales.

Conclusions

The density and quality of waters off the seaward entrance to Warnbro Sound can vary significantly on time scales of order one day as a result of the unsteady wind-driven transport of water masses originating from estuaries, Cockburn Sound and other embayments, or offshore.

The modelling supported Gersbach's (1993) results on the circulation patterns forced by winds of different directions and extended this work to examine the response of the salinity-density structure to wind forcing. This response reflects the operation of several processes, including horizontal buoyancy flux, tilting of the density structure and vertical mixing. The influence of surface heat and evaporative fluxes on flushing has not been considered and requires further investigation.

The model predicted that, under easterly, westerly and southerly winds applied to a stably stratified Warnbro Sound, incoming water downwells on the northern side of the sound and displaces resident basin water to the south. This suggests that the southern bottom zone of Warnbro Sound has on average a longer residence time than the rest of the sound, and this may be a contributing factor to the observed higher levels of phytoplankton found in the southern part of Warnbro Sound (see section 4.8.2).

6.2.5 Dispersion

Far-field dispersion of contaminant in Cockburn Sound

The mixing and dispersion of effluent discharged from a point source outlet can be broadly considered in terms of three zones or regions (Fischer *et al.* 1979). Near the outlet, the mixing is determined mainly by the momentum and buoyancy of the effluent. For a positively buoyant effluent outlet located on the sea floor, for example, this initial mixing occurs as the effluent rises toward the sea surface in the form of a buoyant jet. Far from the outlet, the transport and mixing of the effluent are governed mainly by the water circulation characteristics and turbulence levels of the general area. The effluent plume in the far-field is no longer significantly influenced by the original density of the effluent or the design details of the outlet. Between the near-field and the far-field zones there is a region of transition where effluent dispersion depends both on the discharge characteristics and on the ambient oceanographic conditions. The purpose of this section is not to examine the individual mixing characteristics in the near-field zone of each outfall, which is generally confined to horizontal length scales of order 100 m, but rather to use the three-dimensional hydrodynamic model to simulate the far-field transport of released contaminant over length scales of order 10 km, and to compare the simulation results with historical field data.

In 1978-79 Cockburn Sound surface seawater samples were taken from a grid of stations with 2 km spacing and analysed for cadmium levels (Rosman *et al.* 1980). The purpose of this study was to map the surface distribution of cadmium throughout Cockburn Sound and to relate it to an industrial discharge of gypsum, located south of James Point, which (at that time) constituted the major cadmium input to the sound (Department of Conservation and Environment, 1979). The measured cadmium concentration field of 28 December 1978 has been used as a basis for comparison with simulated dispersion fields produced by the model. Cadmium had been released to the sound at a rate of approximately 4.5 kg d^{-1} for about one month before the time of these measurements (Rosman *et al.* 1980).

The three-dimensional model was set to run barotropically with a constant source of dynamically-passive tracer released at the location of the gypsum discharge. The model was forced with Garden Island wind data from the period 18-28 December 1978. During this period wind speeds were in the range 0-9 m s^{-1} and the wind underwent two anticlockwise cycles, bringing breezes from all directions, before settling between southwesterly to easterly for several days prior to the time of the cadmium field survey. The near-surface dilution contours after 10.5 simulation days, corresponding to the time of the cadmium survey, are shown in Figure 6.2-56b. At this time the simulated plume extended mainly to the north of the source, with highest concentrations over the eastern margin of the sound. Near Woodman Point, the simulated plume was divided into two branches. One branch crossed the eastern Parmelia Bank and extended further northward into Owen Anchorage. The other branch was transported southward over the central deep basin of Cockburn Sound in response to typical flow recirculation (see, for example Figure 6.2-36) under the prevailing summer wind conditions. Simulated tracer was found to the southwest of the source, and very low concentrations were present external to the sound, within a few kilometres of both the northern and southern entrances. These features of the simulated tracer field were also found in the measured cadmium distribution, illustrated in Figure 6.2-57a. The model indicated that, after 11 days simulation, 80% of the contaminant emitted from the source was still resident in Cockburn Sound. This result is generally consistent with the low flushing rates for the sound, derived from the 'summer' barotropic simulation (see section 6.2.4.8). Quantitative comparisons between the simulated and measured cadmium plume results will therefore require that the model be run for a longer period, taking into account the duration of continuous discharge from the cadmium source and the contaminant flushing rate from the basin.

Conclusions

Water-borne contaminants whose zone of environmental influence extends throughout Cockburn Sound need to be identified and their dispersion modelled at the appropriate spatial scale. Sub-basin scale transport models do not account for recirculation of contaminant which occurs within the

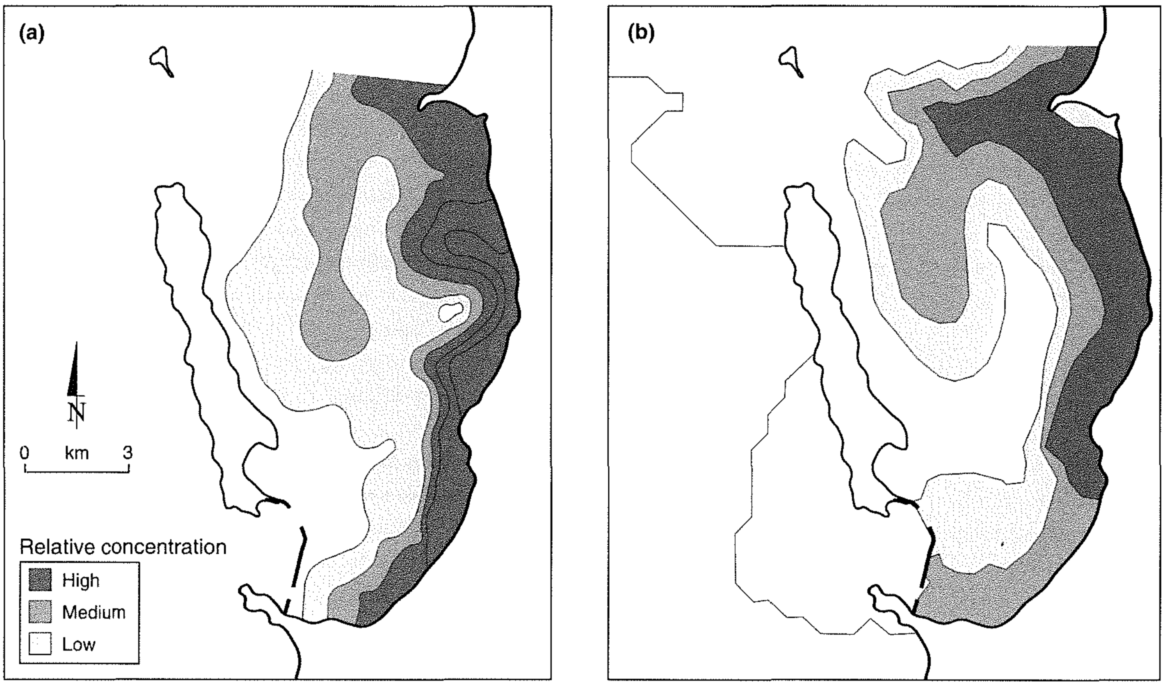


Figure 6.2-56. Far-field dispersion of effluent released from south of James Point: (a) relative concentration of cadmium in surface waters derived from measurements on 28 December, 1978 (adapted from Rosman et al. 1980), and (b) relative tracer concentration patterns in surface waters after a 10.5 day barotropic simulation forced by recorded winds for the period 18-28 December 1978.

sound, but beyond the model domain. Under southerly winds, for example, contaminants may be transported along the eastern margin of the sound before recirculating southward over the central deep basin, or alternatively proceeding further northward into Owen Anchorage.

Contaminant released just south of James Point has a long residence time in Cockburn Sound, under quasi-barotropic conditions. For example, the model predicted that, at the end of an 11-day period in December 1978, 80% of contaminant emitted during that period was still resident in the sound. Modelling of far-field dispersion from a continuous tracer source in Cockburn Sound therefore requires that sufficient simulation time be allowed for the concentration field to develop.

The model was able qualitatively to reproduce the main features of the plume under summer conditions.

The far-field modelling of materials dispersion in Cockburn Sound under conditions typical of the 'autumn' and 'winter-spring' hydrodynamic regimes (see sections 5.1.2 and 6.2.4.8) will require that the model include density effects, to realistically simulate advection and transport fields.

6.3 Ecological modelling

6.3.1 Objectives and approach

The primary objective of the ecological modelling component of the SMCWS was to integrate knowledge of the effects of nutrients on selected aspects of Perth's nearshore marine environment and to use this information to provide estimates of nutrient loads to Cockburn Sound that would be consistent with the draft Environmental Quality Objectives for these waters (see section 3).

A number of complementary approaches to the nutrient-effects ecological modelling were developed in parallel, as part of the overall modelling and integration process. One approach was to utilise the COASEC model (van Senden, 1994) to investigate the effect of a range of nitrogen loading scenarios on sensitive components of the marine ecosystem of Perth's coastal waters (Simpson *et al.* 1993). The COASEC ecological model requires, as input, a simulation of the hydrodynamic behaviour of the study area which is utilised by a 'species transport model' (Knock and Pattiaratchi, 1994) to predict the mixing and dispersion of nutrients and phytoplankton, and hence to simulate the nutrient exposure, algal stimulation and light attenuation conditions in areas of ecological sensitivity. Unfortunately, problems were encountered in the implementation of the species transport model, which meant that the hydrodynamic and ecological models could not be linked within the time-frame available for this study.

As a result, the central ecological sub-models of COASEC were used in conjunction with empirically-derived relationships between nutrient loading and water column response, generated from the results of long-term water quality monitoring programmes in Cockburn Sound (Cary *et al.* 1995a; section 4.7.2), to assist in evaluating management options.

The main ecological sub-models in COASEC were incorporated into a Benthic Site Model (BSM) which operated independently of the species transport model and, instead, received user-specified inputs of key 'forcings' such as nitrogen and chlorophyll *a* concentrations and light (Essers and van Senden, 1994; Masini and van Senden, 1995b). The BSM was originally proposed to provide a means of assessing benthic ecological responses to specified ambient conditions and to assist in the rapid tuning of the model parameters that determine benthic ecological response (Masini and van Senden, 1995b). The general model structure is shown in Figure 6.3-1. The pelagic (planktonic) component of the COASEC model was by-passed in the BSM because the processes of dispersion and advection are critical in determining phytoplankton concentrations in water bodies.

The BSM configuration allows the component sub-models to be decoupled and their performance assessed independently of feedbacks from other components. Combinations of sub-models can be enabled in a sequential manner so that interactions between them can be examined. Through an iterative procedure, the sub-models can be calibrated and validated. Chlorophyll *a* and nutrient exposure can be precisely specified, which facilitates sensitivity analyses and calibration. This capability also allowed the simulated ecological response to specific water quality scenarios to be determined and compared with measured responses to identical scenarios in the field or laboratory which provided a basis on which to assess model performance and reliability. These tests were performed at several levels within the model and many improvements were made on the initial model configuration during this iterative process.

Brief descriptions and the developmental status of the phytoplankton species succession model and the key ecological sub-models in BSM are provided in section 6.3.2. Model development, calibration and validation procedures are presented in section 6.3.3 and the results of ecological simulations for a range of water quality scenarios, appropriate to past, present or target conditions in Cockburn Sound, are presented in section 6.3.4. Section 6.3.5 provides the empirically derived linkages between nitrogen loads and the water quality for Cockburn Sound.

6.3.2 Descriptions of sub-models

6.3.2.1 *Phytoplankton*

The initial phytoplankton model in COASEC was a simple generic formulation, where the nutrient-induced component of growth was directly related to ambient (i.e. external) nutrient concentrations and it did not account for changes in phytoplankton response caused by species succession (Hamilton, 1993). When initially formulated, little was known of the phytoplankton and zooplankton of Perth's coastal waters. The results of phytoplankton and zooplankton studies (sections 4.8, 4.9 and 5.5) and water quality studies (section 4.7) undertaken in the SMCWS provided a characterisation of the dominant phytoplankton assemblages and their seasonality, and indicated that phytoplankton biomass (and hence growth) was not directly related to ambient nutrient concentrations (Masini *et al.* 1992); rather, it appeared that the phytoplankton were able to absorb and store nutrients when encountered (luxury uptake). This internal nutrient store is subsequently utilised for growth when external nutrient concentrations are low. These characteristics were incorporated into a more realistic phytoplankton growth simulation model for Perth's coastal embayments (van Senden, 1995). In this formulation, the silicoflagellate and diatom phytoplankton assemblages are modelled independently and growth is determined by internal nutrient concentrations, photosynthetically available radiation (PAR), water temperature and salinity.

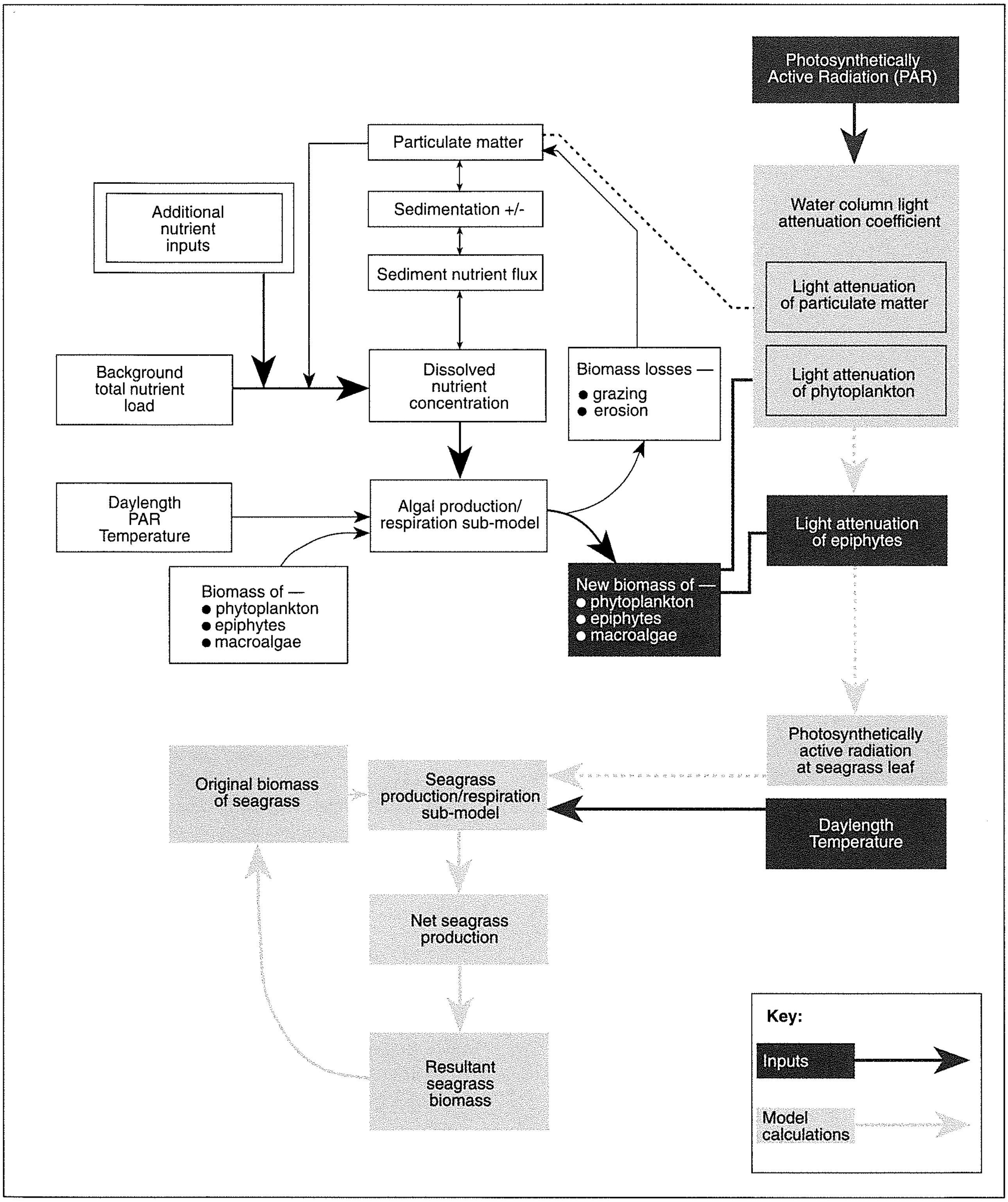


Figure 6.3-1. Conceptual nutrient-effects ecological model. Shaded boxes indicate the Benthic Site Model configuration.

Loss functions include zooplankton grazing, natural mortality and settlement to the seabed. Dead phytoplankton cells contribute to the organic detritus pool in the sediments, and a proportion of their nitrogen content is remineralised and re-enters the water column inorganic nitrogen pool.

Phytoplankton are transported by water movement and, therefore, the phytoplankton species succession sub-model could not be developed and calibrated within the BSM, which has no dispersion component. An alternative test method was devised which uses a constant advection field in a simulated open-ended channel, 10 km long, 2.5 km wide, 15 m deep, and bounded on both sides. The area is divided into cells of dimensions 500 m by 500 m. The water column is divided into two layers and a nitrogen source was inserted into the middle cell in the second row from the upstream end of the channel. This test-bed allows for development and calibration of the model without the need to run the time-consuming transport iterations. In addition, the phytoplankton response, in terms of growth and species succession, to changing parameter settings can be more readily assessed, as the nitrogen source and current velocity are user-defined. A seasonal salinity signal (34-36.6 pss), based on long-term average salinity data for Owen Anchorage/Cockburn Sound (D'Adamo, 1992) was incorporated as a physical forcing.

Initial parameter settings were based on a literature review of available information on the species characterising the dominant phytoplankton assemblages in the southern coastal waters of Perth (Helleren and John, 1994). The initial test results highlighted the sensitivity of the model to the parameter settings for nitrogen uptake rate and the conversion rate of phytoplankton nitrogen to dissolved inorganic nitrogen. The model requires further development, calibration and validation before it can be linked to the hydrodynamic model outputs through the species transport model and used for ecological simulations. When this model is incorporated into COASEC, the sum of the biomass of silicoflagellates and diatoms, expressed as chlorophyll *a*, will provide an input to the light attenuation sub-model described below.

Further development of the phytoplankton species succession sub-model is being undertaken in collaboration with the Centre for Water Research at the University of Western Australia.

6.3.2.2 *Epiphytes*

The epiphyte sub-model is constructed to simulate the growth of the epiphytic algal community growing on seagrass leaves. The ecological importance of the epiphyte community in the model framework is linked to its light absorption characteristics and its effect on the amount of light available for photosynthesis and growth of the seagrass host (Silberstein *et al.* 1986; Burt *et al.* 1995a; see section 5.3.2). Epiphyte communities consist of many different algal species that can change seasonally and have differing responses to physical and chemical conditions found in the environment. In addition, the composition of the epiphyte assemblage will influence its light attenuation characteristics (Burt *et al.* 1995a). Unlike the phytoplankton sub-model, epiphyte assemblages are not explicitly differentiated in this sub-model; instead, the growth algorithms were formulated to include these changes implicitly (see below). Epiphyte growth in the model is determined by ambient nitrogen concentration, PAR and water temperature; losses occur through respiration and by biomass-dependent erosion/abrasion. Grazing by algal herbivores can also be invoked by switching on the grazer sub-model. The output of the epiphyte sub-model is expressed as epiphyte biomass per square centimetre of seagrass leaf, and this output is used in the light attenuation sub-model, described below, to predict light reduction through the epiphyte layer, and hence, the light available for seagrass growth.

6.3.2.3 *Light attenuation*

This sub-model simulates the attenuation of PAR as it passes between the water surface and the seagrass leaf blade. The transparency of the water column is related to the colour of the water and to the concentration of phytoplankton and of organic and inorganic particles in suspension. The epiphytic coating on the seagrass leaf surface also attenuates light and the degree of shading that occurs is related to the biomass of epiphytes present. The amount of PAR reaching seagrasses is related to the combined effects of light attenuation through the water column and through the epiphytic coating on their leaves. This resultant PAR value is used in the seagrass sub-model to calculate photosynthetic production.

6.3.2.4 *Seagrasses*

The seagrass sub-model simulates the growth response of the meadow-forming seagrass *Posidonia sinuosa* which is a dominant meadow-forming species in Perth's coastal waters. This species is sensitive to eutrophication (Shepherd *et al.* 1989) and, like all plants, its survival is dependent on acquiring sufficient PAR (Duarte, 1991). In the sub-model, seagrass growth is determined by PAR at the leaf epidermis, and by water temperature, which influences rates of photosynthesis and respiration (Masini *et al.* 1995b; Masini and Manning, 1995b). Light requirements and metabolic rates are biomass-dependent and seagrass biomass is lost through erosion. Seagrass growth is not affected directly by nutrient concentrations; rather, the seagrass growth sub-model provides a mechanism to integrate and quantify the environmental implications for benthic communities of the direct effects of nutrient enrichment on the phytoplankton and epiphyte communities described in the sub-models above.

6.3.3 Development, calibration and validation

6.3.3.1 *General approach*

The input files for the BSM were constructed to simulate the primary forcings at sites where both the forcings and the biological responses were well documented. The two sites selected were located on the northern and southern slopes of Success Bank, at the local maximum depth limit of seagrass survival (Section 5.3 and 5.4). Input files of global radiation, chlorophyll *a* and inorganic nitrogen concentrations, and water temperature were constructed to represent the conditions measured at these locations. The predicted biological responses (e.g. epiphyte biomass, seagrass growth) to these 'forcings' were compared with corresponding measurements at the sites, and elsewhere, to calibrate and validate the sub-models. The input data files and the procedures involved in developing each of the main sub-models are described below.

6.3.3.2 *Input data files*

The two primary physical inputs to the BSM are photosynthetically available radiation (PAR) and water temperature. PAR is input to the model as 30 minute averages. Comparisons between data from Perth Airport, situated approximately 20 km inland, the Department of Environmental Protection Hope Valley monitoring station, approximately 2 km inland, and measurements at the field monitoring site in Owen Anchorage, indicated that the Hope Valley data were more representative of the conditions in Owen Anchorage. On this basis data from Hope Valley, for the period 1 July 1992 to 30 June 1993 (the main period of biological and physical measurements at the sites in Owen Anchorage; section 5.3), were selected as the annual global radiation input data set (Figure 6.3-2a).

In order to provide a representation of the annual water temperature cycle, a cosine function was fitted, using the method of least squares, to water temperature data from surveys of Cockburn Sound and Warnbro Sound and from current meters stationed in the Cockburn Sound basin (Figure 6.3-2b). The fitted annual temperature cycle was found to range from a minimum of 15.8 °C on day 226 (August 12) to a maximum of 23.35 °C on day 46 (February 15) (Masini and van Senden, 1995b). This functional representation was used to predict daily water temperatures for input to the model. Diel variations in water temperature are not included.

6.3.3.3 *Light attenuation*

Algorithm development

Water column light attenuation is represented in the model as a function of the concentrations of phytoplankton (measured as chlorophyll *a*), organic and inorganic particulates, and of the light absorbing properties of the water itself. The total vertical light attenuation of the water column is the product of the attenuation imparted by each of these components. The relative influence of each component on the total attenuation is determined by the partial extinction coefficient and concentration of that component. In the model, light intensity at any depth is a function of the PAR intensity at the water surface, the total attenuation coefficient of the water column and water depth (van Senden, 1994).

Sensitivity analyses and parameterisation

The results of long-term water quality monitoring programmes (see section 4.7.3.3) and light attenuation process studies (see section 5.3.1) indicate that phytoplankton, measured as chlorophyll *a*, is correlated with the vertical light attenuation in the waters of the study area. On this basis, in the simulations described here, light attenuation through the water column is controlled by water 'colour' and by phytoplankton concentration. Partial attenuation coefficients for water colour and for chlorophyll *a* were established from analyses of water quality data collected at the Owen Anchorage site (Figure 5.3-2; Burt *et al.* 1995b) and during long-term water quality monitoring programs conducted in Perth's southern metropolitan coastal waters over approximately 20 years (Cary *et al.* 1995a).

Time-series of chlorophyll *a* for input to BSM were constructed for the Success Bank and Owen Anchorage (Figure 6.3-2c) monitoring sites by fitting a mathematical function to chlorophyll *a* data collected at approximately five-day intervals over four six-week periods during 1992/93 (Burt *et al.* 1995b). When these chlorophyll *a* time-series were used as input to the light attenuation sub-model PAR intensity tended to be over-predicted by between 4% and 10%, but coefficients of determination (r^2) between predicted and measured average daily PAR were high and similar at both sites, approximately 0.96 for a depth of 3 m and 0.8 at depths of 12 m and 15 m in Owen Anchorage and Success Bank, respectively (Figure 6.3-3).

Simulated water column vertical light attenuation coefficients (LAC) were significantly correlated with measured LAC on a daily basis, and also when expressed as averages for the deployment periods over the annual cycle for the Owen Anchorage site (Figure 6.3-4a-c). The correlations on a daily and average basis for the more exposed Success Bank site were poor (Figure 6.3-4d-f) as suspended particulate concentrations are known to be important in determining light attenuation at this site (Burt *et al.* 1995b) but were not included as forcings in this simulation. The relatively weak correlations between predicted and measured LAC reflect the low seasonal, but high day-to-day variation in LAC at both sites. This high degree of day-to-day variation is not incorporated in the smooth chlorophyll *a* forcing data (Figure 6.3-2c).

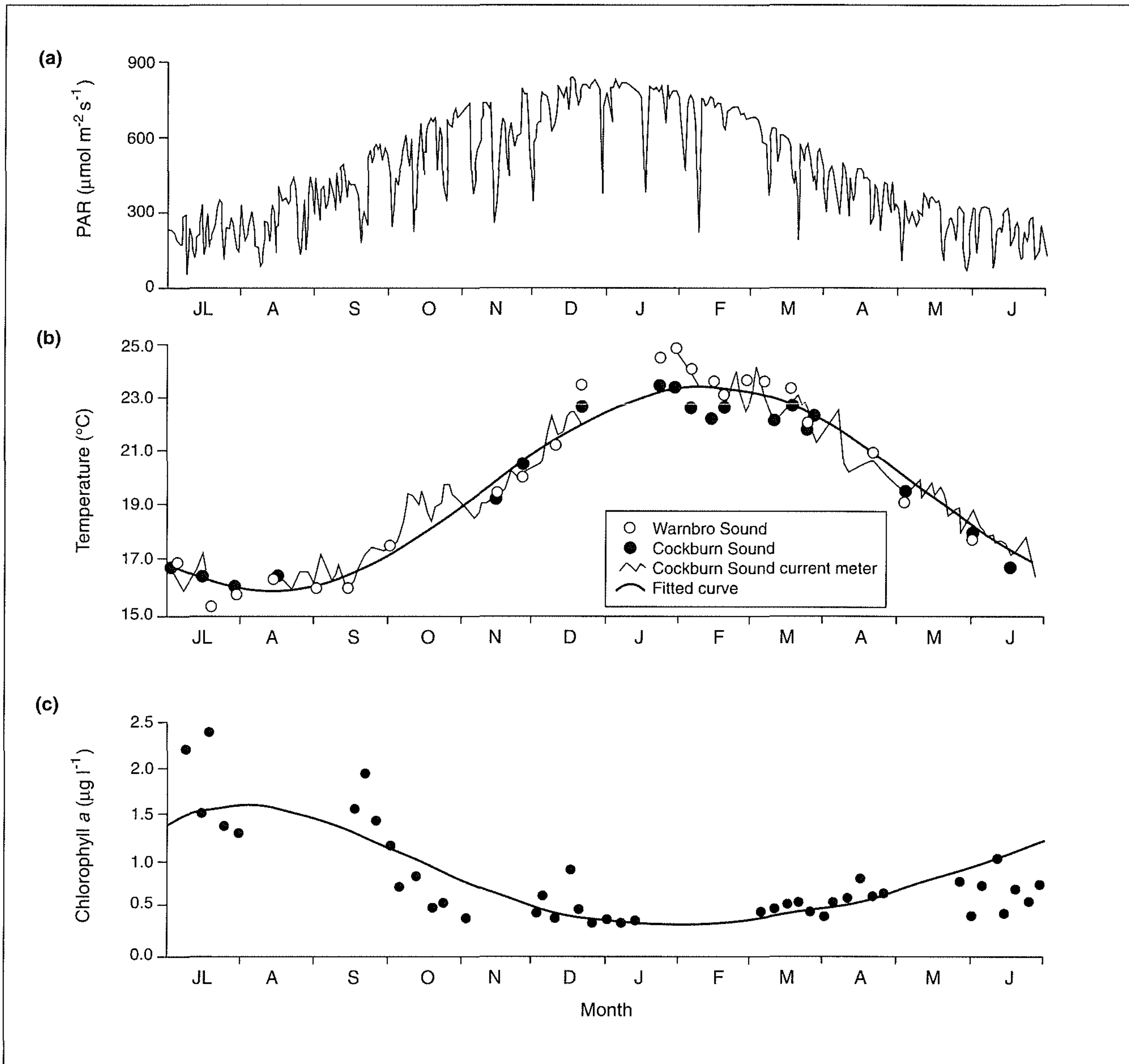


Figure 6.3-2. Input data for the Benthic Site Model used to represent an inshore site in Owen Anchorage: (a) daily average photosynthetically active radiation (PAR) and fitted curves for (b) water temperature and (c) chlorophyll a. Site location is shown in Figure 5.3-2.

To assist in calibrating the seagrass and epiphyte sub-models, another time series of chlorophyll *a* was constructed so as to include day-to-day variation and to reproduce as accurately as possible the measured bottom light field when input to the light attenuation sub-model.

6.3.3.4 *Epiphytes*

Algorithm development

In the model, epiphyte growth is limited by either nutrient or light availability. The light-related aspects of growth are predicted from a set of algorithms describing the effect of temperature on the point of onset of light saturation (I_k), the maximum photosynthetic rate (GP_{max}) and the respiration rate. These algorithms were developed using laboratory-derived data on the photosynthetic responses of epiphyte assemblages collected from the Marmion Marine Park during winter, spring and summer (Manning, 1994). Data on the photosynthesis-irradiance (P-I) responses of the winter assemblage at 13 °C, the spring assemblage at 18 °C and of the summer assemblage at 23 °C, were combined to produce a composite temperature response in an attempt to implicitly incorporate any seasonal changes in epiphyte species composition and/or P-I responses that might have occurred. The mean photometabolic parameter values measured in the laboratory (Manning, 1994) were used for all simulations.

The nutrient-related aspects of growth are predicted by a Michaelis-Menten relationship which is described by a half saturation constant (K_S) which represents the nutrient concentration at which the growth rate is half of the maximum attainable. Field experiments conducted in Cockburn Sound have shown that erosion/abrasion provides a very strong control on epiphyte biomass, even in areas

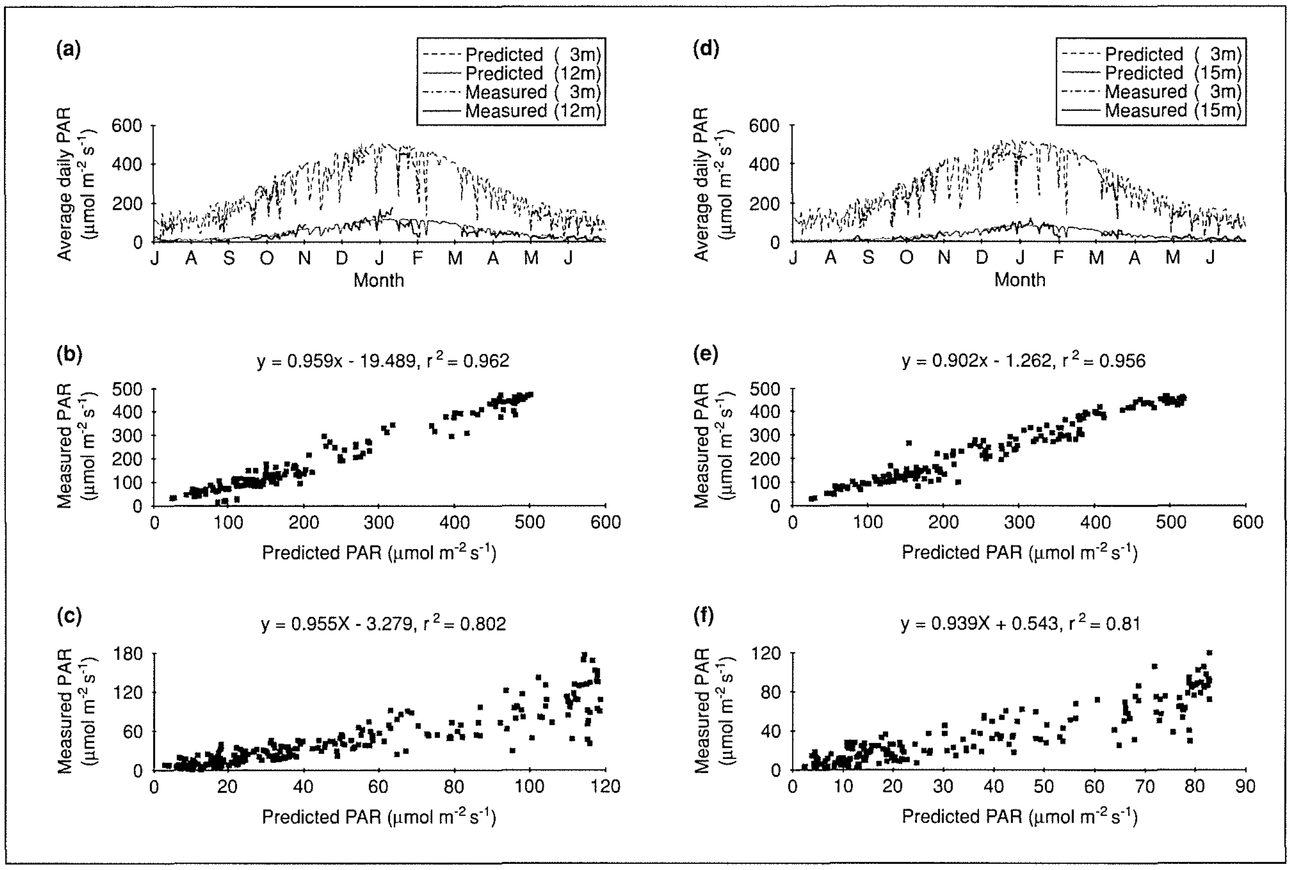


Figure 6.3-3. Light attenuation sub-model results: simulations of an inshore site showing (a) time series of measured and predicted photosynthetically active radiation (PAR) at depths of 3 m and 12 m, and relationships between predicted and measured PAR for depths of (b) 3 m and (c) 12 m. Simulations of an offshore site showing (d) time series of measured and predicted PAR at depths of 3 m and 15 m, and relationships between predicted and measured PAR for depths of (e) 3 m and (f) 15 m. Site locations are shown in Figure 5.3-2.

relatively sheltered from long-period waves (Burt *et al.* 1995c). In a 26-day trial, the biomass of periphyton on artificial seagrass leaves protected from abrasion was up to 100 times greater than on others allowed to move freely. The degree of abrasion is likely to be related to physical energetics at the site (e.g. degree of exposure to waves, turbulence) and this is generally reflected in the structure of the epiphyte assemblage with encrusting algae dominant in high energy environments (see section 5.3.2). The strong influence of physical processes in controlling epiphyte biomass is incorporated in the model as a strongly biomass-dependent erosion/abrasion rate. Light attenuation through the epiphyte layer was predicted from biomass expressed as total dry weight as this provided a relationship that was independent of season or assemblage (Figure 5.3-7, section 5.3.2).

Sensitivity analyses and parameterisation

Incident PAR and chlorophyll *a* input files were constructed to reproduce as far as possible the PAR climate at the Owen Anchorage and Success Bank sites (see above) and three nitrogen concentrations (5, 10 and 15 µgN l^{-1}) were used in separate simulations. Initial coarse scale calibration was performed using these input files, initially to set the P-I parameters, and then to assess the effect of different nutrient concentrations. The half saturation K_S for nitrogen-related growth of an epiphyte community dominated by coralline algae appears to be less than 20 µgN l^{-1}, based on mesocosm experiments (Walker *et al.* 1994b), and was initially set at 6 µgN l^{-1} which is a 'typical' total inorganic nitrogen concentration for offshore coastal waters unaffected by anthropogenic nutrient inputs (Cary *et al.* 1995b). Sensitivity analyses indicated that the average epiphyte biomass was very sensitive to the value selected for the 'E2' constant in the equation describing biomass-controlled erosion rate (see van Senden, 1995) which in turn had the effect of 'capping' the maximum attainable biomass. The effect on epiphyte biomass of increasing ambient nitrogen concentration from 5 to 10 µgN l^{-1} became more obvious and pronounced as depth increased (Figure 6.3-5a). The maximum depth cutoff limit increased by about 4 m (30%) and the resultant biomass after five simulation years was between 25 and 40% higher between 1 m and 12 m depth. The response to increasing ambient nutrient concentration from 10 to 15 µgN l^{-1} was very similar, but the effect was much less pronounced. The magnitude of the seasonal change in biomass over an annual cycle was least at high nitrogen concentrations and at shallow water depths and reflects the biomass 'capping' effect caused by the current form of the relationship for epiphyte biomass-controlled erosion/abrasion (Figure 6.3-5b).

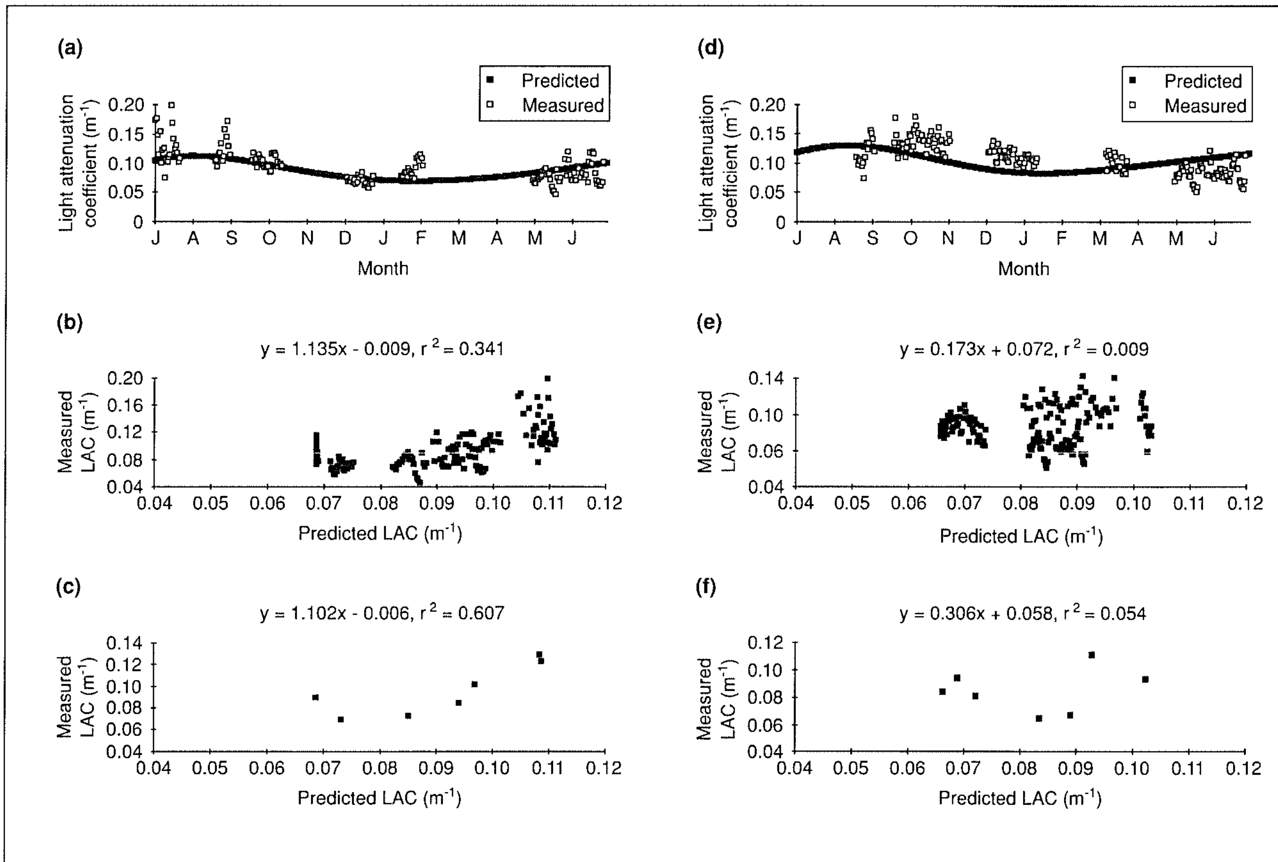


Figure 6.3-4. Light attenuation sub-model results: simulations of an inshore site showing (a) time series of measured and predicted light attenuation coefficient (LAC), and relationships between predicted and measured LAC (b) on all days and (c) as means for seven periods throughout the year. Simulation of an offshore site showing (d) time series of measured and predicted LAC, and relationships between predicted and measured LAC (e) on all days and (f) as means for seven periods throughout the year. Site locations are shown in Figure 5.3-2.

The sensitivity of the model to the K_S parameter setting was assessed by comparing the distributions of epiphyte biomass with depth, using an ambient nitrogen concentration of 10 µgN l^{-1} and K_S settings of 5, 10 and 15 µgN l^{-1}. When K_S is much lower than the ambient N concentration, nitrogen is largely non-limiting and growth rate is primarily determined by PAR availability. In contrast, when the K_S is greater than the ambient nitrogen concentration (e.g. 15 µgN l^{-1}; Figure 6.3-6) nitrogen limitation will restrict gross photosynthetic production to less than 50% of the maximum attainable, even when PAR availability is non-limiting. Respiration is a function of water temperature and can be regarded as a constant in these comparisons as it is not affected by PAR or nutrient availability. The depth distributions in Figure 6.3-6 reflect these characteristics of the current model formulation and show the pronounced effect of the biomass-dependent erosion rate which, in this case, becomes very apparent when biomass increases beyond about 1.1 mg cm^{-2} and limits the maximum attainable biomass to below 2 mg cm^{-2}.

Simulations

Nitrogen concentrations in Perth's coastal waters can vary considerably from one sampling day to the next but a clear seasonal pattern is evident with periods of elevated concentrations occurring during winter (Cary *et al.* 1995b). The effect of variable nitrogen concentrations on epiphyte biomass was assessed using an alternative set of input files that simulated the seasonal trends in nitrogen concentrations measured in bottom waters at the two sites described previously (Burt *et al.* 1995). The annual cycles of light, nutrients and temperature were repeated over five consecutive simulation years and the predicted biomass during the fifth simulation year was compared with the measured biomass at the two sites. Initial biomass was set at 1 mg cm^{-2} of seagrass leaf (i.e. moderate epiphyte biomass), the K_S set at 6 µg l^{-1} and the erosion set to cap biomass below about 1.4 mg cm^{-2}. All other parameter settings were identical to those used in the epiphyte sub-model simulations described above.

Results of simulations

In general, the temporal patterns of simulated epiphyte biomass were similar at the two sites (Figure 6.3-7). Plots of epiphyte biomass at different depths diverged at the beginning of the simulation and stable, depth-related seasonal patterns emerged after about two simulation years. At both sites, the lowest seasonal variations in biomass occurred at shallow water depths with little difference in the seasonal pattern or average biomass at depths down to about 8 m. Biomass tended to fluctuate more over the annual cycle

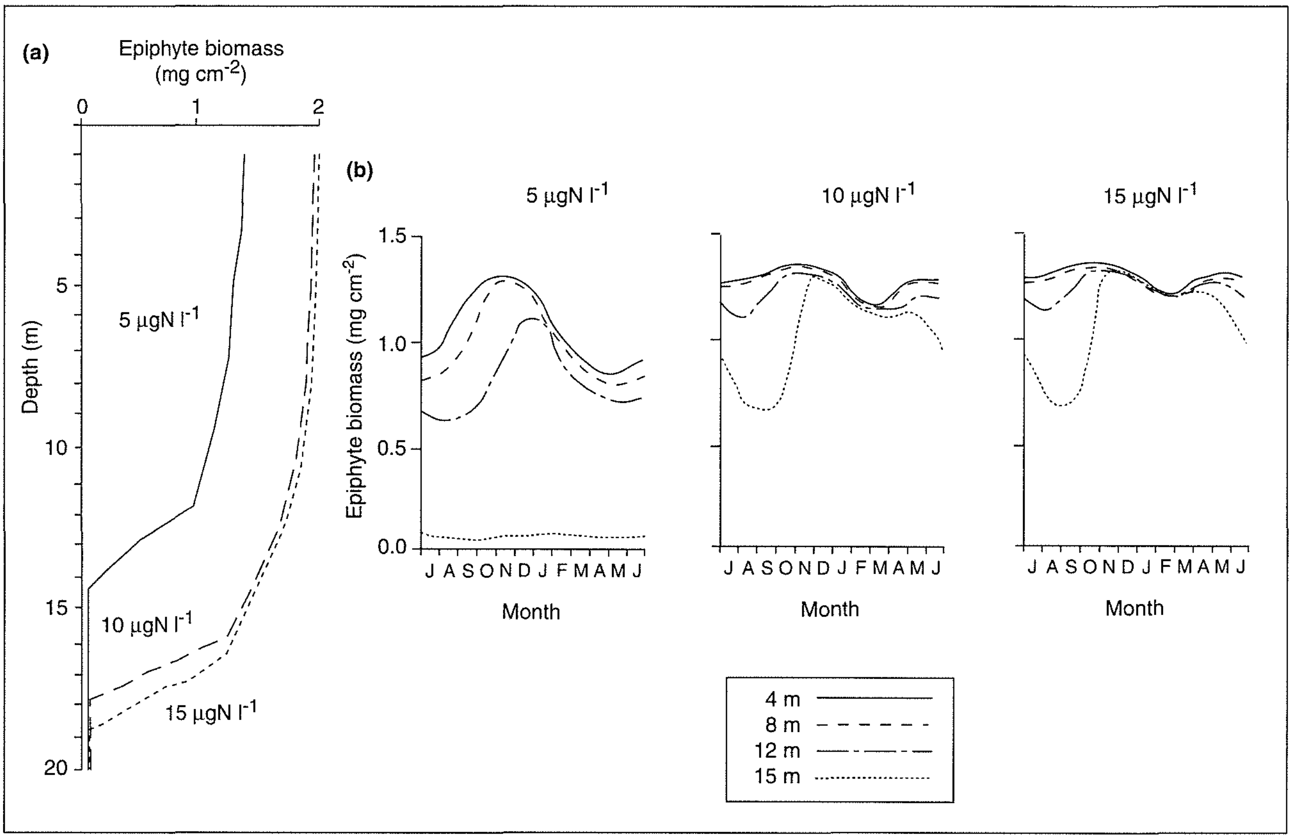


Figure 6.3-5. Epiphyte sub-model results: (a) relationship between epiphyte biomass and depth for three ambient nitrogen concentrations after five simulation years and (b) seasonal change in biomass at four depths for three ambient nitrogen concentrations during the fifth simulation year.

as water depth increased, presumably due to light limitation. There were considerable differences in the seasonal patterns of epiphyte biomass at different depths. A minimum biomass was found in mid-autumn and a maximum occurred in spring or summer, depending on water depth. A secondary seasonal biomass minimum at the end of winter became increasingly apparent below 8 m and by 15 m this became the seasonal minimum (Figure 6.3-7). At the 15 m water depth, the maximum was also better defined and occurred during mid to late summer.

The measured seasonal changes in biomass were different at the two sites; minimum during winter at the Owen Anchorage site and during summer at the Success Bank site. These differences are attributed, in part, to differential exposure to waves and, to a lesser extent, dissimilar patterns of algal propagule establishment (Burt *et al.* 1995a), both of which are not simulated in the present formulation of the epiphyte model. Exactly what comprises a 'typical' seasonal pattern in epiphyte biomass remains unresolved, as there are insufficient data to make generalisations that could be broadly applied. Hillman *et al.* (1994) reviewed available data and found some consistency for epiphyte biomass on *Amphibolis* stems, with winter minima and late spring/early summer maxima, whereas in other studies of periphyton and epiphytes on leaves, minima occurred in late summer and maxima in late winter/early spring.

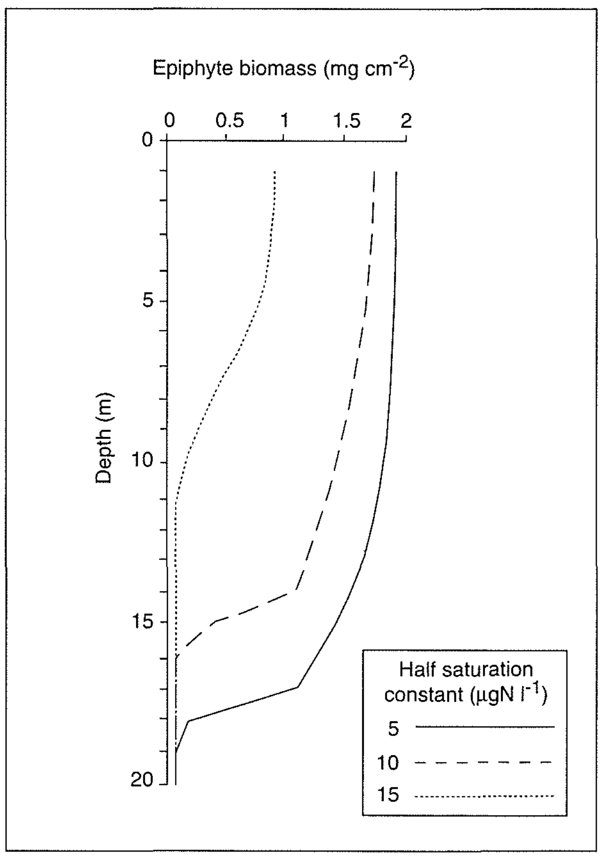


Figure 6.3-6. Epiphyte sub-model results: the relationship between epiphyte biomass and depth for three settings of the half-saturation constant for nitrogen-limited growth after five simulation years.

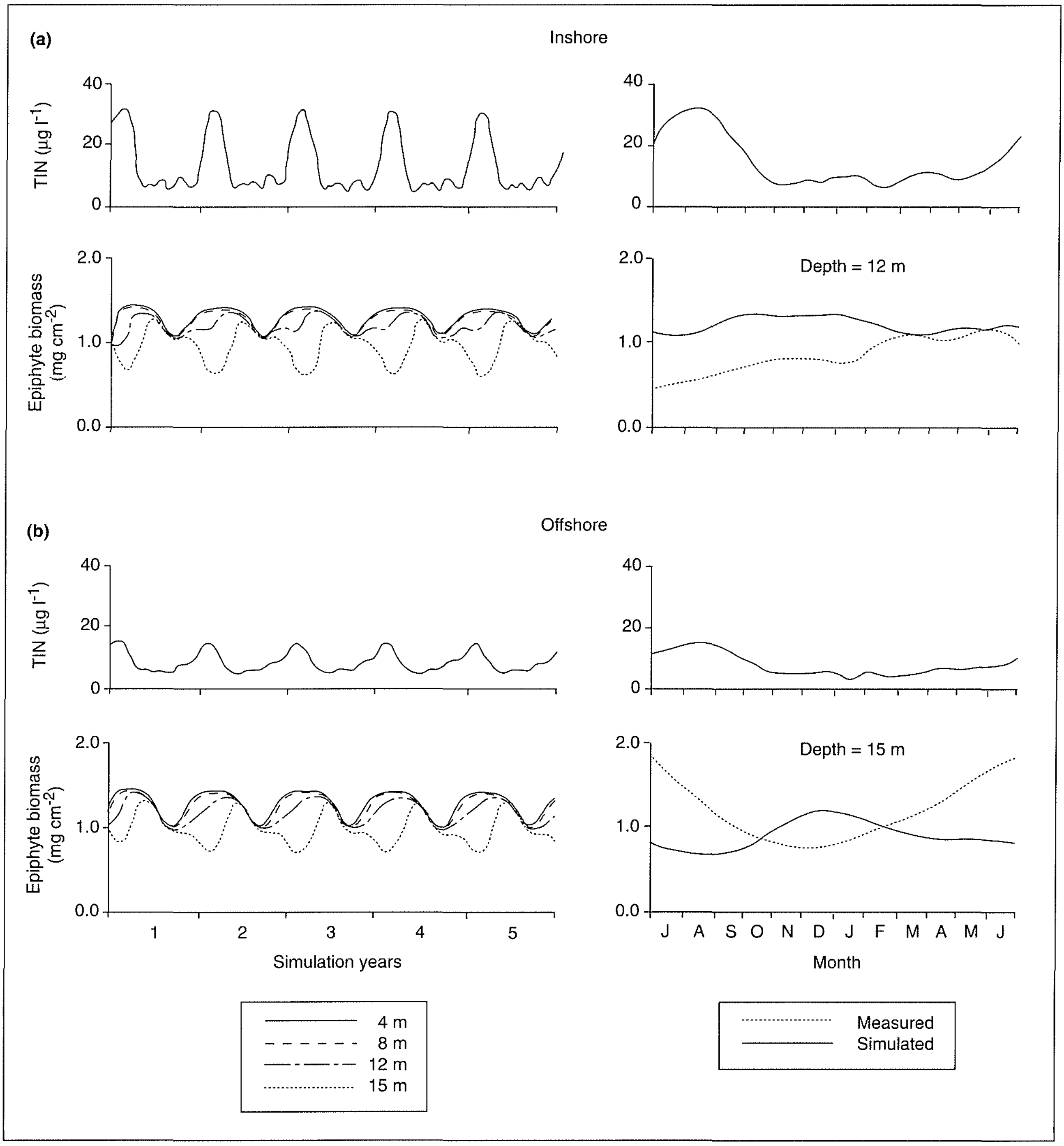


Figure 6.3-7. Epiphyte sub-model results: time series of total inorganic nitrogen (TIN) concentration as model input and simulated epiphyte biomass at four depths over a five year simulation period, and measured and simulated epiphyte biomass for the fifth simulation year for (a) an inshore site and (b) an offshore site. Site locations are shown in Figure 5.3-2.

The output of the epiphyte model is most consistent with the data of Kirkman (1981) and Edgar (1990) as summarised by Hillman *et al.* (1994) for epiphytes growing on *Amphibolis* seagrass stems. The average biomass predicted was in the same order as that measured at the sites but this was largely due to the values selected for the biomass dependent erosion/abrasion parameters.

Although the predicted biomass from the model was of a similar order to that measured (Figure 6.3-7), the reliability of the current formulation of the epiphyte model is considered to be low and results obtained should be interpreted with caution. The influences of erosional/abrasional processes are now known to be critical in controlling epiphyte biomass (Burt *et al.* 1995c) and species composition (Burt *et al.* 1995a). Subsequent modifications of the epiphyte model should take these features into account and include an energy-dependent erosion factor (e.g. related to wave height). Selection of the appropriate K_S value is also important and will require further investigation.

6.3.3.5 *Seagrasses*

Algorithm development

Laboratory-derived photosynthesis-irradiance data for *Posidonia sinuosa* (Masini and Manning, 1995a,b) were analysed to determine the functional form of the P-I relationship for this species that would provide the best fit to the data over a range of water temperatures. A hyperbolic tangent function provided the lowest error (mean and rms) and highest correlation at the three water temperatures examined (Masini and van Senden, 1995a). Algorithms incorporating the effect of temperature on the point of onset of light saturation (I_k), the maximum photosynthetic rate (GP_{max}) and the respiration rate were included (Figure 5.2-1) as well as a biomass dependence effect on I_k and GP_{max} (Masini and van Senden, 1995a). These latter relationships were formulated to describe the combined effect of seagrass leaf density (Masini *et al.* 1995b) and the intrinsic difference in photosynthetic response of plants near the lower end of natural depth gradients (Masini and Manning, 1995b).

Sensitivity analyses and parameterisation

To help develop the seagrass sub-model a version was configured (Masini and van Senden, 1995a) to allow simulations to be performed using light data measured at a 20-minute frequency at the seagrass growth measurement site in Owen Anchorage (Masini, 1994). Sensitivity analyses were performed on each of the tunable parameters and the range of parameter settings used was kept within the 95% confidence bounds of the mean laboratory-derived values.

Photosynthetic rates were most sensitive to respiration and growth rate settings, whereas biomass changes were most sensitive to these and the erosion rate setting. Comparisons between 20 and 30 minute PAR timesteps indicated that the 20-minute timestep produced daily average photosynthetic rates that were 3% to 8% lower than, and closer to, the measured values than the 30-minute timestep (Masini and van Senden, 1995a).

Calibration

The seagrass growth model only accounts for above-ground production whereas the below-ground roots and rhizomes also need photosynthate to meet their respiratory and growth requirements. When this and other aspects of the plant's ecophysiology, such as the seasonal storage and utilisation of carbohydrate reserves and reproduction are taken into account the leaf extension to photosynthetic production ratio is about 0.4 (Masini and van Senden, 1995a).

These results, coupled with the results of the sensitivity analyses were used to establish another set of parameters, and the model was run with physical forcings from the Owen Anchorage site for deployments spanning the summer and winter solstices and the autumn equinox. The predicted seagrass productivity and biomass data were plotted against measured data to compare the degree of correlation within and across the simulation periods (Figure 6.3-8a,c and d).

The simulated seagrass growth rates, expressed as a percentage of biomass present and also on a unit area basis, were very similar in magnitude to the measured data for the summer and autumn simulation periods, however during the winter period, growth rates were underestimated (Figure 6.3-8a,c). Predicted biomass changes were less variable than measured in the field and hence were poorly correlated within sampling periods, but the correlations were strong (r^2=0.91) when data from the three sampling periods were pooled (Table 6.3-1). These results provide a reasonable degree of confidence in the biomass estimates generated by the seagrass growth model over an annual cycle.

The seagrass sub-model was run over five consecutive simulation years in the BSM using an initial biomass of 200 g m^{-2}, no epiphyte shading, and the parameter settings providing the best fit between predicted and measured data for the three simulation periods, as described above. Timeseries of chlorophyll *a* concentrations and Hope Valley PAR data were used to simulate the light climate at the Owen Anchorage and Success Bank monitoring sites. Seagrass biomass followed a typical seasonal growth pattern with late summer maxima and late winter minima at all depths (Figure 6.3-9a). Within one simulation year the seagrass biomass had started increasing in shallow water and decreasing in deeper water. The rate of change in mean annual seagrass biomass at each depth slowed substantially within about three simulation years and was reasonably stable after about five simulation years (Figure 6.3-9b) producing a depth distribution which is consistent with depth distributions determined in field studies in other areas (e.g. West, 1990). The seagrass biomass at the end of the five-year simulation (day 1826, mid-winter) was very similar to the average biomass during the final simulation year which facilitates interrogation of model results (Table 6.3-2).

6.3.3.6 *Evaluation of model utility*

The ecological sub-models described above provide a framework to integrate current understanding of key ecological processes operating in Perth's coastal waters and, through the development and evaluation procedures, serve to highlight inadequacies in this understanding. Once identified, these inadequacies can be redressed by obtaining appropriate information through focussed and tactical monitoring programmes. Qualitative estimates of the utility of the sub-models, in their current forms, for ecological simulation and forecasting are provided below.

Light attenuation sub-model

Although the simulated day to day variability in light attenuation is much lower than measured in the field when 'smoothed' chlorophyll *a* forcings are applied, the seasonal trends in light attenuation coefficient obtained are in good

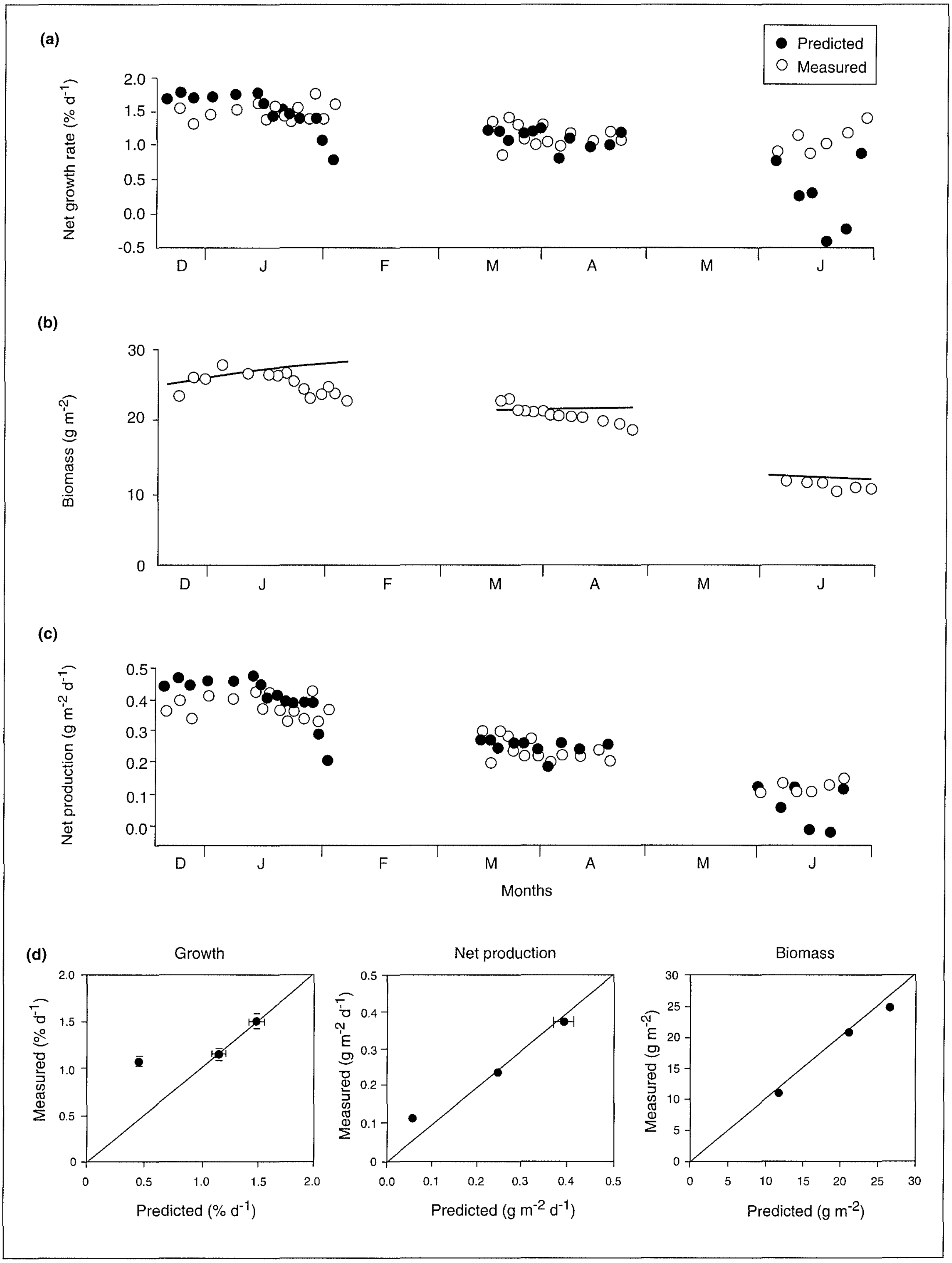


Figure 6.3-8. Seagrass model output: plots of predicted and measured (a) net growth rate (b) biomass and (c) net production for three periods and (d) the relationship between mean predicted and measured seagrass growth, net production and biomass.

Table 6.3-1. Statistical comparison of simulated and measured seagrass productivity and biomass for deployments D1, D2 and D3, and for D1-D3 combined.

Deployment	Productivity (% d-1)		Productivity ($g\ m^{-2}\ d^{-1}$)		Biomass ($g\ m^{-2}$)	
	slope	r^2	slope	r^2	slope	r^2
1	- 0.042	0.012	0.182	0.134	-0.273	0.030
2	0.440	0.111	0.648	0.193	-12.700	0.929
3	- 0.048	0.018	0.011	0.001	3.579	0.805
1 - 3	0.260	0.264	0.681	0.820	0.903	0.914

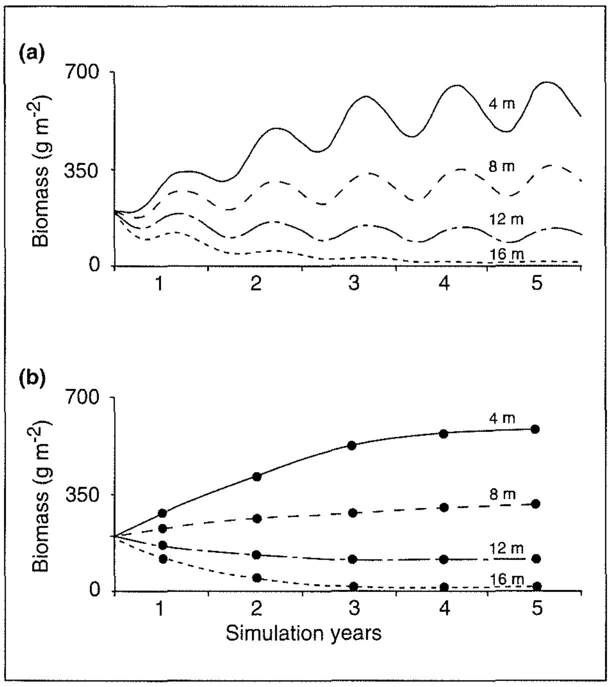


Figure 6.3-9. Simulated seagrass biomass at four depths over a five year simulation period: (a) daily and (b) annual means.

Table 6.3-2. Comparison of seagrass biomass at different depths at the end of simulation year five with the mean biomass for year five.

Depth (m)	Biomass ($g\ m^{-2}$) End of year 5	Biomass ($g\ m^{-2}$) Mean for year 5
4	533	566
8	297	300
12	105	103
16	4	5

agreement with field measurements and provide a realistic estimate of the *in situ* light climate of the study area when irradiance data from the Hope Valley monitoring station are used. As such, the output from this sub-model can be used as a realistic primary forcing for primary production modelling where attenuation is dominated by chlorophyll *a*.

Epiphyte sub-model

The present formulation of the model does not allow site-specific differences in epiphyte biomass to be predicted accurately. Field studies have highlighted wave energy as a major determinant of epiphyte biomass and community type, and sensitivity analyses also show the strong influence of the erosion/abrasion loss function on epiphyte biomass in the model. Therefore, to better predict changes in epiphyte biomass the model requires an algorithm relating erosion/abrasion to wave energy. The lack of data on expected seasonal changes in epiphyte biomass make it difficult to assess the general applicability of the model but the general trends appear reasonable based on sparse information available. The utility of the epiphyte growth simulation sub-model is considered low in its current form. The model does however provide a useful framework for future model refinement and the design of monitoring programmes.

Seagrass growth sub-model

The current formulation of the seagrass model incorporates the latest information on the photosynthesis verses irradiance relationships for the seagrass *Posidonia sinuosa*. The predicted seagrass production rates and biomass changes are in close agreement with measured data at the study site in Owen Anchorage and are consistent with the seasonal trends documented in other parts of southwestern Western Australia. The utility of the current formulation of the seagrass growth model for predicting the effects of epiphyte and phytoplankton biomass increases on seagrass distribution and health (biomass), is considered high.

On the basis of the sub-model evaluations described above, the BSM was configured to link the light attenuation and seagrass models and to incorporate the effect of epiphyte shading as a constant. A description of the simulations conducted and the results obtained is presented below.

6.3.4 Modelled water quality-seagrass scenarios in Cockburn Sound

6.3.4.1 *Seagrass depth limits*

The BSM was used to establish a relationship between light attenuation, depth and seagrass biomass under conditions of low, moderate and high epiphyte loadings. Seagrass biomass was initialised at 200 g m^{-2} and the biomass present at the end of the simulation period (five years) was used to derive the relationships described below. Light attenuation coefficient was expressed as the mean over the simulation period. The six light attenuation coefficients used in the tests were derived from the seasonal chlorophyll *a* time series for the Owen Anchorage site (Figure 6.3-2c) by sequential additions of 0.5 µg chlorophyll *a* l^{-1}. Three levels of epiphyte biomass were used (0.4, 1.0 and 2.5 mg cm^{-2}) corresponding to light reductions of 30, 45 and 60%. Epiphyte biomass was held constant over the simulation period and, for the purposes of this simulation, the epiphyte shading effect was incorporated as an equivalent reduction in PAR immediately below the water surface.

The results of these simulations are shown in Figure 6.3-10. Seagrass biomass declined with depth under all conditions and the rate of decline increased as water column light attenuation increased. Increases in epiphyte shading from 30 to 60% caused large reductions in biomass at each depth and light regime. These data were used to establish the cut-off depths for three arbitrarily selected levels of seagrass biomass shown in Figure 6.3-11. A biomass value of about 20 g m^{-2} could be reasonably considered to be the minimum density of a 'meadow'; a biomass of 1 g m^{-2} would be indicative of scattered plants. These model-derived relationships for a biomass of ≥ 20 g m^{-2} are reasonably consistent with the relationship derived by Masini *et al.* (1995a) from data collected at the deep edge of two seagrass meadows in local waters (Figure 6.3-11d). The influence of water column light attenuation on seagrass depth distribution was most pronounced when seagrass biomass cutoff level was low (Figure 6.3-11a).

These relationships highlight to managers how a given increase in light attenuation coefficient will have a much greater effect on seagrass distribution in naturally clear waters than in waters that are naturally more turbid. In Owen Anchorage for example, a nutrient-induced increase in mean chlorophyll *a* concentration of 0.5 µg l^{-1} would result in an increase in attenuation of nearly 0.02 m^{-1} and an associated reduction in the ≥ 20 g m^{-2} seagrass depth limit of approximately 2.5 m. There is added concern for managers when a large proportion of an ecosystem's seagrass meadows are relatively deep and bathymetrically confined to a narrow depth range. Under these conditions relatively small changes in mean water clarity or epiphyte loading could cause the loss of a large proportion of the area's seagrass meadows. This scenario, which occurred in Cockburn Sound during the early 1970s, is examined further in the simulations described below.

6.3.4.2 *Cockburn Sound hind-casting*

The BSM was configured to represent the Cockburn Sound/Owen Anchorage area at a resolution of 500 m for bathymetry and habitat (Figure 6.3-12). Seagrass biomass was initialised at 200 g m^{-2} and seagrass habitat was assigned to cells with a mean depth ≤ 15 m. Light attenuation was controlled by water colour and a chlorophyll *a* time series (Figure 6.3-2c) that was elevated or reduced by a standard amount to provide mean attenuation coefficients that were representative of the water quality of Cockburn Sound during the 1950s, 1970s and the 1990s (Table 6.3-3). In addition to the hindcasting scenarios described above, a chlorophyll *a* time series was constructed to reproduce the light attenuation coefficient corresponding to the draft nutrient-related criterion for Environmental Quality Objective 2 (i.e. maintenance of ecosystem integrity) for Cockburn Sound (Table 3.5-3) to forecast the likely effects of this 'target' light regime on seagrass biomass over a five year simulation. Final seagrass biomass for the region resulting from the different light attenuation fields used were also compared.

The results of the simulations are shown in Figure 6.3-13. The 1950s simulation is considered to represent pristine conditions. After five simulation years, significant reductions in seagrass biomass were largely restricted to the deepest portions of the Owen Anchorage basin (Figure 6.3-13a) suggesting that it is unlikely that *Posidonia* seagrass inhabited this area to any great extent. Highest average biomass (460 g m^{-2}) was found on east Parmelia Bank and in the Southern Flats/Mangles Bay area. The mean biomass of seagrass on the moderately deep (~9 m) eastern shelf was not as high (~200 g m^{-2}), indicating that seagrasses were partially light-limited in this area even under conditions considered to be pristine.

The water quality of Cockburn Sound, measured as chlorophyll *a* and light attenuation coefficient, was much poorer in the 1970s (Cary *et al.* 1995a) than estimated for the 1950s. Simulations conducted under these conditions (Figure 6.3-13b) show substantial losses of seagrass area and biomass with levels on the eastern margin averaging 2 g m^{-2}, which clearly indicates that the seagrass meadows have been lost. Seagrass meadows were restricted to shallow shoreline sites, Parmelia Bank and Southern Flats/Mangles Bay, but even in these shallow areas the average biomass had declined to about 250 g m^{-2} which is approximately half the biomass density present in the 1950s simulations. The simulated seagrass distribution pattern through the sound was very similar to the distribution mapped from aerial photographs at the time (Figure 4.3-2; Department of Conservation and Environment, 1979; Cambridge and McComb, 1984), to the extent that both showed the survival of a small patch of seagrass on the eastern shelf, northwest of James Point. These changes correspond to a loss of 71% of the biomass and 69% of the area present at the end of the 1950s scenario (Table 6.3-4).

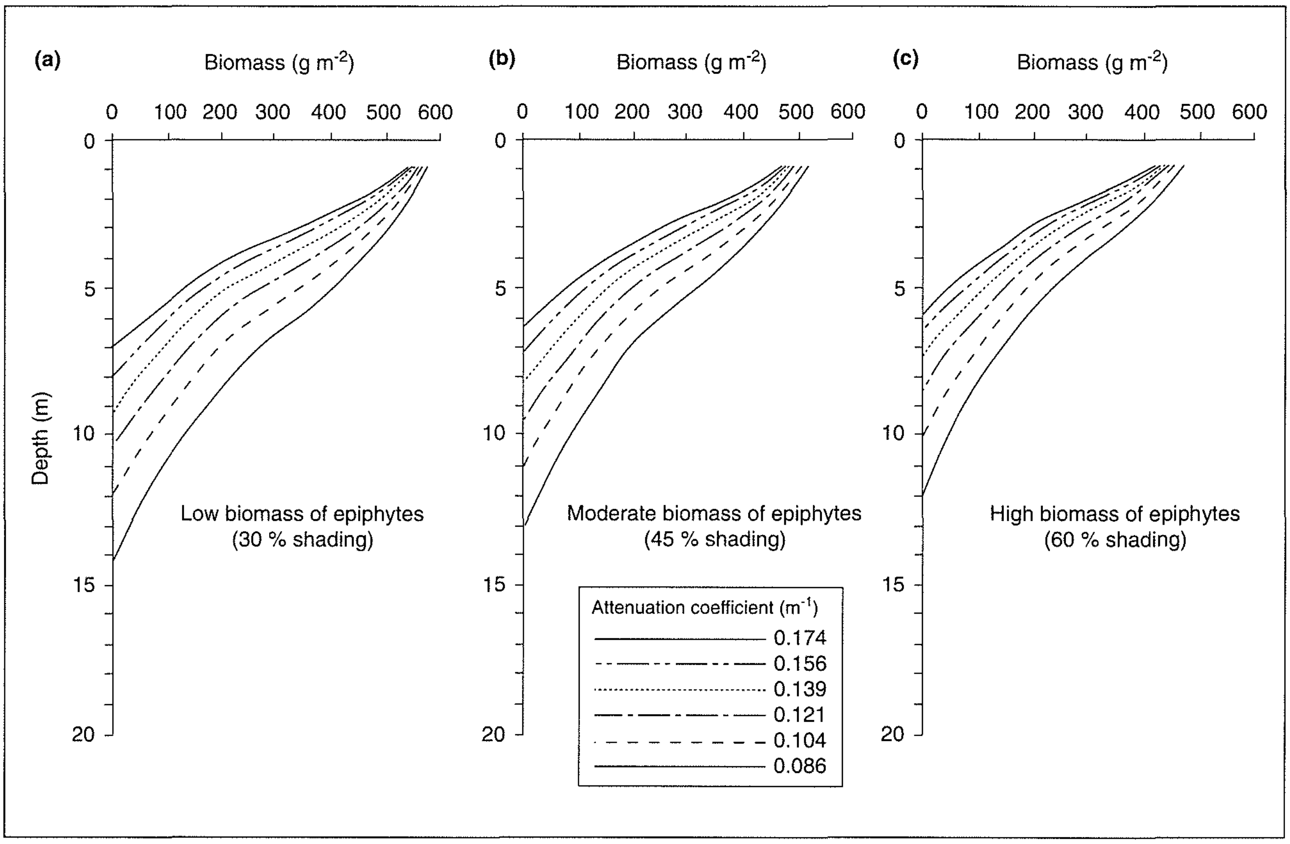


Figure 6.3-10. Benthic Site Model results: the relationships between Posidonia sinuosa seagrass biomass and light attenuation coefficient, depth and (a) low (0.4 mg cm^{-2}), (b) moderate (1.0 mg cm^{-2}) and (c) high (2.4 mg cm^{-2}) epiphyte biomass.

The seagrass biomass resulting from the 1990s simulation (Figure 6.3.13c) was about 350 g m^{-2} for the east Parmelia and Southern Flats/Mangles Bay areas, and about 90 g m^{-2} for the eastern margin of Cockburn Sound. These values are about 100 g m^{-2} lower than the corresponding results from the 1950s simulation. The simulated seagrass distribution is different to the 1993 mapped distribution on the eastern shelf (Figure 4.3-2) because the seagrasses on this shelf disappeared during the 1970s and *Posidonia* seagrasses are not known to re-establish in areas from which they have been lost (Clarke and Kirkman, 1989).

These simulations suggest that if the water quality of Cockburn Sound during the 1970s had only deteriorated to that measured during the early 1990s, the seagrass meadows on the eastern shelf would have thinned substantially but would not have been lost entirely. Under these conditions rehabilitation would have been possible by controlling nutrient loads and hence water clarity (and epiphyte biomass). Once gone however, natural re-establishment of a *Posidonia* meadow is very slow and unlikely on such a scale, even over a timeframe of decades to centuries (Clarke and Kirkman, 1989).

The seagrass distribution for the target water quality (Figure 6.3-13d) is very similar to the distribution resulting from the 1950s scenario. The theoretical seagrass biomass for the 'target' water clarity was 15% lower than under pristine conditions but areal cover was only 9% lower. This represents some loss in the deeper margins of the banks and a general thinning in other areas, predominantly the eastern shelf where biomass was reduced by about 25%. Biomass reductions in shallower areas such as eastern Parmelia Bank and Southern Flats/Mangles Bay were approximately 6%.

The variation of seagrass biomass with depth is shown in Figure 6.3-14 for each of the four water quality scenarios discussed. There was a marked decline in total seagrass biomass and maximum depth of seagrass survival under the 1970s water quality scenario compared to the pristine (1950s) scenario, however the target and pristine water quality conditions resulted in similar seagrass distributions (Figure 6.3-14). These seagrass biomass to depth relationships were used to calculate the lateral distribution of biomass along a west-east transect of Cockburn Sound (Figure 6.3-15).

Table 6.3-3. Key parameters used in ecological model simulations of four water quality scenarios for Cockburn Sound.

Scenario	Attenuation coefficient (m^{-1})	Epiphyte shading (%)
1950s	0.07	30
1970s	0.13	45
1990s	0.11	45
target	0.08	30

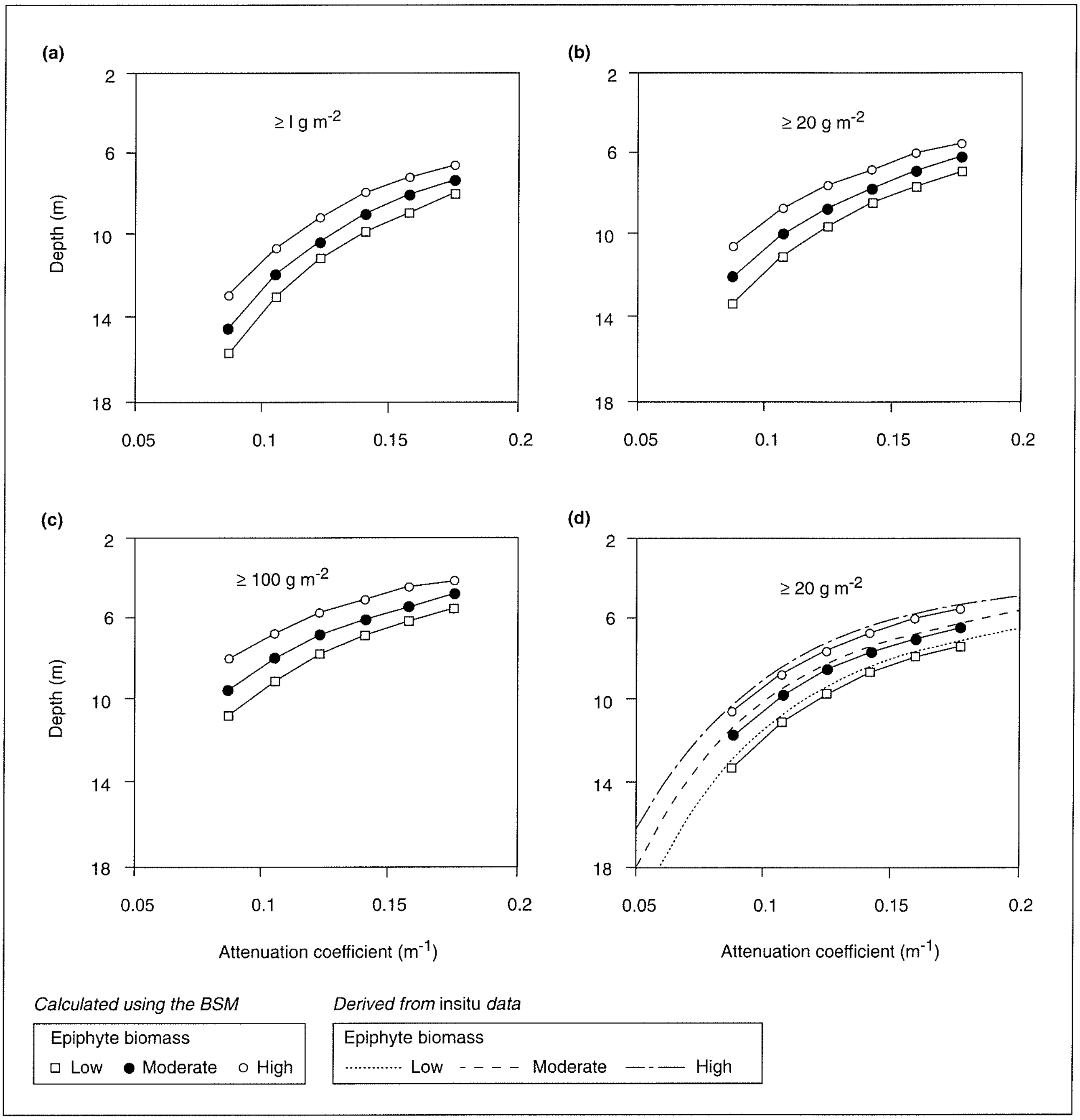


Figure 6.3-11. Benthic Site Model results: the relationships between light attenuation coefficient, epiphyte biomass and maximum depth of survival of Posidonia sinuosa seagrass meadows with biomass of (a) ≥ 1 g m^{-2} (b) ≥ 20 g m^{-2} and (c) ≥ 100 g m^{-2}. (d) Comparison between (b) and relationships derived from in-situ measurements.

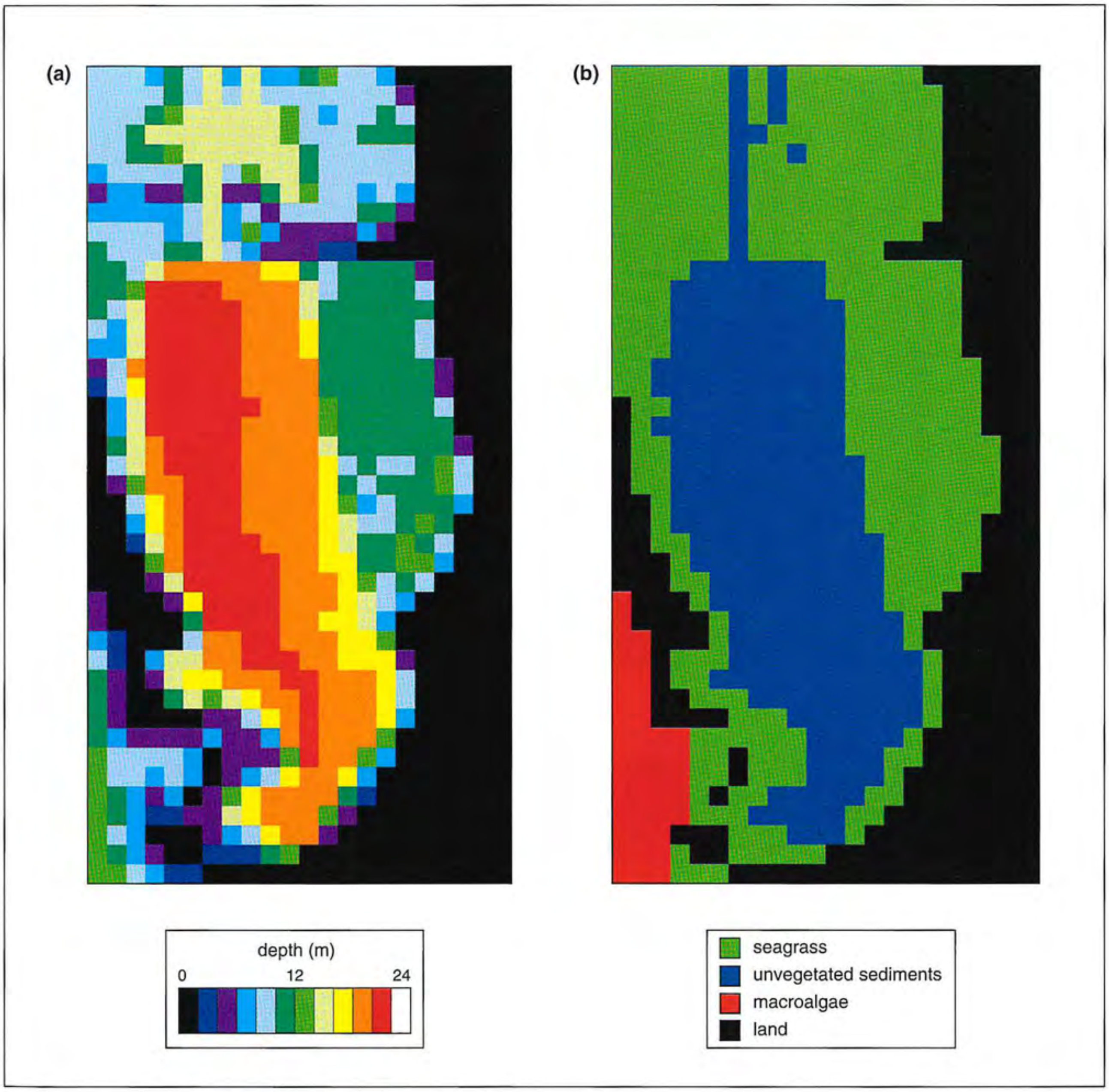


Figure 6.3-12. Input data for Cockburn Sound Benthic Site Model hind-casting: (a) bathymetry and (b) habitat types.

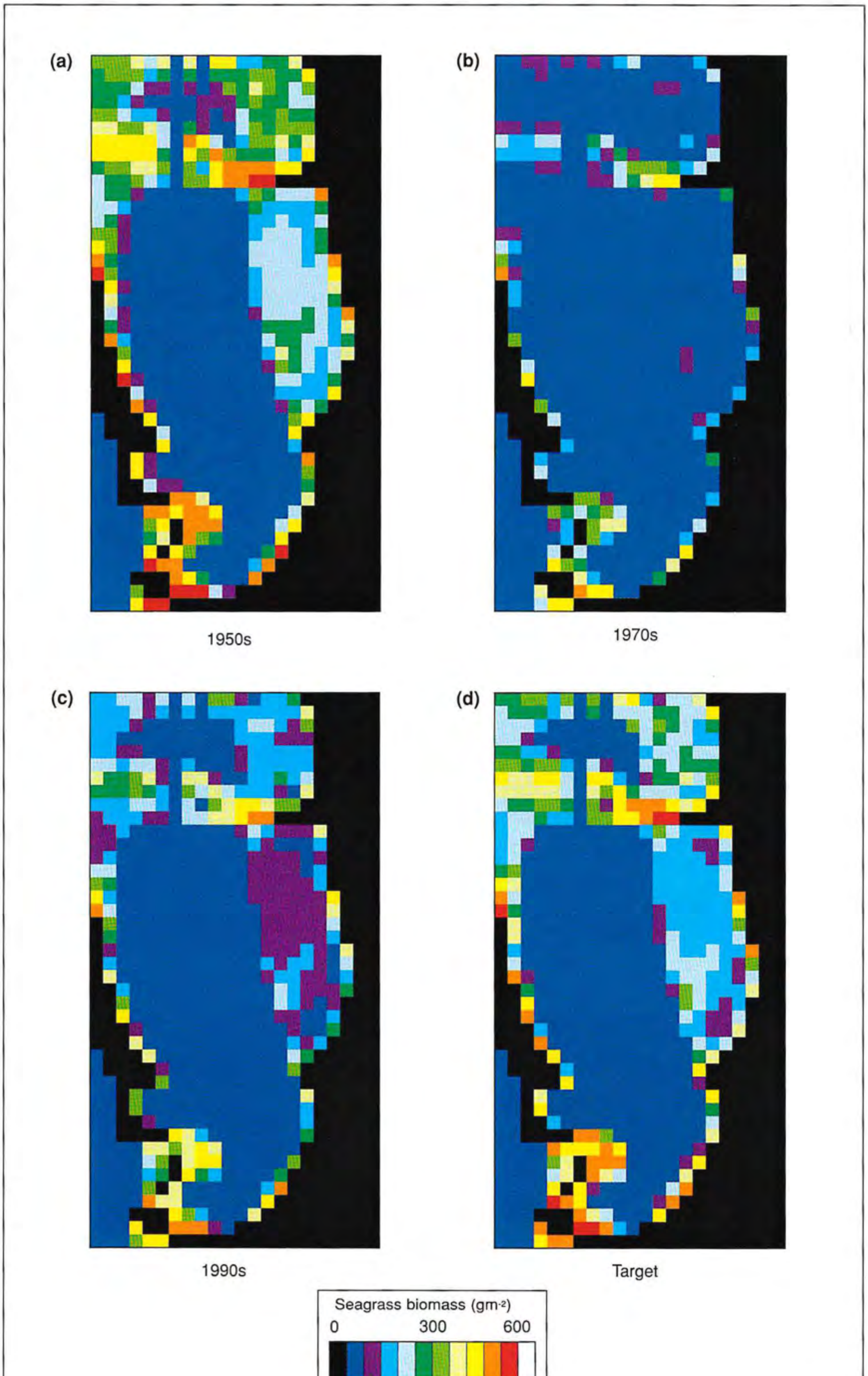


Figure 6.3-13. Benthic Site Model results: seagrass biomass distribution in Cockburn Sound and Owen Anchorage for four water quality scenarios representing (a) 1950s (b) 1970s (c) early 1990s and (d) 'target' water quality conditions.

The characteristics of the seagrass distributions described above, and shown in Figure 6.3-13, are also evident along this transect; notably, the absence of seagrass on the eastern shelf under 1970s conditions, a thin veneer of seagrass on the eastern shelf under the 1990s conditions, and the similarity to the pristine conditions in terms of biomass, but slight decline in depth distribution, under the 'target' light attenuation conditions.

6.3.5 Nitrogen loading and water quality in Cockburn Sound

Simulation modelling based on laboratory and field studies of the seagrass *Posidonia sinuosa* (Masini *et al.* 1995b; Masini and Manning, 1995b; Masini *et al.* 1995a), suggests that an annual average water column light attenuation coefficient of approximately 0.08 m^{-1} would allow the seagrass *P. sinuosa* to survive ($\geq$ 20 g m^{-2}) at depths of between 11 and 14 m, depending on the degree of epiphytic loading (Figure 6.3-11d). Under these conditions reasonably healthy meadows ($\geq$ 100 g m^{-2}), with low to moderate epiphyte loadings would be expected to occur at a depth of about 10 m (Figure 6.3-11c), which is the approximate depth of the eastern margin of Cockburn Sound that was once vegetated with *Posidonia* seagrasses (Department of Conservation and Environment, 1979). Empirical relationships derived from long-term field studies centred on Cockburn Sound indicate that this light attenuation coefficient corresponds to a chlorophyll *a* concentration of approximately 0.8 µg l^{-1} (Cary *et al.* 1995a). These values for chlorophyll *a* and light attenuation coefficient are consistent with the nutrient related draft criteria for Environmental Quality Objective 2 (Class II) established for nearshore coastal waters (see Chapter 3; Table 3.5-3) and the same as measured in Cockburn Sound during the summer period of 1982/83 (Cary *et al.* 1995a).

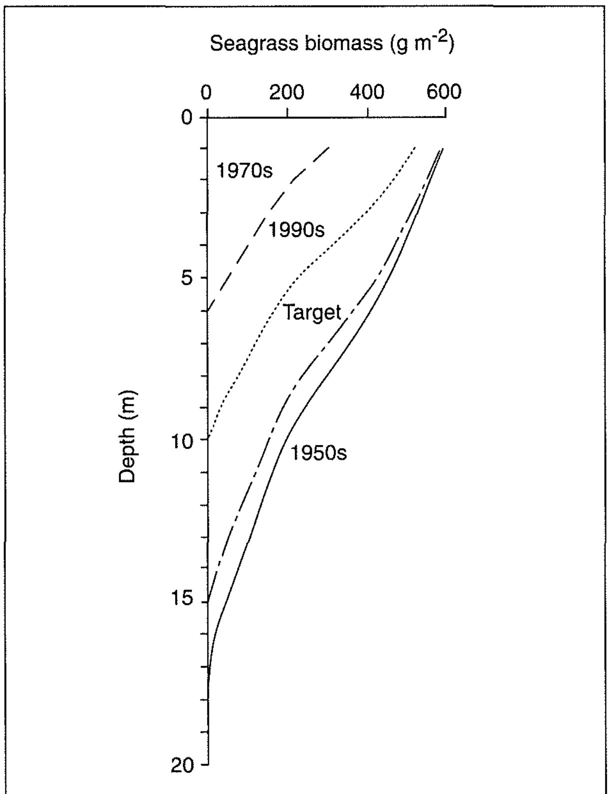


Figure 6.3-14. Benthic Site Model results: relationship between seagrass biomass and depth in Cockburn Sound after five simulation years representing 1950s, 1970s, early 1990s and 'target' water quality conditions.

The results of water quality monitoring programmes conducted in Cockburn Sound during the 1970s and 1980s revealed a significant linear relationship between externally-derived inorganic nitrogen loads to Cockburn Sound during summer and mean chlorophyll *a* concentrations (Cary *et al.* 1995a). This relationship supported the earlier conclusion of the Cockburn Sound Environmental Study (Department of Conservation and Environment, 1979) that external nitrogen loads were a major determinant of water quality in the sound. On the basis of this empirically-derived relationship the draft chlorophyll *a* criterion of 0.8 µg l^{-1} for Cockburn Sound would be achieved by limiting the total external loading of nitrogen to the sound to about 2030 kgN d^{-1} or 7.4 gN m^{-2} y^{-1}, assuming no seasonal variation. This target loading is approximately 40% higher than the 5.4 gN m^{-2} y^{-1}, suggested by Jawowski (1981) as a loading which, if exceeded, may lead to the eutrophication of estuaries and shallow coastal waters. However, it is about 40% lower than the equivalent areal loading in the late 1970s (12.7 gN m^{-2} y^{-1}) when the water quality of Cockburn Sound was considered to be at its worst (Cary *et al.* 1995a).

The response of seagrass communities in Cockburn Sound to the water quality conditions described by the nutrient-related draft criteria for EQO 2 (i.e. maintenance of ecosystem integrity) was assessed using the BSM ecological simulation model. An assessment of the reliability of model predictions for Cockburn Sound was also made by comparing predicted and measured seagrass distributions under three water quality conditions representing: pristine (1950s), moderately degraded (1990s) and severely degraded (1970s) (see section 6.3.4.2).

When the results of these simulations were linked to historical levels of external nitrogen load, seagrass biomass and area were found to decrease non-linearly in response to increases in nitrogen load. The decline in biomass was gradual over the range of loadings examined (Figure 6.3-16a), whereas area declined abruptly as loads increased above about 3200 kgN d^{-1} (Figure 6.3-16b). This latter situation mimics the 'sudden' loss of meadow area on the eastern shelf that occurred between 1968 and 1972 (see sections 4.3 and 6.3.4.2). The simulated seagrass distributions under the historical water quality scenarios (1950s and 1970s) are in close agreement with available information on the actual distributions, thus providing confidence in the predictions of the model under the 'target' water quality conditions.

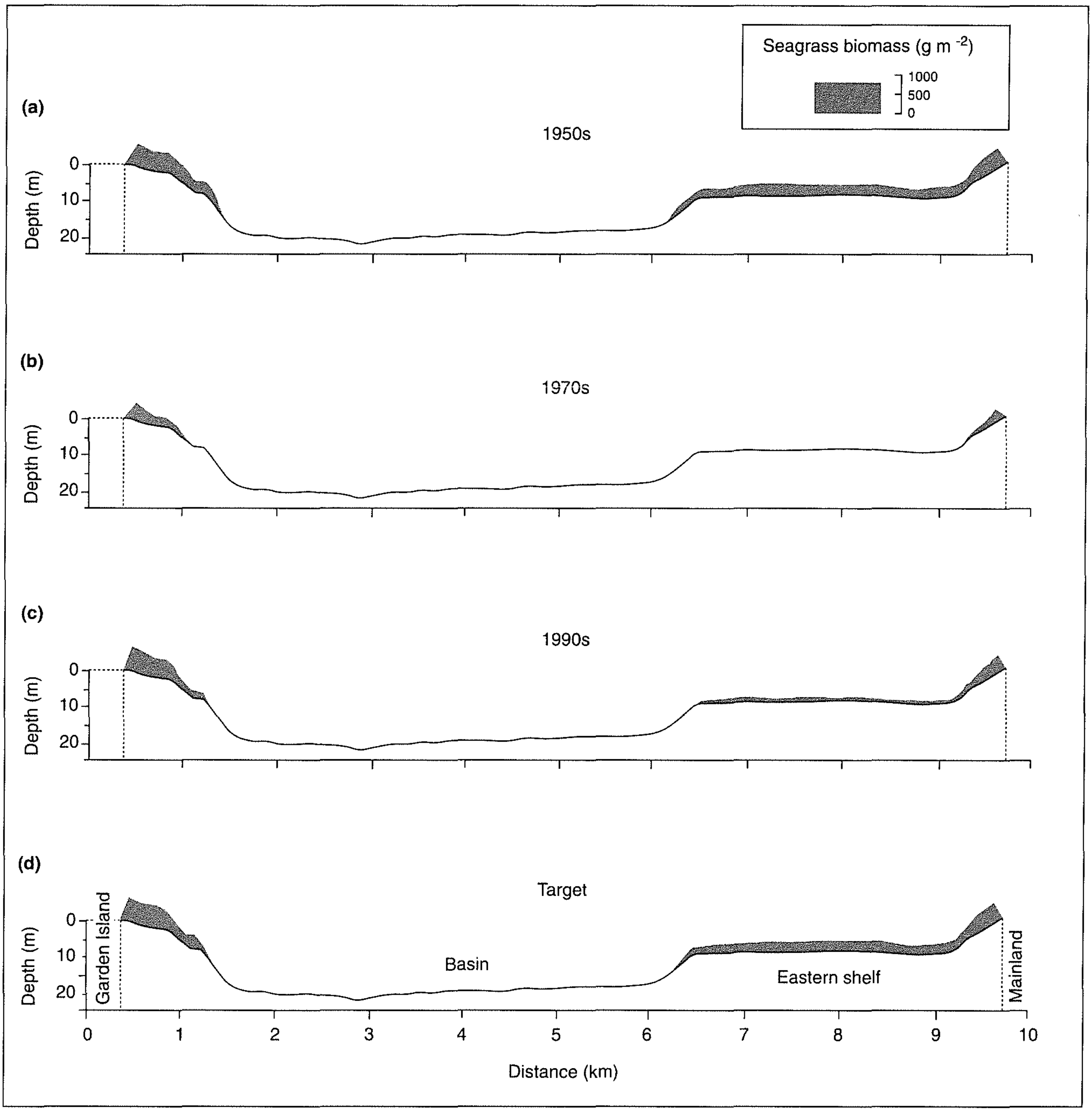


Figure 6.3-15. Benthic Site Model results: cross-sectional schematics along a west-east transect across central Cockburn Sound showing the relationship between seagrass biomass and bathymetry for (a) 1950s (b) 1970s (c) early 1990s and (d) 'target' water quality conditions.

Under the target water quality conditions, seagrass biomass would theoretically be some 85% of that originally present and the meadows would occupy 91% of the original area (Table 6.3-4).

According to the relationship between nitrogen loading and chlorophyll *a* (Cary *et al.* 1995a) mentioned above, the average chlorophyll *a* concentration during the early 1990s should correspond to an external loading to these waters of approximately 3200 kgN d^{-1}. However, the total external nitrogen load to these waters that can be accounted for from known sources was about 1300 kgN d^{-1} (Muriale and Cary, 1995).

Since 1989/90 there has been a major departure from the relationship between nitrogen load and chlorophyll *a*. The water quality of Cockburn Sound, expressed as chlorophyll *a* and light attenuation has remained at about 1989/90 levels even though estimates of nitrogen loads from external sources have declined by over 50%. This departure from the loading-response relationship could be caused by either a change in the key ecological processes, in particular the processes influencing nitrogen recycling (internal sources), or the presence of an additional, but unaccounted for, external source of nitrogen, or an underestimation of loads from current, known sources (see section 4.7.3). Cockburn Sound has a higher pelagic productivity than Warnbro Sound and consequently nitrogen excretion from zooplankton, organic

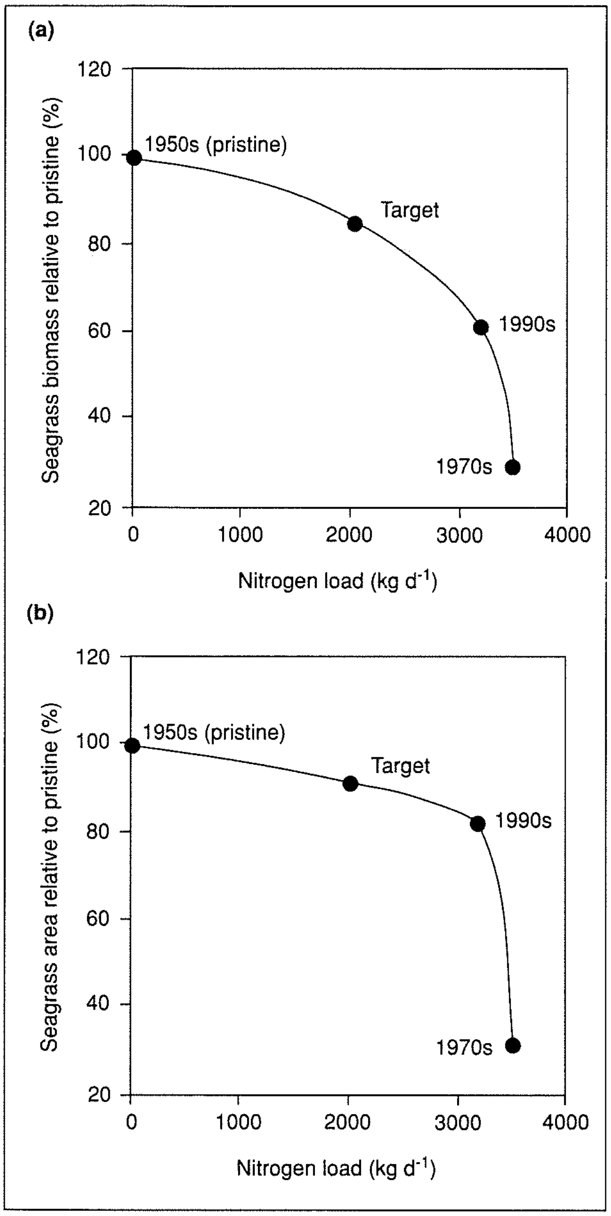


Figure 6.3-16. Relationships between estimated total inorganic nitrogen load and simulated seagrass (a) biomass and (b) area in Cockburn Sound after five simulation years representing 1950s, 1970s, early 1990s and 'target' water quality conditions.

Table 6.3-4. Simulated seagrass biomass and area at the end of year five for four water quality scenarios for Cockburn Sound (see Table 6.3-3 for simulation parameters).

Scenario	Biomass (tonnes)	(%)	Area (tonnes)	(%)
1950s	15 028	(100)	52	(100)
1970s	4 387	(29)	16	(31)
1990s	9 099	(61)	43	(82)
target	12 786	(85)	47	(91)

loading to the sediment and associated sediment nutrient release rates, are higher in Cockburn Sound than in Warnbro Sound (see section 5.5; Tables 5.5-4 and 5.5-5). Although the estimated loads of nitrogen from these internal sources are higher in Cockburn Sound than in Warnbro Sound, the pelagic productivity has been higher for at least two decades and there is no evidence to suggest that a change occurred between the summers of 1989/90 and 1991/92, in terms of productivity or key ecological processes, that would account for the apparent departure in the previously determined nitrogen loading-chlorophyll *a* relationship. The departure occurred abruptly after 1989/90 and is clearly evident in a time series plot of the difference between nitrogen loads from known sources and loads predicted from chlorophyll *a* concentrations and the relationship (Figure 6.3-17). The discrepancy was about 1000 kgN d^{-1} over the summer of 1990/91; it increased to about 2000 kgN d^{-1} in 1991/92 and has remained at that level until at least 1993/94.

Currently it is estimated that approximately 70% of the known cumulative load of nitrogen to Cockburn Sound is from groundwater inflows (Muriale and Cary, 1995). The poor coverage of monitoring bores and large errors associated with estimates of nitrogen loads to Cockburn Sound from inflows of contaminated groundwater make it very difficult to accurately quantify current loads from known sources or to detect new sources (see section 4.7.3). For example in 1990/91, a major nitrogen source to the Sound via contaminated groundwater was discovered near the WMC industrial estate and after further investigations the original load estimates from groundwater in this area were doubled (see section 4.4.1). Studies of groundwater conducted by the Geological Survey of Western Australia highlighted the problems associated with estimating contaminant loads from groundwater sources and concluded that if all the uncertainties are considered, the error associated with estimates of the total mass of nitrogen discharged to Cockburn Sound would be very large (Appleyard, 1994).

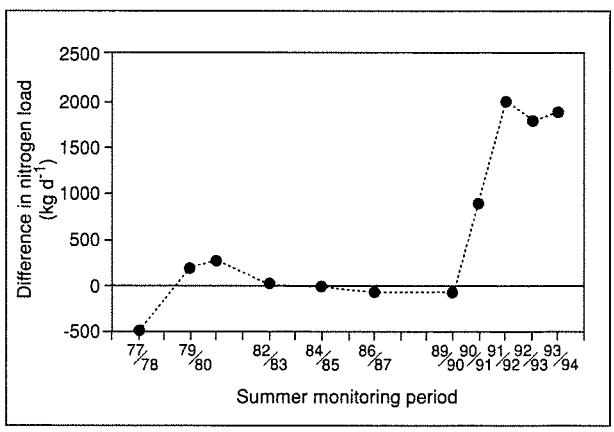


Figure 6.3-17. Difference in nitrogen loading estimates for Cockburn Sound during summer derived from a chlorophyll a : nitrogen loading relationship (Cary et al. 1995) and from the Contaminant Inputs Inventory (Muriale and Cary, 1995).

Conclusions

The ecological modelling component of the SMCWS has provided a vehicle to describe and develop our understanding of the key pathways and processes linking nutrient inputs and their ecological consequences for Perth's coastal waters. The modelling also provided a central focus for many of the ecological process studies undertaken within the SMCWS and the PCWS. The findings of these field and laboratory-based process studies have been incorporated in model parameter settings or algorithms. It was necessary to revise many of the sub-models in response to study findings, however the central stress-response structure was unchanged during the development of the actual model (see Lord and Hillman, 1995) and remains consistent with the original conceptual model (Masini *et al.* 1991).

Although the model structure is relatively simple, comparisons between model output and historical load-response information show that it is sufficient to represent the ecological consequences of chronic nutrient enrichment of Cockburn Sound, supporting the conclusion that the modelled pathways capture the 'key' ecological processes linking epiphyte and phytoplankton biomass with seagrass distribution and biomass in these waters. When the model is used in conjunction with empirically-derived relationships between nutrient load and phytoplankton biomass for Cockburn Sound it is possible to establish a direct link between nutrient load and seagrass 'health' and hence provide a basis for establishing target nutrient loads consistent with maintaining accepted environmental quality objectives for this waterbody (see section 3.1). The results suggest that the current estimate of nitrogen loading to Cockburn Sound is inaccurate and that a load of approximately 2000 kgN d^{-1} is unaccounted for. The most likely source of this 'missing' load is groundwater given the acknowledged uncertainty associated with current estimates.

In summary, the models provide a pertinent and up-to-date representation of the key nutrient effect pathway operating in Perth's coastal waters and is of appropriate complexity to use as a predictive tool to assist in the environmental management of nutrient pollution. In addition, the configuration of the seagrass sub-model provides a high degree of transferability, and as such can be applied to most temperate coastal waters in the State. Knowledge gained through the development, evaluation and implementation of the model will help in the design and interpretation of focussed environmental monitoring programmes in the temperate coastal waters of Western Australia.

6.4 Development of management tools

The use of focussed ecological process simulation models in environmental management studies is increasing as information data bases and computer capabilities develop. These models provide a framework for integrating information concerning key ecological processes and investigating the linkage, via these processes, between an environmental effect of concern and a causative factor that can be managed. Such a framework promotes efficient information exchange between scientists and managers involved in interdisciplinary management-oriented studies. As the models are formulated, specific areas of information deficiency are identified and required outcomes of field and laboratory investigations are defined. Monitoring programmes serve the needs of the model for calibration and verification. Likewise, these models assist in the development and focussing of management-oriented monitoring programmes and help identify key environmental parameters that are responsive to change and/or critical to ecosystem functioning. Used judiciously, these models can provide a reasonable indication of the likely consequences of proposed activities, such as the discharge of pollutants, that may modify the environment.

Once the sub-components of the model have been developed and calibrated for an ecosystem (for example, a particular temperate seagrass ecosystem), they need to be tested over a range of states (environmental qualities) of that ecosystem to ensure satisfactory agreement between model simulation results and field data sets collected under known forcing conditions. The model simulations can then be used to complement the understanding derived from field surveys by providing results at a much finer spatial and temporal resolution than would be economically feasible with field survey techniques. The simulations can also be designed to examine alternative scenarios, not encountered in the field surveys, for example changes in bathymetry due to proposed construction works. In the case of Cockburn Sound, which has a documented history of environmental change, it has been possible to use hindcasting techniques to assess the performance of the ecological model and to gain confidence in its predictive capacity for that system.

When the model has been successfully calibrated and verified for a particular ecosystem, it can then be tested on ecosystems of similar type. The basic configuration of the ecological model will most likely remain the same, however some of the empirical functions may need to be adjusted following further site-specific field studies (e.g. the epiphyte growth and erosion functions and the light attenuation coefficient function may change because the system is more exposed to wave energy or river inflow). Once the model has been verified over a representative range of temperate seagrass-based ecosystems and the sub-model relationships generalised to account for differences between these systems, it should be possible to use it as a transferable model for management purposes, in conjunction with limited field studies.

The modular structure of the ecological model enables various combinations of sub-models to be linked and thereby imparts a great deal of flexibility and general utility. This architecture provides a useful starting point for developing models to assist in the management of other quite different aquatic ecosystems (including estuarine and freshwater ecosystems). For example, the model is currently configured to examine the implications of different nutrient (predominantly nitrogen) loading scenarios on temperate seagrass-dominant marine communities. In tropical ecosystems however, corals often replace seagrasses as the key communities. Corals are also susceptible to nutrient (predominantly phosphorus) enrichment and are vulnerable to reductions in water clarity. Adaptation of the model to coral-dominated ecosystems would involve the development of new sub-models but could also utilise existing components such as the light attenuation sub-model. Advantage can also be taken of the existing data input and graphical output capabilities.

The experience derived from the SMCWS supports the conclusion of GESAMP (1993) that specific ecological process modelling can be of considerable value to environmental management and therefore should be developed further.

PART III

ENVIRONMENTAL PROBLEMS AND SOLUTIONS

This part of the report consists of one chapter. Chapter 7 draws upon the results of the detailed investigations carried out during the Study to define and discuss specific existing and potential environmental problems in Perth's southern metropolitan coastal waters. Problems or issues are considered to arise where the draft Environmental Quality Objectives (EQOs) and associated draft Environmental Quality Criteria (EQC) for these waters (described in Chapter 3) are not being met, or appear unlikely to be met if current trends continue. Findings from the Study have given rise to a number of proposed Department of Environmental Protection (DEP) actions and recommendations to the Environmental Protection Authority (EPA), which are presented in this chapter. The proposed DEP action items list those actions the DEP would take, subject to State Government endorsement, to address existing environmental problems or concerns. The DEP recommendations to the EPA are primarily concerned with the adoption of policy positions aimed at addressing specific environmental issues or problems identified in the Study, as well as the broader need for long-term protection of the marine environment through coordinated environmental management of Perth's coastal waters. The draft EQOs will be finalised after a separate, consultative process being undertaken by the EPA. The desired outcome of this consultative process, which will involve the public and all major users of Perth's marine environment, is an agreed set of EQOs, associated criteria and zones where the EQOs will apply. Some of the following DEP proposed actions and recommendations to the EPA may be modified when the Environmental Quality Objectives for Perth's metropolitan coastal waters are finalised.

7. DESCRIPTION OF PRINCIPAL ENVIRONMENTAL PROBLEMS AND SOLUTIONS

This chapter draws on the results of the detailed investigations carried out during the Study to define and discuss existing and potential environmental problems in the southern metropolitan coastal waters of Perth. Problems or issues are considered to arise where the draft Environmental Quality Objectives (EQOs) for Perth's southern metropolitan coastal waters, as expressed by the associated draft Environmental Quality Criteria, are not being met, or appear unlikely to be met if current trends continue. Proposed management actions by the Department of Environmental Protection and recommendations to the Environmental Protection Authority to address these matters are presented below.

The proposed DEP action items list those actions the DEP would take, subject to State Government endorsement, to address existing environmental problems or concerns. The DEP recommendations to the EPA are primarily concerned with the adoption of policy positions aimed at addressing specific environmental issues or problems identified in the Study, as well as the broader need for long-term protection of the marine environment through coordinated environmental management of Perth's coastal waters.

7.1 Nutrient inputs and water quality

7.1.1 Shelf-scale impacts

Shelf-scale and local-scale water quality surveys supported by water circulation studies show that, during winter, much of the southern metropolitan coastal waters of Perth are periodically influenced by outflows from the Peel-Harvey and the Swan-Canning estuaries. Combined annual nutrient loads from these estuaries to the coastal waters have increased at least four-fold over the past three decades and, as a result, these outflows can elevate ambient nutrient concentrations, particularly inorganic nitrogen, up to 10 times above background concentrations. Additional nutrient inputs can also occur during summer, mostly in the form of organic nitrogen, under conditions where phytoplankton blooms in these estuaries enter the coastal waters.

In winter, chlorophyll *a* concentrations are periodically elevated above background over large areas of Perth's southern coastal waters and, over recent years, phytoplankton populations in the study area have been dominated by a single species of silicoflagellate, *Dictyocha octonaria*. This dominance represents an apparent shift from the late 1970s when a series of surveys in Cockburn Sound showed silicoflagellates to be a minor component of the phytoplankton population. The scientific literature indicates that this group generally comprises less than five percent of phytoplankton abundance in unpolluted waters, and that the ecological significance of silicoflagellates in marine waters is poorly understood. 'Blooms' of a closely related silicoflagellate species, *Dictyocha speculum*, have been linked to broadscale eutrophication of coastal waters in Europe. Whether a link exists between anthropogenic nutrient inputs to Perth's coastal waters and silicoflagellate 'blooms' remains to be demonstrated. However, the apparent increase in abundance of *Dictyocha octonaria* in the coastal waters off Perth during winter is cause for concern.

An increase in the population of radiolarian zooplankton generally followed immediately after these silicoflagellate 'blooms' and is another unusual feature of the plankton of Perth's coastal waters. These animals are commonly found in offshore oceanic waters rather than in coastal areas and their cyclic presence in Perth's waters suggests their occurrence is related to the silicoflagellate 'blooms'.

Total annual nitrogen loading into Perth's coastal waters is currently about 6000 tonnes, mostly from ocean disposal of domestic wastewater, estuarine outflows and, to a lesser extent, from contaminated groundwater inputs and industrial outfalls. Apart from the highly seasonal input from the estuaries, mainly between July and September, all of these sources are continuous throughout the year. Projections based on current wastewater management practices indicate that annual nitrogen loadings would increase to 10 000 tonnes by 2040, predominantly as a result of ocean disposal of domestic wastewater (Lord and Hillman, 1995). The long-term ecological impacts of these inputs are unknown, but the findings of widespread elevated nutrient and chlorophyll *a* concentrations, and the apparently anomalous co-occurrence of silicoflagellate 'blooms' and nearshore radiolarian populations, suggest that shelf-scale, nutrient-related impacts are occurring in these waters during winter.

Recommendation 1

That environmental protection policies and integrated catchment management strategies for the catchments of the Swan-Canning and Peel-Harvey estuaries incorporate the objective of minimising nutrient inputs to marine waters to assist in achieving draft Environmental Quality Objective 2 (i.e. the maintenance of ecosystem integrity) for these waters (excluding designated exclusion zones).

7.1.2 Local-scale impacts

Cockburn Sound

The 1993/94 water quality of Cockburn Sound, in relation to nutrient pollution, was only slightly better than the late 1970s, when the water quality of the Sound was in its poorest recorded state. Gross productivity of phytoplankton in Cockburn Sound is over 10 times that in Warnbro Sound indicating that Cockburn Sound is comparatively eutrophic. The presence of a high biomass of filamentous epiphytes on the leaves of seagrass in Cockburn Sound, particularly on Southern Flats and in Mangles Bay, and the continuing loss of seagrass from these areas provide further evidence of the eutrophic condition of these waters. Based on mean chlorophyll *a* values for the whole of Cockburn Sound during summer, the nutrient-related criteria for draft Environmental Quality Objective 2 (i.e. maintenance of ecosystem integrity) were clearly exceeded in 1993/94 and the four previous summer periods. On this basis, it is clear that a nutrient management strategy is needed to ensure that draft Environmental Quality Objective 2 is met in Cockburn Sound.

Nutrients enter the sound from external sources, mainly via groundwater, industrial outfalls and surface drains, and from internal sources such as nutrient efflux from sediments. Currently known internal and external sources are estimated to contribute a mean of about 7000 - 12 000 kg of nitrogen to the waters of the sound each day. Groundwater sources currently represent 70% of the total external load of nitrogen to Cockburn Sound. Approximately 80% of the total groundwater nitrogen load enters the southeastern part of the sound and originates from the WMC/CSBP industrial estates and a further 20% enters the northeastern part of the sound, probably from the Love Starches site. CSBP and, to a lesser extent BP, are the current major point sources of nutrients to Cockburn Sound. Phytoplankton biomass in waters adjacent to the east coast of Cockburn Sound is chronically high, suggesting that these groundwater and point source inputs are the primary influence on the overall nutrient-related water quality of the sound.

Action 1

Under Part V of the E P Act, the Department of Environmental Protection will require that major contributors of nutrients to Cockburn Sound implement a nutrient management strategy to ensure that the draft Environmental Quality Objective 2 (i.e. the maintenance of ecosystem integrity) for these waters (excluding designated exclusion zones) is achieved by 31 March 2002, with appropriate annual environmental performance indicators.

Action 2

Under Part V of the E P Act, the Department of Environmental Protection will require that major contributors of nutrients to Cockburn Sound jointly undertake annual water quality monitoring programmes in Cockburn Sound until the draft Environmental Quality Objective 2 (i.e. the maintenance of ecosystem integrity) for these waters (excluding designated exclusion zones) is finalised, and the finalised EQO 2 is achieved and maintained for at least two years. Future monitoring requirements will be reviewed at this time.

Recommendation 2

That the Environmental Protection Authority adopt a presumption against any proposals to increase nutrient loads, particularly nitrogen, to Cockburn Sound until the draft Environmental Quality Objective 2 (i.e. the maintenance of ecosystem integrity) for these waters (excluding designated exclusion zones) is finalised, and the finalised EQO 2 is achieved and maintained for at least two years at levels that would permit consideration of further loadings.

Action 3

Under Part V of the E P Act, the Department of Environmental Protection will not issue works approvals or licenses to increase nutrient loads, particularly nitrogen, to Cockburn Sound until the draft Environmental Quality Objective 2 (i.e. the maintenance of ecosystem integrity) for these waters (excluding designated exclusion zones) is finalised, and the finalised EQO 2 is achieved and maintained for at least two years at levels that would permit consideration of further loadings.

Owen Anchorage

The water quality of Owen Anchorage has improved significantly since the 1970s, when the area was in its poorest recorded state. Direct discharges of nitrogen to these waters from industrial sources have reduced dramatically since the 1970s as industries have employed better waste management practices, connected to deep sewer or closed down. Recent (1992/93) water quality measurements in Owen Anchorage during summer indicate that nutrient-related criteria for draft Environmental Quality Objective 2 (i.e. maintenance of ecosystem integrity) were not exceeded. Phytoplankton, light attenuation and nutrient concentrations in Owen Anchorage are significantly higher than background levels during July to September and appear to be associated predominantly with discharges from the Swan-Canning Estuary. The high biomass of filamentous epiphytes on seagrass leaves on the eastern side of Success Bank and the significant decline in areal extent of seagrass meadows on eastern Parmelia Bank over the past decade or so suggest that the seagrasses on the eastern side of Owen Anchorage are stressed by nutrient pollution. This conclusion is supported by the results of hydrodynamic modelling which suggests that eastern Owen Anchorage is exposed to outflows of nutrient-rich water from the Swan-Canning Estuary and Cockburn Sound.

Seagrass meadows in some areas on the western side of Success Bank have also declined significantly in recent years. The reasons for this decline are unclear, although it appears that mobile sand sheets have caused some but not all of these losses. In general, phytoplankton concentrations and light attenuation of the water column and the biomass and species composition of the epiphyte assemblages on the seagrass leaves in this area are more indicative of oligotrophic conditions. However, in recent years several independent water quality surveys have recorded high concentrations of nutrients in the northern Garden Island to west Success Bank area, and large filamentous green algae, usually associated with nutrient-enriched conditions, have been observed on the leaves of seagrasses in this area in late summer and early autumn. Furthermore, numerical modelling demonstrates hydrodynamic linkages between west Success Bank and areas associated with major nutrient sources, including the Cape Peron outfall, eastern Cockburn Sound and the Swan-Canning Estuary. Thus nutrient enrichment may have contributed to the decline in seagrasses on west Success Bank.

Recommendation 3

That environmental protection policies and integrated catchment management strategies for the catchment of the Swan-Canning Estuary incorporate the objective of minimising nutrient inputs to Owen Anchorage to assist in maintaining draft Environmental Quality Objective 2 (i.e. the maintenance of ecosystem integrity) for these waters (excluding designated exclusion zones).

Action 4

The Department of Environmental Protection will request the Water and Rivers Commission to extend its water quality monitoring programmes of the Swan-Canning Estuary to include the waters of Owen Anchorage/Gage Roads.

Warnbro Sound

The current water quality of Warnbro Sound is largely unchanged compared to the late 1970s and data from the summer of 1993/94 indicate that the nutrient-related criteria for draft Environmental Quality Objective 2 (i.e. maintenance of ecosystem integrity) were not exceeded. However, nutrient and chlorophyll *a* concentrations and light attenuation in Warnbro Sound are higher than background levels during July to September and appear to be associated with nutrient inputs from remote sources, particularly the Peel-Harvey Estuary. Water quality surveys have shown that, in winter, contaminants can be transported into the vicinity of Warnbro Sound from the Cape Peron outfall and from Cockburn Sound. The potential influence of pollutants from these external sources is an important consideration for the long-term management of the Shoalwater Islands Marine Park.

Recommendation 4

That environmental protection policies and integrated catchment management strategies for the catchment of the Peel-Harvey Estuary incorporate the objective of minimising nutrient inputs to the coastal waters to assist in maintaining draft Environmental Quality Objective 2 (i.e. the maintenance of ecosystem integrity) for the waters of the Shoalwater Islands Marine Park.

Sepia Depression (North of Becher Point)

Comparisons of historical and recent (1993/94) water quality data for Sepia Depression indicate that, in summer, the waters of Sepia Depression west of Warnbro Sound, have remained largely unchanged since the late 1970s and do not exceed the nutrient-related criteria for draft Environmental Quality Objective 2 (i.e. maintenance of ecosystem integrity). However, in recent years, elevated (above background) nutrient concentrations have regularly been recorded during the non-winter period in Sepia Depression north of the Cape Peron wastewater outfall. Similarly, chlorophyll *a* concentrations were also elevated and exceeded the nutrient-related criteria values for draft Environmental Quality Objective 2 during the summer of 1993/94 at a site approximately four kilometres north of the outfall. Furthermore, data collected in Sepia Depression by the Water Corporation (formerly the WAWA) in early 1995 indicate that nutrient concentrations were significantly higher than background concentrations and chlorophyll *a* concentrations were significantly higher than criteria values for draft Environmental Quality Objective 2 up to five kilometres north of the outfall. The potential long-term ecological significance of these changes is presently unknown.

These data indicate that the nutrient loadings from the Cape Peron outfall have, in recent years, resulted in measurable changes to the water quality of Sepia Depression during summer, to the point where the nutrient-related criteria for draft Environmental Quality Objective 2 are exceeded. Nutrient loads from the outfall therefore need to be reduced and capped if this Environmental Quality Objective is to be achieved and maintained.

Action 5

Under Part V of the E P Act, the Department of Environmental Protection will require that the Water Corporation implements a nutrient management strategy to ensure that draft Environmental Quality Objective 2 (i.e. the maintenance of ecosystem integrity) for waters of Sepia Depression influenced by the Cape Peron outfall (excluding designated exclusion zones), is achieved by 31 March 2002, with appropriate annual environmental performance indicators.

Action 6

Under Part V of the E P Act, the Department of Environmental Protection will require that the Water Corporation revises the environmental monitoring programmes for the Cape Peron outfall, by 31 December 1997, consistent with Action 5.

Recommendation 5

That the Environmental Protection Authority adopt a presumption against any proposals to increase nutrient loads, particularly nitrogen, to Sepia Depression until the draft Environmental Quality Objective 2 (i.e. the maintenance of ecosystem integrity) for these waters (excluding designated exclusion zones) is finalised, and the finalised EQO 2 is achieved and maintained for at least two years at levels that would permit consideration of further loadings.

Action 7

Under Part V of the E P Act, the Department of Environmental Protection will not issue works approvals or licenses to increase nutrient loads, particularly nitrogen, to Sepia Depression until the draft Environmental Quality Objective 2 (i.e. the maintenance of ecosystem integrity) for these waters (excluding designated exclusion zones) is finalised, and the finalised EQO 2 is achieved and maintained for at least two years at levels that would permit consideration of further loadings.

Sepia Depression (South of Becher Point)

During the winter-spring runoff period, nutrient outflows from the Peel-Harvey Estuary can have a broadscale influence on Perth's southern metropolitan coastal water quality. Less information is available on the influence of estuarine outflows for the remainder of the year. Recent preliminary water quality studies by the Department of Transport near the Dawesville Channel indicate that the nutrient-related criteria values for draft Environmental Quality Objective 2 (i.e. maintenance of ecosystem integrity) are regularly exceeded, particularly in the area between the Channel and the southern end of Comet Bay, as a result of the nutrient-enriched outflows from the Peel-Harvey Estuary. These findings suggest that these outflows may be adversely influencing the surrounding marine communities.

Recommendation 6

That environmental protection policies and integrated catchment management strategies for the catchment of the Peel-Harvey Estuary incorporate the objective of minimising nutrient inputs to coastal waters to assist in achieving draft Environmental Quality Objective 2 (i.e. the maintenance of ecosystem integrity) for the coastal waters influenced by the Dawesville and Mandurah Channel outflows.

Action 8

The Department of Environmental Protection will request the Water and Rivers Commission and the Department of Transport to implement monitoring programmes in the nearshore coastal waters influenced by the Dawesville and Mandurah Channel outflows and to provide annual performance indicators in relation to the catchment management objective outlined in Recommendation 6.

Recommendation 7

That the Environmental Protection Authority adopt a presumption against any proposals that would increase nutrient loads, particularly nitrogen, to the nearshore coastal waters influenced by the Dawesville and Mandurah Channel outflows until the draft Environmental Quality Objective 2 (i.e. the maintenance of ecosystem integrity) for these waters (excluding designated exclusion zones) is finalised, and the finalised EQO 2 is achieved and maintained for at least two years at levels that would permit consideration of further loadings.

7.2 Toxic contamination

Extensive baseline surveys of pesticides, polychlorinated biphenyls (PCBs), hydrocarbons, organotin compounds and heavy metals in sediments and the tissue of the blue mussel *Mytilus edulis* were undertaken during this study.

7.2.1 Organic toxicants

Pesticides, PCBs, aliphatic and polycyclic aromatic hydrocarbons in sediments and mussels were detected at only a few of the sites sampled in the study area, and where detected, concentrations were generally close to the limits of detection. Areas where these contaminants could be detected were generally within or immediately adjacent to harbours, marinas and jetties. Apart from PCB concentrations in sediments at less than 1% of the sites and DDT at approximately 15% of the sites, the concentrations of the organic contaminants in sediments and mussels throughout the study area were well below the criteria for draft Environmental Quality Objective 2 (i.e. maintenance of ecosystem integrity) and draft Environmental Quality Objective 3 (i.e. maintenance of aquatic life for human consumption).

7.2.2 Tributyltin

The marine sediments and mussels of Perth's southern coastal waters are significantly contaminated with organotin compounds, with areas of highest contamination occurring in harbours, marinas, the naval area at Careening Bay, and along the eastern shorelines of Cockburn Sound and Owen Anchorage, particularly near industrial wharves. At some sites in Cockburn Sound, sediment concentrations of tributyltin (TBT), the most toxic form of organotin, were among the highest recorded in Australia. The concentration of TBT in the sediments of the study area exceeded the ERL and the ERM criterion values for draft Environmental Quality Objective 2 at approximately 70% and 18% of the sites, respectively. According to indicative values proposed by Page and Widdows (1991), mussel tissue from more than 90% of the SMCWS mussel survey sites can be classed as contaminated with TBT, and at 40% of the sites this contamination is at levels likely to cause physiological stress in mussels.

The frequency of the reproductive disorder *imposex* in females of the mollusc *Thais orbita* collected from intertidal reefs throughout the metropolitan coastal waters of Perth was high, indicating that the biological impact of TBT on this species is widespread. The frequency of imposex exceeded the criterion for draft Environmental Quality Objective 2 at all sites throughout Perth's coastal waters, apart from one site in southern Warnbro Sound. These sites were up to 30 km north and 25 km south of Fremantle and westward to the western end of Rottnest Island. These data confirm the biological implications that can be drawn from the exceedences recorded in the sediment and mussel data mentioned above.

Tributyltin is known to affect over 70 species of marine molluscs worldwide suggesting that other mollusc species in these waters could be affected by this substance. TBT is known to affect the reproductive processes of marine flora and fauna at extremely low concentrations (in water) of around 1-2 ngTBT l^{-1}. The widespread occurrence of imposex in *Thais orbita* indicates that TBT concentrations in much of Perth's nearshore coastal waters are at biologically significant levels and raises the possibility that populations of other marine organisms in these waters are being deleteriously affected by this substance. Furthermore, the presence of imposex in cone shells (*Conus sp.*) from Rottnest Island in 1991, when this disorder was not detected in *Conus* from Rottnest in 1975, suggests that the biological impact of TBT in Perth's coastal waters is a relatively recent phenomenon.

The concentrations of TBT in mussels also exceeded the criterion for draft Environmental Quality Objective 3 (i.e. maintenance of aquatic life for human consumption) at 14 sites, primarily in the harbours and near the wharves in Cockburn Sound. These exceedances indicate that the consumption of mussels from these sites poses a risk to human health. Regulations controlling licensable activities were amended in September 1996 to include ship-building and ship-maintenance facilities which use or remove organotin compounds. The DEP is identifying those premises and will progressively license them to ensure that waste materials containing organotin are disposed of in an environmentally acceptable way.

The use of TBT-based antifouling paint was restricted in Western Australia by regulations, effective from 1 November 1991, which prohibit the application of this substance on boats under 25 m and restrict its usage to low leaching forms on boats over 25 m. However, the regulations do not address the contamination of Western Australian waters arising from visiting ships using TBT-based anti-fouling paint. Comparisons of TBT concentrations in sediments from surveys conducted in 1991 and 1994, that is before and after the introduction of these regulations, indicate that at sites visited predominantly by recreational boats under 25 m (e.g. Rottnest Island), TBT concentrations have remained at, or reduced to slightly below, 1991 levels. This finding suggests that the regulations have been effective in minimising further contamination from small boats. In contrast, sediment TBT concentrations in areas visited predominantly by vessels over 25 m, such as Fremantle Harbour and parts of Cockburn Sound, have increased significantly since 1991, coinciding with an 8% increase in shipping activity over the same period. These findings suggest that vessels over 25 m are the major current source of TBT into the coastal environment of Perth.

Action 9

The Department of Environmental Protection will recommend to the Minister for the Environment that the Western Australian Government request the Australian Government to initiate further action with international agencies to prohibit the use of tributyltin-based antifouling paints on all vessels, or reduce allowable tributyltin release rates to levels that would achieve the criteria for draft Environmental Quality Objective 2 (i.e. maintenance of ecosystem integrity) and draft Environmental Quality Objective 3 (i.e. maintenance of aquatic life for human consumption) in Perth's coastal waters.

Action 10

The Department of Environmental Protection will recommend that the Western Australian Minister for the Environment request ANZECC to review its existing, recommended tributyltin release rate for antifouling paints used on Australian registered vessels (including naval vessels) greater than 25 m in length, with a view to prohibiting the use of this substance or reducing the allowable tributyltin release rate to levels that would achieve the criteria for draft Environmental Quality Objective 2 (i.e. maintenance of ecosystem integrity) and draft Environmental Quality Objective 3 (i.e. maintenance of aquatic life for human consumption) in Perth's coastal waters.

Action 11

The Department of Environmental Protection will recommend to the Minister for the Environment that the Western Australian Government coordinate the implementation of incentives to encourage 'TBT-free' ships to Western Australian ports.

Action 12

The Department of Environmental Protection will identify and license ship-maintenance and ship-building facilities that use or remove organotin compounds so that waste materials containing organotin are disposed of in an environmentally acceptable way.

Action 13

The Department of Environmental Protection will request the Department of Health to investigate the potential health implications of the exceedances of the tributyltin criterion for draft Environmental Quality Objective 3 (i.e. maintenance of aquatic life for human consumption) and, if necessary, implement a public health risk minimisation strategy.

7.2.3 Heavy metals

Heavy metal concentrations in the sediments and mussels of Perth's southern coastal waters are generally low and have decreased significantly in Cockburn Sound and Owen Anchorage since the late 1970s. These changes have coincided with substantial reductions in the discharge of heavy metals from industrial sources to these waters. The concentrations of heavy metals in sediments and mussels show similar distribution patterns with the highest concentrations mainly confined to areas within or immediately next to harbours/marinas and ship maintenance facilities, along the eastern margin and southern part of Cockburn Sound and the shoreline of Owen Anchorage. These patterns reflect the locations of historical and current sources of heavy metals to the sound.

Cockburn Sound and Owen Anchorage

In 1994, the concentrations of most heavy metals in sediments throughout Cockburn Sound and Owen Anchorage were well below the criteria for draft Environmental Quality Objective 2. Generally, the metal concentrations in sediments sampled in the southern half of Cockburn Sound were significantly elevated above concentrations found in the basin sediments in the southern half of Warnbro Sound. At most sites in Cockburn Sound, the arsenic concentrations in sediments exceeded the ERL criterion for draft Environmental Quality Objective 2, and the ERM criterion was exceeded at some sites along the eastern shoreline and in Mangles Bay. Mercury concentrations in sediments exceeded the ERL criterion at sites near the CSBP outfall, in the northeastern and northwestern areas of Cockburn Sound, particularly off the defence area toward the northern end of Garden Island, and eastern and southern Owen Anchorage. The ERM criterion for mercury was exceeded near the Explosives Jetty at Coogee Beach in eastern Owen Anchorage. Furthermore, aluminium concentrations in sediments at some sites in the Sound, particularly near the Alcoa Jetty, exceeded the ERL criterion for draft Environmental Quality Objective 2, but there were no exceedences of the ERM criterion.

Concentrations of heavy metals in mussels, apart from zinc, were below the values proposed by Chegwidden (1979) as indicative of heavy metal contamination at most sites sampled. Zinc concentrations exceeded the indicative value at most sites throughout Cockburn Sound and eastern Owen Anchorage. The concentrations of all heavy metals were below the criteria for draft Environmental Quality Objective 3 (i.e. maintenance of aquatic life for human consumption) at all sites sampled within Cockburn Sound and Owen Anchorage in 1994.

The ecological significance of these findings needs to be determined as a matter of priority; the development of sediment quality criteria to be undertaken as part of the consultative EQO/EQC finalisation process (see section 3.6) will help address this issue.

The generally low concentrations of most heavy metals in sediments and mussel tissue in these waters suggest that, individually, these substances are unlikely to be having widespread biological impacts. However, inverse statistical relationships exist between the concentrations in sediments of most heavy metals and the species richness of the benthic invertebrate fauna. Although correlation does not imply causality, these data suggest that potential links, between heavy metal contamination and the structure of the benthic invertebrate faunal community of Cockburn Sound, should not be ruled out. Synergistic interactions amongst the combination of heavy metals in the sediments of Cockburn Sound may be having adverse effects on the benthic faunal communities despite the concentrations of most metals being less than the draft criteria. Statistical analyses support the hypothesis that sediment contaminants, particularly heavy metals and polycyclic aromatic hydrocarbons, are influencing benthic community structure of the basin sediments of Cockburn Sound.

Currently, the major industrial sources of heavy metals to Cockburn Sound are the point source discharges of CSBP & Farmers Ltd and, to a lesser extent, BP Refinery (Kwinana) Pty Ltd and TiWest Joint Venture. CSBP and BP have an objective of achieving zero discharge of these metals by 2008 and 2011, respectively.

Shoalwater Islands Marine Park

The concentrations of all heavy metals, apart from arsenic and chromium, in the sediments of Shoalwater Islands Marine Park were below the ERL criteria for draft Environmental Quality Objective 2. The concentrations of arsenic in the sediment at most sites in the basin of Warnbro Sound and chromium at one site in the Sound exceeded the ERL criterion for draft Environmental Quality Objective 2. Concentrations of most heavy metals were highest in the northeast part of the Warnbro Sound basin, particularly near the Waikiki main drain, suggesting that this drain is a major source of these metals to the sound. Although the concentrations were below the draft criteria in 1994, heavy metals will continue to accumulate in these sediments if inputs continue.

Sepia Depression

The concentrations of most heavy metals in the water, sediments and biota of Sepia Depression, in the vicinity of the Cape Peron outfall, are below the criteria for draft Environmental Quality Objective 2 (i.e. maintenance of ecosystem integrity) and draft Environmental Quality Objective 3 (i.e. maintenance of aquatic life for human consumption). However the results of the WAWA (now the Water Corporation) 1992 monitoring programme show that the concentrations of mercury in water exceeded the criteria for draft Environmental Quality Objective 2 and draft Environmental Quality Objective 3 at sites up to one kilometre from the end of the outfall. In addition, the concentrations of cadmium and zinc in sentinel mussels exceeded the values proposed by Chegwidden (1979) as indicative of heavy metal contamination, at sites up to four kilometres from the outfall. These findings indicate that proposals to increase heavy metal loads from the Cape Peron outfall should be treated with caution.

Action 14

Under Part V of the E P Act, the Department of Environmental Protection will require that the major contributors of arsenic to Perth's southern metropolitan coastal waters investigate the ecological implications of the current levels of arsenic in sediments with a view to the development of arsenic criteria, and implement (if necessary) an arsenic management strategy, with appropriate annual environmental performance indicators, to ensure draft Environmental Quality Objective 2 (i.e. maintenance of ecosystem integrity), for areas influenced by their discharges (excluding designated exclusion zones), is achieved by 31 December 1999.

Action 15

Under Part V of the E P Act, the Department of Environmental Protection will require that major contributors of zinc to Perth's southern metropolitan coastal waters investigate the ecological implications of the current levels of zinc in mussels with a view to the development of zinc criteria, and implement (if necessary) a zinc management strategy, with appropriate annual environmental performance indicators, to ensure draft Environmental Quality Objective 2 (i.e. maintenance of ecosystem integrity), for areas influenced by their discharges (excluding designated exclusion zones), is achieved by 31 December 1999.

Action 16

That, in relation to possible synergistic effects of heavy metal and polycyclic aromatic hydrocarbon contamination on benthic faunal communities, the Department of Environmental Protection will require that major dischargers of these substances to Cockburn Sound conduct investigations (e.g. ecotoxicological) to evaluate this possibility, develop criteria as appropriate and implement a management strategy as required by 31 December 2001.

Action 17

Under Part V of the E P Act, the Department of Environmental Protection will require that current major contributors of heavy metals and polycyclic aromatic hydrocarbons to Cockburn Sound jointly undertake triennial monitoring programmes of basin sediment contamination and benthic community structure in

Cockburn Sound from 1998 until the relevant criteria are met or until the input of these contaminants to Cockburn Sound from these contributors ceases.

Action 18

The Department of Environmental Protection will request the Water Corporation to implement a contaminant management strategy, with appropriate annual environmental performance indicators, to minimise contaminant inputs to the Shoalwater Islands Marine Park from the Waikiki and Forrester Road main drains.

Action 19

Under Part V of the E P Act, the Department of Environmental Protection will require that, with respect to the operation of the Cape Peron ocean outfall, the Water Corporation investigates the ecological implications of the current levels of cadmium in mussels with a view to developing cadmium criteria and implement (if necessary) a cadmium management strategy, with annual environmental performance indicators, to ensure that draft Environmental Quality Objective 2 (i.e. maintenance of ecosystem integrity) and draft Environmental Quality Objective 3 (i.e. maintenance of aquatic life for human consumption) for areas of Sepia Depression influenced by the Cape Peron outfall (excluding designated exclusion zones) are achieved, by 31 March 2002.

7.2.4 Contaminated groundwater

Contamination of groundwater with toxic substances such as pesticides, herbicides, heavy metals, hydrocarbons and phenolic compounds in the study area occurs principally under the Kwinana industrial area. The two major sites of contamination are under the BP Refinery and Nufarm (previously CIK) industrial estates. Sixty thousand tonnes of hydrocarbons are estimated to underlie the BP site and recovery has been occurring at about 2000 tonnes per year since 1988. Progress of the recovery programme is reported triennially to the Department of Environmental Protection as part of BP's licence conditions. The groundwater under the Nufarm site is contaminated with the herbicides 2 4-D and 2 4 5-T and a toxic plume is currently estimated to reach Cockburn Sound in 20-30 years. The recovery programme and further monitoring will be dealt with under the forthcoming contaminated sites legislation under the Environmental Protection Act 1986-1993. As such, no recommendations are made on the long-term management of these contaminated groundwater plumes.

7.3 Microbiological quality of coastal waters and beaches

Results from microbiological surveys by the Water Corporation (formerly the WAWA) in Sepia Depression indicate that faecal coliform concentrations in water were commonly found to be above the criterion value for draft Environmental Quality Objective 4 (i.e. maintenance of recreational values) at distances up to two kilometres from the end of the Cape Peron outfall. These data indicate that a risk to human health through direct contact recreation exists in these waters. The results also show that the concentrations of faecal coliforms in water occasionally exceed the criterion value for draft Environmental Quality Objective 3 (i.e. maintenance of aquatic life for human consumption) at distances up to four kilometres from the end of the outfall (i.e. the limit of the WAWA sampling grid). These data, although limited, indicate that a potential health risk exists in relation to the consumption of seafood collected in an area approximately prescribed by this radius, including the northern part of the Shoalwater Islands Marine Park. Limited data from the WAWA 'mussel watch' stations off Cape Peron show faecal contamination in mussels to be elevated over controls but not exceeding health criteria.

Surveys of microbiological water quality off Perth's mainland beaches carried out by the Department of Health and the City of Rockingham indicate that faecal coliform concentrations did not exceed the criterion value for draft Environmental Quality Objective 4, indicating that the current microbiological status of the water at these beaches does not pose a risk to human health via direct contact recreation. Faecal coliform concentrations in the waters off Palm Beach and near the Safety Bay jetty exceeded the criterion value for draft Environmental Quality Objective 3, indicating that consumption of seafood collected from these areas may pose a human health risk. Water quality monitoring for faecal coliforms is not undertaken near boat mooring/direct contact recreation areas at Garden Island or Rottnest Island and hence there are no data available to assess potential health risks in these areas.

Action 20

Under Part V of the E P Act, the Department of Environmental Protection will require that the Water Corporation implements a faecal coliform monitoring and management strategy, with annual environmental performance indicators, to ensure that criteria for draft Environmental Quality Objective 3 (i.e. maintenance of aquatic life for human consumption) and draft Environmental Quality Objective 4 (i.e. maintenance of recreation values) for areas of Sepia Depression influenced by the Cape Peron outfall (excluding designated exclusion zones) are achieved by 31 March 2002.

Action 21

The Department of Environmental Protection will request the Department of Health to investigate the cause/s of the chronic exceedance of the faecal coliform criterion for draft Environmental Quality Objective 3 (i.e. maintenance of aquatic life for human consumption) at Palm Beach and Safety Bay and, if necessary, implement a management strategy to reduce faecal contamination to levels consistent with this criterion.

Action 22

The Department of Environmental Protection will request the Department of Health to assess the microbiological water quality at areas commonly used both for boat mooring and direct contact recreation, at Rottnest Island, in association with the Rottnest Island Authority, and at Garden Island, in association with the Commonwealth Department of Defence and, if necessary, to implement a management strategy to reduce faecal contamination to levels consistent with draft Environmental Quality Objective 3 (i.e. maintenance of aquatic life for human consumption) and draft Environmental Quality Objective 4 (i.e. maintenance of recreation values).

7.4 Introduction of foreign organisms

Systematic surveys to determine the extent of foreign organism introductions to Perth's coastal waters have not been undertaken. At present 21 foreign species, the most recent being the marine worm *Sabella* cf. *spallanzanii*, have been recorded in these waters and are considered to have been introduced by the discharge of ships' ballast water and associated sediments or by dislodgement from the hulls of ships. Currently, over 60 foreign species have successfully established in Australian coastal waters following introduction, with some species such as the northern pacific seastar, *Asterias amurensis*, causing widespread biological and commercial impacts in southeastern Australia. The introduction of foreign organisms into Perth's coastal waters is a threat to the maintenance of ecosystem integrity.

In 1994, ships visiting the Port of Fremantle discharged an estimated 3.5 million tonnes of ballast water into port waters. Shipping activity is projected to increase significantly over the next decade and, therefore, the potential for the introduction of foreign organisms into the coastal waters of Perth from this source will increase proportionally if current disposal practices remain. Given the high conservation, recreational and commercial value of these waters, the introduction of foreign organisms via shipping represents a significant threat to the ecological and cultural values of the metropolitan coastal waters of Perth.

The issue of ballast water management is as much an international issue as it is Australian. Hence, action must be global. Australia has developed a "Draft Australian Ballast Water Management Strategy" (Australian Quarantine and Inspection Service, 1994). It is consistent with the International Maritime Organisation (Maritime Environment Protection Committee) "Guidelines for Preventing the Introduction of Unwanted Aquatic Organisms and Pathogens via Ships' Ballast Water and Sediment Discharges". The International Maritime Organisation is working to have those guidelines developed as a mandatory "Annex" to the United Nations International Convention for the Prevention of Pollution from Ships (MARPOL 1973/78). Such an initiative, when implemented, will go a long way to reduce the risk of the introduction of exotic organisms into Australian waters.

Action 23

The Department of Environmental Protection will recommend to the Minister for the Environment that the Western Australian Government request the Australian Government to encourage the International Maritime Organisation to expedite, as a matter of high priority, finalisation of an Annex to the International Convention for the Prevention of Pollution from Ships (MARPOL 1973/78) requiring new vessels, especially bulk carriers and tankers, to have upgraded ballast water management systems.

Action 24

The Department of Environmental Protection will recommend to the Minister for the Environment that the Western Australian Government request the Australian Government to encourage the International Maritime Organisation to research strategies, such as in-transit sterilisation of ballast waters, to minimise risk of introduced organisms from ballast water discharge to Australian waters.

Action 25

The Department of Environmental Protection will recommend to the Minister for the Environment that the Western Australian Government request the Australian Government to implement incentives to encourage ships with appropriate ballast water management systems.

Action 26

The Department of Environmental Protection will request the Department of Transport and Fremantle Port Authority to further encourage ship operators to adopt the guidelines recommended in the Australian *Draft Ballast Water Management Strategy*.

Action 27

The Department of Environmental Protection will request the Department of Transport and Fremantle Port Authority to jointly examine the Australian Draft Ballast Water Management Strategy and implement practical measures as soon as possible.

Action 28

The Department of Environmental Protection will request the CSIRO Centre for Research on Introduced Marine Pests to give high priority to research activities related to the formulation of ballast water risk minimisation strategies.

7.5 Direct impacts

7.5.1 Shellsand mining

Cockburn Cement Ltd (Cockburn) is dredging shellsands from Success Bank for the manufacture of lime and cement. This operation removes seagrasses along with the shellsand and increases the depth from about five to 15 m. Since operations began in 1972, approximately 100 ha of seabed have been modified and 90 ha of seagrass meadows have been removed. The company's lease agreement runs to 2010 with an option to extend the lease a further 11 years. By 2021, 780 ha of seabed will have been affected and 660 hectares of seagrasses will have been removed at projected mining rates using current practices. Further losses of seagrass may also occur due to indirect effects such as bank instability and slumping.

The environmental implications of Cockburn's operation on the Owen Anchorage/Cockburn Sound ecosystem area are currently being determined through a separate Environmental Protection Authority/Department of Environmental Protection process. For this reason no recommendations are made on the long-term impacts of shellsand mining in Owen Anchorage in this report.

7.5.2 Trawling

Scallop and prawn trawling are carried out in the coastal waters of Perth. Since the early 1990s there has been considerable community concern regarding trawling activities in Comet Bay. These concerns centred around the risk of impacts on sensitive benthic habitats, such as seagrass meadows and fragile limestone reef communities, and on non-target fish populations, such as the sand whiting, which are important recreational target species.

Many of these concerns were addressed in the report by the South West Trawl Limited Entry Fishery Working Group (1995) and were the subject of several recommendations in that report. As such, no recommendations are made on the impacts of trawling in this report.

7.5.3 Aquaculture

Mussel farming leases occupy about 70 ha in the northwestern side of Cockburn Sound, with a spat rearing area of 50 ha in the southeastern side of the sound, near the CBH jetty. A smaller lease occupies about eight hectares in the southwestern part of Warnbro Sound. In the absence of detailed historical data the impacts of these operations on the underlying benthic communities cannot be assessed in anything other than very broad terms. In both areas there is some visible evidence to suggest physical impacts on benthic communities, with accumulations of shells on the seabed. In the Cockburn Sound lease area there has been localised loss of seagrasses in the shallow areas. However, most of the lease area is located in deeper waters where abundant seagrasses never existed.

The presence of mussel leases in Cockburn and Warnbro sounds can reduce the aesthetic quality of the surrounding waters by their visible presence and associated infrastructure. More importantly, aquaculture leases reduce access to areas of coastal waters by creating what are essentially 'public exclusion zones' in areas that were formerly 'public open space'. This is an important equity issue in areas of multiple use, particularly when lease areas are relatively large and inevitably occupy relatively calm, sheltered areas where the water quality is high, characteristics often sought by recreational users.

Action 29

The Department of Environmental Protection will request the Inter-Departmental Committee on Aquaculture to develop guidelines to ensure that the cumulative impacts of existing aquaculture ventures are taken into account when evaluating new aquaculture proposals, and to assist in achieving/maintaining ecosystem integrity, recreation values and aesthetic values in the southern coastal waters of Perth.

7.6 Aesthetic quality of coastal waters

The aesthetic quality of the marine environment can be reduced by many factors. These include waste inputs or phytoplankton blooms clouding the water, oils and other materials floating on the water surface, offensive odours from waste discharges, and exclusion from areas by commercial activities. All of the above factors periodically exceed the criteria for draft Environmental Quality Objective 5 (i.e. maintenance of aesthetic values) in parts of Perth's southern coastal waters.

In summer, high phytoplankton (as chlorophyll *a*) concentrations on the eastern side of Cockburn Sound often exceed the criteria for draft Environmental Quality Objective 5. During periods of prolonged easterly winds, these waters are driven westward, and can cloud the waters in the high recreational usage areas along the eastern shoreline of Garden Island. Sheens of petroleum and fine material (e.g. aluminium dust and wheat chaff) are periodically seen on the waters along the eastern side of Cockburn Sound due to accidental spillages/discharges from industries and shipping operations. Due to combinations of these and other factors, the criteria for draft Environmental Quality Objective 5 are frequently exceeded, particularly within one to two kilometres of the eastern shoreline in Cockburn Sound and Owen Anchorage.

The wastewater discharged from the Cape Peron outfall contains, among other things, grease and suspended solids and, on calm days, a sheen can extend for more than five kilometres from the outfall. A plume of turbid water is generally visible within a 500 m radius of the outfall. Odours are frequently present in close proximity to the outfall and can sometimes be detected for up to five kilometres downwind. Together, these factors result in the criteria for draft Environmental Quality Objective 5 being frequently exceeded over a distance of up to five kilometres from the outfall.

Action 30

Under Part V of the E P Act, the Department of Environmental Protection will require that the Water Corporation implements a management strategy, with appropriate annual environmental performance indicators, to ensure that draft Environmental Quality Objective 5 (i.e. maintenance of aesthetic values), for areas of Sepia Depression influenced by the Cape Peron outfall (excluding designated exclusion zones), is achieved by 31 March 2002.

Action 31

The Department of Environmental Protection will request the Fremantle Port Authority, in collaboration with major port users, to implement codes of practice for vessel wash down and cargo handling operations to further reduce and minimise impacts on the aesthetic quality of Perth's coastal waters from these operations in port waters.

7.7 Management

7.7.1 Integrating mechanisms

With more than 70% of the population of Western Australia living within 20 km of the metropolitan coastline, the coastal waters of Perth are the most used and, in that sense, 'valuable' parts of Western Australia's marine environment. Perth's population is expected to increase by more than 50% over the next 30 years resulting in significant urban expansion along the coastal strip and increasing recreational activity in the coastal waters. This growth will inevitably lead to greater demands being placed on Perth's marine environment. As these demands increase, so too does the need to provide sound environmental management.

The primary objectives of the SMCWS and the Perth Coastal Waters Study were to provide the information required for improved management of current impacts and better strategic planning to prevent a repetition of the problems that have occurred in the marine waters of many coastal cities around the world. Currently, there is no formal integrating management framework for these waters; instead, environmental management is presently the responsibility of numerous individual agencies, operating across four jurisdictions, ranging from local Government by-laws to international treaties. The current situation does not provide the level of integration necessary to ensure the multiple use of these waters is both socially equitable and ecologically sustainable.

This study has highlighted the important linkages between a wide range of land-based human activities, occurring both in urban and rural catchments, and the environmental quality of the coastal waters. The study has also demonstrated that the spatial scales of transport and cumulative ecological effect associated with waste discharge to Perth's coastal waters can extend across boundaries between jurisdictions (e.g. between state and commonwealth waters). An integrated management approach would better address these issues.

Recommendation 8

That the Environmental Protection Authority recommend to the Minister for the Environment that the Western Australian Government establish a formal framework to coordinate environmental management within Perth's metropolitan coastal waters (nominally Dawesville Channel to Yanchep) and between these waters and their land catchments.

Action 32

The Department of Environmental Protection will recommend to the Minister for the Environment that the Western Australian Government establish a formal framework to coordinate environmental management within Perth's metropolitan coastal waters (nominally Dawesville Channel to Yanchep) and between these waters and their land catchments.

Recommendation 9

That the Environmental Protection Authority recommend to the Minister for the Environment that the Western Australian Government develop a Memorandum of Understanding with the Australian Government to facilitate coordinated environmental management of State and Commonwealth waters off Perth.

Action 33

The Department of Environmental Protection will recommend to the Minister for the Environment that the Western Australian Government develop a Memorandum of Understanding with the Australian Government to facilitate coordinated environmental management of State and Commonwealth waters off Perth.

Recommendation 10

That the Environmental Protection Authority develop an Environmental Protection Policy for Perth's coastal waters.

Action 34

The Department of Environmental Protection will recommend to the Minister for the Environment that the Western Australian Government consider including the Rottnest Island Aquatic Reserve within the statewide marine conservation reserve system under the CALM Act 1984.

Action 35

The Department of Environmental Protection will support the proposed extension of the Shoalwater Islands Marine Park, northward off the western shores of Garden Island and Carnac Island, and seaward to the limit of the State Territorial Sea, consistent with the report and recommendations of the Marine Parks and Reserves Selection Working Group (1994).

7.7.2 Further studies and monitoring

Further studies are needed to clarify particular issues identified in this report. For example, the apparent link between anthropogenic nutrient inputs to Perth's coastal waters and the winter occurrence of silicoflagellate blooms in these waters requires further study before a cause-effect relationship can be established or dismissed.

Where the causes of environmental problems are cross-sectoral, the responsibility for addressing these problems rests with government. The issues of broadscale tributyltin contamination and the status of seagrass meadows in Cockburn Sound, Owen Anchorage and throughout Perth's coastal waters, are examples of such problems.

The CSIRO's water quality monitoring programme is an extremely valuable long-term data set that provides information on background conditions and interannual variability. This programme should be continued and expanded to include additional parameters (e.g. chlorophyll *a*) and transects (e.g. off Mandurah and Yanchep).

Estimates of individual and total inputs of all contaminants to Perth's coastal waters need to be regularly updated, refined and readily accessible to support environmental management and decision making.

Action 36

The Department of Environmental Protection will ensure further studies are undertaken to determine whether the winter silicoflagellate blooms observed in Perth's coastal waters over recent years are occurring in comparable temperate waters of Western Australia that are not significantly influenced by anthropogenic waste inputs.

Action 37

The Department of Environmental Protection will ensure that monitoring of sediments and biota for tributyltin in Perth's coastal waters is conducted every three years or until draft Environmental Quality Objective 2 (i.e. maintenance of ecosystem integrity) and draft Environmental Quality Objective 3 (i.e. maintenance of aquatic life for human consumption) are achieved.

Action 38

The Department of Environmental Protection will ensure that the elevated mercury levels in the sediments of Cockburn Sound and Owen Anchorage are further investigated, with a view to identifying the sources, the ecological implications and the necessity for remedial action.

Action 39

The Department of Environmental Protection will ensure that the health and distribution of seagrass meadow communities in Cockburn Sound and Owen Anchorage are monitored annually and triennially, respectively.

Action 40

The Department of Environmental Protection will request the CSIRO to continue and expand its long-term water quality monitoring programme in Perth's coastal waters.

Action 41

The Department of Environmental Protection will ensure that an inventory of contaminant inputs to Perth's coastal waters is updated annually and made publicly available.

7.7.3 Review of Environmental Quality Objectives

Urban expansion of Perth is projected to increase rapidly over the coming decades particularly along the coastal strip. This development will increase recreational demand and, to a lesser extent, commercial activity in the nearshore marine environment, thereby increasing the potential for conflict between users. Once finalised, the Environmental Quality Objectives for Perth's coastal waters should be regularly reviewed by the Environmental Protection Authority in collaboration with the Department of Environmental Protection to ensure that these objectives are consistent with the community's current and future aspirations for these waters.

Recommendation 11

That the Environmental Protection Authority formally review the Environmental Quality Objectives for Perth's coastal waters every seven years.

GLOSSARY

Anthropogenic	Created by humans.
Assemblage	Recognisable grouping or collection of individuals or organisms.
Baroclinic	Hydrodynamic processes involving the horizontal movement of water due to differences in water density.
Barotropic	Hydrodynamic processes involving the horizontal movement of water due to factors other than water density.
Bathymetry	The measurement of ocean depths to determine the sea floor topography.
Benthos (benthic)	All marine organisms living upon or in the sediment of the sea.
Bioavailability	That portion of a chemical compound or element that can be taken up readily by living organisms.
Biodiversity	The variety of all life forms: the different plants, animals and micro-organisms, the genes they contain and the ecosystems they form. It is often considered at three levels: genetic diversity, species diversity and ecosystem diversity.
Biological oxygen demand	Measures of oxygen depletion in water due to bacterial decay of organic pollutants. Gives an indication of how much organic matter is in the water.
Biota	The plants, animals and micro-organisms of a region.
Buoyancy flux	The introduction or loss of heat, freshwater or salt to the water column, thereby changing the density characteristics of the water.
Chlorophyll *a*	Chlorophyll *a* is a complex molecule that along with other similar molecules, is able to capture sunlight and convert it into a form that can be used for photosynthesis. All plants contain chlorophyll *a* and the concentration of this molecule in water is commonly used as a measure of phytoplankton biomass.
Community	Ecologically, any naturally occurring group of different organisms sharing a particular habitat.
Contaminant	Any physical, chemical or biological substance or property which is introduced into the environment. Does not imply an effect (see pollution).
Diffuse source contamination	Contamination from multiple and widely-distributed sources, such as urban areas contaminating groundwater and agricultural lands contaminating surface water runoff.
Dinoflagellates	A group of single-celled algae.
Downwelling	The downward displacement of coastal water by the incursion of more buoyant offshore water.
Ecosystem	Unit including a community of organisms, the physical and chemical environment of that community, and all the interactions among those organisms and between the organisms and their environment.
Effluent	A complex waste material which is a by-product of human activity.
Endemic	'Native' species confined to a given region.

Environmental quality objectives (EQOs)	The long-term goals of an environmental management programme in relation to the maintenance of the environmental (ecological and cultural) values of natural systems.
Environmental quality criteria (EQC)	The scientific benchmarks upon which a decision may be made concerning the ability of an environment to maintain certain designated environmental quality objectives.
Environmental Values	Particular values or uses of the environment that are conducive to public benefit, welfare, safety or health and that require protection from the effects of pollution, waste discharges and other human activities.
Epiphytes	Algae that are attached to the leaves of other plants.
Eutrophication	An increase in the rate of supply of organic matter to an ecosystem caused by un-naturally high loads of nutrients to that ecosystem.
Flushing rate	The rate at which the water in a specified volume is replaced by different water.
Grazers	Animals which eat plants, such as algae, by cropping.
Heavy metals	Metals such as zinc, copper, chromium which accumulate in sediments and tissues of animals and may be passed up the food chain.
Hydrodynamic	The movement or mixing of water as a response to applied forcings, such as wind stress at the water surface.
Macroalgae	Group of filamentous and fleshy non-flowering aquatic plants that are found in estuaries and the ocean.
Nitrification	The process whereby compounds like ammonia are oxidised to nitrites and nitrates, especially by bacterial action.
Nutrients	Elements or compounds essential for organic growth and development such as nitrogen and phosphorus.
Oligotrophic	Describes a waterbody with low rates of production of organic matter. Opposite of eutrophic.
Organochlorines	Highly persistent, toxic pesticides such as aldrin, chlordane, DDT, dieldrin, heptachlor and lindane. Bioaccumulates in the fatty tissues of animals and may be passed up the food chain.
Organophosphates	Toxic pesticides such as chlorpyrifos that are generally far less persistent than the organochlorines. If ingested at sub-lethal doses, these compounds are generally broken down by animals and excreted and therefore not passed up the food chain.
Penetrative convection	The downward penetration of the surface mixing layer as the surface water cools, becomes dense and therefore tumbles downward as a turbulent mixing front.
Periphyton	Algae that are attached to abiotic substrates e.g. artificial seagrass leaves.
Photosynthesis	A process, operating in chlorophyll containing plants, which uses solar energy to convert carbon dioxide and water into carbohydrate.
Phytoplankton	Single-celled plants (algae) that live in the water column.
Point-source contamination	Contamination from a localised, well-defined source of contaminants, such as from an industrial outfall.

Pollution	The introduction by man, directly or indirectly, of substances or energy into the environment, which results in deleterious effects to the environment.
Seagrass	Submerged flowering plants that mainly occur in shallow marine areas and estuaries.
Sewage	Domestic wastewater.
Standing crop	The biomass or organic matter present on a given area at a given time.
Stratification	Layering (vertical or horizontal) in a water property such as salinity or temperature.
Suspended solids	Any solid substance present in water in an undissolved state.
Thermocline	Region below the surface layer of the sea, where temperature changes rapidly with increasing depth.
Upwelling	The offshore displacement of coastal water and replacement through the upward movement of less buoyant (deeper) offshore water.
Wastewater	Domestic, industrial and municipal effluent.

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The Directory of Publications provides a summary of the publications from research programmes and studies associated with the Study. The Directory includes technical publications such as scientific papers, technical reports and post-graduate theses that provide the detailed supporting technical information from which the major findings, conclusions and recommendations of the Study are drawn. A list of data reports is also provided. The reports and digital copies of the data for each data report are stored in the DEP library.

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