PORT PHILLIP BAY ENVIRONMENTAL STUDY

FINAL REPORT

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CSIRO
AUSTRALIA

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About the Study

In 1991, CSIRO was approached to design and manage an environmental study of the Bay to seek answers to questions concerning the sustainable use and management of the Bay. The Study began in October 1992 and finished in June 1996.

The Port Phillip Bay Environmental Study was managed by CSIRO and supervised by a Management Committee chaired by the (then) Department of Conservation and Natural Resources. Also represented on the committee were the EPA, Melbourne Water, Melbourne Parks and Waterways and the former Port of Melbourne Authority. The Study was totally funded by Melbourne Water and Melbourne Parks and Waterways.

In all, some 47 research tasks were conducted, ranging from scientific literature reviews to full-scale field surveys. All were contracted out. Of the 30 contractors, six were Victorian State agencies, three Commonwealth agencies, nine universities and 12 consultants. Nineteen contractors were Victorian based, seven from interstate and four from overseas.

The research programs covered the fields of physical oceanography, toxicants, algal nutrients, marine ecology and ecological modelling.

An important feature was the development of a numerical model which integrated data collected during the present and previous studies. This model describes and quantifies the chemical and biological processes in the Bay ecosystem. It enables prediction of the likely effects of gross changes in catchment practices.

The toxicant research program showed the Bay waters, sediments and biota to be relatively unpolluted. Exceptions were found in areas close to sources such as Corio Bay, Hobsons Bay and the mouths of creeks and drains.

The Study confirmed unequivocally that nitrogen is the nutrient element limiting algal growth in the Bay. This was found to be the result of denitrification of most nitrogen entering the Bay from both the catchment and sewage effluent.

The Bay hosts a rich and diverse flora and fauna with very high productivity. Most of this plant and animal production forms a huge recycling system and there is little evidence of nutrients accumulating in the Bay.

The Study was able to identify the critical nutrient loading beyond which irreversible damage to the Bay would occur. Recommendations are made concerning sustainable nutrient loads. The highest priority is to reduce the nutrient loads from stormwater flows in the Yarra catchment and on the eastern shore.

The biodiversity of the Bay is important and must be conserved. The Study has identified a number of issues concerning habitat destruction, resource management, restoration and conservation which need attention. The introduction of exotic species poses a significant threat to the Bay. Monitoring and control of the introduction of such organisms should be introduced.

Finally, the Study makes a number of recommendations about ongoing monitoring and the importance of continued modelling for the sustainable management of the Bay. An ongoing link between science and management is essential.

To ensure the sustainable use of the Bay for generations to come, management of the Bay must be seen in the context of both the Bay ecosystem and its catchments. The entire catchment/estuary system must be managed as a single entity.

The Study Team

Graham Harris (Director)

Graeme Batley	David Fox	Douglas Hall
Peter Jernakoff	Robert Molloy	Alexander Murray
Brian Newell	John Parslow	Graham Skyring
	Stephen Walker	

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Acknowledgements

It is simply not possible to thank individually all the many people who have contributed to this Study. We have lost count of the number of people who have been involved over the past four years. We have enjoyed the most complete co-operation and generous assistance of many State and Commonwealth Agencies, universities, private consultants and individuals. We thank you all for your participation and assistance with the work of the Study. This has been a collective effort; you all share in the success.

We wish to thank all the members of the Management Committee and the Scientific Review Committee for their support and encouragement. In particular, we thank Jeff Floyd, Wayne Chamley and John Bales for their role as chairpersons over the years.

A special vote of thanks is due to Nancy Millis for her role as interlocutor on the committees and for her personal involvement in the scientific work of the Technical Group.

Professor Chris Crossland, now of the Coral Reef CRC in Townsville, conceived the Study and was responsible for the basic design. All his enthusiasm and hard work have been amply rewarded by the successful outcome.

We wish to thank the staff of the Study Office who liaised with the State Agencies, managed the contracts and made all the detailed arrangements. You played an essential role as representatives, diplomats and managers. The Study would simply not have come together the way it has without your efforts.

It is unfair perhaps to single out a few key contributors but the Technical Group does wish to make special mention of John Morgan, Chief Executive of Melbourne Water, for his strong support, and Peter Scott and his staff for their invaluable liaison role throughout.

Our thanks to May Ong and others who provided adminstrative support to the project.

To everyone, too numerous to mention individually, our thanks for your assistance with what has been for all of us a remarkable four years.

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In addition, 11 reviews were commissioned which have been published as PPBES Technical Reports Nos 1, 2, 3, 4, 5, 6, 7, 8, 14, 15 and 16. Another two reviews will appear as Technical Reports in due course.

N.B. Tasks N1.3/T1.2 and P7 both involved two separate exercises making a total of 47 research tasks in all.

A final report on the integrated model and a description of a proposed monitoring program will also be published in due course.

Major Contractors

Australian Geological Survey Organisation, Canberra
Consulting Environmental Engineers, Victoria
Centre for Water Research, University of Western Australia, Western Australia
Division of Fisheries, CSIRO, Tasmania
Division of Water Resources, CSIRO, Canberra
Division of Oceanography, CSIRO, Tasmania
Environment Protection Authority, Victoria
HydroTechnology Pty Ltd, Victoria
James Cook University, Queensland
Lawson and Treloar Pty Ltd, Victoria
Marine and Freshwater Resources Institute (previously the VIctorian Fisheries Research Institute), Victoria
Melbourne Water, Victoria
Monash University, Victoria
Museum of Victoria, Victoria
NSR Environmental Consultants Pty Ltd, Victoria
Patterson Britton and Partners Pty Ltd, New South Wales
University of Melbourne, Victoria
University of Southern California, Los Angeles
University of Tasmania, Tasmania
Victorian Institute of Marine Sciences, Victoria
Water EcoScience Pty Ltd, Victoria
WNI Science and Engineering, Western Australia

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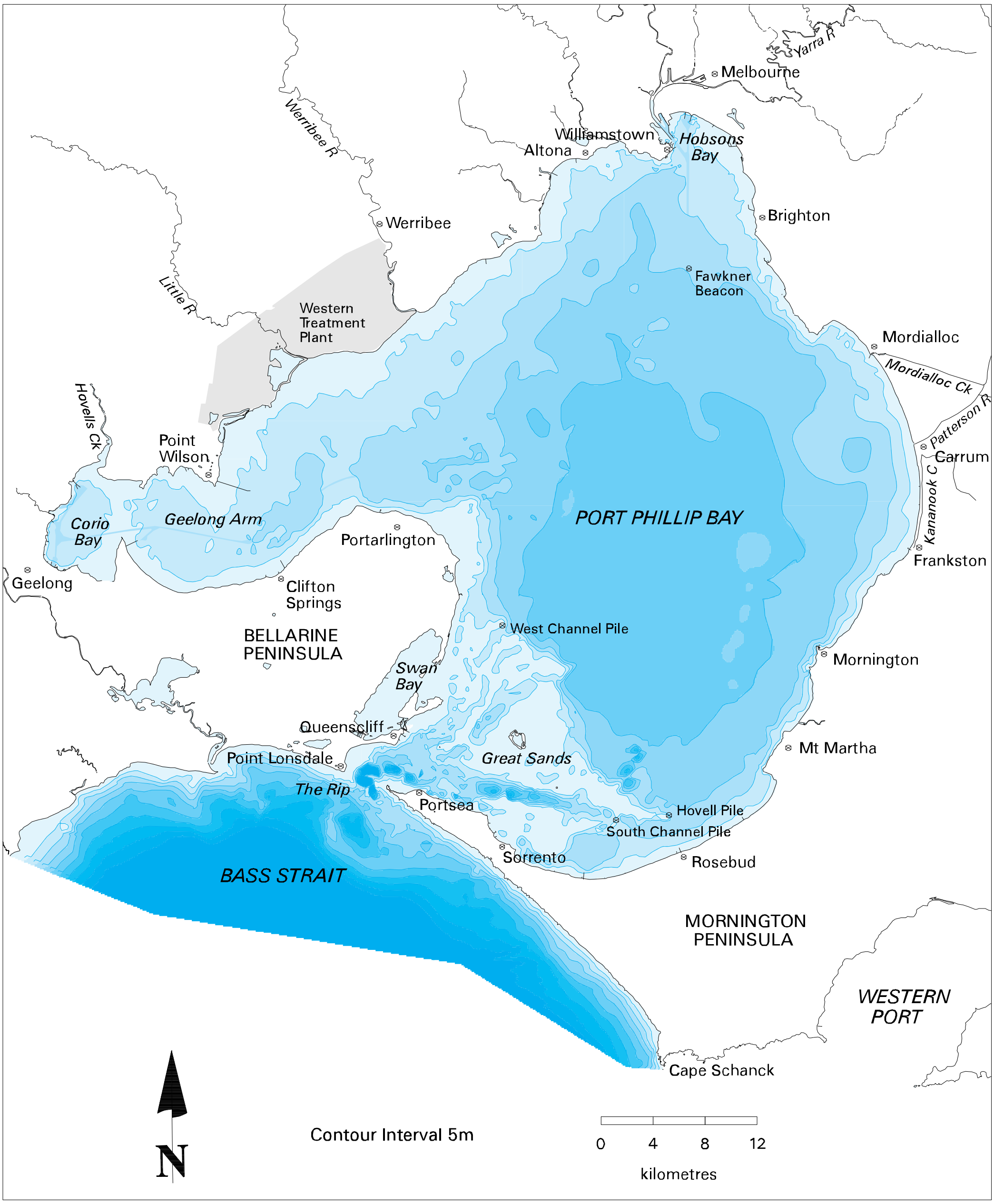

Figure 1.1: Locality map of Port Phillip Bay.

1. Port Phillip Bay

Port Phillip Bay was discovered by John Murray in the 'Lady Nelson' on 5 January 1802.

In October, 1803, an exploration party under the leadership of Lieutenant-Colonel Collins (later Lieutenant-Governor) arrived in HMS 'Calcutta'. Their mission was to examine the potential for establishing a penal colony, but unfavourable reports from the shore parties led Collins to declare the place unsuitable for settlement.

The first European settlement, apart from the ill-fated Sorrento venture which provided only one permanent resident, William Buckley, had to await the arrival of John Batman on the present site of Melbourne in 1834.

Port Phillip Bay is a large marine embayment about 1,930 km² in area and with a coastline length of 264 km. It originated about 8,000 years BP when eustatic rise in sea level following an end to the last ice age resulted in flooding of the delta of the Yarra, Werribee and Little Rivers and Kororoit Creek.

The Bay is extremely shallow for its size. The deepest portion is only 24 m and half the volume is in waters shallower than 8 m. The total volume of water in the Bay is about 26 km³.

The land catchment area of the Bay is 9,790 km² which consists of 21 natural drainage basins.

Of these, only eight deliver runoff directly to the Bay. Of the remainder, six contribute diffuse runoff through drains and seven contribute runoff to the Yarra River (Figures 1.1 and 1.2).

About three million people live and work in the catchments, most of them in Melbourne which occupies the lower reaches of the Yarra and Maribyrnong Rivers and Moonee Ponds and Merri Creeks.

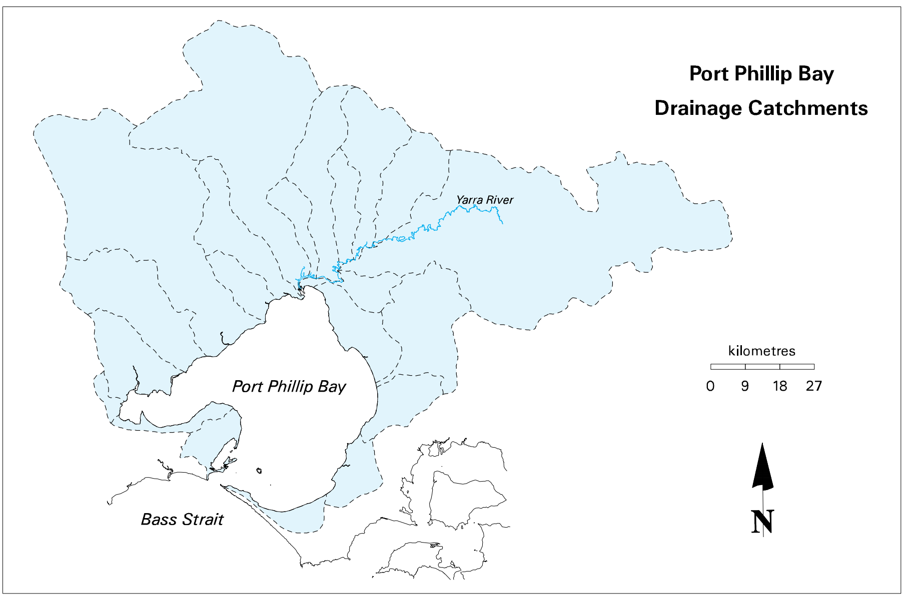

Figure 1.2: The Bay and its catchments.

Despite this large population concentration, only 11% of the catchment area is urbanised. However, the urban runoff to the Bay is highly significant in its contributions of nutrients, heavy metals like zinc and lead and some organic contaminants such as petroleum oils and industrial chemicals.

That part of the catchment utilised for agriculture and horticulture provides major inputs of suspended matter, nutrients and biocides and some fixed nitrogen derives from forested catchments.

Groundwater, with its content of natural and anthropogenic material, provides the base flow of the rivers and creeks but its contribution to loads into the Bay via the Bay floor appears to be negligible.

The importance of Port Phillip Bay to Victoria, and to Melbourne in particular, can hardly be overstated. It is the historic gateway to the city and provides a vista from numerous vantage points. Its degradation by eutrophication or gross pollution would be an aesthetic disaster with profound sociopolitical consequences.

As well, the Bay provides various recreational opportunities for Melburnians and visitors. Use of the Bay ranges from swimming through diving, sailing and pleasure boating to angling. If all recreational pursuits involving the Bay are counted then it receives about 30 million visits per year. Apart from the social benefits, this recreational usage has large economic significance in terms of the purchase of equipment and expenditure on fuel and services. It is not easy to see how this activity would be transferred were the Bay to become unusable.

In addition to recreational fishing or angling, the Bay supports a commercial fishery. From 700 to 2,000 t of finfish are taken annually by commercial fishermen with no net trend of catch over 80 years of record. More than 60 species of finfish are caught for sale with an annual wholesale value of about $3 million. About 600 t of cultured mussels are harvested for sale each year with an average value of $1½ m while abalone, gathered from reefs by divers, amount to 50 t with a value of $1 m.

The Bay is also, of course, the means of access to the Ports of Melbourne and Geelong. Each year about 2,500 ships cross the Bay to the Port of Melbourne and 350 to the Port of Geelong.

Total shipping amounts to 35 m gross registered tonnes a year with an average vessel size of 10,000 t. This traffic handles cargo worth $28 billion a year through the Port of Melbourne. The port industry involves 677 firms, 8,200 employees and a turnover of $975 m.

A significant use of the Bay is the disposal of more than 400 ML/d of treated sewage effluent from the Melbourne Water Western Treatment Plant (WTP) at Werribee. This effluent contains large loads of nitrogen and phosphorus and much lesser loads of common pollutants.

Finally, Port Phillip Bay has an intrinsic value in itself as a unique ecosystem. It supports a diverse community of birds, fish, benthic invertebrates, macrophytes and microscopic algae. Its biochemical and geochemical systems have evolved over more than 8,000 years of terrestrial input and oceanic exchange and even appear to have adapted to the past 150 years of anthropogenic influences. It is one of the largest examples in the world of a shallow embayment in which the benthic ecosystem is predominant.

2. Previous studies of the Bay

The first known studies of the Bay consisted of biological surveys conducted under the auspices of the Royal Society of Victoria in the 1880s and reported in 1890. Several taxa of flora and fauna were collected and identified.

Much later, from 1947 to 1952, CSIRO conducted a program of monthly sampling of Bay waters at six stations, five of which formed an arc from Hobsons Bay to Bass Strait.

Despite the less sophisticated techniques available then, the basic findings remain valid. These were that Yarra River inflows travel down the eastern side and bring nutrients into the Bay, that the salinity of the Bay is, in the long term, similar to that of Bass Strait and that dissolved oxygen levels are high except in bottom waters during periods of stratification.

Interestingly, the levels of nitrate found were similar to those prevailing now, despite the huge increase in settlement of the catchment. The significance of this becomes apparent when the present Study findings are considered.

A decade later, the National Museum of Victoria, in conjunction with the Fisheries and Wildlife Department and assisted by numerous volunteer divers and naturalists, conducted a comprehensive survey of the macro flora and fauna of the Bay.

The whole Bay area was divided into 70 grid squares and each was visited, with specimens being collected by divers or a dredge. Rapid preliminary surveys were made by a manned sled towed behind the research vessel.

Altogether, 172 species of macrophyte algae were identified and 19 macroinvertebrate families were described. Additionally, the first comprehensive descriptions of the geology and geomorphology of the Bay, its surrounds and the bottom sediments were compiled, and an account of the coastal vegetation prepared.

While the biological surveys carried out during this six-year program (1957 to 1963) were not quantitative, they were the first disclosure of the richness and diversity of the Bay ecosystem.

A few years later, from 1968 to 1971, a much more comprehensive environmental survey was mounted. This had its origins in a proposal by the then Melbourne and Metropolitan Board of Works to construct a sewage treatment plant at Carrum, effluent from which would be discharged to the Bay near the outlet of the Patterson River.

Because of public disquiet over the likely effects on the Bay, the Board and the Fisheries and Wildlife Department agreed on a joint study of the physics, chemistry and biology of the Bay to provide base-line data and informed estimates of the effects of existing inputs from the catchment.

In the event, the effluent from the Eastern Treatment Plant at Carrum (commissioned in 1975) was piped to Cape Schanck. This decision was made by Government in 1969 but the study of the Bay was still carried out in order to have a base-line from which to assess long-term effects of urbanisation on the Bay.

The magnitude of this exercise, nowadays called the 'Phase One Study', was remarkable. Not only was a description compiled of the known physical, economic and demographic characteristics of the Bay catchment, but an extensive survey of the physical, chemical and biological characteristics of the Bay itself was also carried out.

To some people's surprise the Bay was found to be in a remarkably healthy state. Although high levels of the nutrient elements nitrogen and phosphorus were found adjacent to the Werribee coastline and in Hobsons Bay, these tapered off rapidly in the Bay to levels akin to those found in Bass Strait. The Bay waters were well mixed and generally saturated with oxygen, and the microalgal population was mostly similar to or lower in magnitude than offshore.

More than 60 species of fish were recorded and the population of benthic invertebrates was rich and diverse. The significance of this latter finding has only become apparent during the current Study.

Also surprisingly, there was little or no ecological effect discernible of the major inputs to the Bay.

Dense stands of macrophytes were found off Werribee, but these were diverse and not indicative of domination by taxa characteristic of pollution. There was some evidence of selection of benthic invertebrates immediately adjacent to WTP outfalls but this seemed due to the fresh water effluent favouring euryhaline taxa and the outflow of organic particulates favouring deposit feeders. There was no evidence of toxicant effects.

The physical measurements confirmed that tidal movements in the Bay were sluggish and that most net movement of water was driven by wind. Estimates were made of the flushing time of the Bay and these converged on a value of just over one year.

While the Phase One Study provided a two-year span of observations, the picture finally presented was a 'snapshot' or static one. Little was deduced of actual processes operating in the Bay ecosystem.

Furthermore, while the approximate magnitude of exchange of water and materials with Bass Strait had been elucidated, the knowledge of inputs to the Bay was scanty. Many inputs had to be derived indirectly. Hence, except for some major inputs such as phosphorus it was not possible to develop Bay-wide budgets of materials.

These deficiencies were recognised by the authors of the Phase One Study report who recommended that a second phase be carried out to address:

- Comprehensive monitoring of inputs, especially in wet periods.
- Further studies on near-shore water movements and exchange across the Bay entrance.
- Further monitoring off the WTP and Yarra River to confirm distribution patterns and assess the effects of diversion of sewage effluent to Cape Schanck.
- Ecological investigations of biological communities to observe interaction between communities, the transfer of nutrients, the detection of indicator species and the reasons for the apparent absence of eutrophication in the Bay.
- An attempt to refine quantitative relationships in the Bay ecology and produce a numerical model able to predict the results of changes to inputs.

In the following years, the first point was addressed somewhat spasmodically, so that the current Study began with still a fragmentary account of inputs to the Bay from rivers, creeks and drains.

The second and third points were addressed only insofar as a consultant study was carried out in 1987-1988 on the behaviour of the Werribee plume. The survey was brief, of only a few days duration, and covered a limited range of weather conditions.

It confirmed previous understanding of plume mixing and quantified this to the extent that it was shown that winds greater than 5 m/s cause turbulent mixing of the plume. In calmer conditions, the plume spreads as a freshwater layer over the salt water of the Bay, mixing by shearing and entrainment.

The Yarra plume behaviour was examined only in Hobsons Bay by the Heated Effluent Study of 1973-1976 and the Webb Dock Study of 1976-1977.

While these two studies were extremely thorough and led to a much greater understanding of the physics, chemistry and biology of Hobsons Bay, they were limited by their spatial terms of reference. They did give the first description of diatom blooms in Hobsons Bay in response to Yarra flow and the present study has shown this to be a significant phenomenon.

The fourth and fifth points, essential to an understanding of the ecological processes and nutrient budget in the Bay were never properly addressed. Limited though valuable studies were carried out during a 'Phase Two' study in the 1970s and a new hydrodynamic model was developed but no integrated modelling was attempted.

Some of the Phase Two studies addressed, for the first time, the rate of production of phytoplankton (microalgae) and macrophytes, although the latter were examined only offshore from Werribee. Phytoplankton growth was measured for a year at three stations extending to 30 km off Werribee and several important features were disclosed.

Chlorophyll *a* values for most of the year were lower than Bass Strait, confirming the low levels measured during Phase One. However, a spring bloom was recorded which raised chlorophyll *a* levels as high as 9 mg/m^3. While productivity rates in the bloom were higher nearer to Werribee, reflecting greater availability of nitrogen, the bloom rapidly attenuated outwards and had diminished to background levels by January.

Only about half of the nitrogen (N) and phosphorus (P) taken up from the water column could be accounted for in the phytoplankton population, so that senescence and grazing were obviously rapid.

As well, the proportion of the annual N and P inputs to the Bay, and the Bay N and P pool, which could be accounted for by microalgae was rather small. The proportion of the Bay pool of phosphorus present in the background level of phytoplankton was only 1%. The proportion of the Bay nitrogen pool present in phytoplankton was slightly higher because nitrate and ammonia levels in the water were so low. This enigma of low phytoplankton and low water column nitrogen in the face of an annual input of 7,000 to 9,000 t remained unsolved until the present Study.

A second productivity study conducted two years later confirmed the general levels found in 1975-76, but also identified the important role of benthic microalgae. Where these were present, their content of chlorophyll per unit area was several times that present in the phytoplankton. Similarly, their contribution to productivity was at least twice that of the phytoplankton, sometimes more. Microphytobenthos were thus found to be an important component of the Bay ecosystem.

A comprehensive survey of macrophytes along the Werribee coastline provided the first estimates of growth and nutrient uptake in this plant compartment. The growth patterns observed suggested that while macrophytes represented only a small proportion of the nutrient pool in Bay waters and a small percentage of annual nitrogen turnover, they did create an important subsidiary pathway for the transfer of nutrients into the Bay ecosystem.

Other minor studies of sediment properties offshore from Werribee, the distribution of fish species and numbers, the dispersion of WTP effluent and the distribution of benthic invertebrates were also made. The results of these added to knowledge of the ecosystem offshore from Werribee but did not help with any understanding of the Bay-wide ecosystem.

Perhaps their most important finding was the absence of any significant deleterious effects of WTP effluent apart from an apparent tendency for adult fish to avoid the plume (though not younger fish). Enrichment studies with WTP effluent on samples of Bay phytoplankton showed growth stimulation rather than any toxic effects.

What is now realised to be the single most important project in Phase Two was a study of bacterial nitrification and denitrification in the sediments off Little River. While relatively high rates were found, these decreased rapidly with distance offshore and were only investigated up to 200 m from the WTP outfall. Hence the Bay-wide significance of the findings could not be perceived.

Nevertheless, these results were the first direct demonstration of nitrification/ denitrification processes in Bay sediments and the first realisation of the high levels of nitrate and ammonia in the interstitial waters of the sediments produced by mineralisation of organic material.

The present Study has extended this work and shown that denitrification is a key process in the nitrogen system of the Bay.

During the 1970s, 1980s and 1990s, physical-chemical monitoring was carried out in the Bay over four periods. The results served to confirm the lack of any overall trend in properties with time and their heterogeneous distribution in space and the short term.

One interesting and important project carried out prior to the PPBES was the measurement of dissolved oxygen in bottom waters in the deep central basin of the Bay for three years. Simultaneous measurement of temperature and salinity enabled periods of thermal or haline stratification to be detected. Readings were logged every two hours giving a fine time scale of change.

Over the whole period only two significant depressions of dissolved oxygen occurred, both during periods of stratification. While the levels reached were not dangerous, only 60%-70% oxygen saturation being reached over a few days, this was a valuable indication of the relatively high oxygen demand in the Bay sediments.

The present Study has shown the significance of this oxygen demand.

3. The state of knowledge in 1992

The first task carried out by the Port Phillip Bay Environmental Study was an extensive review of all available scientific literature relating to the Bay. The task was contracted to Victorian scientists familiar with the disciplines and topics concerned.

This broad effort was carried out very professionally and resulted in eight reviews covering the fields of:

- Physics
- Atmospheric inputs
- Groundwater inputs
- Macrobenthos
- Fisheries and toxicants
- Benthic flora
- Phytoplankton ecology
- Water column trophodynamics

From this material the CSIRO Technical Group supervising the Study was able to compile a Status Review summarising knowledge of the Bay. One field, that of nutrient chemistry, was not reviewed in the first round because of its complexity. Instead, a working group of two Victorian and two CSIRO scientists examined the information available and composed a detailed summary for the Status Review.

It is worth presenting the principal features of this Status Review since it shows the starting point of the Study, setting the scene for the chapters that follow. It will become apparent that the Study has not only added to our knowledge of the state of the Bay but also developed an understanding of the processes which have brought about that state.

3.1 Background

The first chapter of the Status Review gives general background information on the Bay, including a summary of historic inputs, political changes and previous research. It also sets out the objectives of the Port Phillip Bay Environmental Study which were:

General

i) To determine the overall water quality of the Bay and provide scientific information necessary for the development of long-term management options.

Nutrients

ii) To determine the Bay's nutrient status, the effects of existing nutrient loads on the Bay and the potential impact of future nutrient loads. Particular attention will be given to the discharge from the Western Treatment Plant (WTP) and major freshwater inputs such as the Yarra River, Werribee River and Mordialloc Creek.

Toxicants

iii) To determine existing levels of toxicant contamination of water, sediments and biota within the Bay. Particular attention will be given to the areas of influence of the WTP and the major freshwater inputs, to allow evaluation of the long-term implications of these inputs on the environmental status of the Bay.

iv) To determine the effects of dredging operations, particularly in relation to the effects on toxicants associated with sediments.

v) To determine fish health in the Bay.

Monitoring

vi) To determine the monitoring program necessary to ensure that key environmental indicators identified in the study are maintained within the objectives described by the selected management options for the Bay.

The succeeding chapters of this report demonstrate how these objectives have been addressed and achieved.

3.2 Physics

Chapter 2 of the Status Review deals with physical processes in the Bay. These are best dealt with under subject headings.

Meteorology

The wind regime on the Bay is complex with few clear trends. There is a tendency for northerlies to predominate in winter and southerlies in summer, but local winds are affected by topography and land and sea breezes and to some extent by the Melbourne Eddy. The highest proportion of calm weather and light winds occurs in winter and strong winds are associated with storms, mainly in summer. The most frequent wind speeds are, on average, less than 10 m/s and winds greater than 27 m/s occur only about once every two years.

Because of the large area and shallow depth of the Bay, direct rainfall is an important freshwater contribution. Rainfall increases from north-west to south-east across the Bay (mean annual rainfall is 535 mm at Laverton and 735 mm at Mornington) and this also influences riverine input. Hence most river flow to the Bay is from the Yarra and the eastern creeks.

Rainfall is apparently distributed evenly throughout the year on an average monthly basis and this disguises a seasonal difference.

Table 1.1: Inputs (km^3/yr).

All rivers	1.4
Rainfall	1.3
WTP effluent	0.2
	+2.9
Evaporation	-2.3

Groundwater input (0.06 km^3/yr) is insignificant.

Rainfall occurs more heavily on fewer days in summer as opposed to lighter rainfall on more days in winter. As a result, the heaviest river loads enter the Bay during summer storms.

Evaporation exceeds rainfall over the year and nearly equals total freshwater input as Table 1.1 shows.

Water movements

Three forces move water in the Bay – tide, wind and density. Of these, tidal movements are the most consistent and best understood. Both wind effects and density differences are subject to the vagaries of weather.

Tidal movements fall into an unequal semi-diurnal pattern. There is one large and one small tide each day. The amplitude and velocity of the tides are attenuated by the Great Sands stretching across the entrance to the Bay inside The Rip. A mean tidal amplitude of 0.4 m at Point Lonsdale falls to 0.2 m at Williamstown. Correspondingly, maximum ebb and flood velocities of 1 m/s or more at The Rip decline to 0.05 m/s in the centre of the Bay and only 0.02 m/s in the north of the Bay.

Tidal movement seems to be straightforwardly north through the Bay on the flood and south out of the Bay on the ebb, with variable direction at the turn of each tide.

Water movements due to wind are also of low magnitude, commonly 0.05 m/s, and variable in direction. A combination of numerical modelling and field measurements suggests that wind currents follow the usual pattern of such bays, water moving with the wind in the shallows and returning against the wind as a counter current in the deeper centre. With the most common winds, westerlies, the result becomes two gyres, a clockwise one in the north of the Bay and an anti-clockwise in the south. However, strong short-term winds distort this pattern, and the distribution of water properties indicates a complex small-scale circulation pattern.

Density currents arise in the Bay mainly due to freshwater inflow from the Yarra River. The magnitude of such currents depends therefore on the volume of river flow, but is commonly 0.5 m/s in Hobsons Bay during rain events, decreasing southwards as the freshwater plume mixes with saline Bay water.

Except during very rare strong winds from the easterly quadrants, the plume flows down the east coast of the Bay due to the Coriolis force. This flow may be added to by the east coast creeks.

It is probable that some density-driven water movement also takes place between the Bay and Bass Strait due to thermal differences, but the magnitude of this is not known.

Mixing and exchange

The Bay is well-mixed vertically most of the time because of its shallow depth. Occasional episodes of salinity stratification occur as described above when riverine inputs flow over saline Bay waters producing gradients. These may reach six salinity units in the north-east but rarely exceed one unit in the centre of the Bay.

Thermal stratification resulting from surface heating may occur in summer during calm periods but seldom exceeds a gradient of 3°C, usually less. This density gradient sometimes, however, inhibits mixing sufficiently to impede transfer of oxygen to bottom waters.

Thermal and salinity gradients commonly occur horizontally in the Bay, not only because of freshwater inputs at particular locations on the edge but also because of differential cooling and heating with depth.

The shallow peripheral waters are frequently cooler or warmer than the deeper central water because the smaller heat reservoir responds more readily to changes in atmospheric temperature and solar radiation. Gradients of 0.2°C/km and 0.15 PSU/km are common. Few measurements of horizontal mixing had been made prior to this Study but the heterogeneous distribution of properties and the persistence of gradients suggested only slow mixing.

Estimates of the horizontal mixing coefficient have varied from 0.1 to 5 m^2/s in the vicinity of the WTP effluent plume, depending largely on wind strength.

Previous estimates of the rate of exchange of Bay waters with Bass Strait (the flushing rate) depended on either calculated residence times of a conservative solute such as salinity, water or phosphate or upon physical measurements.

The latter, conducted in Phase One, gave artificially high flushing rates because they relied on water movements indicated by the behaviour of drogues released at The Rip.

The average tidal prism is 1 km^3 and from 50% to 90% of this was assumed exchanged based on the proportion of drogues lost to Bass Strait on the ebb. This would imply a flushing time of 28 to 50 tidal cycles.

Calculations from solute inputs and Bay levels give a flushing time range of 12-16 months or 700 to 1,000 tidal cycles. This poor real exchange ratio of 0.035 is due to restriction of mixing over the Great Sands.

Sediments and suspended matter

Numerous measurements of suspended matter by optical or gravimetric methods have been made in the Bay and several descriptions given of the distribution of bottom sediments.

Most suspended matter is inorganic and of the organic content only a small proportion is living material. Total suspended matter ranges from 2 to 30 mg/L whereas suspended organic carbon levels are mainly 0.5 mg/L or less. Only 20% or less of the latter is attributable to living cells.

The distribution of sediment types in the Bay reflects the wave energy pattern. Fine silts and clays and detritus tend to be winnowed out of the shallow margins, leaving coarser sand material behind. The fine material settles in the deep central basin where it forms a loose floc on the bed of the Bay. This fine material has a higher organic content – up to 4% carbon – and also tends to contain slightly higher levels of heavy metals than the coarser peripheral sediments.

The particulate budget of the Bay was not understood. The best measurements of sediment transport brought down by the Yarra, for example, account for only about half the material dredged out of the Port of Melbourne. Inputs from the coastline and Bass Strait have not been quantified.

The knowledge gaps and research needs in physics identified at the outset of PPBES, were:

- More detailed meteorological data.
- Observations of currents, temperature and salinity.
- Estimation of mixing and flushing rates.
- Better measures of catchment input.
- Better measurements on suspended sediment and bed sediment.
- Three-dimensional hydrodynamic, dispersion and transport models with ancillary wave models.

How the knowledge gaps have been addressed

Meteorological data pertinent to the physical oceanography of the Bay was obtained in several ways. A rain gauge, thermometer and relative humidity sensor were added to the Hovell Pile in the south-eastern corner of the Bay to supplement the wind measurements already made there. This program continued from March 1994 to May 1995.

Over a similar period, direct rainfall over the Bay was estimated at 20 minute intervals from Bureau of Meteorology RAPIC weather radar data. This work was carried out by WNI Science and Engineering [Task P1(A)].

Tidal current patterns north of and over the Great Sands were plotted by James Cook University using coastal radar [Task P1(B)].

In co-operation with the Victorian EPA, measurements of wind velocity and direction were extended to 21 stations around the Bay. The EPA also generated diagnostic wind models for the autumn and spring periods of oceanographic measurements in 1994. Two recorders of photosynthetically active radiation (PAR) were installed (at Brighton and Geelong South) and two relative humidity sensors were installed at Point Wilson and Frankston [Task P1(C)].

In April-May and September-October, 1994, Bay-wide measurements of salinity and temperature were made by Centre for Water Research WA at 90 stations twice in each period. At the same time, five paired current meters were moored at sites in the north, centre and east of the Bay by WNI Science and Engineering [Task P2].

In May, traverses were made by Lawson and Treloar Pty Ltd with an Acoustic Doppler Current Profiler for five days, both across the southern edge of the Great Sands and along the three main channels.

Simultaneously, the Centre for Water Research measured salinity-temperature profiles along the same tracks [Task P2].

Mixing rates of the two main freshwater inputs to the Bay were assessed by intensive salinity-temperature measurements in the effluent plume off Werribee and the Yarra River plume [Task P2].

A review of catchment inputs by Natural Systems Research showed the WTP and the Yarra River to be the principal sources of nutrient and toxicant input to the Bay. The WTP output was thoroughly documented, but Yarra River data proved scanty. Accordingly, the Victorian Institute of Marine Sciences (VIMS) was contracted to conduct measurements of flow and loads past the Westgate Bridge, near the Yarra River mouth [Task N1.3/T1.2].

VIMS also conducted all physical sediment research in the Bay with a program which included 240 core samples and eight deployments of its 'SEDCAM' measuring frame. This latter array provides a video record of the bottom, a profile of the bottom, two and three-dimensional current movements, turbidity levels, incident light and water samples [Task P7].

A three-dimensional hydrodynamic model was commissioned from Lawson and Treloar Pty Ltd to provide the base for dispersion and transport models constructed by CSIRO [Task P8].

3.3 Nutrients

Chapter 3 of the Status Review discussed what was known of the nutrient regime of the Bay, examining inputs, distribution, cycling and the overall budget.

Nutrient inputs

All measurements of nutrient (N and P) inputs to the Bay during the Phase One Study and since show two features. One is the dominance of input from the WTP and Yarra River; the other is the temporal variability with season.

The first point is illustrated by Table 1.2.

In the 'All Others' category in 1982, the Yarra River complex contributed 1141 t of nitrogen and 116 t of phosphorus. Hence sewage and the Yarra contributed 78% of total nitrogen to the Bay and 73% of total phosphorus.

During any one year, output of N and P from the WTP varies with season, ranging from about 4 to 14 t N/d and 1-3 t P/d, with maximum loads in late winter around August when hydraulic loads are highest and removal of N is least effective due to the lower ambient temperatures.

The Yarra input of nitrogen shows a similar seasonal pattern, following the annual cycle of flow in the river with maximum flows occurring generally in late winter and spring from August to October. The load cycle is a product of both high spring flows and high spring concentrations of both inorganic and organic nitrogen compounds. On the other hand, there is no consistent seasonal cycle in phosphorus inputs. A combination of higher summer concentrations at low flows with lower winter concentrations at high flows produces an irregular load pattern with no trend.

This indicates different origins for these two parameters. Nitrogen accumulates in the catchment and is washed off in storms. Phosphorus follows a more constant or base flow pattern of input.

The year-to-year variation in discharge of water and nutrients from the Yarra is very marked.

Long-term gauging of the Yarra River at Warrandyte shows a spread of flows from less than 0.07 km^3/yr in droughts to 2.2 km^3/yr in floods, a 30 times range.

During the Phase One Study spring flows in the Yarra downstream at Queens Bridge ranged from 2,700 ML/d in 1967 to 8,000 ML/d in 1970.

The calculated annual inorganic nitrogen load past Queens Bridge ranged from 201 t in 1967 to 1,026 t in 1970. Inputs of phosphate also show wild swings. For example, estimates of total phosphorus input from the Yarra River (at the mouth) range from about 55 t/yr in 1979, to 153 in 1986, 15 in 1991 and 182 in 1992.

Despite these natural fluctuations in input and increases in catchment development the average annual total input of

Table 1.2: Inputs of nitrogen and phosphorus to the Bay (t/yr).

	Total Nitrogen		Total Phosphorus	
	Sewage	**All others**	**Sewage**	**All others**
1969 – 71	4,300	3,191	976	460
1982	4,710	2,800	970	510

Table 1.3: Total nitrogen input to the Bay (t/yr).
Changes in sources over 20 years.

Source	1969-70	1989-90
WTP	3,931	3,781
Other MW plants	156	119
Other authorities	211	700
Sullage	794	113
Urban runoff	562	700
Agricultural runoff	1,174	1,213
Forest/Woodland	33	33
Stormwater drains	635	0
Atmosphere	900	1,050
Groundwater	55	55
TOTAL:	8,451	7,764

nitrogen to Port Phillip Bay has remained remarkably constant at around 8,000 t. This seems to be due to a redistribution of sources. A comparison of load estimates between Phase One and 1989/90 is given in Table 1.3.

The apparent discrepancy of 960 t in 1969-1970 between this and the previous table arises from the inclusion of atmospheric and groundwater inputs in the second table.

The connection of properties and industry to sewer over the 20 years has diverted nitrogen inputs from sullage to treatment plant effluent. There was actually a rise in nitrogen output from WTP to nearly 7,000 t/yr by 1975, but this was reduced by half when the Eastern Treatment Plant at Carrum was commissioned in that year. Stormwater input in 1989-1990 is included in 'Urban runoff'.

Catchment inputs of nutrients other than from WTP and the Yarra River are a minor contribution to the total, about 22% of N and 27% of P.

While the Werribee River provides the next largest flow of fresh water into the Bay after the Yarra, its content of N and P is generally low. Conversely, while the eastern waterways, principally the Patterson River and Mordialloc Creek, often contain high concentrations of nutrients, their catchments are smaller and flows are small and erratic.

The information available places groundwater export of nutrients to the Bay at negligible levels. However, atmospheric input is significant. While direct measurements of wet deposition are scanty, extrapolation of these to the Bay leads to an estimate of 600 t/yr of nitrogen (60% ammonia and 40% nitrate). Theoretical calculations of dry deposition rates increase this to a total of 800 to 1,300 t/yr of nitrogen as ammonia and nitrate.

The pioneering work on nitrification and denitrification carried out during Phase One showed the high levels of nitrogen and phosphorus present in the interstitial waters of the Bay sediments off Werribee. This suggested that flux of N and P from the sediments to the water column might be significant. Subsequent measurements on cores and overlying water taken to the laboratory proved this to be the case. Extrapolating the laboratory results to the whole Bay gave the following flux rates (Table 1.4).

It must be stressed that this flux derives from mineralisation of organic matter generated in the Bay and reaching the bottom. Hence it is not 'new' nutrient input. However, its magnitude indicated the relative importance of sediment processes in the overall nutrient cycle of the Bay. For this reason, equal weight was given to the study of these in the final Project Design, and this proved a profoundly wise decision.

Table 1.4: Bay-wide nutrient flux rates (t/yr).

PO_4-P	NO_3-N	NH_4-N	TIN
-92	3,600	8,200	11,800

The apparent net loss of phosphorus conceals the fact that most sequestering of phosphorus actually occurs in the nearshore sediments – the central sediments do produce a net efflux of P.

Nutrient Distribution

The Phase One Study showed that despite the heterogeneous distribution of Bay water properties in space and time, some zonation could be perceived.

Nutrient levels are high along the Werribee shoreline and often in Hobsons Bay and along the eastern shore. This reflects the influence of catchment inputs. The centre of the Bay shows low and relatively uniform levels while gradients in properties again occur in the southern region across the Great Sands and The Rip. The latter reflect mixing and exchange with Bass Strait oceanic waters.

The compounds of nitrogen and phosphorus in the Bay behave quite differently. Almost all phosphorus is present in the water as inorganic orthophosphate with very little organically bound, as Table 1.5 (adapted from the Phase One report) shows.

In 1969 about 77% of all phosphorus occurred in orthophosphate form. In 1970 the proportion was 85%. The levels of orthophosphate found have not changed in subsequent surveys to the present, probably because they approximate to the maximum solubility of orthophosphate ion in seawater.

Nitrogen to phosphorus ratios in the Bay are very low if organic nitrogen is excluded. On a long-term average basis, the N:P atomic ratio is only 0.25:1 compared to the Redfield ratio for optimum nutrient availability for algal growth of 16:1. Hence there is a *prima facie* case for inorganic nitrogen being the limiting plant nutrient in the Bay, and this was confirmed by enrichment experiments carried out in the 1970s.

Phosphate is present in excess and behaves as a quasi-conservative element. In fact, the estimated annual input of total phosphorus (1,300 t) compared to the Bay pool (1,900 t) gives a residence time of about 16 months, similar to that calculated from the salt balance of the Bay.

The nitrogen system in the Bay waters presents a different picture. There are large inputs of ammonia from the WTP and nitrate from rivers and drains, and also large inputs of organically combined nitrogen. In the Bay waters, however, levels of inorganic nitrogen are extremely low and organic nitrogen dominates. This may be illustrated by comparing the pool sizes calculated from surveys made over the whole Bay from 1980 to 1984 (Table 1.6).

In other words only about 5% of the total dissolved nitrogen in the Bay is inorganic. The organic nitrogen is an unknown complex mixture of compounds. Since the Bay exchanges continually with Bass Strait it is possible that some of the organic nitrogen may be imported from there, while some could be generated from algal breakdown in the Bay in addition to organic nitrogen derived from the catchment. The role, if any, of organic nitrogen in the Bay ecosystem is unknown.

On the other hand, all available data on phytoplankton productivity from surveys in 1975-1978 indicated rapid uptake of both

Table 1.5: Phosphorus forms in the Bay (µM/L).

	1969	1970
Total P	2.6	2.7
Total soluble P	2.3	2.5
Orthophosphate	2.0	2.3

Table 1.6: Nitrogen pool in the Bay (t).

Inorganic	231
Organic	3,960

forms of inorganic nitrogen. The annual turnover of nitrogen through the phytoplankton community was estimated at around 10,000 t. Given a phytoplankton nitrogen pool of 200 t, this meant the pool must turn over 50 times a year. Present Study results suggest that growth may occasionally be even more rapid. Given also that the annual input of inorganic nitrogen to the Bay is about 3,000 t or more, it is clear that inorganic nitrogen is extensively recycled (see Chapter 6).

Furthermore, with a Bay pool of only 200-300 t and an exchange time with Bass Strait of just over a year, the maximum export of DIN could only be 200-300 t/yr. Against this, there is an input from the catchment of more than 3,000 t of DIN a year. The missing nitrogen can be explained by three possible mechanisms.

The first of these is uneven export. Major inputs of inorganic nitrogen occur during rainy periods when flows into the Bay increase. The major freshwater flow is from the Yarra River through Hobsons Bay and along the east coast where it subsumes other river and drain flows.

The resultant freshwater plume has occasionally been detected passing along South Channel and even out through The Rip. This plume transports high concentrations of most catchment inputs including inorganic nitrogen so that some of the annual input could ‘by-pass’ the main Bay. Since this mechanism probably operates only once a year, in the spring, and not in every year, it may not be significant.

The second mechanism is accumulation in the sediments. The total nitrogen content of the Bay sediments amounts to about 1,250 t for each millimetre depth. The annual sedimentation rate, so far as we can measure it, is probably less than 1 mm/yr overall. Hence only about 1,000 t at most of annual input becomes sequestered in the sediment. This still leaves a large part of annual input unaccounted for. It is now known that most of this missing nitrogen results from a third mechanism – denitrification in the sediments, through the processes described in Chapters 3 and 6.

While mineralisation and denitrification are widespread phenomena in water bodies, the extent and efficiency of the processes in Port Phillip Bay is unusual.

The knowledge gaps and research needs identified in the Status Review were:

- A better knowledge of catchment inputs.
- A better understanding of the exchange processes between the Bay and Bass Strait.
- A fuller understanding of sediment nutrient fluxes and processes.
- Measurement of phytoplankton grazing rates.
- Development of models describing nutrient dynamics in the Bay, based on a three-dimensional hydrodynamic model.
- Assessment of the significance of macrophytes, seagrasses and microphytobenthos (MPB) in the nutrient system.
- More comprehensive measurement of phytoplankton productivity accompanied by simultaneous measurement of ambient nutrient levels in the water column.
- Examination of sedimentation processes.

How the knowledge gaps have been addressed

The matter of inputs proved difficult in that the resources required in time and money to thoroughly monitor all influent streams for all nutrients for all conditions were beyond those available to the Study.

Instead, attention was focused on the Yarra River as the major contributor of nutrients and the least monitored in terms of net input to the Bay. This work was conducted by the Victorian Institute of Marine Science.

As well as a physical program designed to generate a numerical model linking gauged flow and tide heights to loads, the concentrations of nutrients in a cross-section across the river below Westgate Bridge were determined [Task N1.3/T1.2].

Sediment nutrient processes have been examined in a program which measured the distribution of several properties with depth. Cores were taken and sub-samples analysed. The resultant depth profiles gave a measure of efflux (or influx) from passive diffusion.

In addition, direct measurements of flux were made by following the changes of relevant properties in water enclosed within chambers set onto the sediment. These fluxes were always greater than those calculated by passive diffusion, but inorganic N fluxes were usually less than those calculated from oxygen consumption.

The inference was that denitrification was occurring and this was later demonstrated by direct measurement of nitrogen:argon ratios in gases drawn off the enclosed water. The occurrence of higher rates of flux than could be accounted for by passive diffusion now seems almost certainly due to bioirrigation of the sediments.

This program was carried out by the Victorian Fisheries Research Institute (VFRI) in co-operation with CSIRO, the Australian Geological Survey Organisation and the Universities of Tasmania and Southern California [Tasks N3.1 and N3.2].

Phytoplankton grazing rates were measured in a research task undertaken by Monash University [Task E10.5].

The integrated model of the Bay has been developed by CSIRO and more will be said on this in the final section of this introductory chapter and Chapter 6 [Task G8/9].

Assessment of the biomass and productivity of MPB was carried out by Monash University [Task N2.4], while Consulting Environment Engineers undertook surveys of macrophyte biomass [Task N2.3].

Monash University also carried out the phytoplankton investigations. A large and complex program of measurement of biomass and productivity in three size classes was conducted at several sites over two years [Task N2.2] in conjunction with monthly surveys of nutrient levels at the same sites [Task N2.1].

Sedimentation processes proved difficult to examine. The University of Melbourne carried out analysis of isotopes in several cores and CSIRO examined vertical distribution of biomarkers. However, interpretation of the results was confounded by the difficulty of obtaining undisturbed cores and the burrowing and mixing effects of invertebrate fauna.

3.4 Toxicants

Chapter 4 of the Status Review described the toxicant status of the Bay as it was known in 1992. The chapter dealt with inputs, levels in Bay waters, sediments and biota and toxicant dynamics or processes.

Toxicant inputs

It is inevitable that toxicants, as anthropogenic substances, will enter the Bay from the catchment via rivers, creeks, drains, effluent streams and atmospheric deposition. There are also smaller inputs from boating, shipping and dredging.

Examination of the available data showed that although most catchment waterways had been monitored at some time in the two decades prior to the PPBES, the toxicant situation was not well described.

The monitoring was often infrequent or confined to one or few periods. Similarly, the parameters measured often did not include toxicants. When these were measured, analyses were generally confined to a few metals. Water flow was often not recorded simultaneously, so that toxicant loads could not be calculated, and in any case, monitoring was never designed to catch storm events when the heaviest material loads enter the rivers.

However, an examination of the best available monitoring data was able to generate approximate loads of the common metal toxicants from the principal rivers and drains entering the Bay.

These are shown in Table 1.7, together with estimates of annual loads from the WTP based on analyses of effluent from its four outlets.

Table 1.7: Metal inputs to the Bay. Mean annual totals (t)

Metal	Yarra	All others	WTP
Chromium	7.8	4.8	3.9
Copper	6.6	8.0	2.7
Nickel	6.3	2.9	4.9
Lead	19.0	11.0	1.1
Zinc	63.0	58.0	5.8
Cadmium	0.17	0.14	-
Mercury	0.04	0.04	-

(Cadmium and mercury are at or below detection limits in WTP effluent.)

It can be seen that Yarra River inputs of metals are approximately the same as the totals from all other rivers, creeks and drains. The two together constitute the major input of metals to the Bay. Up to 90% of metals in sewage arriving at WTP remain there.

Former point sources have largely been diverted to sewer. Corio Bay suffered most from such discharges in the past, receiving cadmium, lead, mercury and hydrocarbons.

No direct measurements of toxicant discharge from shipping had ever been made in the Bay, and estimates of toxicant input from groundwater were non-existent even though some aquifers were known to be contaminated. Nor were any estimates of direct atmospheric deposition available.

Information on organic toxicant inputs was non-existent apart from a few measurements of organochlorines and dioxins in WTP effluent. The levels found were negligible.

Toxicant distribution

In contrast to the situation with inputs there was considerable information available on toxicant levels in Bay waters, sediments and biota. Of course, comparison of results over time is not always valid because of changes in sampling techniques, sample treatment and analytical methods.

The levels of metals found in the waters of three zones of the Bay from 1975 to 1989, from a variety of publications, are shown in Table 1.8 on the next page. Also shown are global ocean levels and the ANZECC water quality guideline levels for the protection of aquatic life in marine waters.

It can be seen that zinc and lead levels declined slightly over the period while other metal concentrations remained fairly constant. There was no clear trend of concentration with location. All metals except copper were well below ANZECC guidelines. Only lead levels are higher than in the open ocean.

Only total and petroleum hydrocarbons had been measured in Bay waters. While some high values for total hydrocarbons (up to 226 µg/L) were reported in 1977, by 1986 all analyses showed less than 1 µg/L. Values for polycyclic aromatic hydrocarbons (PAHs) were very low, in the order of less than a nanogram per litre.

Toxicants are more easily measured in sediments because the levels per unit volume of sample are higher.

This is because many metals, in their ionic form, are absorbed onto particles of suspended silt or clay which eventually find their way downwards to the bed of a waterway. Also, most metals, while soluble in fresh water in ionic form or as inorganic and organic complexes, tend to precipitate out when mixed into saline water (see Figure 1.3).

Hence the sediments form a useful indicator of inputs, and the only high levels of toxicants found in the Bay sediments up to 1992 (and since) are at locations adjacent to inputs. Such locations are western Corio Bay, Hobsons Bay and the mouths of Kororoit Creek, Mordialloc Creek and Patterson River.

Of course, the concentration of metal found depends on the severity of the extraction method used. Since the objective of most surveys is to determine the labile fraction of metal (i.e. the part likely to be assimilated by aquatic biota), dilute (0.5M) acid leaching is a common technique. If more severe methods are used (e.g. digestion with aqua regia), then

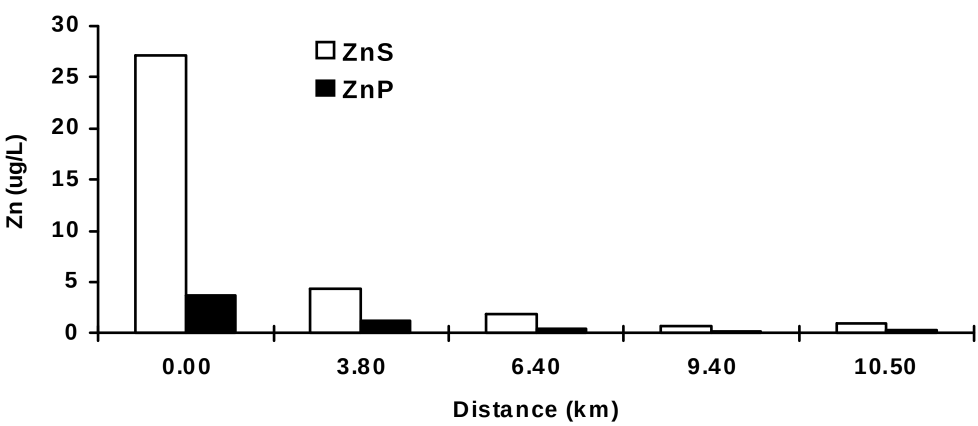

Figure 1.3: Concentration changes of dissolved and particulate zinc species in the Yarra River and Hobsons Bay with distance downstream from the Westgate Bridge.

Table 1.8: Total trace metal concentrations in Port Phillip Bay waters (µg/L).

Site	Year	Zinc	Nickel	Lead	Copper	Arsenic	Cadmium	Mercury
Hobsons	1980-82	9	1.1	3.6	–	2.3	<0.1	0.06
Bay	1983	2	1.2	0.7	0.8	1.1	0.1	0.006
	1984	<2	1.4	0.7	1.0	1.1	0.1	0.005
	1985	3	1.6	1.1	1.5	1.6	<0.05	0.003
	1986	<2	1.4	<0.8	0.8	–	<0.05	–
	1987	15	3.4	3.2	2.0	2.0	0.3	0.007
	1988	4	1.0	1.4	1.0	3.1	0.1	0.003
	1989	3	1.1	<0.8	1.1	3.0	0.1	<0.002
Corio Bay	1975	–	–	1.5	3.2	–	0.3	–
	1977	30	–	0.8	5.6	–	0.4	–
	1980-82	<2	1.5	1.6	–	2.6	0.3	0.05
	1983	4	0.8	1.4	0.7	0.9	0.2	0.003
	1984	4	0.6	1.4	0.5	0.8	0.2	0.004
	1985	<2	1.8	<0.8	1.3	2.2	0.3	0.004
	1986	3	1.8	<0.8	1.6	–	0.4	–
	1988	<2	1.8	<0.8	1.1	3.3	0.3	0.004
	1989	<2	1.5	<0.8	1.1	3.2	0.2	<0.002
Central	1980-82	<2	0.9	1.1	–	2.6	<0.1	0.05
	1983	<2	1.0	<0.8	0.9	1.2	0.1	0.003
	1984	4	1.2	<0.8	1.2	1.0	0.1	0.004
	1985	<2	1.2	<0.8	0.9	2.3	<0.05	<0.002
	1986	<2	1.1	<0.8	0.7	–	0.1	0.004
	1987	6	1.4	1.2	1.3	2.2	0.1	0.003
	1988	<2	0.8	<0.8	5.5	3.0	0.1	<0.002
	1989	<2	0.9	<0.8	0.6	2.8	<0.05	–
Ocean		10	2	0.03	3	3	0.1	0.03
ANZECC		50	15	5	5	50	2	0.1

higher metal recoveries are obtained, but the significance of these is uncertain.

Data prior to 1992 suggested that most of the sediments of Port Phillip contain metal concentrations well below hazardous levels, and subsequent work in the Study has generally confirmed this with some exceptions.

Table 1.9, abbreviated from the Status Review, compares levels of metals in sediments from Port Phillip Bay with those found in some NSW bays and with USEPA guidelines. Some of the NSW examples are from contaminated sites like Corio Bay. However, metal levels in Port Phillip Bay were similar to those in Botany Bay and lower than the USEPA guidelines.

Very few cores had been analysed to give depth profiles. Some taken in Corio Bay showed surface enrichment with copper, lead, zinc and cadmium due to anthropogenic inputs. Other metals showed complex behaviour as a result of diagenic processes and sediment mixing.

One survey of tributyl tin (TBT) levels was undertaken which found elevated concentrations near marinas but low or undetectable levels elsewhere.

Several surveys of organic toxicants were carried out on Bay sediments prior to 1992, all on surface samples. The pattern was similar to that for metals. High levels of hydrocarbons were found in Corio Bay and Hobsons Bay reflecting industrial input and urban runoff. These results included petroleum hydrocarbons as a class, polycyclic aromatic hydrocarbons (PAHs) and polychlorinated biphenyls (PCBs).

Some traces of chlorinated hydrocarbons (dieldrin and PCBs) were found in sediments off the WTP. An extensive Bay-wide sampling for dioxins and furans found only traces of these compounds and then mainly the non-toxic congeners. The origin of these latter compounds was considered to be urban runoff and the petroleum industry with some contribution from WTP.

Several taxa of marine fauna had been analysed for toxicants, principally the commercially harvested species because of concerns for human health. However, concern for the

Table 1.9: Metal concentrations in sediments from various locations (µg/g).

Site	Zinc	Lead	Copper	Cadmium
Corio Bay	4-400	2-210	2-50	0.1-13
Port Phillip Bay	21-40	8-22	1-8	0.8-2
Lake Munmorah	150	40	70	
Tuggerah Lake	110	40	30	
Lake Macquarie N	2,400	1,200	170	140
Lake Macquarie S	150	68	20	4
Port Jackson	1,150	520	180	3
Port Kembla	380	113	113	2
Botany Bay	25	10	3	0.5
USEPA Guidelines	90	40	25	6

health of fish in the Bay arose on two occasions in 1984 when kills of fish occurred.

Flathead and flounder showed skin lesions and spiky globefish showed interstitial haemorrhage. The symptoms were consistent with toxicant damage, and analyses for metals in fish tissues (muscle, liver and bone) did show elevated concentrations of lead and cadmium, but not of other metals.

However, any connection of the moderate levels of lead and cadmium with death of the fish was considered tenuous.

Some flathead have also been found with haemosiderosis which could have been caused by PAHs, but once again this was not confirmed. An extensive survey conducted in 1976-1977 of more than 15,000 specimens of several fish species found only two true neoplasms. The only examples of teratogenic defects reported have occurred in spiky globefish.

The earliest analyses for toxicants in edible seafood were for mercury in fish and cadmium in mussels. Levels of mercury above the NH&MRC limit of 0.5 µg/g were found in flathead and snapper but not in whiting. Later analyses in the 1980s showed levels declining to well below NH&MRC limits.

Organic toxicants in fish were not investigated until the late 1980s. Flathead and mullet tissues were analysed for phthalate esters, chlorinated hydrocarbons, PCBs, phenols, organochlorine pesticides and PAHs, and traces of some of these were found.

A comparison study examined sand flathead tissues for petroleum hydrocarbons and chlorinated hydrocarbons.

Dieldrin, DDT, DDD and DDE, lindane and PCBs were found in muscle and liver of fish throughout the Bay, although levels of concern to human health were only found in Geelong Arm and Corio Bay.

PAH concentrations in either muscle or liver were very low or undetectable. Fish from the Geelong Arm contained petroleum hydrocarbons at levels above that resulting in taste tainting.

Mussels are commonly used as indicators of pollution because of their abundance and well established bioaccumulation of toxicants. Not surprisingly, therefore, a considerable body of data on levels of toxicants in mussels from Port Phillip Bay had been collected prior to 1992.

In 1974, severe cadmium contamination of mussels from Corio Bay was reported. Levels as high as 41.5 µg/g had been found. However, cadmium concentrations in mussels from elsewhere in the Bay were well below NH&MRC guidelines. The Corio Bay contamination was traced to an industrial effluent which later ceased. After that time, cadmium levels in Corio Bay mussels decreased to below the NH&MRC limits and levels in mussels from the rest of the Bay also decreased.

Levels of metals such as copper and zinc were never above NH&MRC limits. Lead levels were significantly high in mussels in Corio Bay in 1977-78 and remained above the maximum permitted concentration of 2.5 µg/g up to the last measurements made prior to 1992.

No significant levels of metals were found in scallops, oysters or abalone.

The situation with organic contaminants in mussels was less favourable. In the late 1970s, levels of petroleum hydrocarbons and derivatives were found to be extremely high in mussels from Corio Bay and Hobsons Bay – with means of 141 µg/g and 128 µg/g respectively.

The picture had not improved by the late 1980s with Corio Bay mussels still showing petroleum hydrocarbon levels of 114 µg/g. Levels elsewhere in Port Phillip Bay were much lower (12-19 µg/g) but still significant, although there are no guidelines for petroleum hydrocarbons in seafoods other than a tainting level of 10 µg/g.

It would seem that biota in Port Phillip

Bay were generally contaminated with petroleum hydrocarbons albeit at low to moderate levels over most of the Bay.

Other organic toxicants had also been found in mussels but at trace concentrations well below maximum permitted concentrations. Those tested for were PCBs, chlorinated pesticides, organophosphate pesticides and TBTs.

A summary of the state of knowledge of contamination in Port Phillip Bay seafoods prior to 1992 is given in Table 1.10. The apparently high levels of arsenic in shellfish are of the non toxic form.

The knowledge gaps and research needs in toxicants identified by the Status Review, were:

- Previous studies have been fragmented. A synoptic survey of water, sediments and biota is needed with a full suite of toxicants measured.
- A better knowledge of inputs from streams, drains, groundwater and the atmosphere is required.
- A more detailed study of toxicants in Bay sediments is required including extractable species, total metals and depth profiles.
- Sediments should be screened for toxicity by bioassay.
- The impact of shipping must be investigated, including navigation dredging.
- The results from toxicant tasks must be integrated with those from physical, nutrient and ecology programs.

Table 1.10: Toxicants in fish muscle and shellfish compared against health standards (in µg/g wet weight) prior to 1992.

Toxicant	Fish		Shellfish	
	NH&MRC	Bay	NH&MRC	Bay
Cadmium	0.2	N/A	2.0	0.3
Copper	10.0	N/A	70.0	1.1
Lead	1.5	N/A	2.5	0.3
Mercury	0.5	0.2	0.5	0.02
Zinc	150.0	N/A	150.0	32.0
Arsenic	1.0	N/A	1.0	1.5
Petroleum Hydrocarbons	—	Up to 46.0	—	Up to 500
PCBs	0.5	0.02	0.5	0.166
Total Pesticides	—	0.02	—	—
DDT+DDD+DDE	1.0	0.03	1.0	0.006
Dieldrin	0.1	0.01	0.1	0.005
PAHs	—	0.0002	—	0.003

NH&MRC (1987, 1988) *N/A Not available* *— No Standard*

How the knowledge gaps have been addressed

A completely synoptic survey for water, sediments and biota did not prove possible for practical reasons. However, water samples for Task T2 were collected in June 1993, and sediment samples for Task T7 in July 1993. The Yarra event program under Task T2 was scheduled for late 1993, but because of the drought a suitable flow in the Yarra River was not encountered until April, 1995.

The suite of toxicants measured in Task T2 (water) was: cadmium, chromium (Cr VI), copper, iron, nickel, lead, zinc, arsenic, mercury and selenium, and the organics in Table 1.11.

Table 1.11:

Hydrocarbons :	Normal petroleum HC Biogenic hydrocarbons Oils
Polycyclic aromatic hydrocarbons (PAHs)	
Organochlorine insecticides:	HCB BHC Lindane Heptachlor Aldrin Heptachlor epoxide Dieldrin ppDDE, ppDDD, ppDDT
Polychlorinated biphenyls (PCBs):	Arochlor 1232 Arochlor 1254 Arochlor 1260
Triazine herbicides:	Atrazine Hexazinone Metriburzin Prometrin
Organophosphorus compounds:	Diazinon Chlorpyriphos methyl Demeton – s – methyl Fenthion Malathion Parathion Dibutyl methyl phosphate Tributyl phosphate Fyrol PCF
Chlorinated phenols Sulphide Cyanide	

The suite of toxicants measured in Task T7 were (Table 1.12):

Table 1.12:

Metals:	As for Task T2 plus manganese.
Both extractable and total metals were determined on 3 size classes (less than 63 micron, 63-200 micron and coarser than 200 micron grain size).	
Organics:	As for Task T2 minus sulphide and cyanide.

Several cores were taken to obtain depth profiles.

Measurement of toxicants in biota was not started until mid 1994.

Samples of flathead muscle and liver, mussels and macrophyte taxa were collected from six areas around the Bay.

Analysis of this material was conducted for the heavy metals cadmium, chromium, copper, mercury, nickel, zinc, lead and arsenic.

Measurements were also made of several organic toxicants – petroleum hydrocarbons, PAHs, PCBs and organochlorine pesticides [Task T8].

It was beyond the resources of the PPBES in time and money to monitor all the above toxicants in all or even most of the influent streams and drains, especially if event sampling was to be carried out.

Instead, attention was focused in the Yarra River. Under Task N1.3/T1.2, a section across the river just below the Westgate Bridge was monitored to capture the total load entering the Bay from the Yarra River.

Two surveys were carried out in 1993. One, in May, monitored a low flow situation while a second, in September, captured a high flow event. The parameters measured are listed in Table 1.13.

Table 1.13: Toxicants measured in the Yarra River, 1993.
Cadmium, chromium, copper, iron, lead, nickel, zinc, mercury, selenium and arsenic.
Chloroethers
Organochlorine pesticides
Chlorobenzenes
Polycyclic aromatic hydrocarbons
Organochlorine pesticides
Chlorinated aromatic hydrocarbons
Phthalate esters
Total petroleum hydrocarbons
PCBs

In conjunction with the physical measurements made under this Task, it was intended to calculate loads of toxicants entering the Bay, as for nutrients. When monitoring recommenced in 1995 after a delay necessitated by the drought, zinc was chosen as the most appropriate indicator of urban pollution.

Task T6-10 was devoted to the impact of shipping and dredging. Sediment cores were collected in the Ports of Geelong and Melbourne and at the spoil ground in northern Port Phillip Bay.

Water samples were also collected in the port areas and in the wake of bulk carrier ships. Analyses were conducted for the nine heavy metals cadmium, chromium, copper, iron, lead, manganese, nickel, zinc and mercury, and for total petroleum hydrocarbons, PAHs and TBT.

Because the levels of toxicants were in general so low in all compartments of the Bay, no bioassay tests were conducted. The behaviour of toxicants after entering the Bay depends largely on sediment processes and will be described in the Task P8(B) Transport Model and Task G8/9 Integrated Model.

It does not appear from evidence available that toxicants have affected the ecology of the Bay.

3.5 Ecology

Chapter 5 of the Status Review described the state of knowledge of the ecology of the Bay in 1992. This knowledge had been gathered together in five Technical Reports (Nos 1, 2, 6, 7 and 8) commissioned from various authors as part of the Study Literature Review.

Taking phytoplankton first, as the lowest trophic level, only 16 publications could be found that contributed directly to knowledge of the microscopic algae in the Bay whether in terms of taxonomy, abundance or productivity.

Collections of phytoplankton had been made from 1987 by the Victorian Fisheries Research Institute under the Shellfish Sanitation Program and these had furnished 282 species ranging through all the phyla of marine microalgae.

The dominant taxa found in these and earlier collections were diatoms, followed by cryptophytes and dinoflagellates. However it seemed that the smaller organisms (the nano and picoplankton) had not been regularly identified or enumerated.

There were no descriptions of seasonal or spatial variation in the phytoplankton, but extensive measurements had been made of chlorophyll *a*, over both 20 years and the whole Bay. At least two attempts had been made to interpret the chlorophyll *a* data record but with no clear results. Levels in the central Bay had remained about the same (1-2 mg/m^3) over the two decades. No seasonal pattern emerged but year-to-year fluctuations were common.

There were spatial differences in that chlorophyll *a* levels were generally higher in the vicinity of the Werribee coastline and in Hobsons Bay. This was consistent with the knowledge that photosynthetic activity had also been shown to be higher in those two regions, which are close to nutrient inputs. Chlorophyll *a* levels were claimed in one investigation to be rising off Werribee over the period 1970-1986 at a rate of about 6% per year. The other examination found a much smaller trend. However, these trends must be viewed against the wide scatter of values, with most grouped at low levels of 1-2 mg/m^3 and the means skewed by occasional excursions to higher values (up to 15 mg/m^3).

Chlorophyll *a* is a very crude measure of phytoplankton abundance. Overall levels give no indication of taxonomic distribution and relation to cell density varies with species and the physiological condition of cells. However, a comparison of chlorophyll levels in coastal waters around the world is quoted in Technical Report No. 8. Table 1.14, taken from that source, shows that Port Phillip Bay falls very near the bottom on any scale of eutrophication.

Table 1.14: Chlorophyll *a* in coastal waters (mg/m^3).

Site	Chlorophyll *a*
Stockholm Archipelago	Up to 60
Portsmouth Harbour	Up to 96
Swanpool	5-540
Northern Adriatic	Up to 13
Ebrie Lagoon	Up to 56
New York Bight	45-80
Chesapeake Bay	3-33
Potomac Estuary	25-125
James River	50-100
Dickinson Bayou	120-500
Portage Inlet	2-20
Kaneohe Bay	0.2-5
Tokyo Bay	Up to 60
Cockburn Sound	Up to 100
Port Phillip Bay	1-20

Notwithstanding the generally low level of phytoplankton in the Bay most of the time, some spectacular algal blooms have been reported occasionally. Some have been composed of noxious species.

For example, *Alexandrium catenella*, thought to have been introduced to the Bay, bloomed in 1986, 1988, 1991 and 1992. This alga produces a toxin and necessitated public health alerts. However, no cases of human illness or faunal deaths were reported.

Alexandrium tamarense bloomed for a period of three weeks off the Bellarine Peninsula in the winter of 1993. Although this species is also toxic, no fish kills or human health hazards occurred.

Rhizosolenia chunii, a diatom, bloomed in 1987-1988 and in 1993. This organism imparts a bitter taste to mussels.

Pseudonitzschia pseudodelicatissima, another diatom, bloomed in large numbers for five months in 1991-1992. Fortunately, this was a non-toxic strain.

There have also been occasional plagues of *Noctiluca scintillans*. This is a dinoflagellate which is not autotrophic but feeds on phytoplankton and the smaller zooplankton.

The reasons for these occasional outbreaks are not fully understood but are thought to result from a combination of physical factors (e.g. increased river flow, calm sunny weather) and biological interactions. Such outbreaks occur world-wide in many coastal waters and are not peculiar to Port Phillip Bay.

In contrast to the situation with phytoplankton, there was an abundance of literature on macroalgae and seagrasses. The authors of the literature review (Technical Report No. 6) found 58 references in the form of scientific articles, theses, reports and data listings.

These authors point out, however, that the information had been collected over a span of many years on separate occasions at disparate localities. There had been no systematic spatial and temporal survey of the whole Bay.

Nevertheless, knowledge of the benthic flora of the Bay was extensive, if not comprehensive. It was known, for example, that there are 64 species of chlorophytes recorded from the Bay, 109 species of Phaeophyta, 276 species of Rhodophyta, 12 species of seagrass and 29 species of non-planktonic microalgae.

The distribution of the various taxa was also known in a general way, associated with environmental factors such as bottom sediment type, wave action, light availability, nutrient sources, salinity, currents and temperature.

For example, some taxa are found only or mainly in the calmer waters of Corio Bay and on the western and northern coasts, others are never or seldom found in those waters or the central Bay while some taxa are ubiquitous. No attached macroalgae have been reported below 15 m depth.

Substrate plays an important role in the distribution of the benthic flora. While the densest stands of macroalgae occur on reefs, these occupy only a small area of the Bay floor so that the dominant species over much of the Bay is the green alga *Caulerpa*. This alga is able to flourish in soft unstable substrata. Its dominance lessens with increasing depth.

Some knowledge of seasonal patterns had also been gained.

In the case of macroalgae, seasonal changes in species composition were observed at all sites where observations extended long enough in time.

The most thorough studies were off Werribee where it was noted that species diversity decreased through winter and increased through spring and again decreased in summer as larger plants dominated the system. Quite dramatic changes in biomass of most species were recorded with season both on a reef habitat and on sand. The magnitude of material transfer, in terms of carbon and nitrogen, was of the same order as that occurring through the phytoplankton.

A significant feature in the Bay is the occurrence of large quantities of 'drift algae' of heterogeneous composition. Some of this plant material derives from broken fronds of attached algae such as *Caulerpa* or *Ulva*. However, much of the drift material consists of free-floating plants from all three major phyla. It is this drift algae which accumulates on beaches as a result of wind and wave action.

Of the 12 species of seagrass present in the Bay, *Heterozostera tasmanica* is the most dominant and grows from the intertidal zone to 9 m deep. However, 95% of seagrass in the Bay grows in depths less than 5 m.

Aerial photography spanning three decades showed minor changes in extent of seagrass beds, with accretion in a few areas and losses in others. Most seagrass is found along the southern shores of Corio Bay and the Geelong Arm and in Swan Bay, but there are scattered or sparse stands along the remaining shallow zones of the Bay.

Seagrasses in the Bay undergo a seasonal cycle of growth above ground. Maximum biomass usually occurs in summer followed by shedding of leaves through autumn. There can be a 40-fold increase in shoot biomass from winter to summer though growth is usually less than this. Some species, however, show a bimodal growth cycle, with minimum biomass in both mid-winter and mid-summer.

Non planktonic microalgae, or microphytobenthos, had been investigated in only one exercise although their presence had been noted by several observers. Most of this group are diatoms. Their productivity per unit area of Bay floor was greater than that of the phytoplankton in the water column, but there was no knowledge of their extent.

Mangroves are known to occur in the Bay at scattered locations, principally Hovells Creek, although a remnant stand persists in Kororoit Creek.

A review of the benthic invertebrate fauna of the Bay began with the advantage that some areas of the Bay had been examined for temporal change and spatial variability.

The foundation for this systematic knowledge base was the zoobenthos survey of the Phase One Study, 1969-1971. Collections were made of 0.1 m^2 sediment grab samples at 86 stations on a 5 km grid over 3¼ years. Only organisms larger than 1mm were studied.

A total of 713 species were distinguished though not all could be identified. Overall, about 15% were molluscs, 33% annelids, 35% crustaceans, 3% echinoderms and 14% other taxa.

The benthic fauna was very diverse compared to similar coastal areas elsewhere, but this diversity did not apply to the whole Bay. For example, 206 species occurred at only one station and 77 species at only two stations. Only 32 species occurred at more than half the stations. The most abundant species were generally found over muddy sediments while nearshore sandy sediments provided more numerous rarer species, especially along the north-west and south coasts.

On the other hand, the density of animals was not unusually high, and their distribution was very patchy, varying from 140 to 14,270 individuals/m^2. As with diversity, the highest densities were generally found nearshore.

An examination of the distribution patterns produced nine station-groups:

- The central muddy basin deeper than 15 m.
- Areas adjacent to this basin and the entrance to Geelong Arm.
- Corio Bay entrance, shallow water north of Point Wilson and Altona.
- Corio Bay muddy sediments.
- The Great Sands.
- The Rip region.
- Offshore from Werribee, Portarlington and similar shallow areas.
- Swan Bay, the Clifton Springs region and seagrass beds.
- Shallow water off Werribee and the east coast.

The only recognisable determining factor in this distribution pattern appeared to be the nature of the sediments. An examination of the distribution of one phylum, the molluscs, gave a slightly different zonation compared to the above list. It was related to sediment type but with some effects thought to be due to predation, food supply or salinity variation.

While the fauna offshore from the WTP was abundant and diverse, there was no detectable effect of WTP effluent on community structure other than immediately adjacent to the outfalls where there appeared to be some selection due to lower salinity.

There was a preponderance of filter feeders in the northern parts of the Bay, presumably reflecting the generally higher phytoplankton population there. Grazers and surface deposit feeders were more abundant off the Bellarine shore.

A later study extending over three years involved quarterly sampling of muddy sediments in the centre of the Bay, in Corio Bay and off Martha Point. In all 249 species were collected.

This survey disclosed that while there was some variation in numbers and taxa between stations, variability with time was more significant, especially year to year.

However, the two dominant species of these muddy environments found in 1969-71, *viz.* the bivalve *Theora* and a crustacean *Callianassa* (now *Neocallichirus*) were still the dominant taxa six years on.

A survey of Hobsons Bay and the Yarra River found a similar range of species as in the main Bay – 248 – but the number of species was significantly lower in the Yarra River than in Hobsons Bay.

Surveys in Corio Bay indicated that the densities of macrofauna may have declined over 20 years although species number has been maintained. However, two introduced molluscs and one polychaete have grown in large numbers.

The magnitude and diversity of the benthic fauna in the Bay led the author of Technical Report No. 2 to suggest that the benthic fauna might play a key role in nutrient turnover in the Bay.

A later review, published as Technical Report No. 10, examined the likely magnitude of trophic processes through the zoobenthos. This review used the existing knowledge of zoobenthos in the Bay in terms of speciation, biomass and feeding style with literature values for turnover rates to estimate material transfers. The result was confirmation of the significant role of the benthic fauna in secondary production in the Bay.

It was estimated, for example, that the deposit feeders in the Bay could process the top 13 mm of sediment in one year. The suspension feeders were thought to filter the entire volume of the Bay in about 16 days. The two categories processed about the same mass of organic matter in their diet.

The net annual secondary production from the whole benthic fauna was calculated as 62,700 t C/yr.

Knowledge of the zooplankton community in the Bay was very scanty. In fact only four surveys of zooplankton had been carried out. Unfortunately, not all used the same sampling technique so that comparison of results can only be general.

All surveys agree in assessing the zooplankton population to be low compared to similar waters elsewhere.

Mesozooplankton, those organisms larger than 150 microns, amounted, on average, to 3,000-4,000 organisms/m^3. The mesozooplankton contained mainly crustaceans (copepods, cladocerans, malacostracans and cirripedians) with some bivalve larvae.

The microzooplankton, those less than 150 microns in size, were more numerous – up to 200,000 organisms/m^3. The microzooplankton consisted of ciliates, tintinnids, copepod larvae, bivalve larvae and rotifers.

Most organisms underwent annual cycles of growth, with some reaching maximum numbers in summer.

During the Phase One Study, the total crustacean standing crop ranged from a maximum of nearly 7,000 organisms/m^3 in early spring to only 1,200 organisms/m^3 in the late summer of 1970.

No consistent relation was found between numbers of animals and position in the Bay i.e. there was no evidence of either deleterious or stimulatory influences. There were, however, qualitative differences in speciation.

Several studies of particular taxa had been carried out, including observations of feeding habits. There was also some work on ichthyoplankton.

A later review, Technical Report No. 11, assessed the magnitude of zooplankton trophodynamics in the Bay. The approach was the same as that used for zoobenthos, *viz.*, literature values of clearance and assimilation rates were applied to the zooplankton taxa known to be present in the Bay. It was calculated that the whole Bay volume was cleared between 11 and 42 days.

Attention was drawn to the probable importance of the microzooplankton and the 'microbial loop' and for this reason grazing measurements were recommended.

Finally, one objective of the PPBES [Task E8, Fish Productivity] was to survey the distribution, species composition, abundance and biomass of demersal fish in Port Phillip Bay. Hence a review was commissioned of the existing knowledge of fish stocks in the Bay.

The commercial fishery in Port Phillip Bay is the largest in Victoria, having a value of \$30-\$40 million a year. Paradoxically, the finfish production rate in terms of catch per unit effort or biomass per unit area is one of the lowest in Australia, but the large area of the Bay compensates for this.

On the other hand, the filter-feeding shellfish (i.e. scallops and mussels) display an extremely high rate of growth and abundance. In fact, of the total fish production from the Bay of 35 kg/ha, shellfish provided 24 kg/ha or two-thirds. The actual abundance of demersal fish caught by the trawl survey during the Phase One Study was in the range 9-21 kg/ha. There is no estimate of the pelagic fish population. Of course, a large part of the commercial catch and some of the recreational fish catch consists of pelagic species.

The Phase One trawl surveys disclosed considerable variation in fish abundance with time and in space.

Highest abundance was in late summer and late winter overall, but catches were always highest in the south-west off the Bellarine Peninsula. On the other hand, yields in the north and east were higher than the Bay average so that no untoward effects of Yarra River or WTP outflow could be detected. In fact, the highest commercial catches of fish are in the north-east of the Bay and the eastern portion of Geelong Arm. The highest recreational catches are taken in the north-east and east of the Bay, but this probably reflects such factors as population distribution and the availability of boat ramps, piers and jetties.

The Phase One trawl survey found 68 species of fish in the Bay. A repeat survey at similar locations with the same trawling gear in 1990 found the same number of species.

On both occasions, the three varieties of flathead dominated, providing more than half the overall yield, with stingarees the next most abundant. The Phase One Study report concluded that distribution of demersal fish species in the Bay depended on depth and substrate. These two factors are, of course, inter-related, as is the distribution of the macroinvertebrates providing the food for most species of fish. Because the height of the trawl used was 1.5 m, some species of bathypelagic fish were also taken.

The general biology of the principal commercial fish species was known, such as their distribution, growth patterns, habitat, migration, feeding and spawning behaviour. A description of these factors for flathead, snapper, King George whiting, pilchard, anchovy, calamari, mussel, scallop and abalone is given in Technical Report No. 7.

Chapter 5 of the Status Review set out nine knowledge gaps and research needs:

- Measurement of phytoplankton productivity including nano and picoplankton.
- Sampling for toxic and non-toxic algal bloom species.
- Assessment of the rate of grazing of phytoplankton filter-feeders.
- Measurement of light attenuation in Bay waters.
- Mapping of seagrass meadows and macroalgae beds and measurement of their productivity and nutrient turnover.
- Determination of the distribution and productivity of MPB.
- Determination of the species distribution, biomass and abundance of the dominant macroscopic benthic invertebrates.
- A review of the trophic relationships of the benthic invertebrates.
- A survey of the distribution, species composition, abundance and biomass of demersal fish species and their trophic relationships.

How the knowledge gaps have been addressed

Task N2.2, carried out by Monash University, measured the chlorophyll content and productivity of three size classes in the phytoplankton over two years. The field work was carried out in conjunction with Task N2.1 so that simultaneous measurements of nutrients, light, physical parameters and phytoplankton were made.

The VFRI Shellfish Sanitation Program had been collecting microalgae at several sites around the Bay for four years. PPBES provided financial assistance for the collation, statistical treatment and publication of the accumulated data [Task E10.4].

A literature review of zooplankton grazing rates extrapolated to the Bay had been published as Technical Report No. 11. Confirmatory field and experimental work was then conducted by Monash University as Task E10.5.

Habitat Mapping has been pursued in three modes. Tasks G2.1 and G2.2 developed a remote sensing approach using CASI (Compact Airborne Spectral Imaging). Ground truthing was carried out both by underwater video camera [Task G2.3] and diver observations [Task N2.3]. The first two approaches were pioneered by CSIRO while the diver surveys have been carried out by Consulting Environmental Engineers Pty Ltd.

The microphytobenthos in the Bay have been examined for distribution, abundance and productivity by Monash University as Task N2.4.

The benthic macroinvertebrates of the Bay have been re-surveyed by the Museum of Victoria [Task E5.2]. The triple survey pattern was developed after statistical examination of all pre-existing data [Task E5.1].

By arrangement with VFRI, surveys of demersal fish distribution were carried out together with an examination of fish diets [Task E8].

4. Design of the Study

The objectives of the PPBES have been described earlier in this introductory chapter. It is now appropriate to examine why these objectives were chosen and how the Study was designed to achieve them.

The objectives of the Study resulted from a process of consultation with officers of relevant government agencies, academic and other research scientists and interested members of the public, either as individuals or representatives of interest groups.

A description of the process and those involved is given in the Study Project Design, issued in 1992.

Such consultations gave rise to what were termed 'issues' which were either environmental or managerial.

Environmental issues are really questions about the state of the Bay. Many of these had not changed since the genesis of the Phase One and Phase Two Studies, although some had been answered by those Studies, at least up to 1978.

The number of issues raised was legion, but a condensed version would be:

- What are the sources, nature and magnitude of nutrient and toxicant inputs to the Bay?
- What processes of mixing, transport, uptake, sequestration or export of these inputs occur?
- What effects do inputs have on the physico-chemical status of the Bay and its biota? For example:
    - Is there any danger to biota from existing toxicant levels?
    - Do existing toxicant levels present a hazard to human health by consumption of seafood from the Bay?
    - Is the Bay in danger of eutrophication from excess nutrient loads?
    - Has there been a deleterious change in either the flora or fauna of the Bay?
- Is a description of the condition of the waters of the Bay sufficient to judge the health of the Bay – what about sediment processes?
- Is it sufficient to describe the Bay as a whole or are there parts more threatened than others?
- What are the key inputs in terms of effects?

Managerial issues depend to some extent on the outcome of the PPBES in that key inputs and effects need to be known. However, some issues could be foreshadowed:

- If key inputs are discovered, what threats do they pose to the Bay and how can they be controlled?
- Are there other influences of a lesser nature but significant when taken in total? Should they be controlled?
- What time scale do we have for remedial action if this is required?

The Study was designed to answer these issues, and took the form described on the following page. The division into subject areas was convenient given the different professional skills required.

Physics

- Collection of meteorological and oceanographic data.
- Measurement of mixing coefficients and processes.
- Collection and analysis of data on inputs of heat, water and suspended matter from rivers, the land, groundwater and point sources.
- Collection and analysis of data on the nature and behaviour of suspended and bed sediments.
- Development of numerical models to describe the water circulation patterns, exchange processes and sediment behaviour in response to forcing factors.

Nutrient Chemistry

- Collection and analysis of data on inputs of nutrients from rivers, the land, groundwater, atmosphere and point sources.
- Collection of data on the temporal and spatial distribution of phosphorus and nitrogen compounds and silicate in the waters of the Bay. This to be in conjunction with measurements of the basic physical parameters of salinity, temperature, dissolved oxygen, turbidity and light penetration.
- Collection of data on nutrients and associated parameters (salinity, iron, dissolved oxygen, sulphate) in Bay sediments. In particular, knowledge was required of nutrient distribution with depth, influx or efflux from sediments to water and the extent of denitrification.

Toxicant Chemistry

- Collection and analysis of data on toxicant inputs from rivers, the land, the atmosphere, groundwater and point sources.
- Measurement of toxicant levels in the water column, the sediments and selected biota.
- Observation of any toxicant transfers due to shipping and dredging.

Ecology

- Measurement of phytoplankton productivity and biomass and their changes with time and place.
- Assessment of the distribution, biomass and productivity of seagrasses, macroalgae and microphytobenthos.
- A survey of the benthic fauna providing data on species composition, abundance and distribution compared to past surveys.
- Assessment of the species composition, distribution, abundance and diets of the demersal fish population in the Bay.
- Examination of the species composition and abundance of phytoplankton in blooms.

General

- Setting up of a database to hold all data generated by the Study in a GIS framework.
- Mapping of benthic habitat by remote sensing with ground truthing.
- Conduct of an intensive and extensive review of existing knowledge of the Bay.
- Development of integrated models of the nutrient and toxicant processes in the Bay.

5. The Integrated Model

One important feature of the PPBES was the underlying concept of a numerical model which would integrate all data by processes. This concept was built into the Project Design and influenced the nature of the research tasks as well as generating two Tasks, G8 and G9. The objectives of these two Tasks were given as:

- To form an on-going interdisciplinary modelling capability to be used for experimental design, to improve process understanding and to serve as a guide for future model developments.
- To form the basis of a range of complex scientific models describing the transport and fate of nutrients and toxicants. The outputs of these models would be presented in report form and would constitute the basis for tools to be used directly by management.
- To make an assessment, possibly using predictive models, of the biological impacts of nutrients and toxicants in PPB and to advise managers of potential impacts of various management scenarios.
- To design realistic and informative field experiments to test the models.
- To develop a range of computer-based management tools and to design field experiments to test the models.

A symposium was held in November, 1992, at which the various options for modelling were canvassed.

Reference was made to the two earlier hydrodynamic models of Port Phillip Bay, the Beenyup Eutrophication Model and Jervis Bay Model in Australia, and several models developed overseas.

The presentations made at this symposium, and the conclusions reached, are set out in Chapter 6 of the Status Review.

Briefly, the symposium discussed the value and purpose of an integrated model, its feasibility and the form it should take.

The principal conclusions were:

Issues to be addressed fall into two categories. The first of these may be described as indicators, criteria or standards. They consist of readily measurable physico-chemical parameters where levels reflect the first result of management changes to inputs. Such indicators are dissolved oxygen, toxicants, nutrients, pathogens, turbidity and chlorophyll. These can be modelled on a ‘process’ basis.

The second category are called ecosystem indicators or environment quality objectives. They represent the effect of the first category as perceived by a user, and might include water clarity, nuisance algal blooms, contaminated seafood, fish and shellfish yields, abundance and health of flora and fauna, odour and anoxia. The processes linking these effects are not well quantified by theory and require empirical modelling based on observed relationships.

The underlying physical features of Bay processes could be fully described by a 3D model of quite fine resolution, say a 300 m grid with 10 layers in the vertical. This resolution could be even finer in limited areas close to inputs. This model would generate advection and mixing coefficients for a transport model.

A nutrient model could be much coarser in the central Bay, but would need finer resolution near inputs. It might also require vertical resolution because of occasional stratification in the Bay. The time scale could vary from hours to weeks. The model would need to extend into the sediments. As well as covering primary production, the model will have to take grazing and sedimentation into account.

The toxicant model presents a special case because of the unique physico-chemical transformation of toxicants entering the marine environment. Precipitation, partition and decay would have to be considered as well as uptake through biological pathways. The time scale would range from hours to years. The spatial resolution would be fine (of the order of 100 m) near inputs but coarser (some kilometres) in the central region of the Bay.

In the case of ecology, it was felt that modelling possibilities were more limited. Empirical models might be used to predict toxic algal blooms, contamination of fish and shellfish or the fate of valued habitats. Higher levels such as birds, mammals or fish yield would not be attempted.

Finally it was seen that a combination of model types would be required. These would be process, empirical, statistical and budget or inverse. Their roles would be description, explanation, prediction and recommendation.

6. Sources

The material presented in this chapter was mainly derived from the Status Review and the references listed in Chapter 7 of that Review.

Historical material was drawn from the Memoirs of the National Museum of Victoria, Volume 31, 1971.

Some relevant information from PPBES Technical Reports Nos. 10, 11, 12 and 13 was also used.

Contents

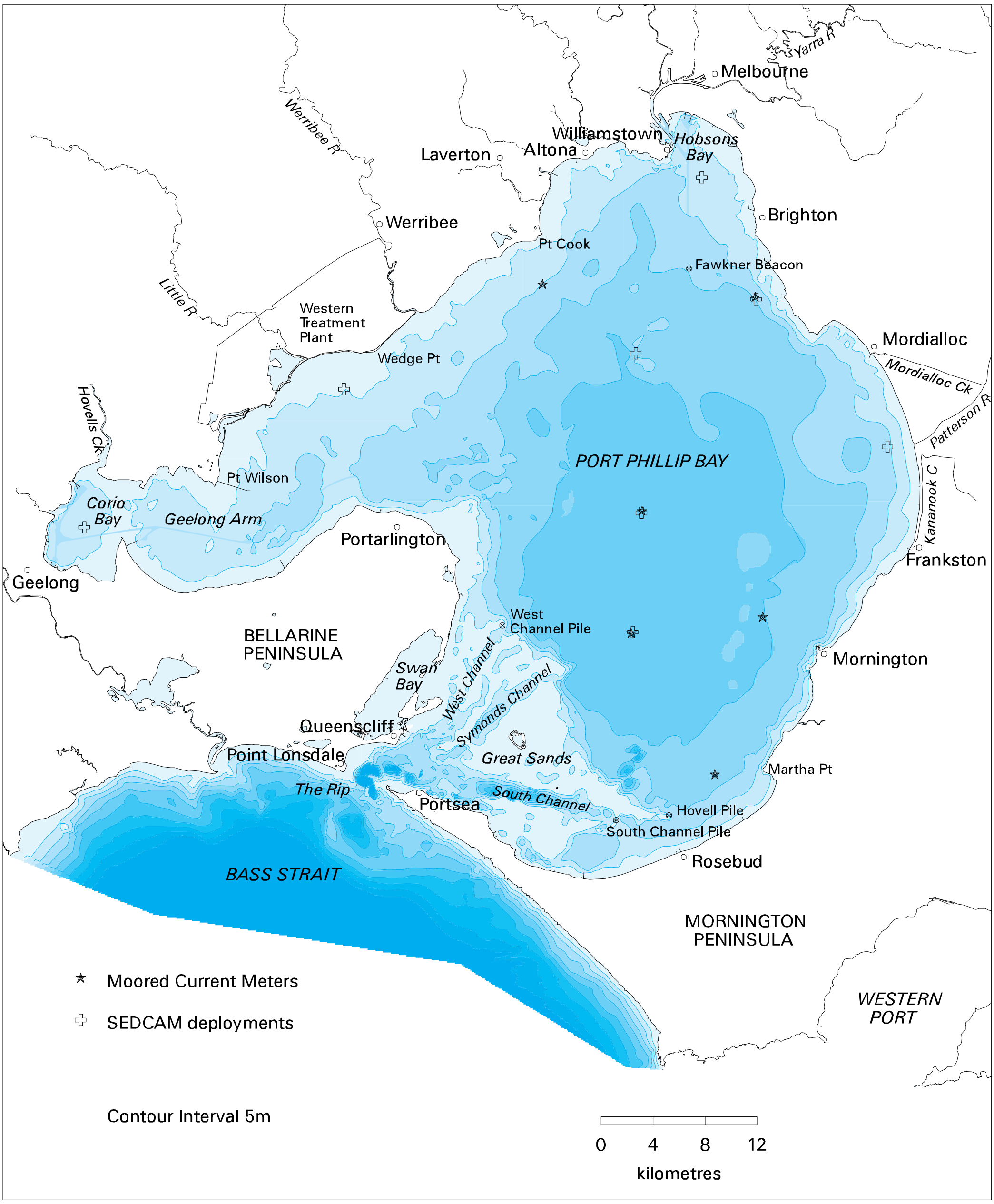

Figure 2.1: Physics tasks locality map.

1. Introduction

Given the large population surrounding Port Phillip Bay and the range of activities which take place on and near the Bay, it was not surprising that many aspects of the Bay's physical environment had been well studied prior to this Study.

The physics program of the Port Phillip Bay Environmental Study was designed to improve the understanding of physical processes in the Bay, to fill gaps in existing knowledge and to provide physical data and models which were capable of supporting the integrated ecological modelling of the Bay. It was vital that the physical data sets and models cover as wide a range of conditions as possible so that comprehensive scenarios could be modelled. As a result, extensive data sets were collected, both historical and newly observed.

A number of reviews were commissioned to define the existing knowledge of the physics of the Bay. These included a general review of the physics of the Bay (Black and Mourtikas 1992), a review of atmospheric inputs to the Bay (Beer *et al.* 1992) and a review of physical processes (Hunter 1992).

Initially, the Study Design described eight physics tasks (CSIRO 1992). Some of the activities in these tasks were incorporated into tasks elsewhere in the study, leaving four major physics tasks to be conducted for the Study. These were:

Task P1 The collection of a wide range of meteorological and oceanographic data from the Bay. This task involved the collation of existing data sets and an extensive field program. Data were obtained from Melbourne Water, Victorian EPA, Bureau of Meteorology, National Tidal Facility and the Port of Melbourne and Port of Geelong Authorities. Field work was conducted by Oceanroutes Pty Ltd (now WNI Science and Engineering), Lawson and Treloar Pty Ltd and James Cook University Department of Physics.

Task P2 The collection and analysis of mixing data in the Bay. This task involved a review of mixing processes in the Bay (Pattiaratchi *et al.* 1995a) and a number of CTD and microstructure surveys in the Bay. This work was conducted by the University of Western Australia Centre for Water Research (CWR).

Task P7 The collection and analysis of suspended and bed sediment data in the Bay. This task involved the collection and analysis of sediment cores from 80 sites, as well as the deployment of an instrumented tripod (referred to as SEDCAM) at eight sites on the bottom of the Bay. This work was carried out by the Victorian Institute of Marine Science (VIMS).

Task P8 The development, calibration and validation of numerical models of the Bay. This task involved developing, calibrating and validating wind, wave, hydrodynamic and transport models for the Bay. The work involved Lawson and Treloar Pty Ltd and the CSIRO Division of Oceanography.

This chapter provides only a summary of the physical knowledge developed and data sets gained during this Study. A vast amount of data has been collected. Much more detailed information can be found in the various Technical Reports and Task reports from this Study.

2. Bathymetry

Port Phillip Bay is a large, relatively shallow body of water with a restricted connection to Bass Strait.

The main part of the Bay is almost circular, sloping gently on the western side and more steeply on the south and east sides. The maximum depth in the centre of the Bay is about 24 m.

To the south of the main body lies the Great Sands region, a flood tide delta consisting of substantial sand bars and shallows dissected by deeper channels. This region greatly restricts exchange between the main Bay and Bass Strait.

Further south again lies the entrance to the Bay, a relatively narrow opening (about 3 km) with strong tidal currents. In fact, the deepest parts of the Bay are found in this area (chart AUS158) with deep scour holes in the South Channel (36 m) and in The Rip (almost 100 m).

To the west of the main Bay lies the Geelong Arm and Corio Bay. Exchange between Corio Bay and the main body of the Bay is restricted by a shallow bar through which a channel has been dredged to allow shipping access to the Port of Geelong. The Port of Melbourne is also accessed via dredged channels which traverse Hobsons Bay at the northern end of the main Bay.

Using coastline data obtained by the Study, the surface area of the Bay is calculated to be about 1930 km^2. Bathymetric data obtained from the Royal Australian Navy show that the mean depth of the Bay is about 13.6 m, so that the Bay has a total volume of about 26.3 billion (10^9) m^3. A plot of the bathymetry of the Bay is shown in Figure 2.2.

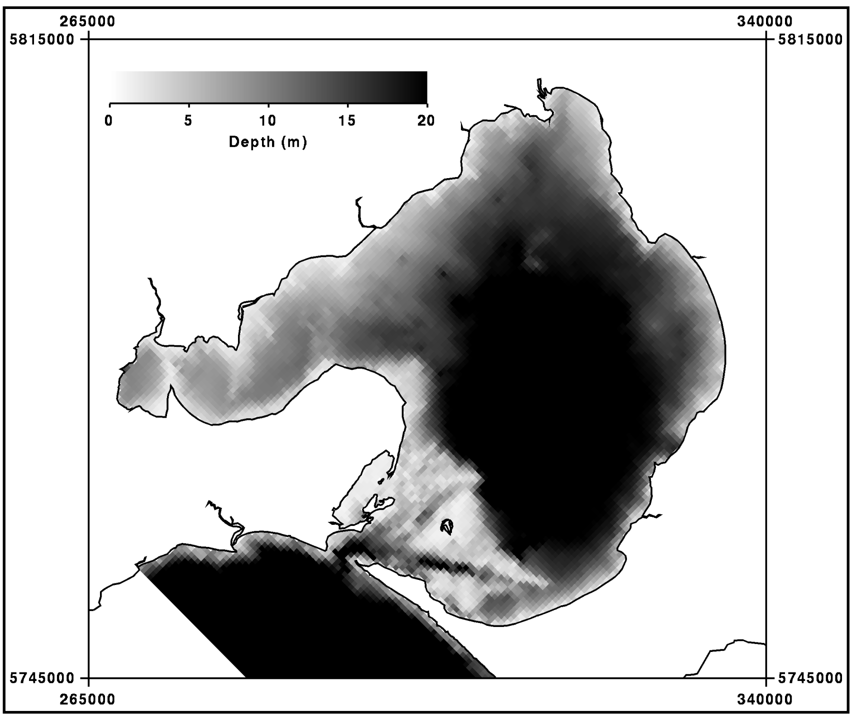

Figure 2.2: Bathymetry of Port Phillip Bay. Co-ordinates on this and other figures are in metres, referenced to AMG Zone 55.

3. Sea-level

Measurements of sea-level were not taken for this Study because there are a number of permanent tide gauges both near and inside the Bay.

These gauges are primarily operated by the Port Authorities, with data archived at the National Tidal Facility (Flinders University).

Hunter (1992) listed the sites of tide gauges which have been installed at various times in the Port Phillip Bay region.

Sea-level observations were obtained from the National Tidal Facility for sites shown in Table 2.1.

Data from Lorne were used to specify sea-level in Bass Strait for the open boundary of the physical models of the Bay.

Data from other locations were used to calibrate and validate the sea-level predictions made by these models inside the Bay.

It was also found that the differences in levels between gauges inside the Bay were well correlated with winds over the Bay (Section 5).

Table 2.1: Sea-level data obtained by the Study.

Site	Period	Resolution
Breakwater Pier (Williamstown)	1994	hourly
Geelong	1994	hourly
Hovell Pile	1994	hourly
Point Lonsdale	1985-1994	hourly
Lorne	1993-1994	hourly
Lorne	1994	six minute
Queenscliff	1994	hourly
Victoria Dock	1994	hourly
West Channel Pile	1994	hourly

3.1 Tides

The most obvious water level variations in the Bay are caused by the tides. Tidal amplitudes in the Bay are less than half of those in northern Bass Strait and there is a large phase difference between the main body of the Bay and Bass Strait. Tides are semi-diurnal (two tidal cycles per day) with a large diurnal inequality. For each of the tide gauge locations listed in Table 2.1, tidal constituents were also provided by the National Tidal Facility, calculated over a duration of at least one year.

The amplitudes (in metres) and phase (in degrees, UTC + 10 hours) of the major constituents are shown in Tables 2.2 and 2.3.

As the tables show, most of the attenuation of the tides occurs near the entrance to the Bay. Compared to its amplitude at Lorne, the M2 component drops to about 70% at Point Lonsdale and about 40% at Queenscliff. The remaining attenuation occurs across the Great Sands region. North of the Great Sands, the tides are reasonably uniform throughout the Bay, with some increases in amplitude in both Corio Bay and Hobsons Bay, resulting from the shape of the Bay.

Table 2.2: Amplitudes of major tidal constituents (m).

Gauge	M2	K1	O1	S2	N2
Lorne	0.615	0.209	0.145	0.193	0.117
Point Lonsdale	0.438	0.144	0.102	0.128	0.084
Queenscliff	0.259	0.111	0.085	0.073	0.051
West Channel Pile	0.205	0.096	0.067	0.046	0.035
Hovell Pile	0.202	0.096	0.070	0.049	0.035
Geelong	0.268	0.100	0.072	0.066	0.047
Breakwater Pier	0.236	0.098	0.068	0.054	0.039

Table 2.3: Phase of major tidal constituents (deg).

Gauge	M2	K1	O1	S2	N2
Lorne	319.1	57.1	26.1	88.6	272.1
Point Lonsdale	325.6	71.4	39.6	97.5	279.9
Queenscliff	344.8	95.0	62.2	119.7	297.3
West Channel Pile	57.0	131.1	94.9	193.5	10.2
Hovell Pile	60.4	133.4	96.0	199.9	14.1
Geelong	61.3	133.3	96.1	198.1	17.4
Breakwater Pier	60.4	131.8	95.1	197.6	12.4

3.2 Non-tidal motions

Variations of water level in the Bay are also caused by lower frequency motions in Bass Strait, winds, atmospheric pressure variations and horizontal density gradients. A discussion of these effects can be found in Hunter (1992) and Black and Mourtikas (1992).

In general, the low-frequency (periods of several days or more) variations in water level in the Bay are closely correlated with those in Bass Strait. In effect, The Rip and Great Sands act as a low-pass filter for sea-level so that the relatively higher frequency tides are attenuated as they enter the Bay, but lower frequency variations are not.

Particularly high water levels in the Bay are associated with low pressure systems and winds with a strong westerly component. Low pressure systems over Bass Strait will tend to raise sea-level generally, and westerly winds tend to raise the sea-level on the northern side of Bass Strait (due to the rotation of the earth). There may also be contributions from low-frequency waves generated in or propagating through Bass Strait.

A good example was seen in May 1994 when water levels in the Bay reached about 0.9 m above mean sea-level. Plots of atmospheric pressure, wind and sea-level during this period are shown in Figure 2.3. High levels are seen to occur on 18-19 May and 26-27 May in conjunction with strong north-westerly winds and relatively low air pressure. A decrease in air pressure of 1 hPa corresponds to an increase in sea-level of about 1 cm, so that the air pressure changes account for about 0.25 m of the changes in water level seen in the Bay during these periods.

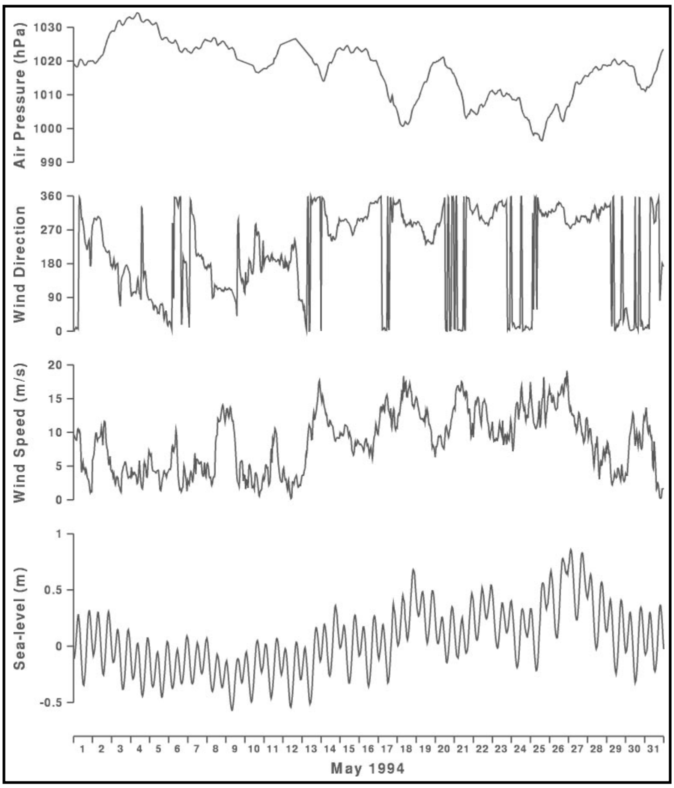

Figure 2.3: Plots of sea-level, wind speed and wind direction measured at Hovell Pile during May 1994. Air pressure data were recorded at Moorabbin.

4. Currents

Currents are of prime importance in understanding the Bay because they are the primary agents which carry nutrients, sediments and other substances from place to place in the Bay.

Tidal currents dominate the circulation in the southern-most parts of the Bay with speeds of 2.5 m/s or more during peak flood and ebb flows through The Rip and speeds up to 0.8 m/s in the major channels through the Great Sands. Currents are weaker in the main body of the Bay, typically 0.05 to 0.1m/s, and predominantly driven by the wind. These wind-driven currents show the following features (Hunter 1992):

- Horizontal gyres with the currents directed with the wind in shallow areas and against the wind in deeper areas.
- An overturning pattern with currents directed with the wind near the surface and against the wind near the bottom.

Density-driven currents may also be important in the region affected by the Yarra plume (Hunter 1992).

Current measurements described in this section include those collected using moored current meters, coastal radar (COSRAD) and a vessel-mounted acoustic doppler current profiler (ADCP). Supplementary current measurements were collected in conjunction with sediment tasks (SEDCAM in Section 7) and nutrient input tasks (N1.3, T1.2).

4.1 Moored current meters

Moorings were deployed at five sites in the Bay for two, six-week periods. Two current meters were located at each site, at about one quarter and three quarters of the prevailing water depth.

The sites were all in the main body of the Bay because the primary purpose was to monitor the net (non-tidal) water movements around the Bay and also to provide data for model calibration and validation.

The first current meter deployment covered the period from 21 April 1994 to 2 June 1994, with meters near the northern edge of the Great Sands, the Bay centre, Martha Point, Point Cook, and Red Bluff.

Data were recovered from all meters during the first deployment period.

The second deployment covered the period

6 September 1994 to 14 October 1994, with meters at the same sites except that the Great Sands site was replaced by a site near Schnapper Point. Data were not recovered from the lower meters at the central Bay and Red Bluff sites during the second period.

As an example of the data, Figure 2.4 shows mean velocity vectors at all meters over a two-day period (17-18 May 1994) when strong north-westerly winds were present.

The plot clearly suggests a gyre type circulation in the eastern half of the Bay, with currents of about 0.1 m/s directed with the wind at the Red Bluff site and of about 0.05 m/s directed against the wind in the centre of the Bay.

Somewhat surprisingly, the mean currents at the lower meters are stronger than those nearer the surface over this period.

Instantaneous speeds are similar at lower and upper meters, and this effect seems primarily to be because the near surface currents are more variable in direction over this period.

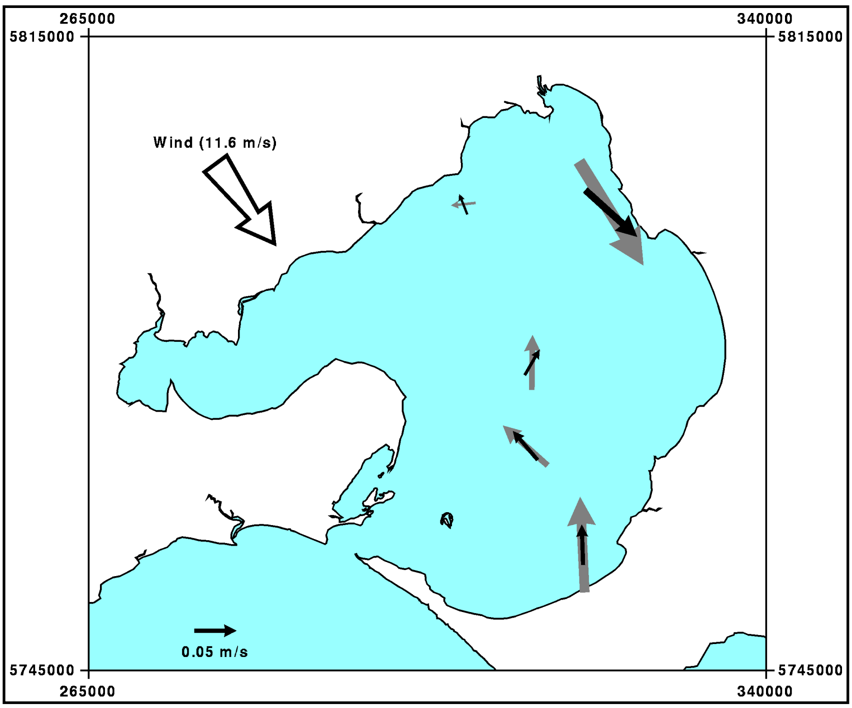

Figure 2.4: Mean currents measured for 17-18 May 1994. Data from lower meters are shown as grey arrows, upper meters are shown as black arrows. Arrow length is directly proportional to current speed. Mean wind velocity (measured at Hovell Pile) is also shown.

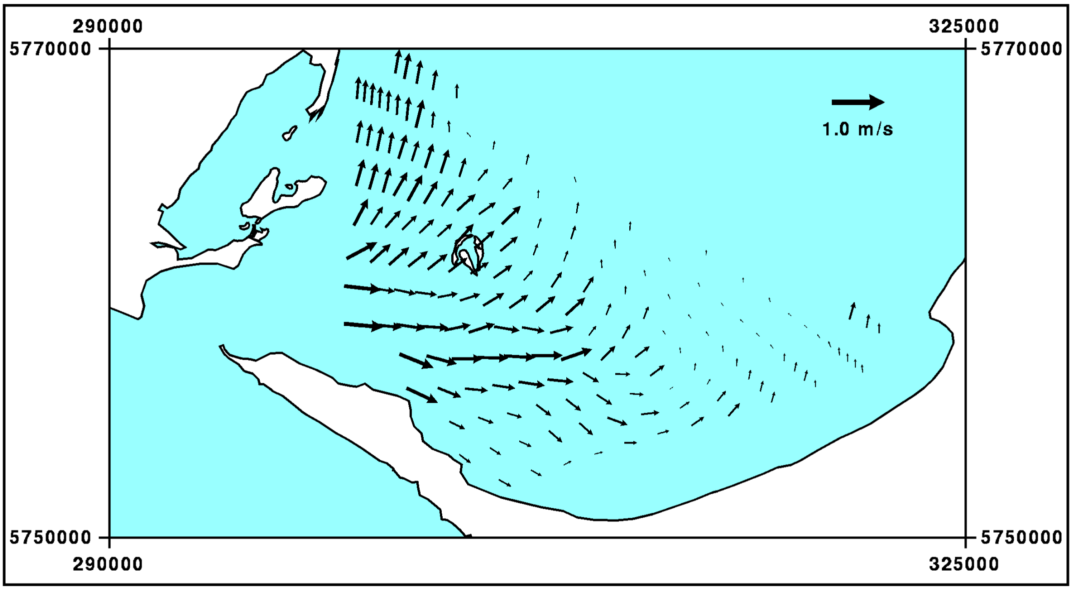

Figure 2.5: Surface currents measured by COSRAD at 2300 on 9 May 1994 (approximately peak flood flow).

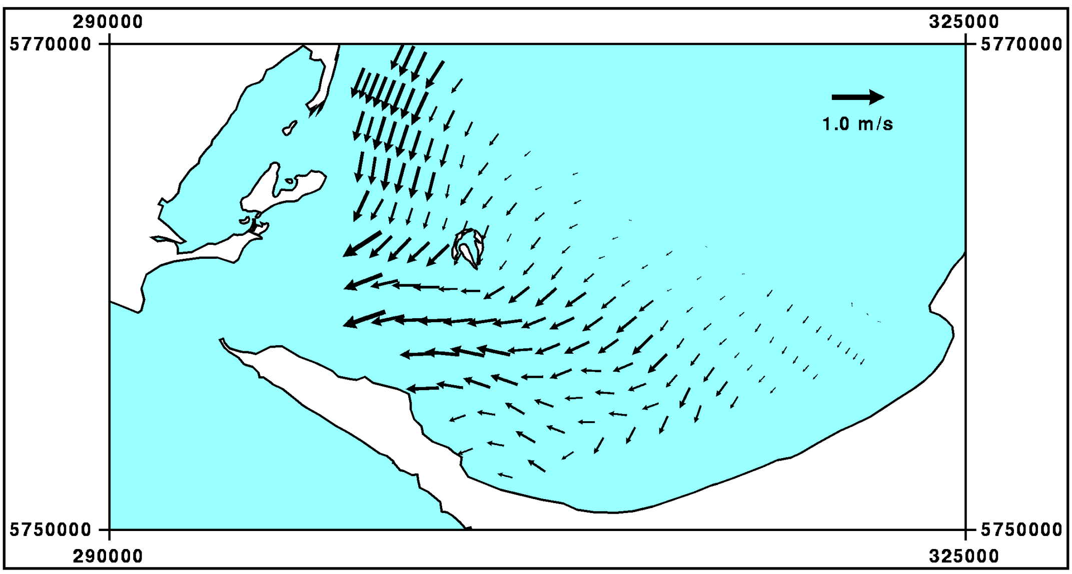

Figure 2.6: Surface currents measured by COSRAD at 0500 on 10 May 1994 (approximately peak ebb flow).

4.2 COSRAD

The COSRAD HF radar system developed by the Physics Department at James Cook University was used to study the complex flow patterns in the southern part of the Bay. The COSRAD system provides continuous measurements of surface currents over large areas using radar techniques.

Two COSRAD deployments were conducted. The first spanned from 2000 on 29 April 1994 to 0800 on 5 May 1994 and covered the area just to the north of the Great Sands. The second deployment spanned from 1800 on 5 May 1994 to 1200 on 12 May 1994 and covered the Great Sands region itself.

The data collected from these deployments provided hourly surface currents over the coverage areas with a spatial resolution of about 3 km. They form an excellent data set with which to compare numerical model output. The deployments and data are described in detail in Prytz and Heron (1995).

Plots of surface currents in the Great Sands region during flood and ebb flows are shown in Figures 2.5 and 2.6.

4.3 ADCP

A vessel-mounted ADCP was operated in the Great Sands area from 7 to 12 May 1994. This was coincident with the period when COSRAD data was collected over the Great Sands area.

The ADCP measured the vertical structure of flows through the major channels in the Great Sands and hence gave a measure of the total transport of water through these channels.

These data were used in conjunction with high resolution CTD surveys carried out in this area to obtain an estimate of the flushing time of the Bay (Section 10).

5. Wind

As mentioned in Section 3, wind plays an important role in determining sea-levels within Port Phillip Bay. However, for the physics of the Bay, the major effects of wind are to generate currents, which circulate material around the Bay, and waves, which may resuspend sediments into the water column in shallower parts of the Bay.

Beer *et al.* (1992) described the main features of the wind climate over the Bay. In summary, the winds are quite variable with few obvious patterns. There are significant effects from cold fronts, sea breezes, local topography and the 'Melbourne Eddy'.

The aim of the study was to obtain wind data suitable for physical modelling for the period from 1985 to 1994 (inclusive). Winds over the Bay were interpolated from observations at three sites within the Bay (Hovell Pile, Point Wilson and Fawkner Beacon) and one site on land (Point Cook – a well-exposed site with a long record of observations). Raw wind data measured at Hovell Pile during the study are shown in Figure 2.7.

Correlations between points on the Bay and Point Cook were calculated for periods when all sites were available, and these correlations were used to calculate Bay-wide winds when only Point Cook data were available.

The end results of this process were continuous hourly wind data on a 10 km resolution grid covering the entire Bay.

An example of the result is shown in Figure 2.8, showing the interpolated wind field at midday on 19 September 1994, when strong south-westerly winds caused large waves in the north and east of the Bay (Section 6).

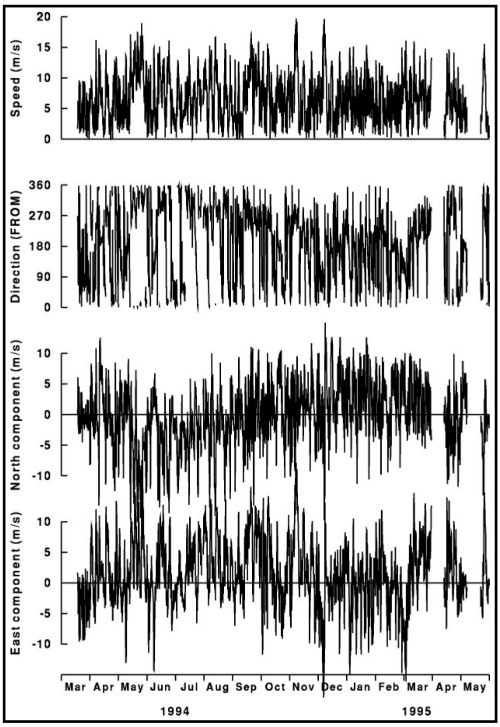

Figure 2.7: Wind data recorded at Hovell Pile from March 1994 to May 1995.

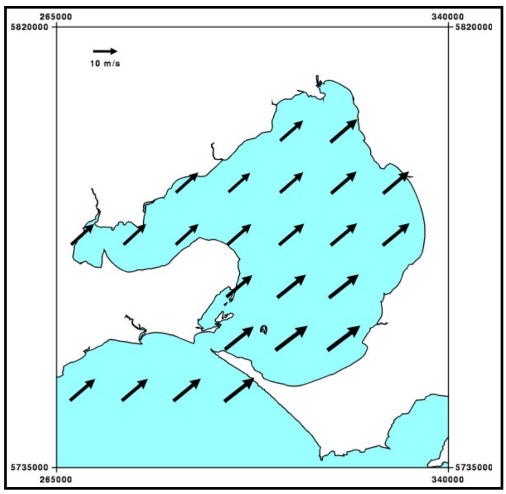

Figure 2.8: Winds interpolated over the Bay at 1200 on 19 September 1994.

6. Waves

To obtain some idea of the effects of waves, particularly on sediment resuspension, a wave model (Section 11.2) was developed for the Bay. This model takes wind data as input and provides outputs of various wave statistics on a 1,500 m resolution grid covering the entire Bay.

Figure 2.9 shows modelled wave heights at 1600 on 19 September 1994. This plot shows large waves (1.3 m significant wave height) in the north-east of the Bay during a period when the turbidity in this region was observed to rise significantly (Section 7).

Orbital velocities associated with waves were measured near the bottom during some SEDCAM deployments (Section 7). These data provide an indirect method of checking wave model results.

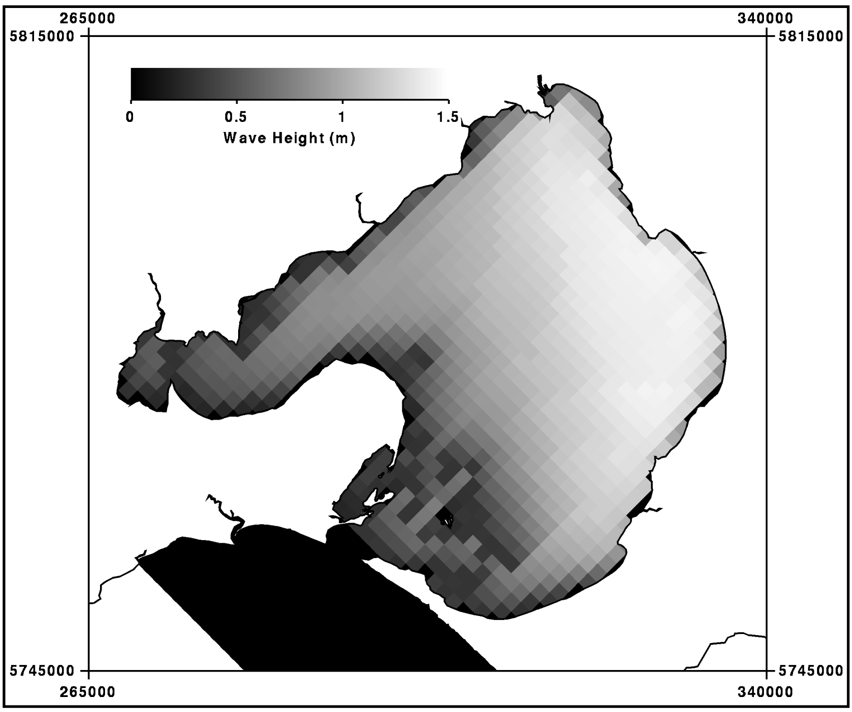

Figure 2.9: Wave heights calculated by the Wave Model at 1600 on 19 September 1994. The model does not perform calculations for Bass Strait.

7. Sediments

The sediments of Port Phillip Bay are a mixture of fine sands, silts and clays, with coarser sediments in shallow areas and the Great Sands region and finer sediments towards the deeper central portion of the Bay.

Sediment studies carried out prior to this study are summarised in Black and Mourtikas (1992).

Physical measurements of sediments undertaken during the study included the collection and analysis of cores from 80 sites in the Bay (Greilach *et al.* 1996), and the deployment of an instrumented tripod (known as SEDCAM) at eight sites on the bottom of the Bay (You *et al.* 1996).

The cores were analysed for grain size, density, porosity, viscosity, permeability and consolidation characteristics. These data form the basis for initialising the sediment properties of the transport model (Section 11.4).

Plots of the percent distribution of sand and silt+clay are shown in Figure 2.10.

SEDCAM deployments were made at eight sites in the Bay during the period June 1994 to September 1995, each deployment lasting about a month.

The tripod carried a video camera, current meters, optical backscatter sensors, PAR sensors, an automatic water sampler, a fluorometer and sediment traps.

The complete suite of data were not always available owing to some instrument failures, damage sustained when the frame was dragged during the third deployment and fouling.

The data obtained included information on currents, wave orbital velocities, suspended sediment concentrations, light levels, attenuation and bed forms (You *et al.* 1996). These data form a valuable calibration and validation set for the sediment component of the transport model.

Much SEDCAM analysis and sediment model calibration and validation work re-

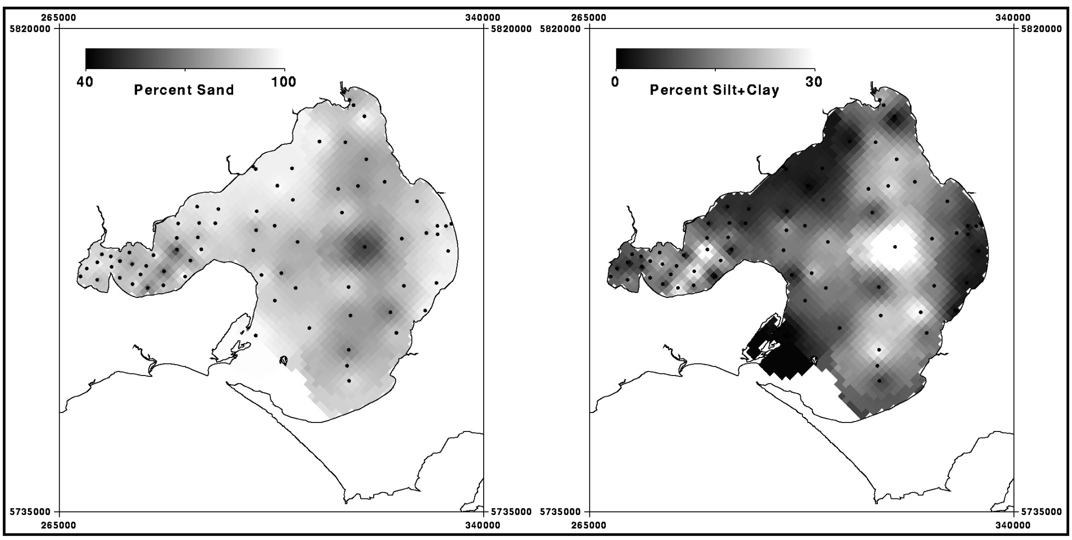

Figure 2.10: Distribution of sand and silt+clay in sediments in Port Phillip Bay. These data were interpolated to a 1 km grid from sampling sites shown as black dots. No data was present for and to the south of the Great Sands region.

mained to be done at the time this report was written.

Turbidity was measured during the four Bay-wide CTD surveys in May and September 1994. These data complement the SEDCAM data, providing a Bay-wide snapshot of suspended sediment concentrations.

The September 1994 surveys showed that turbidity increased significantly in the north and east of the Bay between mid and late September 1994. During that period SEDCAM was at a site near Red Bluff (in the north-east of the Bay).

Figure 2.11 shows near-bottom suspended sediment concentrations measured by SEDCAM, wind speed and direction measured at Hovell Pile and total flow from the Yarra.

Suspended sediment concentrations increased strongly on 19 and 20 September in conjunction with strong south-west to north-west winds.

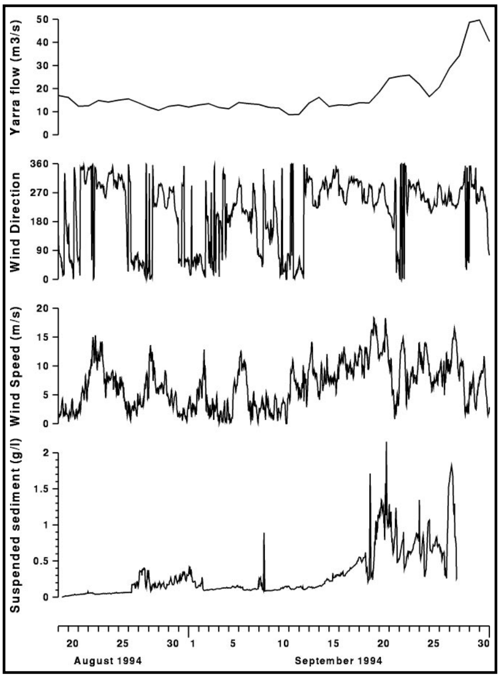

Figure 2.11: Suspended sediment concentration measured by SEDCAM near Red Bluff, wind speed and direction measured at Hovell Pile and total Yarra flow for the period late August-September 1994.

These winds presumably generate waves (see Figure 2.9) which help cause the resuspension of the sediment. There is a moderate increase in flow from the Yarra during this time which may also contribute some suspended matter.

Particulates enter the Bay from a number of sources. No data is available for inputs from coastal erosion, or from Bass Strait (which might provide sand to the southern-most parts of the Bay).

Other inputs are dominated by the Yarra River. Here, inputs were estimated as part of the monitoring of nutrient inputs to the Bay (Chapter 3). These estimates show a total non-filterable residue (NFR) input of about 145,000 t over the period January 1993 to October 1995 (inclusive) which corresponds to an annual average of about 51,000 t.

Average figures do not give a good idea of the variations in input, however, and closer examination of the data shows that most input occurs in just a few short events.

Particulate inputs from rivers, creeks and drains (excluding the Yarra and WTP) are described by NSR Pty Ltd (1993). These data indicate a total annual input of about 14,000 t NFR from these sources, based mainly on an analysis of historical data collected from 1979 to 1981.

Atmospheric inputs of particulates were roughly estimated by Hunter (1992) to be about 13,000 t/yr based on atmospheric concentrations of particulates given in Beer *et al.* (1992).

Total suspended solids data obtained from the Western Treatment Plant show an annual input of about 6,000 t.

The total of all the above inputs comes to about 85,000 t/yr. If it is assumed that the material is deposited uniformly over the Bay then the equivalent sedimentation rate for that area is 0.04 kg/m^2/yr. Assuming a grain density of quartz (2,650 kg/m^3), and a deposit porosity of 0.5, this is equivalent to a build-up of sediment of 0.03 mm/yr.

This calculation is likely to underestimate the actual sedimentation rate.

Another estimate of sedimentation rate can be made from sediment trap data collected during SEDCAM deployments.

Traps were placed on the tripod at 0.55 m, 1.2 m and 1.6 m above the bottom.

At five of the sites, insufficient material or fouling prevented the calculation of deposition rates.

At the remaining three sites, the traps 1.6 m from the bottom showed rates equivalent to 1.2 kg/m^2/yr (Central Bay), 3.5 kg/m^2/yr (Corio Bay) and 70 kg/m^2/yr (Hobsons Bay).

These rates overestimate the build-up of sediments because the traps collect sediment resuspended from the bottom as well as new inputs of particulates. Also, they represent rates over different periods of only about a month each.

Using the deepest site (Central Bay), where resuspension is likely to be minimal, and using the assumptions about grain density and porosity detailed above, the equivalent build-up of sediment is calculated to be about 0.9 mm/yr.

From analyses of ^{210}Pb and TBT profiles in a core taken from the northern part of the Bay it was calculated that the build-up of sediments was less than 1.5 mm/yr.

The long-term build-up of sediments in the Bay is roughly 2 m over the past 7,000 years or about 0.3 mm/yr.

A summary of sedimentation rate estimates is shown in Table 2.4.

Table 2.4: Bay-wide sedimentation rates.

Method	Sedimentation rate (mm/yr)
Particulate inputs	0.03
Long-term sediment depth	0.3
Sediment Trap (Central Bay)	0.9
Core profiles	0 to 1.5

8. Fresh Water

On average, the amount of fresh water lost by evaporation from the Bay surface almost equals the inputs to the Bay from rivers, creeks and direct rainfall, so that the net fresh water input to the Bay is very small (Hunter 1992).

Nevertheless, fresh water inputs, particularly from rivers, creeks and drains, are very significant because they carry much of the load of particulates, nutrients and toxicants entering the Bay.

Fresh water inputs also dilute 'oceanic' tracers (such as salt) and alter the density of the receiving waters.

Changes in salinity can be used to estimate a flushing time for the Bay (Section 10). Horizontal density differences can drive currents in the Bay (Section 4).

8.1 Rivers, creeks and drains

The total flow rate into the Bay from rivers, creeks and drains was estimated by the Phase One Study (MMBW/FWD 1973) to be about 44 m^3/s, most of which enters the Bay at the mouth of the Yarra River.

The present study obtained daily flow data for those rivers, creeks and drains over the periods shown in Table 2.5.

Table 2.5: Flow data obtained by the study.

Site	Period
Yarra River (Chandler Highway)	1976-1995
Maribyrnong River (Keilor)	1980-1995
Merri Creek	1980-1995
Gardiners Creek	1980-1995
Moonee Ponds Creek	1990-1995
Mordialloc Creek	1988-1994
Patterson River	1988-1994
Dandenong Creek	1994
Eumemmerring Creek	1994
Kororoit Creek	1993
Werribee River	1993-1994

Table 2.6: Annual mean flows, 1990-1995 (m^3/s).						
Input	**1990**	**1991**	**1992**	**1993**	**1994**	**1995**
Yarra River (total)	23.9	22.5	36.4	39.0	18.3	30.0
Mordialloc Creek	2.0	2.0	2.8	2.1	2.0	-
Patterson River	4.2	5.4	7.1	6.6	2.0	-
Werribee River	-	-	-	4.6	0.25	-

The data obtained show that inflows vary substantially from year to year.

For example, Table 2.6 shows mean annual flows from some of these inputs over the years 1990 to 1995. Here, the data for Chandler Highway, Keilor, Moonee Ponds Creek, Merri Creek and Gardiners Creek have been combined to give a single total flow for the Yarra River.

Flow data for the last two months of 1995 were not available at the time of writing so that the 1995 estimate for the Yarra is an average from the first 10 months of that year.

Substantial variation is evident for all inputs except for Mordialloc Creek where the flow is controlled.

Compared with the other years in the 1990s, 1994 was a dry year, however, the early 1980s were much drier – in 1982 the mean flow from the Yarra was only about 6 m^3/s.

8.2 Western Treatment Plant

Inputs of fresh water enter the Bay from the Western Treatment Plant via four outlets (15E, 145W, Lake Borrie and Murtcaim). These outlets are located over about 10 km of coastline (Murray 1994). Flow data for these outlets were obtained for the period 1980-1995. Mean flow rates for each year over the period 1990-1995 are shown in Table 2.7. A fifth outlet (Little River) ceased operation in 1989.

These data show that flow rates from the WTP have remained fairly constant over the past six years, but that the distribution of flow rates between the outlets has changed, with increased flows from the 15E outlet and reduced flow rates from the other outlets.

Table 2.7: Annual mean flows from the Western Treatment Plant, 1990-1995 (m^3/s).						
Outlet	**1990**	**1991**	**1992**	**1993**	**1994**	**1995**
15E	1.26	1.11	1.54	2.26	2.20	2.74
145W	1.29	1.24	1.08	1.11	0.90	0.85
Lake Borrie	0.91	1.09	1.13	0.97	0.81	0.60
Murtcaim	1.58	1.26	1.44	0.88	0.71	0.87
TOTAL	5.04	4.70	5.19	5.22	4.62	5.06

8.3 Rain

Rain data were acquired by the Study in a number of forms. A rain gauge was installed on Hovell Pile and recorded data over the period March 1994 to May 1995. A number of operational problems caused significant gaps in this data set.

Rain was also estimated from Bureau of Meteorology RAPIC radar data. This data set covered the Bay on a 2 km resolution grid, spanning from late February 1994 to late February 1995.

The total RAPIC estimated rainfall for the 10 months March to December 1994 is shown in Figure 2.12.

The RAPIC data for this period show rainfall increasing from west to east across the Bay, with about 300 mm of rain in the Geelong Arm increasing to about 500 mm near Mordialloc.

Extreme values shown in the very north of the Bay are most likely to be aberrations resulting from the small distance from the radar site.

Rainfall data were also obtained from the Bureau of Meteorology for the period 1985-1994 at 13 sites surrounding the Bay and a site at Barwon Heads. These data were interpolated onto a 10 km resolution grid covering the Bay. The resulting rainfall for the 10 months March to December 1994 is shown in Figure 2.13.

This figure shows the spatial pattern described by Beer *et al.* (1992) with rainfall increasing from north-west to south-east.

During this period total rainfall varied from about 300 mm in the north-west of the Bay to almost 500 mm in the south-east of the Bay.

These values are quite similar to those obtained by the RAPIC data presented above, but the large-scale pattern is slightly different.

The RAPIC data also show finer scale spatial detail over the Bay itself as well as a high level of very fine-scale noise.

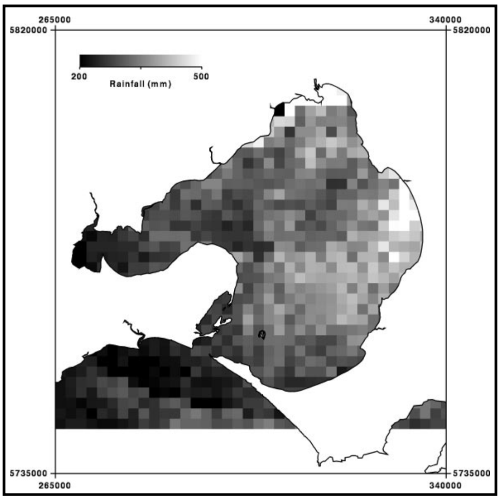

Figure 2.12: Total RAPIC estimated rainfall, March-December 1994.

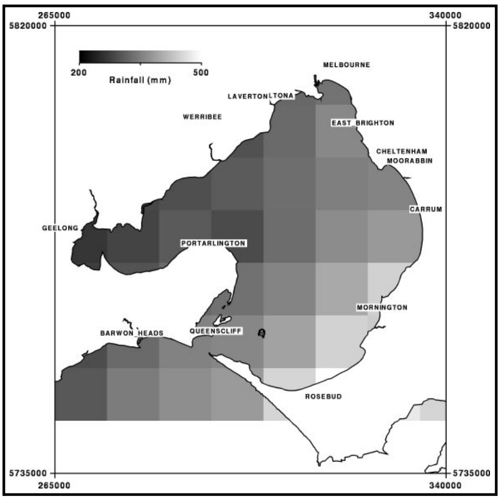

Figure 2.13: Total rainfall March-December 1994 as interpolated from the sites shown.

CHAPTER 2

Calculated annual rainfall (from Bureau of Meteorology data) over the Bay for each of the years 1985 to 1994 is given in Table 2.8.

Also shown are equivalent flow values obtained by multiplying annual rainfall by the area of the Bay and dividing by the number of seconds in a year.

The mean annual rainfall over the 10-year period is 677 mm.

Substantial variations occur from year to year with the wettest year (1993) receiving 849 mm and the driest year (1994) receiving 484 mm.

8.4 Evaporation

Evaporation is difficult to measure directly, and is recorded at only two stations in the Melbourne area, both to the north of the Bay (Hunter 1992).

As a consequence, evaporation rates for this study were calculated using a method which requires knowledge of the wind speed, air and water temperatures and relative humidity (WMO 1966).

These calculations covered the period 1985 to 1995, providing evaporation values on a 10 km resolution grid covering the entire Bay.

Table 2.9 shows annual mean evaporation values calculated over the Bay for this period. There is little variation in the calculation from year to year.

When used by the physical models (Section 11), these evaporation rates resulted in predictions of salinity in the Bay which were somewhat higher than observed values.

Table 2.8: Annual rainfall over the Bay.

Year	Rainfall (mm)	Equivalent flow (m^3/s)
1985	647	40
1986	612	38
1987	678	42
1988	661	41
1989	721	44
1990	606	37
1991	712	44
1992	797	49
1993	849	52
1994	484	29

Table 2.9: Annual mean evaporation over the Bay.

Year	Evaporation (mm)	Equivalent flow (m^3/s)
1985	1,418	87
1986	1,430	88
1987	1,433	88
1988	1,456	89
1989	1,421	87
1990	1,414	87
1991	1,393	85
1992	1,369	84
1993	1,397	86
1994	1,403	86

A problem with the method used to calculate evaporation is that it is highly sensitive to the humidity value used.

A humidity sensor was deployed on the Hovell Pile so that evaporation could be estimated by 'bulk' formulae (Beer *et al.* 1992). Data were recorded from March 1994 to May 1995 with substantial losses resulting from instrument failure.

Unattended measurement of humidity over extended periods is a difficult task, particularly for sites located on the water.

Despite elaborate precautions, the accuracy of the data collected remains in some doubt.

The observations show a mean humidity of about 70% over the measurement period. Increasing this value to 80% leads to a reduction in evaporation of about 30% which gives better agreement between modelled and measured salinities (Section 11).

Beer *et al.* (1992) describe some of the difficulties of obtaining reliable evaporation measurements and quote annual evaporation values measured at Aspendale of between 1,095 mm and 1,548 mm depending on the measurement technique used.

It is possible to estimate evaporation numerically using a one-box implementation of the transport model (Section 11), given long time series of fresh water inputs, salinity and possibly other tracers in the Bay.

8.5 Groundwater

Groundwater inputs to the Bay are estimated to be greater than 5.5 x 10^7 m^3/yr, or 1.7 m^3/s (Otto, 1992). However, groundwater is assumed to have little effect on the physics of the Bay (Hunter 1992).

9. Temperature and Salinity

9.1 Temperature

The temperature of the Bay is determined by the following processes:

- Solar radiation input (heating from the sun). This obviously only occurs during daylight hours and is a larger effect in summer than winter (due to both the longer days and the higher sun angle). Solar radiation input is also strongly affected by clouds. Hourly solar radiation data were obtained for Brighton during 1994. They show daytime values peaking at about 1,100 W/m^2 during December-January, decreasing to about 550 W/m^2 during June. Values averaged over a 24-hour period range from about 350 W/m^2 in summer to about 100 W/m^2 in winter.
- Long-wave radiation between the water surface and the sky. This cools the water surface and its magnitude is dependent on the water temperature, relative humidity and cloud cover. No direct measurements were available for use in this Study. For modelling purposes this effect was calculated using the variables mentioned above.
- Turbulent transfer of heat between the water and air. The magnitude of this transfer depends on the water and air temperatures and wind speed. For modelling purposes, it was calculated using a 'bulk' formula.
- Evaporative heat loss. As water evaporates from the Bay, the water left behind is cooled. The magnitude of this effect can be estimated directly from the evaporation rate, if it is known (see Section 8.4).
- Inputs of water from rain, rivers, creeks and drains. These often have a temperature different from the Bay and so alter the temperature of Bay water.
- Exchange with Bass Strait. Bay water is usually warmer in summer and cooler in winter than the waters of Bass Strait. Water exchange between the Bay and Bass Strait will tend to reduce the difference.

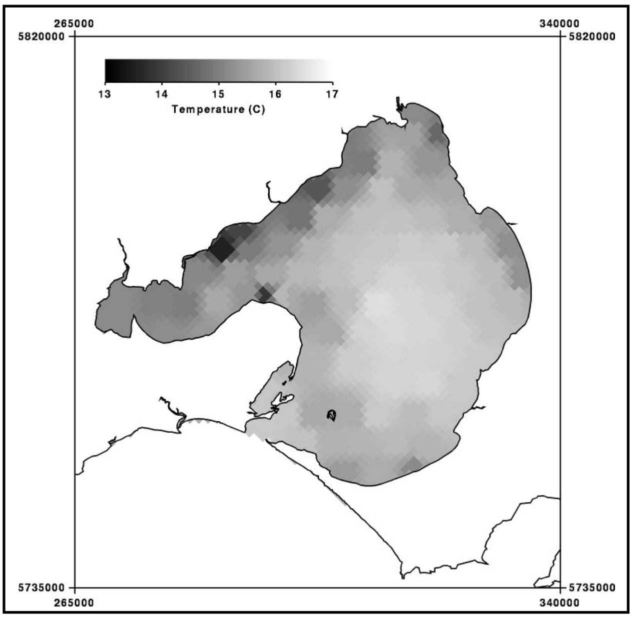

Figure 2.14: Temperature averaged between the surface and 1 m depth, interpolated from a CTD survey of 85 sites on 3-4 May 1994.

Continuous measurements of both water and air temperature were made at the Hovell Pile during the Study. These data show water temperatures varying from about 21°C in summer to about 11°C in winter with little day to day variation.

Air temperatures vary widely from day to day, but on average are almost 2°C lower than the water temperatures.

Water temperature was measured monthly during the Bay-wide nutrient surveys (Chapter 3). These provide an indication of temperature variations around the Bay. Very detailed measurements were taken during the four intensive physical surveys – two in May and two in September 1994. Details of these measurements can be found in Svenson and Pattiaratchi (1994 a and b). Figure 2.14 shows an example of these measurements.

The Victorian Fisheries Research Institute (VFRI) has conducted a range of measurements at a number of sites around the Bay fortnightly since 1990. More details of these are given in Chapter 3. Temperature measurements from VFRI sites 1 (Blairgowrie), 4 (Wooley Reef), 6 (Sandringham), 9 (Williamstown), 11 (Werribee River) and 13 (Clifton Springs) are shown plotted over each other in Figure 2.15. As in the Hovell Pile data, the seasonal signal is obvious, with water temperatures ranging from about 21°C in summer to about 11°C in winter. There are minor temperature differences from site to site. The distribution of temperature over the Bay from May 1993 to March 1995 is shown in Plate 1.

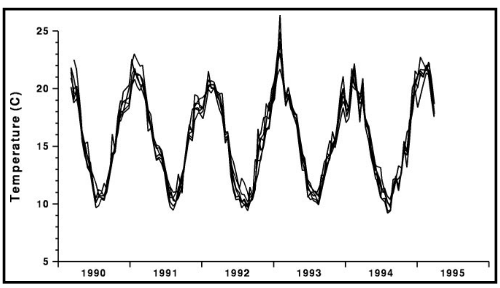

Figure 2.15: Surface temperature measured at six sites in Port Phillip Bay from 1990 to 1995.

9.2 Salinity

Salinity in the Bay is controlled by the following processes:

- Inputs of fresh water from rain, rivers, creeks and drains lower the Bay salinity. The effect is most noticeable near significant localised sources of fresh water. For example, surface waters near the mouth of the Yarra may be substantially less saline than the main bulk of the Bay. This can in turn lead to the formation of density driven currents.
- Loss of fresh water by evaporation increases the salinity of the Bay. The effect occurs over the entire Bay surface and is particularly noticeable in the Geelong Arm and Corio Bay where evaporation losses tend to exceed fresh water inputs. This area is often more saline than the rest of the Bay (and sometimes more saline than Bass Strait).
- Exchange with Bass Strait. Based on maximum salinities observed in The Rip region the salinity of northern Bass Strait varies little from a value of about 35.4 PSU. Exchange tends to change the salinity of the Bay towards this value. Mostly, the Bay is slightly less saline than Bass Strait, so that exchange tends to increase salinity in the Bay. More details on exchange can be found in Section 10.

Monthly measurements of salinity were conducted during the Bay-wide nutrient surveys (Chapter 3)(see Plate 2). These provide a good idea of salinity variations around the Bay on a monthly time scale. Detailed measurements were taken during the four intensive physical surveys mentioned above. Details of these measurements can be found in Svenson and Pattiaratchi (1994 a and b).

Figure 2.16 shows an example of these measurements. Obvious features include the Yarra plume in Hobsons Bay and the influence of Bass Strait water to the south of the Great Sands region.

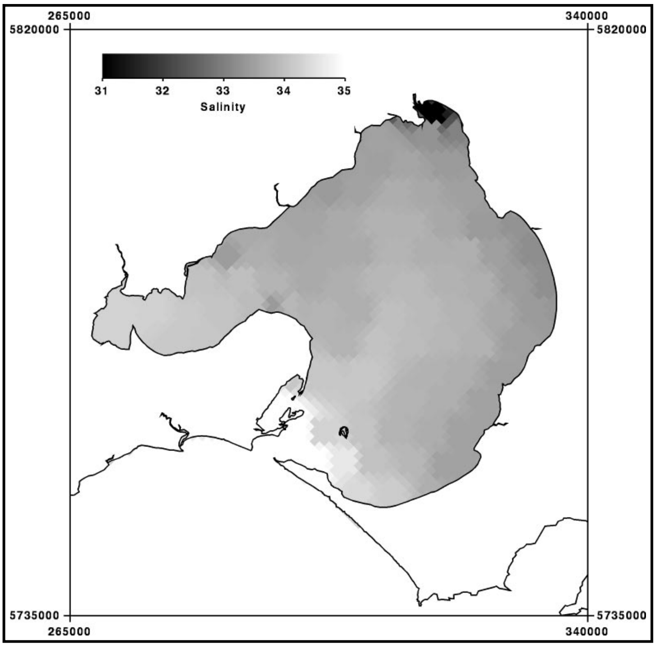

Figure 2.16: Salinity averaged between the surface and 1 m depth, interpolated from a CTD survey of 85 sites on 3-4 May 1994.

Salinity measurements were also conducted at the VFRI sampling sites. Observations for site 11 are shown in Figure 2.17.

Data from other sites show broadly similar features, but the behaviour at each site depends to some extent on the site's location in relation to river inputs and to Bass Strait.

Site 1 in the south of the Bay, for example, shows less variation in salinity than the other sites because of its proximity to Bass Strait. Sites 9, 6 and, to a lesser extent, 4 all show sudden decreases in salinity associated with the Yarra plume and east coast inputs during floods.

Bay-wide, the salinity drops markedly in late 1992 and again in late 1993 as a result of large fresh water inputs.

The year 1994 was relatively dry and salinity increased steadily due to both exchange with Bass Strait and evaporation.

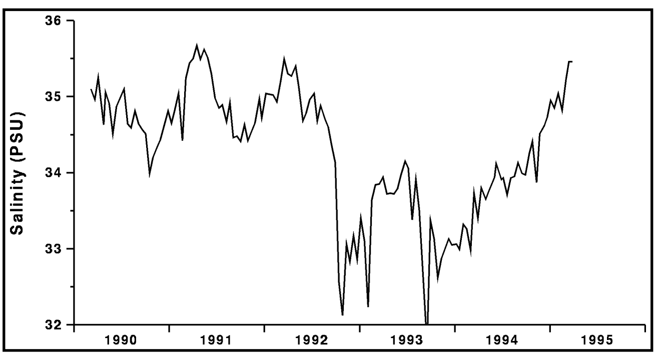

Figure 2.17: Surface salinity measured at site 11 (near the Werribee River), 1990-1995.

10. Mixing and Flushing

Understanding of the mixing and flushing processes in the Bay is essential to understanding water quality in the Bay. As stated by Hunter (1992) and Black and Mourtikas (1992), few observations of mixing processes have been carried out in the Bay.

This Study surveyed the Yarra and WTP plumes in order to better understand how these inputs mix into the Bay. Detailed surveys of the Great Sands region were also undertaken to quantify how the Bay exchanges water with Bass Strait. Descriptions of these surveys can be found in Pattiaratchi *et al.* (1995b).

10.1 Yarra plume

CTD surveys were conducted over two periods, 17-22 September 1994 and 5-6 October 1994. The survey grid contained 82 points and covered the mouth of the Yarra, Hobsons Bay and about 12 km into the north-east part of the Bay. Microstructure data were obtained at a limited number of sites during the period 19-23 September 1994.

The fresh water from the Yarra River mixes with saline Bay water in the Yarra estuary and in Hobsons Bay to form a buoyant surface plume. The plume can be readily discerned by its salinity and to a lesser extent its temperature (operation of the Newport Power Station tends to locally increase water temperature in the plume).

The plume's shape and extent depends strongly on river flow and wind speed and direction. Higher flow rates increase the extent and thickness of the plume. Northerly winds moved the plume to the western side of Hobsons Bay; during westerly winds the plume moved in a south-east direction along the coast. Southerly winds have a long fetch across the Bay to this area and so have a larger capacity for wind stirring. Mixing processes observed in the area were associated with wind mixing, interfacial shear and bottom boundary shear.

Microstructure data indicated vertical eddy diffusivity in the range 10^{-6} to 10^{-3} m^2/s, although the fact that bad weather sometimes prevented work implies that higher mixing rates sometimes probably occurred but were not observed. At the time of writing, horizontal mixing rates had not been estimated from the observed data. However, previous estimates in the Bay (near Werribee) are of the order of 1 to 5 m^2/s (Pattiaratchi *et al.* 1989).

10.2 Western Treatment Plant plume

CTD surveys and microstructure measurements were conducted over the period 23-26 September 1994.

The survey grid was near the western shore of the Bay, extending about 3 km offshore from just south of the Little River to just south of the Werribee River. Westerly winds with speeds greater than 5 m/s dominated this period.

In general, the water in the area was found to be vertically well mixed.

Horizontally, a less saline plume of water extended along the coast around Wedge Point and to the north-east towards the mouth of the Werribee River.

The eastward extent of the plume was greater than observed by Pattiaratchi *et al.* in 1989. This difference may be partly explained by the increasing use of the 15E outlet since that time (Table 2.7 in Section 8).

Vertical eddy diffusivity observed in this area again ranged from about 10^{-6} to 10^{-3} m^2/s (Pattiaratchi *et al.* 1995b).

10.3 Exchange with Bass Strait

The 'flushing time' of the Bay is of interest because it is one of the factors which determine concentrations of substances in the Bay.

It must be said that the flushing time is not a well-defined quantity. For example, a water parcel originating in the south of the Bay near The Rip will exchange with Bass Strait quite quickly (leading to the high exchange ratios found by the Phase One drogue study (MMBW/FWD 1973). A water parcel in the centre of the Bay will exchange with Bass Strait more slowly, and one in Corio Bay more slowly still.

Nevertheless, a global estimate of the flushing time can be very useful.

Hunter (1992) quoted several estimates of flushing time for the Bay ranging from about 10 to 16 months.

In this Study, CTD surveys in the Great Sands area were conducted during the period 7-11 May 1994, in conjunction with the COSRAD and ADCP measurements described in Section 4. Transects were located at the northern and southern edges of the Great Sands and along the West, Symonds and South Channels through the Great Sands.

These surveys give a detailed picture of the movement of Bass Strait water and Bay water across the Great Sands.

The two water masses were distinguished by their salinity, with Bass Strait water having a salinity greater than 35.3 PSU and Bay water having a salinity of less than 34 PSU at the time of the surveys.

Intermediate salinities were found in the Great Sands area, representing mixtures of Bass Strait and Bay water.

The area is well mixed vertically, with horizontal salinity distributions determined by the state of the tide.

At the conclusion of the flood tide, Bass Strait water extends about 6 km north and east past the entrances to the channels, but with little reaching the main body of the Bay. Similarly, little Bay water reaches Bass Strait at the end of the ebb tide.

The flushing time of the Bay was estimated from CTD and ADCP data by calculating salt and water fluxes at the southern end of the channels through the Great Sands.

Figure 2.18 shows flow and salinity at the western end of the South Channel over a single tidal cycle (observed on 10 May 1994). Flow out of the Bay is positive in this plot.

The cycle begins with Bass Strait water present at the start of the ebb tide. The salinity drops as the ebb tide progresses, indicating the presence of Bay water.

At the end of the ebb tide, the water is about 70% Bay water. The flood tide begins, and salinity rises. By half-way through the flood tide the water present is again Bass Strait water. The other channels show similar behaviour.

By carefully integrating these curves for all channels, the net amount of Bay water lost can be calculated. This calculation was done for cycles on 7 and 10 May, giving an average flushing time for these two examples of about 380 days (Pattiaratchi *et al.* 1995b). This figure compares well with previous estimates, but it is important to note that the calculation is based on data from only two tidal cycles.

It is also possible to estimate flushing time numerically using the transport model. An example is given in Section 11.

CHAPTER 2

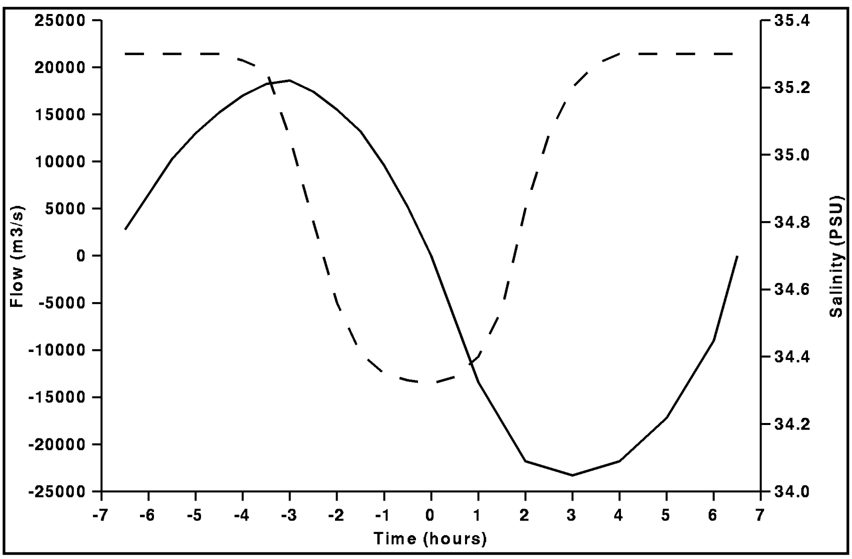

Figure 2.18: Flow and salinity observed in the South Channel on 10 May 1994. Flow (solid line) was obtained from ADCP measurements; salinity (dashed line) was obtained by CTD measurements.

11. Models

A number of numerical models were developed as part of the Study. These models provide a predictive capability, allowing the transport of water, dissolved substances and particulates to be calculated on space and time scales which extend those achievable by direct observation.

11.1 Wind Model

During the course of the study, a wind model was developed based on the existing Environmental Protection Authority of Victoria (EPAV) diagnostic wind model. The EPAV model was originally developed for urban air pollution studies in which episodes of calm or light winds are most significant. The reverse was true for this Study – strong winds are most significant for the dynamics of the Bay causing currents, waves and enhanced mixing.

The model obtained its input data from a large number of observation stations, mostly located on land. However, land-based measurements of wind generally show lower speeds than measurements taken over water. This sometimes caused the model to give unrealistic results, particularly during strong wind events when the model only showed higher speeds over the water very near to the few observation sites located on the water.

In the end, the model was abandoned and winds over the Bay were calculated using a simpler interpolation scheme. This scheme was based on observations at three sites within the Bay (Hovell Pile, Point Wilson and Fawkner Beacon) and one site on land (Point Cook – a well-exposed site with a long record of observations).

11.2 Wave Model

A commercial second generation discrete spectral wave model was implemented for the Bay on a 1,500 m resolution grid. This model was verified against Waverider buoy measurements obtained near Point Wilson (the buoy data is described in Lawson and Treloar Pty Ltd 1995).

Two verification periods were used, one from 7 to 11 May 1994 with strong easterly winds and one from 16 to 20 May 1994 with strong westerly winds. Both periods show good agreement between measured and modelled wave heights and periods.

Details of the wave model can be found in Lawson and Treloar Pty Ltd 1996.

The wave model was forced with interpolated winds and provided hourly outputs of wave heights, periods and directions over the Bay. These data were then used in conjunction with currents from the hydrodynamic model to estimate bottom stresses for the sediment transport component of the transport model.

At the time of writing this report, hourly wave fields had been calculated for all of 1994. An example is shown in Figure 2.9.

11.3 Hydrodynamic Model

A number of hydrodynamic models of the Bay were implemented during the course of the Study. The models were fully three-dimensional, non-linear and included density effects. Horizontal model resolutions ranged from 500 m to 3,000 m. Vertical resolution was generally set to 2 m.

Most simulations were performed using a model with 1,000 m horizontal resolution. This model was calibrated against data obtained during the May 1994 field period and was validated against data obtained during the September 1994 field period.

The hydrodynamic model provides three-dimensional distributions of velocity, temperature and salinity as well as concentrations of passive tracers given input fluxes of water, salt, heat and passive tracers, and forcing by winds, atmospheric pressure gradients and tides. At the time of writing this report, runs had been performed to simulate all of 1993 and 1994.

Model development, calibration and validation and results of these runs are described in detail in Lawson and Treloar Pty Ltd (1996) and Walker (1996a).

Figure 2.19 shows a comparison of low-pass filtered modelled and measured currents at the Central Bay upper meter during the May 1994 field measurement period. The measured currents are quite well represented by the model at this site. Some sites show better agreement than this, some worse.

In general, this model represents the current meters near the bottom better than those near the surface. Agreement is generally better when currents are larger (during episodes of strong winds, for example). It becomes extremely difficult to represent current directions well when speeds fall to small values. Disagreements between the measurements and model represent errors in the way the model is forced (specification of wind, for example), limitations in the way the model represents the real world (limited grid resolution, for example) and errors in the current meter measurements themselves.

The main purpose of the hydrodynamic model is to provide information to the transport model on the movements of water around the Bay. This is done by tracking a very large number of 'particles' to yield time dependent exchanges of water between the cells in the transport model.

This exchange data becomes an input to the transport model, along with bottom stress values calculated from the combined effects of bottom currents from the hydrodynamic model and waves from the wave model.

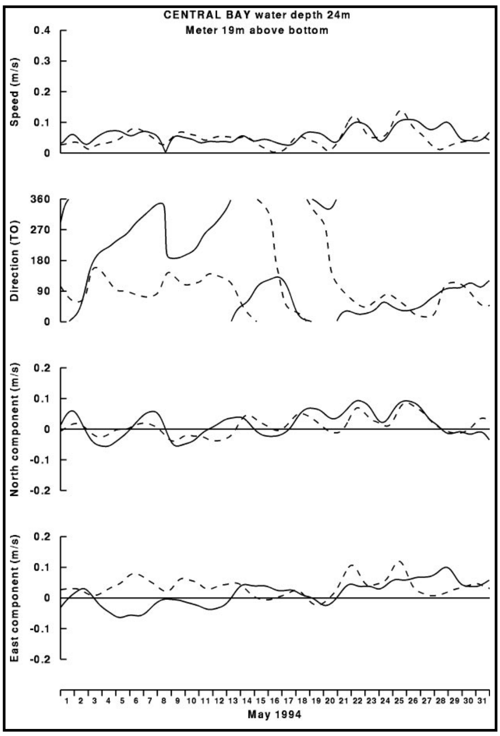

Figure 2.19: Measured (solid lines) and modelled (dashed lines) currents at the upper meter, Central Bay site, May 1994.

11.4 Transport Model

The transport model was required to simulate the movement of ecologically significant substances (nutrients, pollutants, plankton and fine sediments, for example) around the Bay. It is a generalised box model, designed to be applicable over a wide range of space and time scales.

The transport model forms the link between the hydrodynamic model and ecological models (Chapter 6).

The transport model moves substances around the Bay in accordance with water movements prescribed by the hydrodynamic model. These exchanges represent the effects of both advection and diffusion of material between cells in the water column.

The transport model implements these exchanges, accounting for the corresponding changes in concentrations of substances.

It also explicitly simulates physical processes which are not included in the hydrodynamic model, such as sources (or sinks) of materials, sediment transport, exchange of material between the water column and sediments, and mixing within the sediments. The model is described in detail in Walker (1996b).

The application of the transport model to simulate the ecology of the Bay is discussed in detail in Chapter 6.

To demonstrate flushing of the Bay, a model with 59 boxes in the Bay was initialised with a single tracer having unit concentration everywhere in the Bay and zero concentration in Bass Strait.

The model was run for two years, using physical forcing and inputs from the 1993-1994 hydrodynamic model runs.

The flushing time for each model cell was calculated as the time taken for the tracer concentration to reach 1/e (0.368) of its initial concentration. The resulting times are plotted in Figure 2.20.

Modelled flushing times vary from near zero at The Rip to about 260 days in the main body of the Bay and about 350 days in the Geelong Arm and Corio Bay.

These times are somewhat smaller than other estimates of the flushing time (Section 10) and are quite dependent on the resolution of the model, particularly in the Great Sands region.

A coarser model with seven boxes covering the Bay showed flushing times of about 105 days. Splitting the single Great Sands box in this model into two boxes increased flushing time to about 165 days.

A simple one-box inverse model has also been used to estimate both evaporation and flushing rates for the Bay, using long time series of freshwater inputs and salinities in the Bay. This model gives a value of about 400 days for the flushing time of the Bay (Walker 1996b).

As a simple test of both the hydrodynamic model and the transport model, the transport model was used to simulate salinity in the Bay for a period of two years (1993 and 1994 – the current range of hydrodynamic model output).

Figure 2.21 shows measured and modelled salinities at eight sites in the Bay over this period.

Modelled salinities are somewhat lower than observed in the Geelong Arm (Clifton Springs), probably due to the fairly uniform distribution of evaporation over the Bay used to drive the model. There is generally good agreement at other sites.

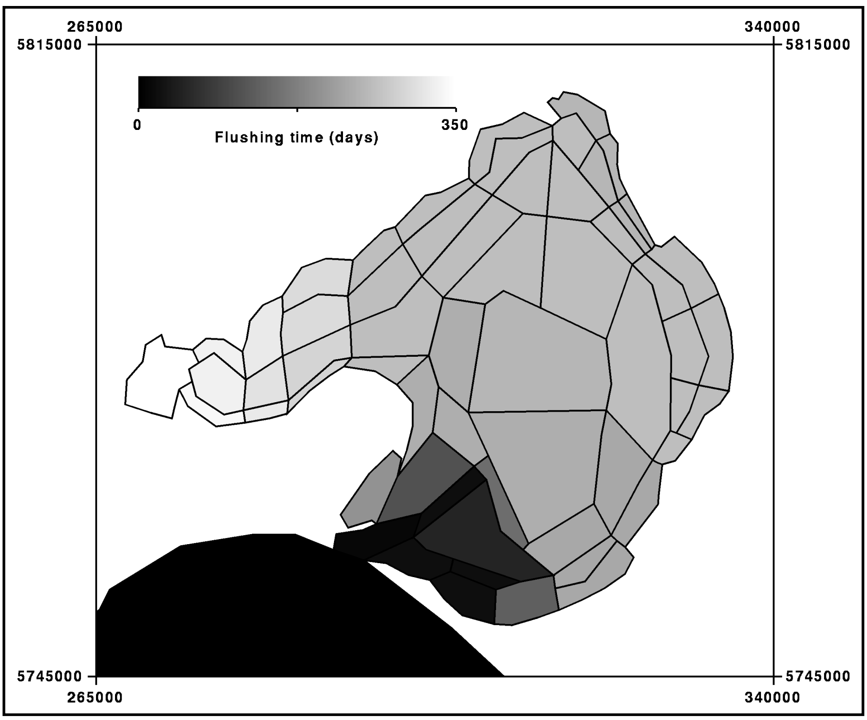

Figure 2.20: Flushing times for each box in the transport model with respect to Bass Strait.

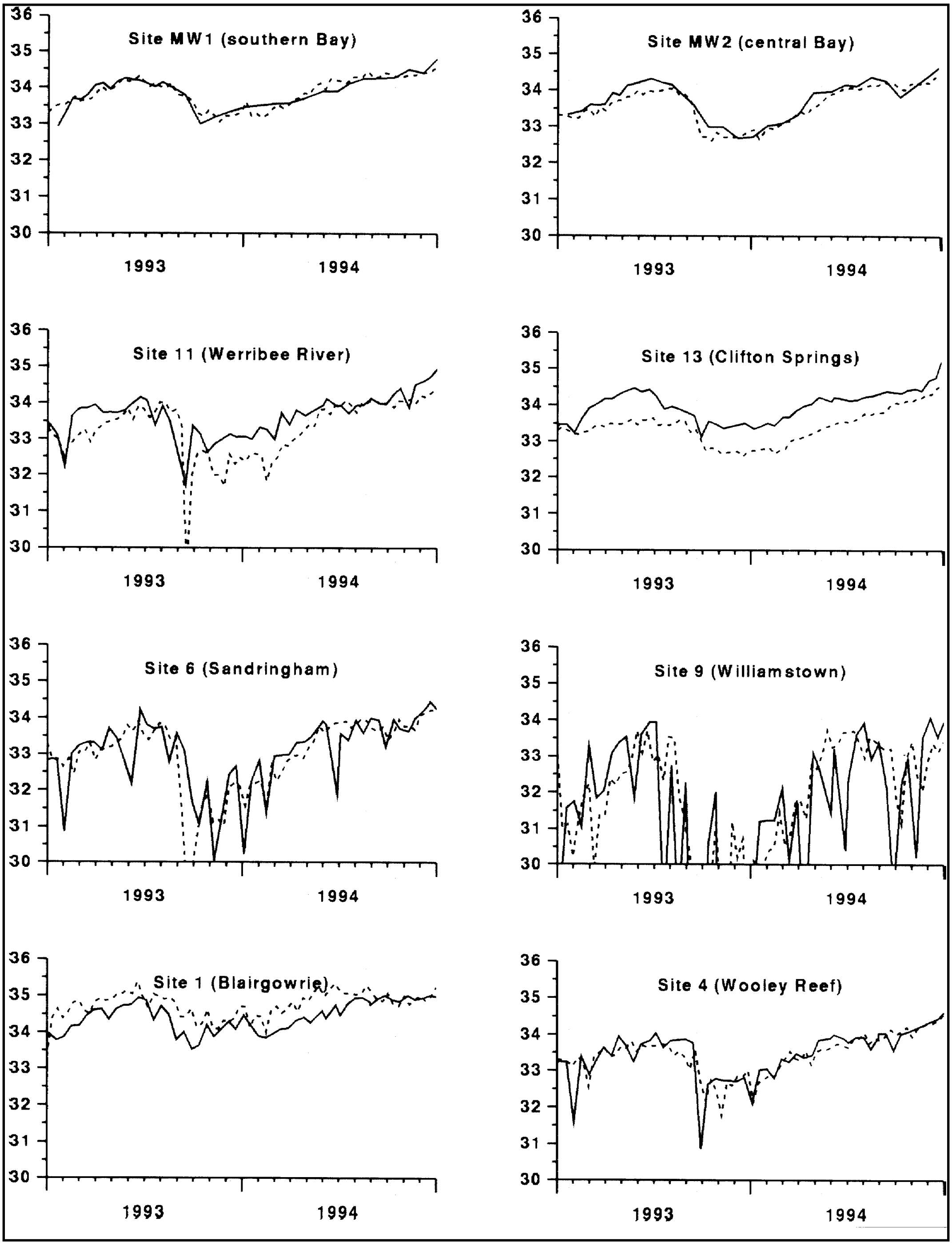

Figure 2.21: Measured (solid lines) and modelled (dashed lines) salinities for 1993 and 1994 at eight sites in the Bay. Salinity values (vertical axes) are in PSU.

Contents

1. Where do nutrients come from?

Most of the information presented in this chapter is concerned with the various chemical forms of nitrogen (N) and phosphorus (P) which are nutrients for the plants and subsequently the animals in Port Phillip Bay.

There is also some discussion of silicate (as Si) which is an essential nutrient for diatoms, the major microscopic plant types in the Bay.

Depending on bioavailability, the organic carbon compounds entering the Bay will be used as a nutrient by some of the heterotrophic bacteria.

Nutrient inputs into the Bay come from sewage treatment plants (STP), rivers and drains, groundwaters and the atmosphere. The rivers, creeks and drains convey nutrients arising from urban and agricultural runoff.

Table 3.1 gives various estimates of the amounts of P and N which each source has contributed from 1967 to 1995. Inputs from the catchment are identified by the conduit rather than the primary source because this Study had a brief to investigate only the Bay, not the catchment.

Table 3.1: Annual nutrient inputs (t) to Port Phillip Bay from all sources.

Source	1967-71[1]		1979-81[2]		1989-90[3]		1993-94[4]		1994-95[4]	
	P	N	P	N	P	N	P	N	P	N
WTP	1,083	3,931	900	4,500	1,200	3,781	1,254	3,571	1,091	3,147
Yarra & Maribyrnong		635	114	1,048		813	672	1,929	521	672
Other rivers and drains	454	2,136	374	1,636		1,246		1,689		1,218
Atmosphere						1,050		1,000		1,000
Totals	1,467	7,491	1,388	7184		7,764	1,926	8,189	1,612	6,038

Notes:
1. MMWB/ FWD, (1973).
2. NSR (1993) and Blutstein et al. *(1982)*
3. Peck and Leticq (1991) and Chiffings et al. *(1992)*
4. Sokolov (1996).

1.1 The Western Treatment Plant

Since its opening in 1896, the WTP (variously known as 'The Farm', the Werribee Treatment Complex and the Western Treatment Plant) has treated a large proportion of the wastewater and sewage from Melbourne and discharged the effluent to Port Phillip Bay.

The amount of waste treated by the WTP was reduced by about 40% in 1975 when the South Eastern Purification Plant, currently known as the Eastern Treatment Plant (ETP), at Carrum was commissioned. The treated effluent from the ETP is discharged to Bass Strait at Boags Rocks.

Accurate records of nutrient loads discharged to Port Phillip Bay from the WTP have been made only since 1967.

Figure 3.1 shows some annual N and P loads from 1967 to 1995. The dramatic effects of the ETP coming on line in 1975 are clearly evident (Chiffings *et al.* 1992).

Murray (1994) comprehensively analysed the WTP weekly loads of C, N and P to Port Phillip Bay from 1980 to 1989.

The seasonal patterns for N and P are presented in Figure 3.2 (following page).

The major points deriving from this analysis and others (MMBW/FWD 1973, Chiffings *et al.* 1992 and Skyring *et al.* 1992) are:

1. There is an annual cycle of N and P discharge from the WTP which peaks in August; the lowest discharge rates are in January.
2. Ammonia is the major N compound discharged and the highest concentrations in the effluent occur in winter and spring. Discharges range from 2 t/d in January-February to 11 t/d in July-August. The organic N discharge varies from 2 t/d to 3 t/d but with considerable scatter; there is no clear annual cycle. Nitrate discharge is about 0.4 to 0.6 t/d with no annual cycle. Nitrite discharge is a fairly constant 0.1 t/d.
3. The P discharge is greatest during the

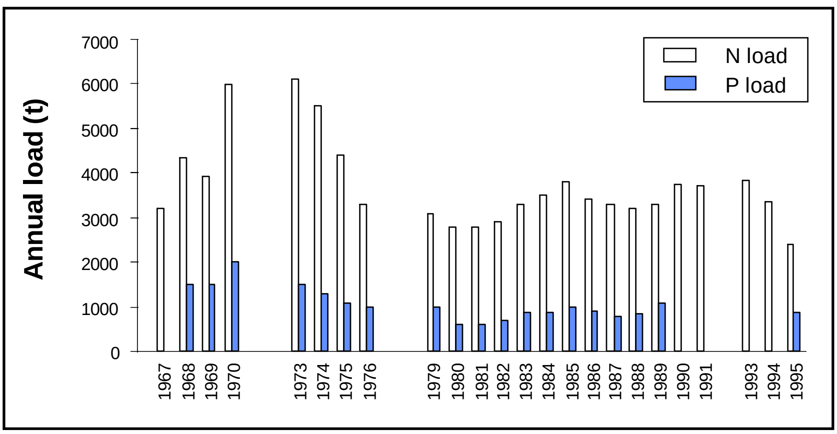

Figure 3.1: Annual N and P loads from the Western Treatment Plant 1967 to 1995 (t/yr).

winter following the same temporal distribution as ammonia. The total P loads increase from about 1 t/d during the summer to 3 t/d during the winter. The total P load increased from 600 t/yr in 1980 to 1,000 t/yr in 1989. Most (80%) of the P from the WTP is discharged as orthophosphate (PO_4).

4. The discharge of dissolved organic carbon declined from around 7,000 to 4,000 t/yr from 1980 to 1989 and has remained fairly constant since.

5. The discharge of silicate from the WTP was in the order of 800 t/yr in 1969-1970 (MMBW/FWD 1973). Best estimates from measurements in 1994 suggest the current discharge is about the same.

It is a tribute to the original 'farm' planning that the WTP has maintained a high efficiency in removing nutrients (and most toxicants) from the incoming sewage and wastewater for 100 years of operation.

The efficiencies of N and P removal from the incoming wastewater and sewage have been 65% and 63% respectively, even during periods of high input such as 1970-1975.

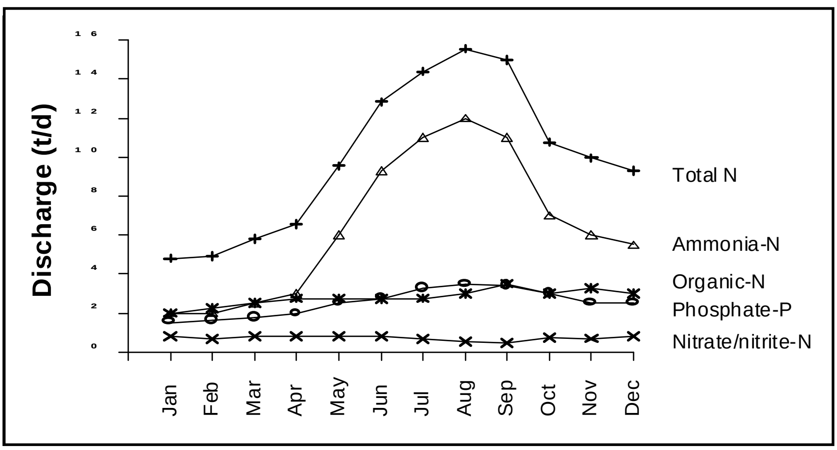

Figure 3.2: Mean monthly discharge of N and P from the Western Treatment Plant (Murray 1994).

1.2 The Yarra and Maribyrnong Rivers

The estimated inputs of nutrients (Sokolov 1996) from the Yarra/Maribyrnong Rivers into Port Phillip Bay from May 1993 to March 1995 are shown in Figure 3.3. The loads shown are calculated from a model and represent totals for the time intervals between cruises. They show the variability of inputs with time.

The flow from these rivers is seasonal and varies interannually. Peak flows usually occur from September to October.(MWWB/FWD 1973, Beckett *et al.* 1982, see also Chapter 1).

The combined runoff from the catchment for these rivers, which includes metropolitan and suburban Melbourne with a population of 3.2 million, contributes around 66% of the surface water entering Port Phillip Bay.

The nutrient loads from the Yarra estuary have a profound effect on water quality in Port Phillip Bay.

As part of the PPBES, nutrient inputs to the Bay from the Yarra/Maribyrnong River systems were measured directly by the Victorian Institute of Marine Science and Melbourne Water in 1993 and 1995 (Black *et al.* 1993a and b, Sokolov 1996).

This intensive investigation determined the net flows and loads during high (September 1993) and low (May 1993) flow periods along a transect of the Yarra estuary downstream of the West Gate Bridge.

Water samples were taken at very frequent

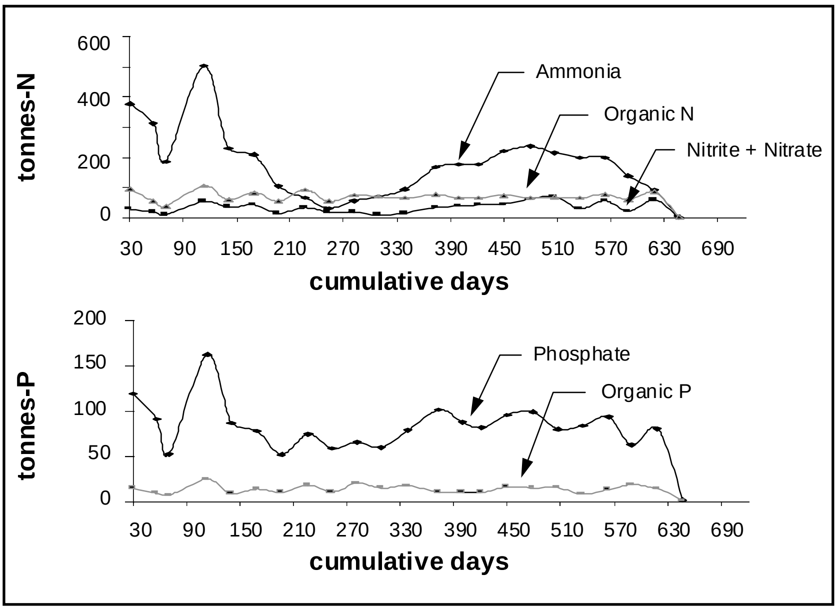

Figure 3.3: Modelled N and P loads from the Yarra/Maribyrnong, May 1993 to March 1995.

intervals (every one to two hours) at five depths and three positions across the transect. All contributing sources of the Yarra/Maribyrnong systems were thereby integrated and net loads to Port Phillip Bay could be calculated.

The exercise was repeated over two, three-month periods in 1995 with continuous monitoring of the physical parameters and weekly sampling for nutrients (Sokolov 1996).

A similar but smaller-scale study was carried out during April and May 1970 for the Phase One Study of Port Phillip Bay (MMBW/FWD 1973).

The results of the 1970 and the 1993 studies are compared in Table 3.2. It contains information about high and low flow conditions in the Yarra estuary and also about the variability in loads. These may represent change over 23 years or simply different antecedent catchment conditions. There is no obvious explanation for the very high organic loads in the September 1993 period.

Table 3.2: Nutrient loads (kg/d) in the Yarra estuary.

Input	Net Load 1970		Net Load 1993	
	Low	High	Low	High
Water (ML)	500	7,500	1,080	27,000
Ammonia – N	109	2,270	130	1,285
Nitrate/nitrite – N[1]	14	4,222	547	9,588
Total Organic N	272	4,830		64,294
Phosphate – P	257	923	107	674
Total phosphate – P		1,500		5,772
Total Organic Carbon				389,270
Si		17,000		
Solids[2]	2,497	214,288	-36,800	

Notes:
1. TON = TKN - NH_3
2. Negative sign indicates upstream transport

1.3 Minor rivers, creeks and drains

The nutrient loads of minor rivers, creeks and drains to the Bay for the period from 1979 to 1981 were estimated by NSR (1993). It should be noted that this was a dry period in which average annual flows were around 70% of normal years. The NSR estimates are given in Table 3.3.

The wastewater treatment plant at Dandenong (a major contributor to the Patterson River/Mordialloc Main Drain system) was decommissioned in 1994, resulting in estimated decreases of 30% N and 20% P being discharged into Port Phillip Bay by this route.

Although the Werribee River did not emerge from the NSR review as a significant nutrient source, Longmore *et al.* (1996) determined that the Werribee River may be a significant N source (particularly ammonia) to Port Phillip Bay during high runoff.

They estimated an input of 75-120 t ammonia-N over a flood period of a few days during September 1993 and noted that 11 of 25 of their monthly cruises from May 1993 to March 1995 indicated a recognisable ammonia input from the Werribee River.

Table 3.3: Annual inputs to Port Phillip Bay (t) from minor rivers, creeks and drains (1979-1981) (NSR 1993).

Source	TP-P	TKN-N	NO_3/NO_2-N
Patterson River	111	326	172
Mordialloc Main Drain	90	229	111
Kororoit Creek	56	171	74
Mordialloc Settlement Drain	37	133	34
Kananook Creek	32	65	14
Dunlops Drain	9	28	13
Saltworks	5	16	12
Heatherton Road Drain	4	15	9
Laverton Creek	3	10	7
Chinaman's Creek	3	10	6
Others (189 points)	2	118	77
Werribee River & WID	–	–	9
NSR Annual Total	352	1,121	538

1.4 Groundwater

Estimated nutrient inputs into Port Phillip Bay via groundwater are small compared to the inputs from surface waters; the information in Table 3.4 records the most recent estimates by HydroTechnology Pty Ltd (1993) and Otto (1992).

While the present load of N and P nutrients from groundwater may not be large, the concentrations of nitrate in some aquifers are particularly high (1 to 200 mg/L) and both reviewers predict a minor rise in nutrient load via groundwater as the nutrient fronts resulting from historical inputs move slowly to discharge into Port Phillip Bay.

1.5 Atmospheric inputs

Airborne nutrients can be introduced to the Bay by wet or dry deposition.

Wet deposition results from the scavenging of gases or particulate matter by water droplets in clouds or by rain, fog or snow. Dry deposition is the fallout of gaseous or particulate matter at the earth's surface. Dry deposition is less efficient than wet deposition but its effects are continuous.

The deposition of ammonium and nitrate-N into Port Phillip Bay from the atmosphere is currently estimated at 800 to 1,300 t N/yr (Carnovale *et al.* 1992). This is a very large amount considering that the total N input to Port Phillip Bay is only 6,000 to 8,000 t/yr (see Chapter 6). As the Melbourne population increases, nitrogenous contamination of the atmosphere will also increase. The estimated N load to Port Phillip Bay in 2005 will be from 900 to 1,600 t/yr (Carnovale and Saunders, 1988).

Carnovale *et al.* (1992) did not estimate an atmospheric P load to Port Phillip Bay because of the lack of local measurements but they calculated that the atmospheric contribution of P would be marginal. Episodic events of wind-blown dust may add significant amounts of P to the Bay.

Table 3.4: Annual nutrient inputs (t) from groundwater to Port Phillip Bay.

Source	Otto (1992)		HydroTechnology (1993)	
	P	N	P	N
North-east Werribee Delta	8	34	1-3	10-24
South-west Werribee Delta			3	7-22
Nepean Peninsula		0. 3	17	19
Western Suburbs			2	12-17
TOTAL	8	34	Max 25	Max 82

2. Where do nutrients go in the Bay?

2.1 Soluble nutrients

When the soluble compounds of nitrogen, phosphorus and silicate are discharged into Port Phillip Bay, they are distributed and rapidly mixed into the very large volume (26.3 km^3) of water (Chapter 2).

The mixing processes in Port Phillip Bay take various forms. For example, Longmore *et al.* (1996) illustrate mixing during periods of high runoff by plotting nutrient concentrations against salinity obtained during short-term cruises off the north coast of the Bay. Each salinity/nutrient data set was identified by a unique location and, because of this, dilution curves could be generated that identified mixing gradients and the sources of the nutrients.

Figure 3.4 shows a group of 'end member' plots for phosphate, ammonia, nitrate and silicate, identifying the WTP, the Yarra estuary and the Werribee River as significant nutrient sources. The lines in Figure 3.4 are not regression curves but merely link the extreme observations recorded. A full description is given in Longmore *et al.* (1996). Nutrient discharges from the eastern side of Port Phillip Bay were not detected.

An important objective of the PPBES was

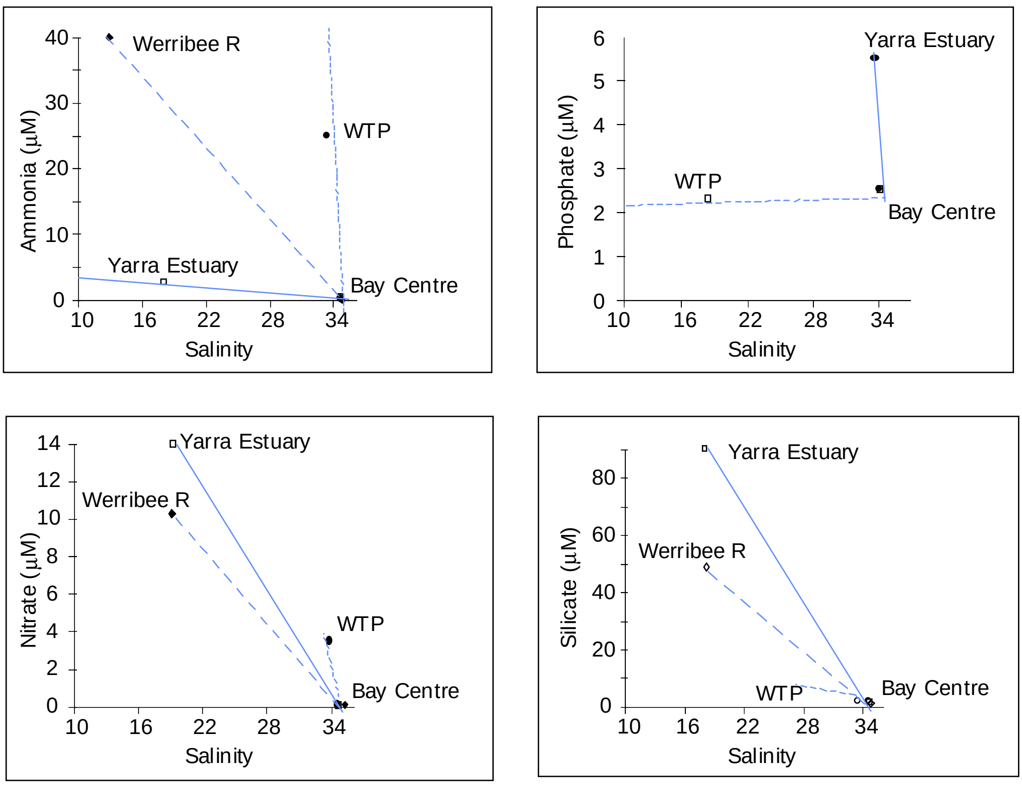

Figure 3.4: Identification of multiple nutrient sources from salinity/nutrient gradients (Longmore *et al.* 1996).

to determine the current nutrient status of Port Phillip Bay.

The results of this large task are reported by Longmore *et al.* (1996), who measured the concentrations of a suite of nutrient and chemical components at monthly intervals (25 cruises) from May 1993 to March 1995. This was achieved using a system of continuous shipboard chemical analysis of surface waters and a fixed station program for surface, mid and bottom waters.

Variations in nutrient concentrations with depth were usually the result of fresh water inputs floating on the denser seawater of the Bay or fluxes from the sediments into bottom waters. Stratification of the water column was most noticeable in the Yarra estuary, Hobsons Bay and in the waters offshore from Werribee. A typical cruise path and the fixed positions are shown in Figure 3.5.

This work provided a very large database for ammonia, nitrate, nitrite, phosphate, silicate, chlorophyll *a*, dissolved oxygen, salinity and temperature.

Similar measurements were made over much shorter periods (four hours to five days) with several circuits of a cruise path off the Werribee coast and Hobsons Bay.

Both these studies provided spatial and temporal distributions over varying scales for water quality characteristics.

A valuable complementary database giving long-term interannual variations, was available from water quality monitoring at six fixed sites every 14 days from February 1990 to March 1995. The sites were Clifton Springs, Werribee, Williamstown, Sandringham, Wooley Reef and Blairgowrie (Figure 3.5).

The total of Longmore's data and analyses

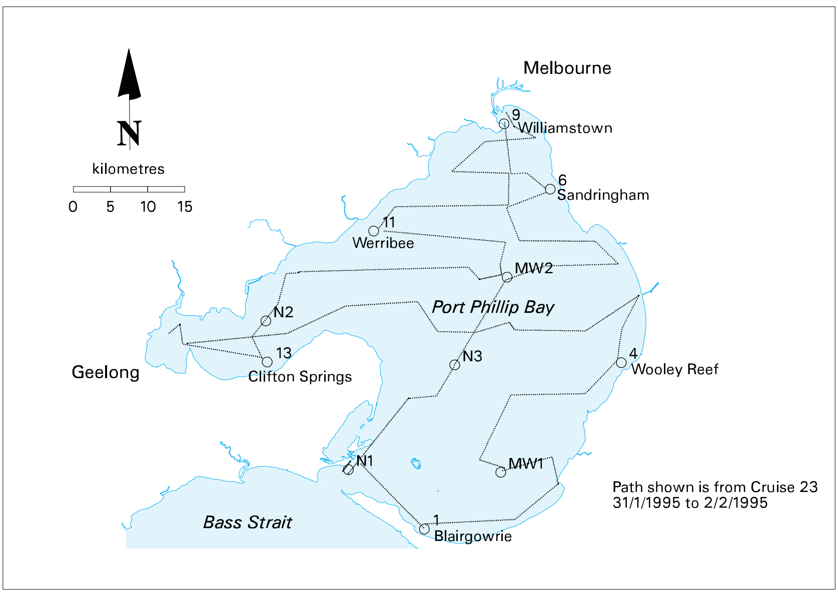

Figure 3.5: A typical cruise path and station locations from the water nutrient task [N2.1] (Longmore *et al.* 1996).

(Longmore *et al.* 1996) are too numerous to include but Plate 3 shows examples of statistically derived concentration contours for the waters of Port Phillip Bay after a period of flood rains in 1993 and during the 1994-1995 drought; distributions of the biennial averages are also presented.

Table 3.5 shows the amounts of nutrients in Port Phillip Bay waters for each of three periods of measurement (1969-1970, 1980-1984, 1993-1995).

It is immediately apparent from the entries in Table 3.5 that all pool sizes have decreased since 1969-1970. The pool of P is about the same size as the input load (Table 3.1) but the total N pool is now about half of the input load. The simplest interpretation of these data is that:

(i) P is in an apparent steady state with the pool size reflecting exchange, relocation and transformations due to physical and biogeochemical processes.

(ii) N is involved in biological and biogeochemical processes which deplete the soluble pool faster than the exchange process with Bass Strait.

These grand averages must be taken with some caution because there are several constraints on extrapolating from spot measurements to Bay-wide pool sizes. There is further discussion of these data in sections 3.4 and 3.5.

Table 3.5: Bay-wide annual average pool sizes (t), 1969-1995 (Longmore *et al.* 1996).

Nutrient	1969-1970		1980-1984	1993-1995
Ammonia – N	1,250[1]	358[2]	140	163
Nitrate+Nitrite – N	123		91	90
Total – N	8,505		4,165	3,025
Phosphate – P	1,891		1,473	1,460
Silicate – Si	3,640		2,520	3,303
Chlorophyll *a*	33		24	21

Notes:
1. This was an overestimate due to the inappropriate method used.
2. This estimate was calculated from the averages of ammonia as a percentage of Total-N for the periods 1980-1984 and 1993-1995.

2.2 Particulate nutrients

Nitrogen, phosphorus and silicate also enter Port Phillip Bay as particulates, both organic and inorganic. For example, the export of particulate N (as Total-N less the ammonia and assuming that DON is small) from the Yarra estuary during the high flow period of September 1993 was around six times the amount of inorganic N and total P was nearly 10 times the amount of soluble orthophosphate (Table 3.2). Because of the long residence time and the topography and size of Port Phillip Bay, the export of particulates to Bass Strait is negligible.

The transport, distribution, settling and resuspension of particulate material in coastal embayments like Port Phillip Bay are complex processes. They are considered in more detail in Chapter 2 and Walker (1996b).

2.3 Soluble nutrients in sediment porewaters

Nicholson *et al.* (1996) measured the porewater nutrient concentrations in integrated layers at seven sites representing the major sedimentary types in Port Phillip Bay. An example of the downcore nutrient distributions in the sediments of the centre of the Bay is given in Figure 3.6.

The soluble N, P and Si in the sediment porewaters do not get there by passive diffusion from the water column because the concentration gradients are from the sediment to the water. All the nutrients have downcore distributions which indicate two important processes:

(i) production in the sediment and

(ii) bioirrigation.

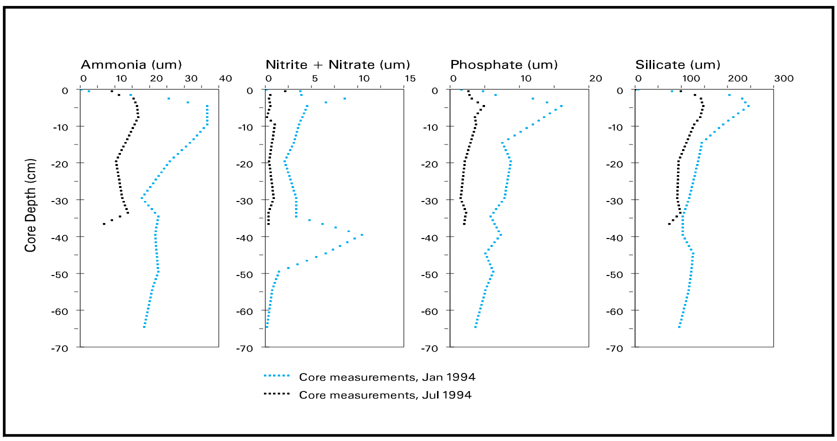

Figure 3.6: Mean porewater depth profiles of ammonia, nitrite plus nitrate, phosphate and silicate for summer and winter taken at site 37, central Port Phillip Bay (Nicholson *et al.* 1996).

The important effects of bioirrigation and bioturbation on nutrient dynamics in Port Phillip Bay are discussed in Section 3.1.

The pool sizes of dissolved and solid phase nutrients in the four zones of the Bay distinguished by Nicholson *et al.* (1996) are shown in Table 3.6. The Bay-wide pool size for ammonia and nitrate/nitrite in porewaters (0-20 cm) was found to be around 160 t N, which is slightly less than that in the water.

The amount of inorganic orthophosphate-P is approximately 125 t, a very small amount compared with the average water column pool size of about 1,400 t.

The DON and DOP are around 18% and 10% of the DON and DOP, respectively, in the water column (see Table 3.5).

The bioavailability of the dissolved organic compounds in the porewaters of Port Phillip Bay sediments is not known.

Solid C, N and P contents continue below 20 cm depth so that most organic matter found in Bay sediments must be relatively refractory.

The pool of soluble Si in the porewaters (1,140 t Si) is around 30% of the annual average in the water.

Table 3.6. Mean pooled values from the sediment of dissolved nutrients (DIN, DIP, DON, DOP) and solid phase nutrients (SedN, SedP and SedC), in tonnes, from all regions and the Bay-wide total in PPB to a depth of 20 cm (Nicholson *et al.* 1996).

Region (km^2)	DIN	DIP	DON	DOP	SedN	SedP	SedC
Central (506)	25	20	60	1.5	40,000	14,000	480,000
Northern (240)	60	45	130	3	29,000	12,000	315,000
Surround (919)	45	25	150	3	37,000	30,000	270,000
Western (284)	30	35	120	3	41,000	15,000	360,000
Baywide (1949)	160	125	460	10.5	147,000	71,000	1,425,000

3. What happens to nutrients in the Bay?

When the soluble inorganic forms of N and P enter Port Phillip Bay they are immediately available for the flora as nutrients. Some of this nutrient is transferred to the resident fauna via the food chains.

Some of the organic forms of N and P entering the Bay from STPs, rivers and drains may be assimilated directly by the Bay's biota, but it is probable that most undergoes microbial mineralisation to simple inorganic forms of N and P. Silicate is assimilated only by the diatomaceous phytoplankton and microphytobenthos.

These biological processes are discussed in detail in Chapter 6.

Port Phillip Bay is shallow and generally well mixed with a continuous but slow exchange with the waters of Bass Strait, thereby giving adequate time for the nutrients to be converted into phytoplankton biomass. All other biological and biogeochemical processes in the Bay are dependent on this photosynthetic process. From the integrated work of the PPBES there is now a comprehensive picture of the biological and biogeochemical processes which keep the Bay waters in an oligo- to mesotrophic (low nutrient) condition.

The key process is bacterial denitrification in the sediments by which means nitrogen is removed from the trophic cycle. The major biological and biogeochemical processes, shown in Figure 3.7, are:

1. Ammonia, nitrate, silicate and some of the phosphate entering the Bay are taken up rapidly by phytoplankton in the water column and by the microphytobenthos (MPB).
2. The nano- and picoplankton and the

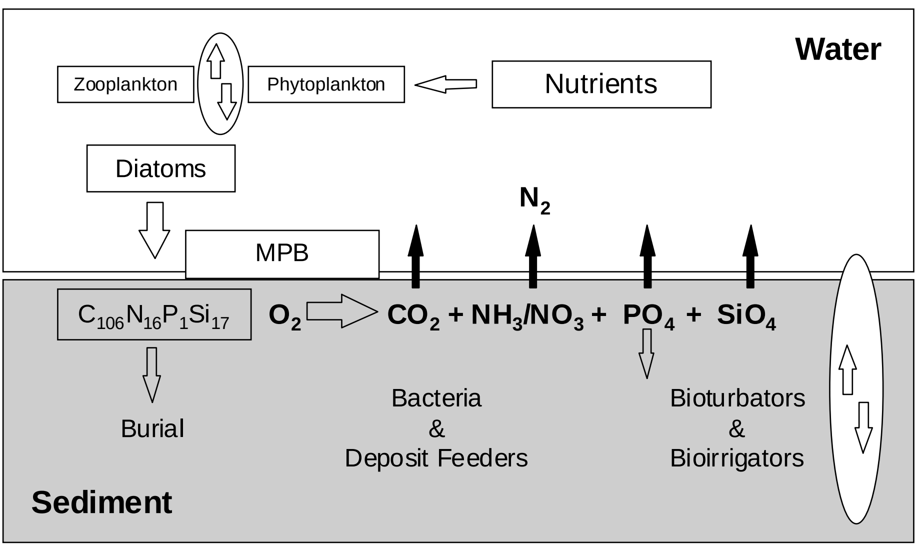

Figure 3.7: A process for nutrient dynamics in Port Phillip Bay.

zooplankton grazers rapidly recycle nutrients in the water column. It is the diatoms (microplankton and MPB) which constitute the major organic input to the sediment. Diatoms are made dense by their silica frustules causing them to sink.

3. The diatoms (including those in the microphytobenthos) are degraded in the sediments by a variety of organisms, resulting in the uptake of oxygen and the production of carbon dioxide-C, ammonia-N, phosphate-P and silicate-Si in the same atomic ratio as the diatomaceous algae.
4. The net metabolic activity in the sediments is due almost entirely to bacteria. Nitrifying bacteria in the sediments oxidise some ammonia first to nitrite and then to nitrate. Denitrifying bacteria then convert the soluble nitrate to relatively insoluble and inert nitrogen gas (N_2). The nitrification/denitrification processes are very tightly coupled.
5. The silica frustules, freed from cell substance, are dissolved rapidly and silicate is not precipitated in the sediments.
6. Some sedimentary fauna bioturbate and irrigate the upper layer of the sediments, providing active aeration and a flushing of the soluble nutrients back into the water column. At the same time, refractory or very stable compounds may be transferred into the sediment by this process from the water column.

3.1 Discoveries: denitrification, silicate cycling and bioirrigation

The single most important discovery of the PPBES is that denitrification in the sediments prevents soluble and available N (ammonia and nitrate) from accumulating in the water column because it is removed from the main Bay environment as nitrogen gas (N_2).

By contrast, in the sediments of the Yarra estuary and Hobsons Bay, ammonia, not nitrate, is the major N product of organic breakdown. N is returned to the water column as ammonia providing N for a new round of phytoplankton production.

The recycling of the ammonia between the sediments and the water column may occur over and over again.

If this recycling of ammonia were to happen Bay-wide, the ecosystem would rapidly degrade into a eutrophic state. The exchange of water with the ocean via Bass Strait would be too slow to prevent an accumulation of N in Bay waters and there is an ample supply of phosphate to sustain high rates of phytoplankton productivity.

This aspect of nutrient dynamics in Port Phillip Bay is dealt with in more detail in Chapters 6 and 7.

A key feature disclosed by PPBES is bioirrigation of the sediments with oxygen rich water through the activities of benthic invertebrates. This irrigation process may increase the rates of nutrient transformations and also the fluxes from the sediments by an order of magnitude or more over simple diffusive fluxes (Berelson *et al.* 1994 and 1996, Nicholson *et al.* 1996).

One of the important results of bioirrigation is to carry dissolved oxygen to sites deep in the sediments (at least to 50 cm) thus facilitating the aerobic degradation of organic material carried into the sediments by burrowing and grazing animals.

Nicholson *et al.* (1996) show that silicate fluxes are:

(i) always enhanced by bioirrigation,

(ii) greater at all sites during the warmer months of the year and

(iii) higher in the Yarra estuary and coastal Werribee sediments than in the central basin sediments.

They also show that ammonia fluxes from the Yarra estuary and coastal Werribee sediments were around 10 times the fluxes from the central Bay sediments.

Although the bioirrigators are very active in Yarra and Werribee sediments, they cannot keep up the supply of dissolved O_2 to promote high denitrification efficiencies.

The PPBES has, for the first time, demonstrated the prime importance of sedimentary processes in keeping the waters of the Bay clean. The following sections describe the important results and interpretations of the extensive studies which have enabled this new and dramatically different conceptual model of nutrient dynamics in Port Phillip Bay to be developed.

3.2 Sedimentary processes

The investigation and quantification of sedimentary processes in Port Phillip Bay was done by placing electronically controlled benthic chambers on the surface of the sediment over periods from 4 to 24 hours.

Fluxes (to and from the sediment) of oxygen, carbon dioxide, ammonia, nitrite/nitrate, dissolved organic nitrogen (DON), N_2, phosphate, and silicate were measured.

Bioirrigation was also quantified by measuring the radon flux from the sediment.

These important studies have been reported by Berelson *et al.* (1994) and by Nicholson *et al.* (1996).

Direct measurements of N_2 fluxes were made by Berelson *et al.* (1996). Important complementary studies were carried out on biomarkers (O'Leary *et al.* 1994) (Plate 4), sedimentation rates (Chapter 2) and bioturbation (Bird 1994, Bird *et al.* 1996).

Figure 3.8 shows the kind of results which were obtained using the benthic chambers. Fluxes are calculated from the rates of uptake or production.

All of the fluxes are calculated from the initial rate (i.e. the graphs were extrapolated to zero time). The major conclusions of the benthic chamber measurements and the complementary studies are:

1. The sediments have a daily net oxygen uptake. Oxygen and carbon dioxide fluxes in summer are generally two to three times higher than in winter. The stoichiometry of oxygen uptake and carbon dioxide output (138:106) is consistent with the degradation of diatomaceous plant material according to the Redfield equation (Redfield *et al.* 1963, Richards 1958). Specific biomarker measurements (O'Leary *et al.* 1994) and long-term phytoplankton studies (Magro *et al.*

CHAPTER 3

1996) show that diatoms are the predominant phytoplankton in Port Phillip Bay (Plate 4).

2 The sum of the ammonia and nitrite/nitrate flux is much less than that expected from Redfield stoichiometry. This is due to bacterial denitrification. Direct measurements of denitrification (using N_2:Ar ratios) confirm that high denitrification rates occur in the sediments of Port Phillip Bay. The sequence of biochemical steps leading to denitrification are: organic-N to ammonia (NH_3) to nitrite (NO_2) to nitrate (NO_3) to dinitrogen (N_2) (Figure 3.7).

Figure 3.9 (following page) shows the theoretical and measured N_2 fluxes.

Table 3.7 shows the variation of denitrification rates in the Bay.

In the Yarra estuary and Hobsons Bay, the ammonia flux was often consistent with Redfield stoichiometry (106C:16N) indicating that, in these areas, nitrification and denitrification did not occur to any significant extent. The nitrogen transformation stops at:

organic-N $\Rightarrow$ ammonia (NH_3)

3. DON fluxes were difficult to measure because of the small concentration differences but the data suggest that the

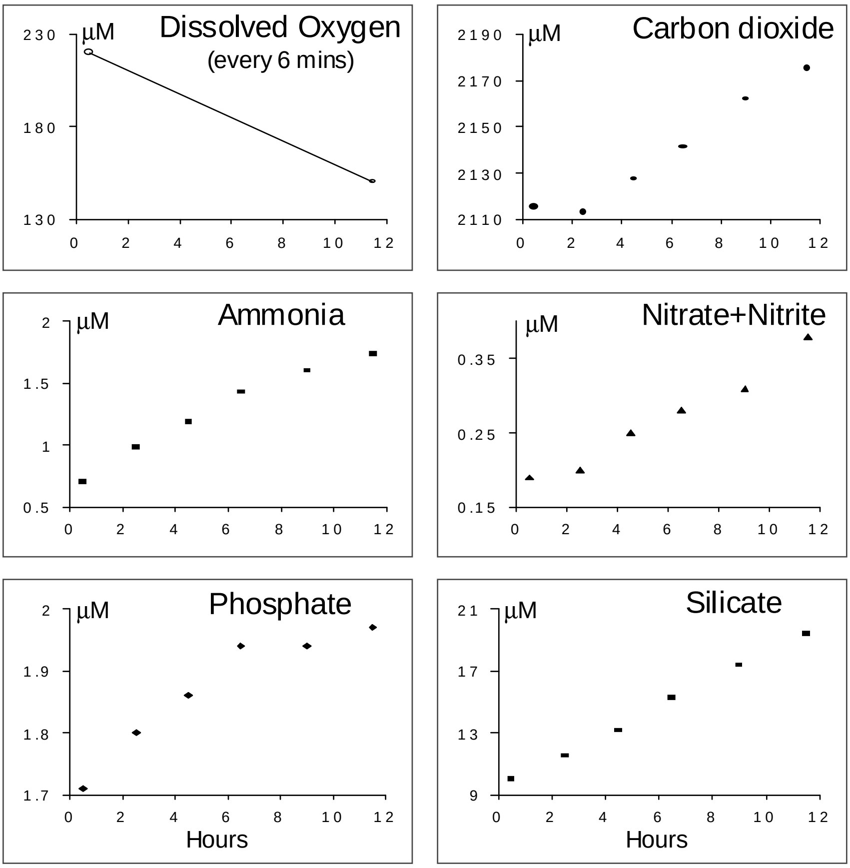

Figure 3.8: An example of benthic measurements over a 12-hour period for central Bay sediments, January 1994 (Berelson *et al.* 1994).

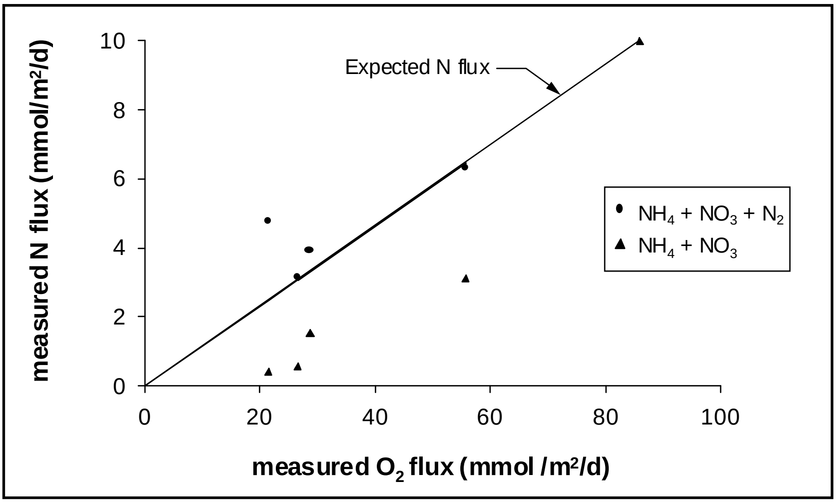

Figure 3.9: An example of nitrogen balance in benthic chambers. Elemental N_2 was measured directly. Expected N flux is from the Redfield ratio 138 O_2:16 N.

sediments are not a major source of DON to the water column.

4. The P flux was 30 to 90% of that expected from Redfield stoichiometry so that some P may be precipitating in the sediment. The Yarra estuary and Hobsons Bay sediments show different behaviour and may be sources of phosphate.
5. The fluxes of naturally occurring radon indicated that bioirrigation is an active process in all sediments.
6. Sedimentation is slow (probably much less than 1.5 mm/yr) and bioturbation dominates over settlement. N retention in the sediments is not yet quantified satisfactorily (see Chapter 6).
7. The ratio of silicate efflux to oxygen uptake indicates a constant C:Si ratio in the diatomaceous material in the sediments.

All these observations are pertinent to understanding nutrient dynamics, but it is the discovery of the quantitative importance of denitrification, the unique role of the diatoms and the importance of bioirrigation which provides the key to understanding the carbon, nitrogen, phosphorus and silicate dynamics and budgets in Port Phillip Bay.

Table 3.7: Denitrifying efficiencies in the major sediments of Port Phillip Bay.

Sediment	Area (km²)	Denitrifying Efficiency (%)
Central Basin	800	90-100
Northern Silty Sand	240	0-30
Coastal Sandy Gravel	475	50-70
Corio Bay	200	85

3.3 Primary and secondary producers: N pool sizes

The assimilation of nutrients into biomass of the primary producers constitutes a partial diversion into a standing-stock pool.

Table 3.8 shows the amounts of N contained in the major biomasses in Port Phillip Bay.

Table 3.8: Nitrogen pool sizes (t) in the primary and secondary producers in Port Phillip Bay.

	Average N-pool sizes
Phytoplankton[1]	210
Microphytobenthos[2]	380
Macrophytic Algae[3]	600
Benthic Fauna[4]	3,000

Notes:
1. Calculated from the monthly Bay-wide averages estimated by Shao and Fox (1996) from measurements made by Longmore et al.*(1996).*
2. Beardall and Light (1996).
3. Chapter 6.
4. Wilson et al. *(1993).*

3.4 Productivity

Productivity by the phototrophs and the secondary producers has a much greater significance than the nutrient pool sizes of the various biomasses with respect to understanding nutrient dynamics in Port Phillip Bay.

For example, the phytoplankton in the water column of Port Phillip Bay 'turn over' around 40,000 t N/yr and the microphytobenthos around 21,000 t N/yr (Chapter 6).

A total of 61,000 t N/yr is required to support the phytoplankton and the MPB alone. Yet only 6,000 to 8,000 t N is discharged into the Bay each year and we now know that most of this is lost as N_2 by denitrification in the sediments.

This seems an insoluble paradox, but the N balance is maintained by rapid recycling, which is discussed in more detail in Chapters 6 and 7. This rapid recycling system results in tight coupling of processes. For example, Figure 3.10 shows the positive linear relationship between the productivity of the phytoplankton and the activity of the benthic community in Port Phillip Bay (Nicholson *et al.* 1996).

Another important observation of the PPBES may be that high phytoplankton productivity in the Yarra estuary and Hobsons Bay occurs with lower denitrifying efficiencies in the sediments (Nicholson *et al.* 1996).

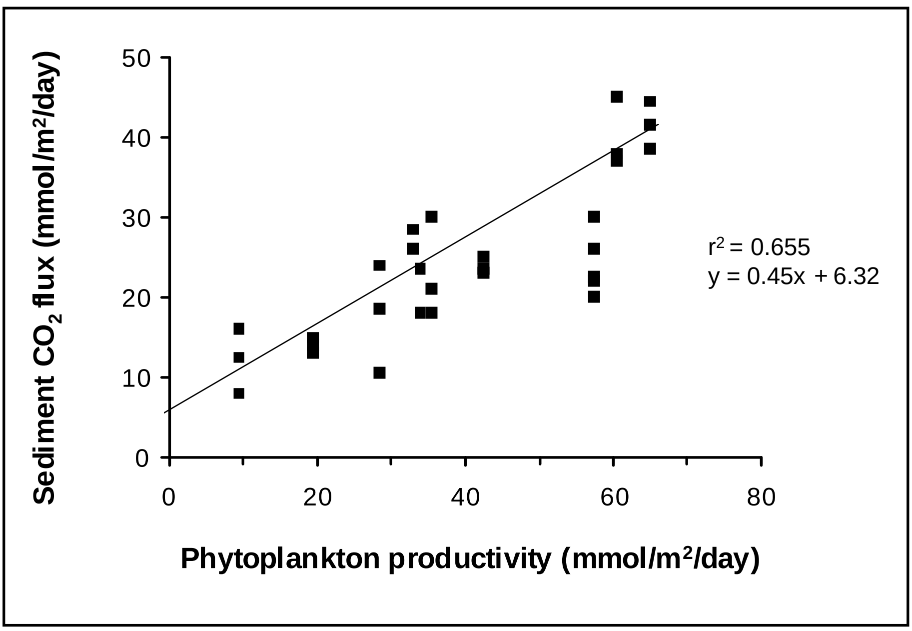

Figure 3.10: Carbon dioxide flux from the sediments versus phytoplankton productivity in the water (Nicholson *et al* 1996).

3.5 Bay-wide processes and budgets

Although the spatial distribution maps compiled by Longmore *et al.* (1996)(Plate 3) and Figures 3.3 and 3.4 show the large inputs of materials from the catchment during heavy run-off, these inputs are eclipsed in magnitude by the nutrient cycling processes referred to in the previous section.

Figure 3.11 shows the calculated pool sizes of N, P, Si and chlorophyll *a* for the period of monthly sampling, May 1993 to April 1995.

The fluctuations in pool sizes represent the net result of varying catchment inputs and cycles of phototrophic and secondary producers in the Bay system.

The dramatic changes in Si pool size may be contrasted with those of P. Only a small proportion of the P pool is involved in the Bay production system whereas the Si pool is at times almost totally contained in the diatom population.

These complex relationships will be further interpreted and reported elsewhere, but a brief illustration of a Si relationship is given below.

Relating water column productivity by the phytoplankton and microphytobenthos, and the benthic Si flux measurements to Bay-wide rates is problematic because of the uncertainties of the areas (in km^2) of sediments represented by the experimental sites and also the

CHAPTER 3

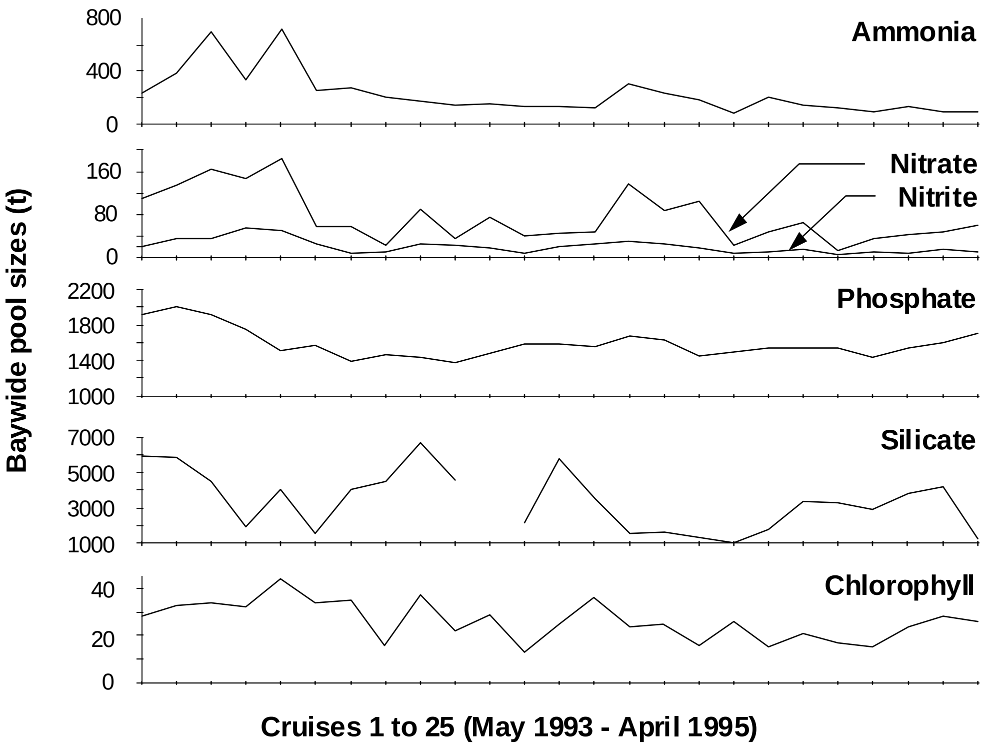

Figure 3.11: Bay-wide pool sizes of nutrients and chlorophyll, calculated from monthly cruise data, May 1993 – April 1995 (Longmore *et al.* 1996).

CHAPTER 3

Table 3.9: A comparison of measured and estimated silicate fluxes, 1994-1995.

Period		Silicate data[a]				Date	Si chamber flux[b]
Dates	Days	µM start	µM finish	µM difference	Flux mmol/m²/d		mmol/m²/d
4-18/1/94	14	6.8	10.1	3.3	2.9	10/1/94	2.7
12-25/10/94	13	2.6	3.8	1.2	1.2	11/10/94	0.8
16-27/2/95	11	6.1	8.2	2.1	2.4	23/2/95	3.1
28/8-11/9/95	13	3.4	2.7	-0.7	-0.7	31/8/95	2.9
15-30/1/96	15	10.5	10.8	0.3	0.3	23/1/96	2.8

Notes:
a. The silicate data are from the Wooley Reef fixed site (Longmore et al. 1996) because the silicate analyses were done every 14 days; the silicate concentrations for the Wooley Reef site were almost identical to those for the central station (MW2) for the same sampling periods. The silicate fluxes were calculated from the difference data, the period in days, and the total volume and area of the Bay.
b. Data from Berelson et al. 1994 and Nicholson et al. (1996).

actual duration of the measured rates.

Because of the predictable nutrient stoichiometries established from the benthic chamber measurements, silicate fluxes in the central waters may be used to compute Bay-wide nutrient fluxes for long periods.

For example, the fluctuating silicate concentrations measured at the Wooley Reef station every two weeks should be generally compatible with measured chamber rates and the results in Table 3.9 show this to be true for only three periods out of five.

A more precise process model for silicate in the Port Phillip Bay sediments is required.

3.6 Balancing the budget

In balancing the nutrient budget for Port Phillip Bay there are three major accounts; nitrogen, phosphorus and silicate. Balancing for each of these accounts requires detailed estimates for total inputs (point and diffuse sources from the catchments, atmospheric and groundwater sources), unrecoverable losses (export to Bass Strait, burial in the sediments and losses to the atmosphere) and temporal perturbations in the standing biomass of the flora.

3.6.1 The nitrogen budget

Balancing the N budget is extremely difficult because it is involved in so many biological, biogeochemical and physical processes.

Taking into account measured primary productivity rates of the phytoplankton and MPB, and the rates of remineralisation and denitrification, we conclude that the Bay-wide N budget is balanced by denitrification and export of about 1,000 t/yr DON to Bass Strait.

The N in the Bay may pass through the biomasses and biogeochemical pathways many times before it is finally converted to N_2 by the denitrifying bacteria in the sediments. For example, around 210 t N is cycled through the phytoplankton every two days and around 370 t N is cycled through the diatoms of the MPB every eight days but, eventually, their biomass is degraded and the N (as nitrate) arrives at a denitrifying site in the sediments.

The measured denitrifying efficiencies have a grand average of 50% (with the range shown in Table 3.7). Model calculations require a theoretical denitrification efficiency of 65% to balance the N budget. The net annual N budget is now balanced.

Chapter 6 presents a quantitative theoretical treatment of the N budget.

3.6.2 The phosphorus budget

Like N, P passes through many biomasses and biogeochemical and chemical pathways during residence in Port Phillip Bay and it is subject to the same temporal perturbations.

Unlike N, P is not biologically converted to a volatile product and it is involved in a more diverse range of chemical reactions involving precipitation and remobilisation, as well as turnover through phototrophic biomasses.

The net P budget is balanced by geochemical precipitation and burial within the sediments and exchange with Bass Strait.

3.6.3 The silicate budget

The silicate budget should be relatively easy to balance because loads in river runoff can be calculated from the flow and concentrations and the loads from the WTP are also known.

Silicate is taken up only by the diatomaceous phytoplankton and MPB and the solubility of free frustules ensures that the sediments of Port Phillip Bay are not permanent sinks for silica (Spencer 1983).

Export to Bass Strait can be calculated with the integrated model (Chapter 6).

3.6.4 A concluding note

The PPBES has succeeded in making all of the measurements necessary to close the N, P and Si budgets and account for the quantitative effects of all of the major physical, biological, biogeochemical and chemical reactions occurring in the waters and sediments of Port Phillip Bay.

The major distinction of the PPBES compared with previous environmental studies is that enormous effort and emphasis was given to sedimentary processes. This strategy succeeded because sedimentary processes are now recognised as essential for modelling the nutrient budgets and understanding how Port Phillip Bay works as an ecosystem.

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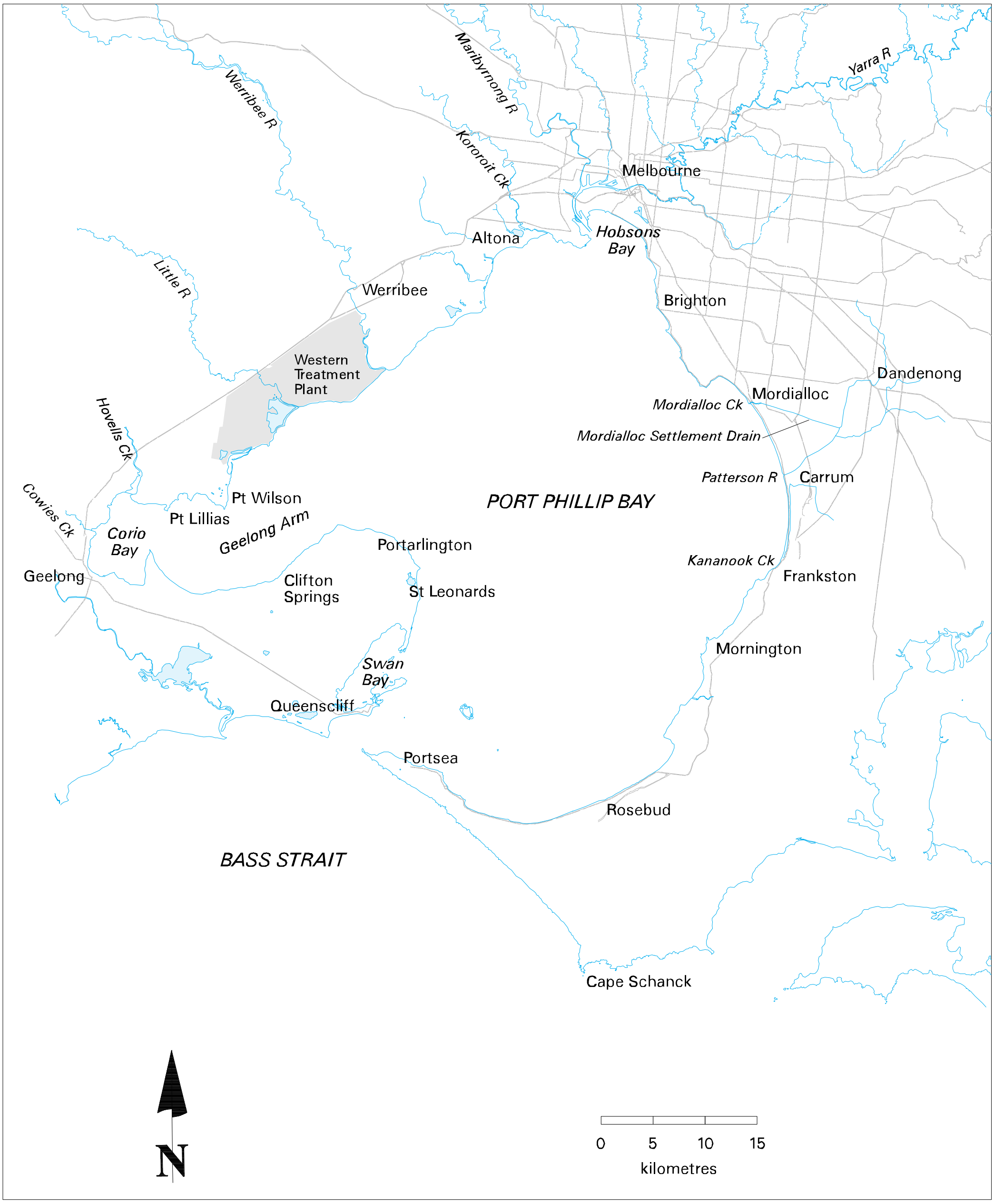

Figure 4.1: Major rivers, streams and drains that discharge into the Bay.

1. Introduction

The development of major population and industry centres around Melbourne and Geelong has resulted in a variety of potentially toxic heavy metal and organic compounds entering the waters of Port Phillip Bay.

If they remain in solution they are available for bioaccumulation by aquatic organisms such as algae, fish, mussels, scallops and seagrasses, all important components of the food chain with a possible impact on human consumers.

Natural processes may transform and transport many of these toxicants and, in many cases, they end up accumulating in bottom sediments where they can have longer-term effects on organisms inhabiting these sediments.

The study of metals and organics in Bay waters, sediments and biota was largely ignored by the Phase One Study (MMBW/FWD 1973) in favour of the more immediate concern for nutrients.

Since then, an awareness of the environmental importance of toxicants has led to a large number of studies and, for some toxicants, to regular monitoring programs.

Many of these studies followed early findings of elevated concentrations of metals such as cadmium, mercury, zinc and copper in biota (e.g. Kerr 1974, Walker 1976 and 1979).

Subsequent studies also addressed organic contaminants and, not surprisingly, found elevated concentrations of polycyclic aromatic hydrocarbons (PAHs), petroleum hydrocarbons and polychlorinated biphenyls (PCBs) in parts of the Bay (Phillips *et al.* 1992).

The results of these studies have been the subject of a number of extensive reviews (Coller *et al.* 1991, Batley 1992, Phillips *et al.* 1992).

Over the past 20 years our ability to measure ultratrace concentrations of toxicants has improved dramatically, as has our understanding of the chronic and acute effects of these low concentrations on key biota.

Regulatory guidelines that are protective of aquatic biota are now well established for waters and are well understood for sediments.

There are also defined maximum permitted concentrations of these toxicants in biota that are for human consumption.

It was important that the waters, sediments and biota in the Bay be examined using the latest analytical techniques to detect the concentrations of expected toxicants in relation to the guideline values.

This objective was addressed in the Toxicants Program in the Port Phillip Bay Environmental Study through a range of specific research and monitoring tasks, refined from those in the original Study Design (CSIRO 1992).

Particular attention was given to the inputs from the Western Treatment Plant and the major freshwater inputs from the Yarra and Maribyrnong River systems, as well as other sources, to allow evaluation of the long-term implications of these inputs on the environmental status of the Bay.

Processes that transport, degrade or redistribute these toxicants between waters, sediments and biota were also examined to enable the incorporation of toxicant behaviour into a model of the Bay.

From these studies a monitoring program was designed that would identify changes in toxicant concentrations that might be detrimental to the health of the Bay.

The toxicants of concern are those recognised by environmental agencies world-wide as being a problem in waterways near industrialised urban or agricultural environments.

Trace metal contaminants studied comprised cadmium (Cd), copper (Cu), lead (Pb), zinc (Zn), nickel (Ni), chromium (Cr), iron (Fe), manganese (Mn) and mercury (Hg), as well as the metalloid elements arsenic (As) and selenium (Se).

Organic compounds included petroleum hydrocarbons including polycyclic aromatic hydrocarbons (PAHs), polychlorinated biphenyls (PCBs), polychlorinated dibenzo-p-dioxins and dibenzofurans, organochlorine pesticides (such as aldrin, dieldrin, isomers of benzene hexachloride, p,p'-DDT, p,p'-DDD), and chlorophenoxy herbicides (2,4,5-T and 2,4-D), chlorophenols and chlorobenzenes (including mono and dichlorinated benzenes).

2. Metals

2.1 Inputs

Table 4.1 shows the estimated inputs of metals to Port Phillip Bay from all sources. Most groundwater inputs (column 5) are included in columns 3 and 4 (see Section 2.1.4).

2.1.1 Western Treatment Plant

Industrial discharges are a major source of toxicants to Port Phillip Bay. However, apart from fugitive emissions that may enter rivers, creeks and drains, the majority of Melbourne's industrial wastewaters are discharged under licence into the sewage system. The Western Treatment Plant (WTP) is therefore a potential contributor of contaminants to the Bay via discharges at the end of its treatment system.

The concentrations of metals in the waste-waters from WTP are regularly monitored by Melbourne Water. Data for the period 1980-1989 showed that the annual discharges of lead, copper and nickel have been relatively constant at 1.1, 2.7 and 4.9 t/yr respectively (Murray 1994). Chromium discharges declined from 4.5 t/yr before 1986 to 2 to 3 t/yr afterwards. Zinc discharges increased from 4 to 10 t/yr in the period 1987-1989, but returned to about 6 t/yr by 1993. The annual discharge of iron increased steadily during the 1980s from about 150 to 200 t/yr. Seasonal fluxes of copper and lead were noted, with discharges in autumn, winter and summer near the ratio 3:2:1, with a similar pattern for chromium, nickel, zinc and iron.

Concentrations as well as annual loads of metals in these inputs are important. As discussed by Batley (1992), concentrations of copper, chromium and lead exceeded the

Table 4.1: Loads of metals into Port Phillip Bay from input sources.

Metal	Estimated Annual Load (t)			
	WTP[a]	Rivers, creeks and drains (excluding Yarra)[b]	Yarra River[b]	Groundwater[d]
Fe	180	1,235	2,373	136-187
Cr	3.9	4.8	7.8	0.9-1.24
Cu	2.7	8.0	6.6	0.49-0.68
Ni	4.9	2.9	6.3	1.07-1.48
Pb	1.1	11	19	1.03-1.42
Zn	5.8	58	63	1.75-2.41
As		0.9	1.4	0.07-0.09
Cd	0.02[c]	0.14	0.17	0.21-0.30
Hg	0.02[c]	0.04	0.04	

a. Murray (1994)
b. from NSR (1993) (Table 7)
c. Melbourne Water (unpublished data)
d. HydroTechnology (1993)

ANZECC (1992) water quality guidelines before 1990, although the dilution in the Bay, and the fact that only a fraction of these metals are bioavailable, make these values less of a concern.

Mixing of the freshwater input with the saline water of the Bay leads to a natural process of precipitation and removal from the water column of many metals, including iron, copper, zinc, lead, nickel and chromium.

Changes in WTP management practices, or in industrial discharges, have led to a further decrease in the concentration of a number of metals in the 1990s (Murray, 1994).

Although the WTP is a significant source of a number of heavy metals to the Bay, for no metals, other than nickel, do the loads exceed those from the Yarra River catchment or from other creeks and drains (Table 4.1).

2.1.2 Rivers, creeks and drains

Five major rivers, numerous creeks and more than 300 drains empty into Port Phillip Bay (Figure 4.1). Collectively, they represent the largest source of metal contaminants (Table 4.1), with the Yarra River alone carrying more than half the total load. Their flows are variable and contaminant transport includes a contribution from high flow rainfall events, which transport surficial catchment contaminants, as well as the base flows which contain more constant inputs from industrial, domestic and agricultural sources, and groundwater.

Non-Yarra inputs

Metal loads to the Bay from 106 drainage sources, excluding the Yarra River, were surveyed as part of the Study (NSR 1993). The data was obtained from a three-year EPA monitoring program between 1979 and 1981, from sites on streams and drains around the Bay shore.

The major metal contribution was found to originate from streams and drains on the south-eastern side of the Bay (Figure 4.1).

Metal loads were calculated on the basis of flow-weighted mean concentrations and average daily flows at each site (NSR 1993).

An important finding was that a relatively small number of the streams and drains surveyed contributed most of the loads of metals that enter the Bay (excluding the Yarra River and WTP). The top 10 sites ranked on the basis of metal loads were the source of between 75 and 90% of the total metal loads (NSR 1993).

The Mordialloc Settlement Drain at Canterbury Road was consistently a main contributor of metal loads into the Bay being ranked first for Cr (33% of estimated stream/drain inputs), Cu (53%), Ni (23%), Pb (22%), Zn (33%) and As (23%).

The Patterson River was also important being ranked first for suspended matter (Non-Filtrable Residues, NFR) (48%) and Fe (29%), and second for Cd (19%), Cr (15%), Cu (15%), Ni (20%), Pb (21%), Zn (17%), As (17%) and Hg (22%).

Kororoit Creek (Altona) was ranked third for Hg (12%) and Cr (16%).

The review showed that the Patterson River and Mordialloc Drains, which are hydrologically linked, contribute between 35% and 75% of metal loads (excluding iron) into the Bay from rivers and drains and are the most significant input into the Bay after the Yarra River system and the WTP discharges. Their large contribution, >54% of the total flow from minor sources entering the Bay, was a principal reason for this.

Although the loads were calculated from data collected between 1979 and 1981, a comparison with 1989 data suggests that the input water quality had not changed. The cessation, during 1994, of the discharge from the Melbourne Water Wastewater Treatment Plant at Dandenong into the Patterson River/Mordialloc Drain system resulted, however, in a 15% reduction in total flow. This would have reduced metal loads to the Bay from this source.

Yarra River inputs

The 1979-81 data were also used to estimate total metal loads into Port Phillip Bay from the Yarra River catchment (Table 4.1).

Load estimation is critical to modelling inputs to the Bay, and a major concern was that if first flush concentrations after storm events were not measured the resulting load estimates could be subject to large errors.

It has been suggested that storm events contribute the majority of the river-borne metal contaminants to the Bay and that the first flush may contain concentrations an order of magnitude greater than low-flow concentrations. Because of this, a more detailed study of inputs from the Yarra River was conducted over a rain event in March 1995.

Water samples collected during the event were analysed for Cd, Cr, Cu, Fe, Ni, Pb, Zn, As and Hg (Fabris and Monahan 1995). The mean concentrations of dissolved and particulate cadmium, arsenic and mercury in Yarra water during the rain event were comparable to the concentrations found in the Bay.

On the other hand, mean river water concentrations for particulate Cr, Cu, Fe, Ni, Pb and Zn were from eight (Fe) to 39 (Zn) times the respective Bay concentrations. The corresponding concentrations for dissolved species ranged from 4 (Cu) to 42 (Zn) times their respective Bay-wide concentrations.

Estimated loads entering the Bay during the 1995 rain event were eight to 25 times base flow loads but comparatively small compared to total annual loads estimated from earlier studies. However, earlier load estimates were subject to errors in stream flow estimates during sampling, possible non-representative sampling and errors in the measurement of metal concentrations (NSR 1993).

The 30-h rain event studied in 1995 was found to contribute less than 0.5% of the Bay's historic annual load from the Yarra River system, varying with each metal (Fabris and Monahan 1995). This was despite the 1995 event occurring after a prolonged dry spell.

2.1.3 Shipping and dredging

Shipping has the potential to contribute contaminants to the Bay, in the form of metals and antifouling compounds such as tributyl tin (TBT) leached from hulls, and through the discharge of petroleum hydrocarbons or metals in port and during the passage of ships through the Bay.

To assess the importance of this source, water samples collected in the wake of ships were analysed (Fabris *et al*. 1995).

No detectable increases over ambient Bay concentrations of metals, TBT or hydrocarbons were found and it was concluded that, in the absence of accidental spillages, ships do not pose a significant source of toxic substances to the Bay. As expected, however, measurements of dockyard waters did show marginally higher concentrations of TBT, lead and zinc.

Dredging and dumping of dredged materials into the Bay also have the potential to release soluble toxicants.

Dumping operations clearly add contaminated material to the Bay floor, but elutriate testing showed that inputs to the Bay waters from this source are not significant (Fabris *et al*. 1995).

2.1.4 Groundwater

Groundwaters flowing through contaminated soils have the potential to leach toxicants and transport them to the Bay.

Groundwaters in the western suburbs had a high risk of being polluted with toxic substances because land use in this region includes a high density of manufacturing industries (HydroTechnology 1993).

The North Shore area of Geelong was also a potential concern.

Pollution of groundwaters results both from discontinued bore disposal and from existing

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industrial operations that contribute soluble organics such as phenols and surfactants, as well as metals.

Estimates of the groundwater inputs from the western suburbs, including the baseflow component to streams, suggest that only 2-12% flows directly into the Bay. The remainder flows into the stream drainage system (HydroTechnology 1993). Fluxes of toxicant inputs calculated from surface waters therefore include a groundwater component which could be a substantial proportion of the total (see note in Section 2.1).

Although groundwater loads of toxicants were predicted to be relatively high it is quite likely that the results are overestimates because the sampling bores were sited close to known pollution sources (HydroTechnology 1993). The assumptions used in calculating the load may also have produced biased results (O'Rourke *et al.* 1995).

It was estimated that the groundwater component of the total discharge of metals from the WTP into the Bay is less than about 1% (O'Rourke 1995).

2.1.5 Atmospheric inputs

Given its widespread use as a gasoline additive, lead is likely to be the only significant metal input to the Bay from atmospheric sources. Although no reliable estimate has been obtained for dry or wet depositional loads of lead to the Bay, an annual emission of about 500 t has been estimated for the Port Phillip Bay Control Region (Carnovale *et al.* 1992).

As will be seen, Bay-wide concentrations of dissolved lead are low and it is most likely that any lead reaching the Bay does so attached to particulates, either deposited directly or transported to the Bay via stormwater runoff.

2.2 Metals in Bay waters

Systematic water quality monitoring of metals in Bay waters was commenced in 1980 to ensure that the provisions of the State Environment Protection Policy (SEPP) were being met and to provide baseline data so that future changes in water quality could be assessed.

The initial sampling design was based on a stratified random sampling where the Bay was divided into 16 strata selected on the basis of perceived beneficial uses as specified in the SEPP. Currently, monthly samples are collected at only six fixed sites, most of which are located close to shore.

As part of the PPBES [Task T2], a Bay-wide survey of dissolved and suspended particulate metals was carried out in 1993, covering 25 sites as shown in Figure 4.2. The results enabled a comparison with previous monitoring data to determine the significance of the current concentrations and to provide data for modelling toxicant dispersion and fate (Fabris and Monahan 1995). Summary data are presented in Table 4.2 and compared with typical data for coastal and estuarine waters and ANZECC guideline concentrations (ANZECC 1992).

The measured concentrations in Bay waters of all elements except arsenic, were low compared to the ranges for other estuaries and were in many cases close to the values for relatively pristine coastal waters. Values were highest near the potential point source areas such as Hobsons Bay and Corio Bay, as shown with the example for zinc (Figure 4.3), but even at these sites they were well below accepted water quality guideline concentrations for marine waters for all of the metals examined (Table 4.2).

Arsenic concentrations in the Bay are at the high end of those reported internationally for near-shore and estuarine environments. In particular, they are high in comparison with arsenic concentrations observed from Victoria's other coastal environments.

Available data for arsenic inputs to the Bay indicate that these are not the current source of the elevated arsenic in the water column. Data for arsenic inputs from the WTP were unavailable, but are unlikely to explain the difference since the input from this source would have to be massively high compared to all other inputs. It is more likely that the excessive arsenic load in the Bay is related to the natural sediment mineralogy, with diagenetic processes in the sediments resulting in the release of arsenic into the water column.

Arsenic concentrations in surface sediments of the Bay are depleted relative to deeper sections and this suggests that arsenic is mobilised to the water column (Fabris *et al.* 1994). The origin of the dissolved arsenic is of academic interest only, given the low arsenic concentrations relative to ANZECC guidelines, but further investigation would be necessary for the issue to be resolved.

Knowledge of the distribution of metals between dissolved and particulate phases and the fraction bound as organic complexes is also needed to determine the significance of the measured metal concentrations in relation to their transport and bioavailability.

It was found that most of the cadmium (70%), copper (94%), nickel (92%), zinc (82%) and arsenic (99%) was present as dissolved species. As would be expected in estuarine waters, most of the iron (97%) and chromium (70%) was present in particulate forms, and these are able to absorb other trace metals. Lead, for example, closely followed the pattern for iron, and similar correlations were found between iron and other metals.

Temporal trends in metal concentrations can be estimated from the EPA Fixed Sites Monitoring Program data. Although this program measured only total (acid-labile) metals rather than dissolved metals (after 0.45 µm filtration), it was possible, even allowing for variations in the particulate loads as a contribution to this total, to see useful temporal trends in metal concentrations. Moni-

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Table 4.2: Dissolved metal concentrations in Bay waters (Fabris and Monahan 1995).

Metal	Port Phillip Bay Data Mean (range) (µg/L)	Nearshore and Estuary Data Range (µg/L)	ANZECC Guideline Concentration for Protection of Marine Life (µg/L)
Iron	0.76 (0.17-1.8)	<0.04-13.7	–
Chromium	0.04 (0.01-0.09)		50
Copper	0.47 (0.40-0.63)	0.06-1.3	5
Nickel	0.70 (0.54-1.10)	0.06-1.6	15
Lead	0.06 (0.02-0.13)	0.02-0.06	5
Zinc	0.47 (0.25-1.05)	0.01-3.3	50
Arsenic	2.8 (2.2-3.2)	1.0-3.3	50
Cadmium	0.026 (0.015-0.070)	0.007-0.10	2
Mercury, ng/L	1.7 (0.8-4.9)	0.7-3	100

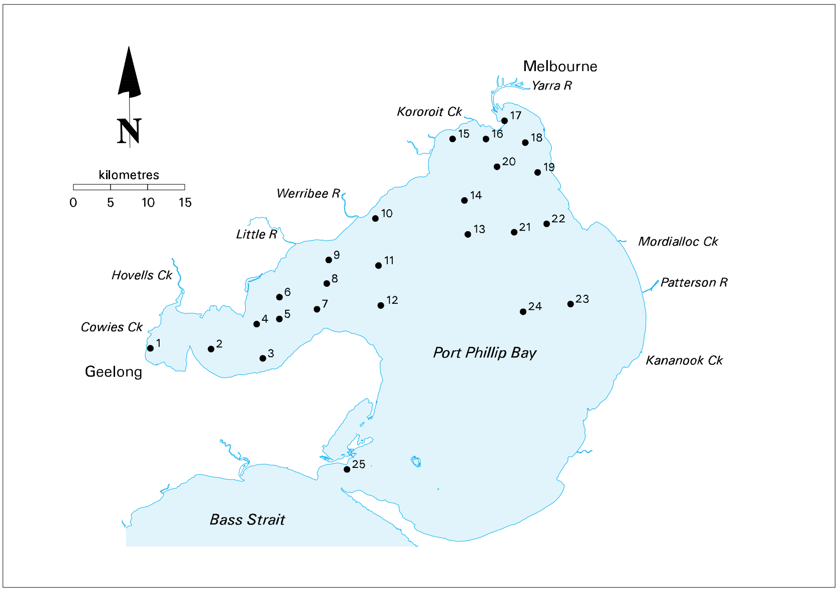

Figure 4.2: Sampling sites for toxicants in Bay waters (Fabris and Monahan 1995).

toring data for Hobsons Bay, Central Bay and Corio Bay provided a reasonably continuous time series from 1980.

Results showed that, prior to 1990, cadmium and nickel concentrations in Corio Bay had been significantly higher than at the other two sites. In Hobsons Bay, copper, zinc and iron concentrations have been gradually decreasing, but concentrations were significantly higher than those of the Central Bay

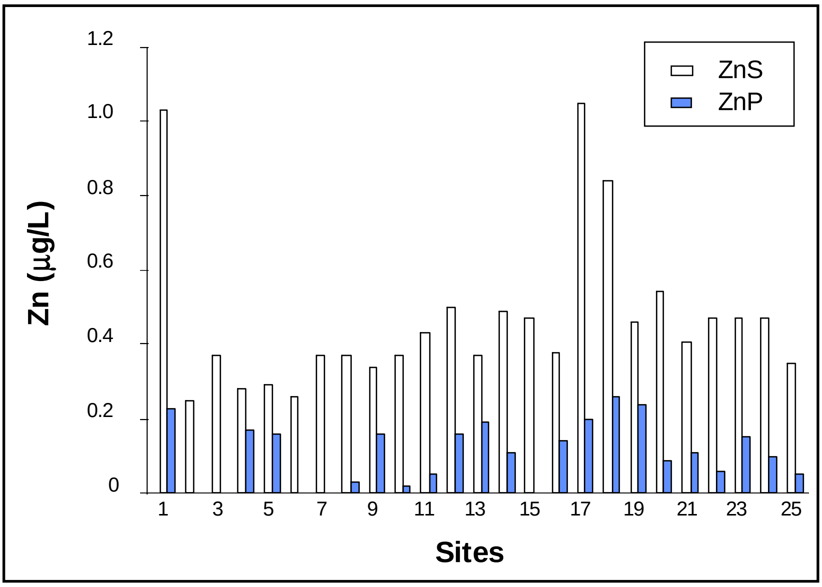

Figure 4.3: Zinc concentrations in Bay waters (Fabris and Monahan 1995). ZnS = dissolved zinc species, ZnP = particulate zinc species.

site, and in the case of iron, higher than both of the other sites. This reflects the importance of the Yarra River as a source of dissolved iron, much of which is in particulate form by the time it reaches the Bay.

An interpolated plot of the mean annual cadmium concentration in waters from Corio Bay, Hobsons Bay and the Central Bay for the period 1980-1995 suggests that concentrations have been decreasing steadily since about 1988 (Figure 4.4).

The industrial contamination of Corio Bay by cadmium had been identified more than 20 years ago. Cadmium forms stable complexes in saline waters and would be readily transported throughout Port Phillip Bay. Once the contaminant source had been identified and remedial action initiated, concentrations would have been expected to decrease.

Mercury was another element that was elevated in biota and waters in 1980-1982.

Apart from an isolated contamination event in Corio Bay in 1987, mercury concentrations decreased in 1983 and have remained comparatively constant, although there is a suggestion that concentrations may have been increasing slightly since about 1992. Nevertheless, mercury levels in the Bay are close to values for open ocean waters and are of no environmental concern.

Since anthropogenic inputs to the catchments of the Bay, and hence to the input streams, are likely to have increased over the past 15 years, it is important that the concentrations in Bay waters of the more ubiquitous metals such as lead, zinc, copper, iron and nickel have remained relatively constant over that period.

Apart from some fluctuations at sites (such as Hobsons Bay) that are closest to sources whose inputs are flow or event dependent, these findings indicate that, in addition to the regulation of discharges, removal processes such as sedimentation and Bay flushing are effective in maintaining dissolved metals in Bay waters at relatively pristine concentrations.

A comparison of the metal pools in the waters of the Bay with the estimated annual loads to the Bay from the major input streams and drains (assuming a residence time of one year), found that chromium, iron, lead and zinc inputs exceed the standing pools in the Bay (EPA miscellaneous data). These elements are therefore not remaining in solution or attached to suspended particles in the Bay waters, but are most likely being removed by sedimentation to the Bay floor.

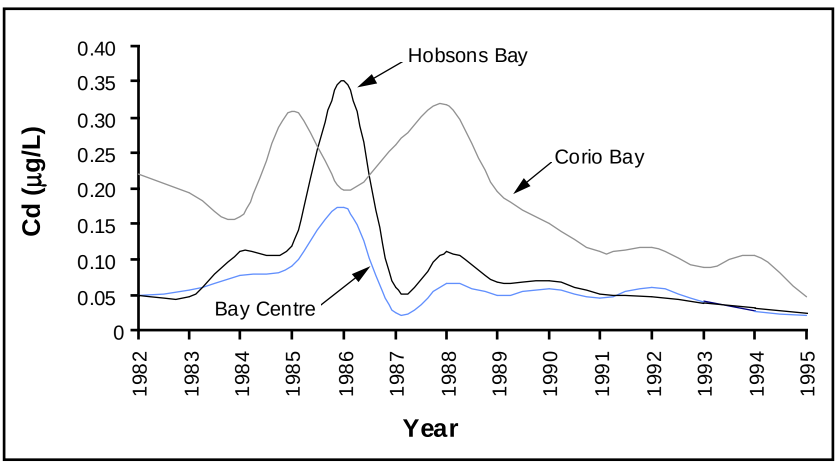

Figure 4.4: Temporal changes in total cadmium concentrations in waters from Hobsons Bay, Corio Bay and the Bay centre.

2.3 Metals in Sediments

Sediments are the ultimate sink for most toxicants entering Port Phillip Bay. There they remain potentially available to benthic organisms, including filter feeding, burrowing and deposit feeding biota, as well as to seagrasses and other rooted plants.

Except in the very surface oxic layers, most heavy metals are present in the sediments as insoluble sulphides.

Metals are bioaccumulated by ingestion of particles or as dissolved metal from the sediment porewaters.

The dissolution of metals from particles to porewaters is usually very small, but will be greatest in oxic sediments. It can be enhanced in the deeper anoxic zones through the introduction of dissolved oxygen by burrowing animals or plant roots.

Depth profiles of trace elements in sediment cores can provide information on the rate of sedimentation and the historical trends in contaminant accumulation, provided mixing of sediments by burrowing animals is not significant.

Knowledge of the patterns of sedimentation and sediment movement in the Bay enable modelling of the ultimate fate of toxicants entering the Bay.

The rates of sediment accumulation in Port Phillip Bay are low and the changes in metal concentrations in the surface 10 cm of sediments are minor.

Previous data have been summarised in Batley (1992). Melbourne Water and the EPA have extensively studied 18 Bay sites (EPA 1981, MBW 1991), while localised studies, such as those in Corio Bay (Fabris *et al.* 1986, Nicholson *et al.* 1992), have been carried out in response to specific pollution issues.

Sediment analyses have also been required prior to dredging operations (PGA 1993), but these studies have been specifically targeted at shipping channels and dredge spoil grounds.

Typical total metal concentrations found in Bay sediments in 1976 are shown in Table 4.3.

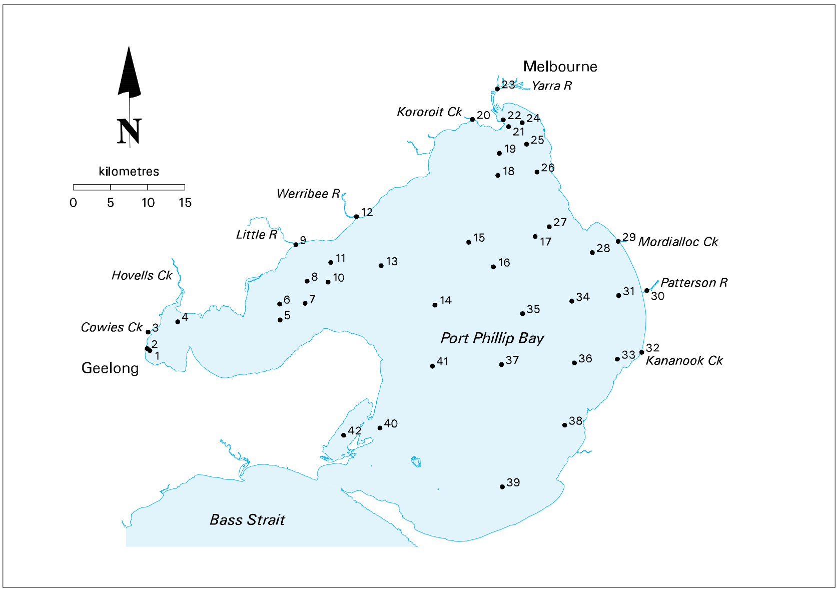

Figure 4.5: Sampling sites for toxicants in Bay sediments (Fabris *et al.* 1994, Good and Gibbs 1995b).

As part of the PPBES [Task T7], sediment samples were obtained from 42 widely-distributed sites within the Bay, with three replicate sediment samples collected from each of 33 sites within the Bay and from nine widely-spaced input streams and drains (Sites 2, 3, 9, 12, 20, 23, 29, 30 and 32) (Figure 4.5).

The sites within the Bay were primarily located in grid patterns near the Western Treatment Plant (Sites 5-8 and 10, 11), Hobsons Bay (Sites 18, 19, 21, 22, 24, 25, 26) and the centre of the Bay (Sites 14-17, 27, 28, 31, 33-41).

Corio Bay was not sampled intensively because this had been done in recent studies.

Surface sediments (0-10 cm deep) were collected at all sites and, additionally, deeper cores were obtained from four sites.

The samples were analysed for dilute-acid-labile (0.5M HCl) metal concentrations in the fine, silt/clay (<63 µm) fraction, 63-200 µm (fine sand) fraction and >200 µm (coarse sand) fraction.

Because international sediment quality guidelines specify total metal concentrations,

Table 4.3. Mean total metals in whole sediment from Port Phillip Bay in 1976.

Site (Fig 4.5)	Cd	Cu (µg/g dry weight)	Pb	Zn	Hg
2[a]	60	57	250	148	0.06
3[a]	3.6	31	83	420	0.06
3[b]	6.2	25	126	125	
9[a]	0.6	41	38	63	0.11
9[b]	1.3	14.8	39	94	
12[a]	1.1	7.2	15	30	0.1
12[b]	1.5	3.3	17	87	
20[a]	4.2	330	350	420	20.7
22[b]	4.3	23	112	119	
23[a]	1.6	21	55	132	0.38
29[a]	23.6	860	325	1400	0.39
29[c]	12.5	450	340	1150	
30[a]	0.7	14.6	31	135	0.06
30[b]	0.7	9	5	8	
32[a]	3.1	61	213	320	0.21
Central Bay[a]	0.6-2.4	5-16.4	18-52	35-123	0.05-0.12

a. EPA 1976
b. Talbot et al. *1976 (<200 µm size class)*
c. Smith 1976. Mean values from sites 3 and 4 are given.

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Task T7 also analysed for all metals in the <63 µm, 63-200 µm and >200 µm grain size fractions by aqua regia digestion (Fabris *et al.* 1994).

An analysis of the results for differences between metal concentrations in sediments near input sources (WTP, Hobsons Bay) and central areas of the Bay (Fabris *et al.* 1994), showed that within the Bay, sediments from Hobsons Bay sites had significantly higher total concentrations of copper, lead, zinc and arsenic than the other Bay sites.

As expected, sediments from input sites were higher in cadmium, chromium, copper, iron, nickel, lead, arsenic, selenium and mercury than those from the Bay sites.

Data from other studies indicate that sediments from Corio Bay had metal concentrations that were similar to those from the Hobsons Bay sites.

The PPBES data from the Corio Bay sites confirmed the findings of recent studies indicating that contamination by cadmium has decreased, although lead and zinc concentrations were still elevated.

Additionally, WTP sites had iron and selenium concentrations that were significantly lower than other Bay-wide sites, and the central sites had the highest concentrations of manganese.

The low metal concentrations in the sediments from sites near the WTP suggests that metal concentrations in particulate loads from the WTP are either low or they are transported to other parts of the Bay.

In Corio Bay, sediment transport is restricted by the sand-bar which forms its eastern boundary.

However, cadmium is readily leached from contaminated sediments and once it enters the water column is exported from Corio Bay into Port Phillip Bay by means of tidal exchanges.

Significant inter-element correlations were noted. These relate principally to the nature of the sedimentation processes where many anthropogenic contaminants in a dissolved state in the input waters, coprecipitate with iron and dissolved organic matter in the mixing zone of the estuary (Fabris *et al.* 1994).

The nature of the sediment deposition behaviour near the Yarra River as a major input source is seen with lead as an example, in Figure 4.6 (Fabris and Monahan 1995).

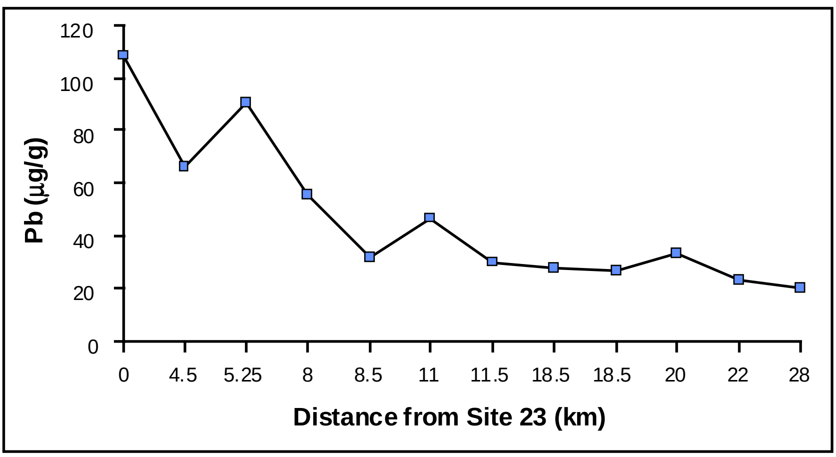

Figure 4.6: Reactive lead concentration in the silt/clay (<63 µm) fraction of suspended sediments with distance from the Yarra River input site (Site 23, Figure 4.5) (Fabris and Monahan 1995).

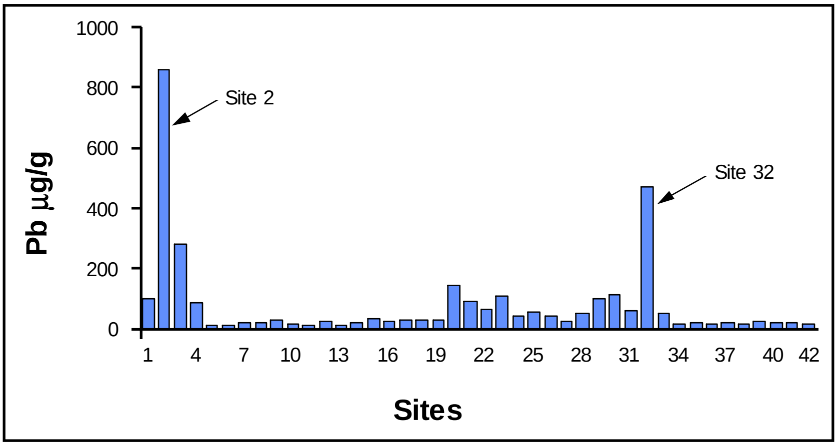

Figure 4.7: Reactive lead concentrations (µg/g dry weight) in the silt/clay (<63 µm) size fraction of sediments from Bay sites (Figure 4.5) (Fabris *et al.* 1994).

Reactive lead in the clay/silt fraction clearly decreases with distance from the source.

High metal concentrations at specific sites within the Bay were evidence of toxicant inputs. For example, lead concentrations in the clay/silt fraction were extremely high at two input sites in Corio Bay and at Site 32 close to Kananook Creek (Figure 4.7).

At Site 21 in Hobsons Bay, mercury concentrations were anomalously high compared to adjacent sites, suggesting that the additional contamination of the sediments at this site is unrelated to discharges from the Yarra River system.

As already discussed, the biological impact of metals in contaminated sediments is assessed on the basis of total metal concentrations. Comparisons have been made between the measured concentrations and the sediment quality guideline values for each metal

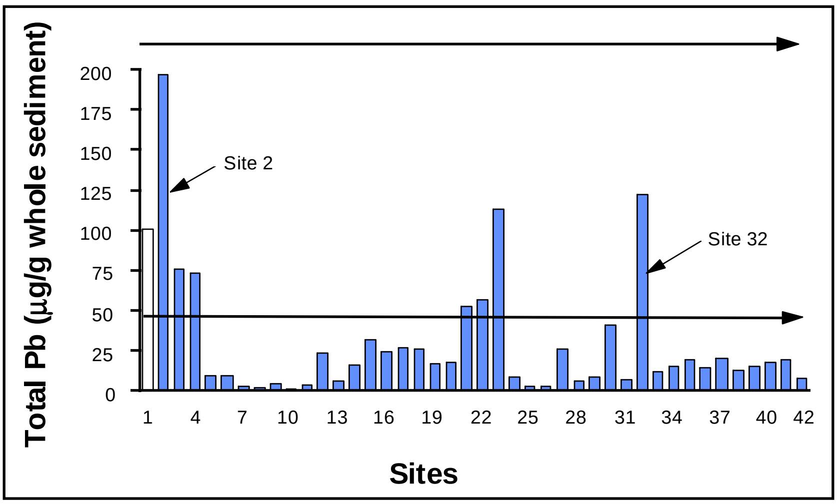

Figure 4.8. Concentrations of total lead in whole sediment from Bay sites. Concentrations below the lower horizontal line (ERL) are not expected to cause adverse biological effects (Table 4.4). The upper horizontal line (ERM) indicates the concentrations which could cause adverse biological effects (Fabris *et al.* 1994).

(Table 4.4), and the example of lead is given in Figure 4.8.

Two guideline values are indicated which correspond to the screening value, threshold effects level or target value, and probable effects value at which adverse biological effects might be expected (Batley, 1994).

Most sites within the Bay had metal concentrations well below levels considered to cause adverse biological effects. However, exceptions were noted for Rippleside Beach (lead and zinc), Cowies Creek, the Yarra River system and Kananook Creek (zinc). The lower guideline value was exceeded in Corio Bay (cadmium, copper, lead, and zinc), the Yarra River system (copper, lead, zinc and mercury) and Kananook Creek (copper, lead and zinc).

The history of sediment contamination in the Bay is best evaluated from the analysis of metal concentrations in the vertical profiles of sediment cores. A number of cores were examined but did not meet the criteria of being both undisturbed as well as sufficiently deep.

A typical profile for lead from Site 37 in the centre of the Bay is shown in Figure 4.9.

The core data indicated that contamination of the Bay by copper and zinc as well as lead is continuing.

Arsenic concentrations in the upper sections of the cores were depleted relative to the deeper sections. It is possible that the elevated arsenic concentrations in the waters of the Bay are related to this apparent leaching of arsenic from surface sediments (Figure 4.10).

The use of lead-210 dating and the presence of specific contaminants such as TBT whose date of introduction is known, was investigated in order to determine sedimentation rates in the Bay.

It would appear that sedimentation is greatest in the central Bay sites, and less closer to Hobsons Bay. The latter is likely to accumulate sediment during periods of low flow, but may be substantially scoured by the high flows associated with storm events. Elevated metal concentrations in the clay/silt fraction of the central Bay cores support this argument.

Table 4.4. Range of metal concentrations (µg/g) in whole surface sediments of the Bay (from Fabris *et al*. 1994) and sediment quality guidelines.

Site No. (Figure 4.5)	Fe (%)	Cr	Cu	Ni	Pb	Zn	As	Cd	Hg
1-5	3.5-5.5	36-66	7-51	16-66	9-197	68-830	7-30	0.2-6	0.03-0.3
6-13	0.3-5.6	8-97	1-36	2-26	2-23	13-171	3-12	0.1-0.8	0.07-0.16
14-20	1.3-5.4	16-64	9-14	3-41	16-27	122-197	9-13	0.1-0.2	0.06-0.16
21-23	5.1-8.8	52-94	9-62	13-42	52-113	288-640	15-25	0.1-1	0.2-0.5
24-29	0.8-4.7	12-57	1-11	4-40	3-26	42-162	4-13	0.1	0.07-0.12
30-32	2.3-3.2	24-31	3-46	5-8	6-122	99-780	5-22	0.1-0.5	0.08-0.12
33-42	0.8-5.9	17-115	3-10	5-49	7-20	39-161	10-20	0.1-0.4	0.03-0.07
ERL[a]	–	81	34	21	47	150	8.2	1.2	0.15
ERM[a]	–	370	270	52	218	410	70	9.6	0.71

a. ERL is the Effects Range Low and represents the lower 10th percentile of effects data. The median, or 50th percentile is the Effects Range Medial (ERM) (Long et al. *1995).*

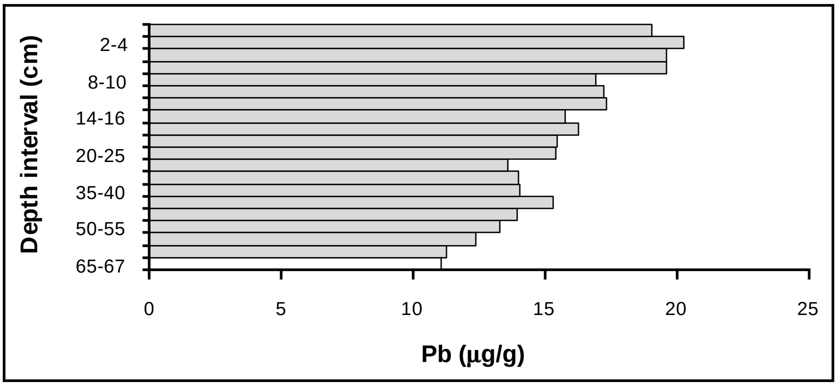

Figure 4.9. Lead concentration profile in a sediment core from Site 37 (see Figure 4.5) (Fabris *et al.* 1994).

Sediment dredging and disposal

Metal concentrations are elevated in sediments from the dockyard areas. These areas are subject to maintenance dredging by the port authorities with disposal elsewhere in the Bay.

The dumping of sediments from the Port of Melbourne to the spoil grounds in the northern part of the Bay has resulted in elevated concentrations of cadmium, lead, zinc and tributyl tin (TBT) in spoil ground sediments (Fabris *et al.* 1995).

Nickel and mercury exceeded the accepted sediment quality guidelines in some of the spoil ground sediments, but these concentrations are lower than those found in the more contaminated port areas.

Elutriate testing of these sediments suggested that it is unlikely that leaching of metals from the sediments into the water column during dredging and disposal operations would result in elevated concentrations of dissolved metal species in the water column, although particulate metal species may be spread to other parts of the Bay (Fabris *et al.* 1995).

Monitoring programs, to be implemented during large scale dredging operations, such as those involving the Geelong Channel Improvement Program and developments for the Point Lillias Chemical Storage Facility and East Coast Armaments Complex (ECAC), are designed to detect any unforeseen adverse effects.

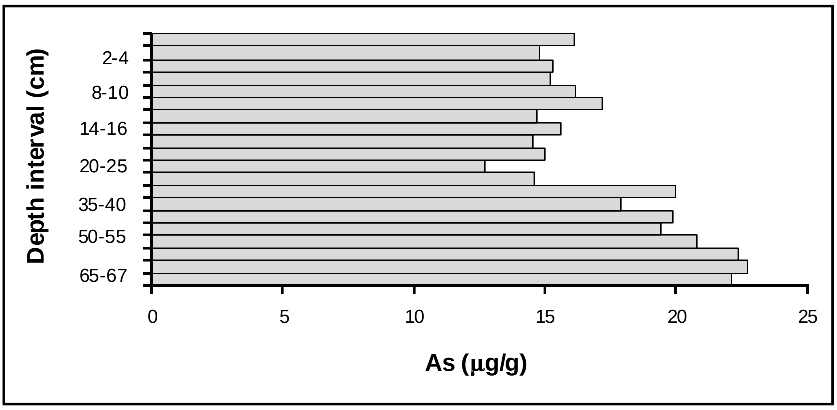

Figure 4.10. Arsenic concentration profile in a sediment core from Site 37 (see Figure 4.5) (Fabris *et al.* 1994).

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2.4 Metals in biota

Aquatic biota are integrators of toxicants both from the water column and the sediments.

Bioaccumulated toxicants have the potential for both chronic and acute impacts on organisms. Effects are predictable on the basis of quality criteria for both waters and sediments.

Biomagnification can occur through the food chain and an ultimate concern is for the impact on human consumers. The latter can be judged against the Food Standards Code (1995) derived from a consideration of an average daily intake by consumers (Table 4.5).

Monitoring of fish and molluscs in the Bay for heavy metals and organic toxicants has been undertaken routinely, and in response to specific pollution incidents, by the Department of Conservation and Natural Resources. Additional studies have been undertaken by Melbourne Water and other research groups and considerable data have been accumulated over the past 20 years (Coller *et al.* 1991, Batley 1992, Phillips *et al*. 1992).

As part of the PPBES [Task T8], a survey was conducted of heavy metals in flathead,

Table 4.5: Maximum permitted concentrations (MPC) of heavy metals in seafood (Section A12 of the Food Standards Code 1995).

Element	Food type	Maximum Permitted Concentration (µg/g wet weight)
Arsenic (As)	Fish and molluscs	1.0 (inorganic As only)
	Seaweed (edible kelp)	1.0 (inorganic As only)
Cadmium (Ca)	Fish (and fish contents	
	of products)	0.2
	Molluscs (and mollusc	
	content of products)	2.0
	Seaweed (edible kelp)	0.2
Copper (Cu)	Molluscs (and mollusc	
	content of product)	70
	AOF[1]	10
Lead (Pb)	Fish in tinplate containers	2.5
	Molluscs	2.5
	AOF[1]	1.5
Mercury (Hg)	Fish and molluscs	A mean of 0.5 (the standard specifies a sampling plan)
	AOF[1]	0.03
Zinc (Zn)	AOF[1]	150

1. AOF = All other foods (applicable to seafood when no specific reference is made in the standard) mussels and dominant benthic flora (excluding seagrasses) from six offshore areas in the Bay, comprising the Geelong Arm, WTP, Hobsons Bay, Eastern Bay off Mordialloc Creek, Southern Bay and St Leonards as shown in Figure 4.11 (Fabris 1995). These were considered to be representative of the important biota in the Bay.

2.4.1 Molluscs

Blue mussels (*Mytilus edulis planulatus*) are among the most common molluscs in the Bay, occurring on shallow rocky reefs, channel marker piles and piers. A mussel-culture industry has been operating since 1982 and now includes culture sites near Clifton Springs in the Geelong Arm and Beaumaris in the central east of the Bay. Cultured mussels are monitored regularly for conformance with the United States Food and Drug regulations. Monitoring includes the determination of toxicant levels to assess compliance with the Food Standards Code (1995) (Table 4.5).

Mussels have been used as biomonitors of metal and organic pollution of marine environments for more than 20 years.

During the 1970s, surveys of the Blue mussel population of the Bay showed there was widespread contamination of many shoreline areas of the Bay by cadmium, lead and zinc. The most contaminated areas were those subject to urban and industrial inputs.

A long-term survey of metal concentrations in offshore mussels from fixed sites in the north of the Bay and Corio Bay between 1980 and 1986 found no temporal trends in metal concentrations, with the exception of cadmium in Corio Bay which, in keeping with the concentrations in waters and sediments, decreased after the industrial source had been identified and controlled (Longmore *et al.* 1989).

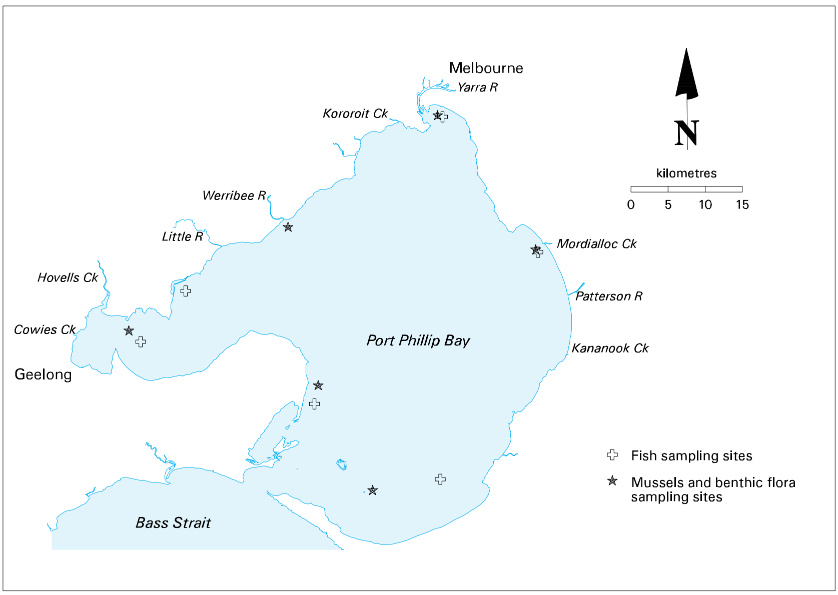

Figure 4.11: Sampling sites for fish, mussels and benthic flora in the Bay (Fabris 1995).

These results support the finding of the sediment studies that most metals entering the Bay through streams and drains are deposited close to the point of input.

The metal content of mussel samples analysed as part of the PPBES (Table 4.6) were all within the concentration ranges found in earlier studies and below the maximum permitted concentrations (MPC) of metals in seafood (Fabris 1995).

Cadmium concentrations were still higher in mussels from the Geelong Arm (and the Southern Bay site) compared with samples from the rest of the Bay, but were well below the MPC.

Cadmium contamination of the Bay has decreased significantly since the mid-1970s, when mussels from intertidal zones around the Bay contained cadmium from 0.6-9.5 µg/g and 1.3-1.7 µg/g in offshore areas.

The higher nickel concentrations in mussels from the Geelong Arm and Werribee sites are consistent with the relatively high input of nickel into the Bay from the WTP (4.9 t/yr) compared to other inputs (9.2 t/yr for all other sources) (Murray 1994).

Zinc concentrations in mussels exhibit a wide inherent variability (Lobel and Wright 1982), so differences between sites should not be taken as an indication of differences in ambient zinc concentrations, particularly since the variation is less than a factor of three.

Since 1974, zinc concentrations in mussels from the Bay have ranged from 13 to 32 µg/g in offshore sites and from 13 to 76 µg/g along the intertidal zone (Phillips *et al.* 1992).

Most of the arsenic in the mussels from the Bay is most likely present in non-toxic organic forms such as arsenobetaine (Edmonds and Francesconi 1981) and so does not pose a health risk to consumers.

Surveys of other commercially-exploitable mollusc species such as oysters (*Ostrea angasi*), scallops (*Pecten alba*) and abalone (*Haliotis rubra*), sampled in the 1980s (Phillips *et al.* 1992), showed mean metal concentrations generally below the MPC.

The mercury concentrations in scallops decreased from north to south and from west to east of the Bay, with high values in Corio Bay (range 0.01 to 0.2 µg/g wet weight).

Other metal concentrations in these samples followed the same regional patterns as those seen for mussels.

Table 4.6: Metal concentrations (µg/g wet weight) in soft tissues of mussels (*Mytilus edulis planulatus* Lamarck) (n=15) in Port Phillip Bay.

Site	Geelong Arm	Werribee	Hobsons Bay	Mordialloc	Southern Bay	St Leonards
Cadmium	0.64	0.23	0.17	0.20	0.54	0.33
Chromium	0.29	0.36	0.29	0.27	0.37	0.33
Copper	1.16	1.06	0.86	0.93	1.10	0.86
Nickel	0.44	0.50	0.19	0.23	0.34	0.36
Lead	0.17	0.09	0.37	0.43	0.21	0.09
Zinc	30.7	24.6	34.6	27.3	47.4	42.9
Arsenic*	3.67	2.56	2.56	2.81	3.50	2.86

** mostly in non-toxic organic forms*

2.4.2 Fish

Sand flathead support an important recreational and commercial fishery in the Bay. In the early 1990s this species was as much as 66% of the recreational fish catch of 316 t/yr (Coutin *et al.* 1995).

The commercial catch adds about 10 to 15 t/yr to this total (Victorian Fisheries Research Institute unpublished data). Consumption of fish from the Bay is considerable and it is important for regulators to assess the potential risk of adverse effects to consumers from toxic substances accumulated by the fish.

Most available data for metals in fish from the Bay relate to mercury concentrations, largely in response to world-wide human health concerns in the early 1970s.

Two flathead species – sand flathead (*Platycephalus bassensis*) and yank flathead (*Platycephalus speculator*) – occur commonly in the Bay.

Sand flathead are the more abundant species and have been shown to be useful biomonitors of ambient mercury concentrations in Port Phillip Bay (Fabris *et al.* 1992a). During the late 1970s sand flathead from the Bay had a mean mercury concentration of 0.5 µg/g wet weight, and in some locations the concentrations exceeded the MPC. Industrial inputs were identified as the source of mercury.

By 1990 mercury concentrations in flathead had declined to a Bay-wide mean of 0.23 $\pm$ 0.18 µg/g and results from the PPBES study indicate that the decline has been sustained (Fabris 1995).

Analyses of sand flathead in PPBES Task T8 showed that levels of metals were well below health guidelines and indicated no threat to human consumers (Table 4.7).

Arsenic concentrations did exceed food-guideline values, but these are based on inorganic arsenic and, as with mussels, it is well established that arsenic in fish is present mainly as non-toxic organic forms. Metals other than mercury in Bay fish have not been studied extensively.

As an indicator of the presence of metal

Table 4.7: Metal concentrations in sand flathead muscle (n=15) from six sites in Port Phillip Bay. Data are µg/g wet weight (Fabris 1995).

Site	Geelong Arm	Werribee	Hobsons Bay	Mordialloc	Southern Bay	St Leonards
Arsenic	6.1	9.7	9.2	7.4	9.0	8.3
Cadmium	0.05	0.05	0.05	0.05	0.05	0.05
Chromium	0.06	0.05	0.06	0.07	0.08	0.07
Copper	0.41	0.25	0.26	0.66	0.23	0.32
Mercury	0.11	0.31	0.04	0.09	0.12	0.11
Nickel	0.05	0.05	0.05	0.06	0.06	0.05
Lead	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05
Zinc	5.5	6.8	4.6	4.8	4.4	3.8

toxicants in the Bay, analyses of fish showed some differences between sites.

Fish near Werribee were significantly higher in mercury and at the Mordialloc site were higher in copper. As expected, low concentrations were found in fish from the Southern Bay and St Leonards sites.

Fish livers preferentially concentrated zinc, copper, cadmium and mercury, and the same spatial differences were evident. MPC concentrations for cadmium were often exceeded, but fish livers are rarely consumed by humans.

2.4.3 Benthic flora

Benthic plants accumulate metal contaminants from both the water column and from sediments and, although not an important component of human diets in Australia, are nevertheless part of the food chain and have been used in the past as indicators of metal contamination (Werner 1992).

A range of dominant macroalgal species at six sites were sampled and analysed as part of the PPBES (Fabris 1995). These included the macroalgae *Ecklonia radiata* (Southern Bay and St Leonards), *Caulocystis* sp. (Geelong Arm), *Rhodymenia australis* (Werribee) and *Polysiphonia isogona* (Hobsons Bay), and the sea lettuce, *Ulva* sp. (Mordialloc).

Comparisons between sites were not possible because of differences in the species studied.

However, the algae from Werribee and Hobsons Bay contained higher concentrations of most metals than species from the other sites, consistent with the findings for water samples and for mussels and fish. The results were similar to those reported elsewhere for similar species and were not a cause for concern.

Apart from zinc, which was present in the highest concentrations at all sites, nickel was more than an order of magnitude higher in the Werribee sample than for other sites.

Arsenic was high in all samples, but as with fish and mussels, most of this occurs as non-toxic organic forms.

3. Organics

Organic contaminants (including petroleum hydrocarbons (TPHs), polycyclic aromatic hydrocarbons (PAHs), polychlorinated biphenyls (PCBs), organochlorine (OCs) and organophosphorus (OPs) compounds) enter the Bay from oil refineries (e.g. Corio Bay), river and urban runoff (particularly from Melbourne and Geelong), boating and shipping activities and from groundwater.

Previous monitoring of organic contaminants in the Bay waters, sediment and biota has been well documented (Phillips *et al.* 1992).

For this Study, organics were investigated in all of the samples collected for metals.

3.1 Inputs

3.1.1 Western Treatment Plant

The most toxic organics are those which have low water solubility and are therefore accumulated in the high lipid containing tissue of aquatic biota.

In natural waters, these compounds partition preferentially to particulate matter, associating with particulate organic matter in clay and silt particles.

The concentrations of organics in the effluent from WTP are typically low because most are either removed by sedimentation, in the case of hydrophobic compounds, or degraded by photolytic and microbial processes during their passage through the lagoon system.

The concern for organics in WTP effluent arose in relation to the discharge to WTP of 2,4-D from an industrial plant.

Organochlorines and dioxins were of primary interest because of their known toxicity (Bremmer *et al.* 1990). Only minor concentrations were able to be detected which were below thresholds of concern.

3.1.2 Rivers, creeks and drains

As with metals, inputs from rivers, creeks and drains are likely to represent the principal source of organic contaminants. Verification of this proved more difficult.

A study of inputs from the Yarra River in 1993 focused mainly on nutrients and to a lesser extent metals, examining the fluxes through a cross-section of the Yarra before it entered the Bay.

Limited data were collected for TPHs, PAHs, total organochlorine pesticides, total chlorinated aromatic hydrocarbons and phthalate esters (dissolved plus particulate), but the results are of doubtful value (Black *et al*. 1993b).

A more reliable study over a rain event in March 1995 (Good and Gibbs 1995a) showed a peak in TPH about 12 h after the rain event commenced and coincident with the flow maximum. A second peak occurred after about 38 h. Total PAHs also peaked at 12 h but the concentrations of more toxic PAHs such as benzo(a)pyrene were low.

Dieldrin, DDT, DDD, and lindane peaked in the outflow at concentrations which often exceeded the ANZECC guideline values.

Loads of organic toxicants delivered to the Bay over the 60 h event were as high as 590 kg of TPH and 2 kg of PAHs, with only 120 g of organochlorine insecticides recorded. These figures were used as a representative input to the toxicant model.

The study also showed that, as might be expected, the concentrations of the more hydrophobic organics decreased significantly with distance from the shore into the Bay, due to attachment to sedimenting particulates, whereas hydrophilic contaminants tended to stay in solution.

Organic loads from other inputs were not measured and very little previous monitoring data were found for these sites.

The NSR study (1993) of surface inputs other than the Yarra River and WTP provided historical data only for surfactants as organic contaminants. Their input amounted to 53 t/yr and was equal to the estimated load from the Yarra River.

3.1.3 Shipping and dredging

Petroleum hydrocarbons are the most likely organic compounds to enter the Bay as a result of shipping activities. As will be discussed later, sampling of ports and spoil grounds found significantly higher concentrations of total petroleum hydrocarbons in the Port of Melbourne (including North Wharf and Swanson, Victoria, Appleton, Yarraville and Holden Docks) than for the spoil grounds or other Bay sites (Fabris *et al.* 1995).

As expected, the concentrations at some dock sites were almost twice the spoil ground concentrations.

Total PAHs, which will be a component of these oils, had a mean value close to the effects range low sediment quality guideline value, but well below the effects range median value at which significant biological effects might be expected.

Elutriate tests on sediment samples indicate that leaching of petroleum hydrocarbons from sediments into the water column during the dredging and disposal operations is not an issue (PGA 1993, Fabris *et al.* 1995)

3.1.4 Groundwater

No detailed study of groundwaters for organics has been conducted. The review by HydroTechnology (1993), conducted for the PPBES, suggested that phenols might be the major organic compounds of concern. The load from the western suburbs aquifer was estimated to be 6.1 to 8.4 t/yr, although this estimate was qualified by the comment that the bores were most likely non-representative of the area, being too close to historically polluting sites. The ANZECC guideline concentration for phenol is quite high at 50 µg/L.

3.1.5 Atmospheric inputs

It has been estimated that more than 167 kilotonnes of volatile organic compounds are emitted annually in the Port Phillip Bay region (Carnovale *et al*. 1992), but the amount reaching the Bay is unknown. In terms of toxicants of concern, PAHs would be prominent, however, again it is not possible to estimate their concentrations. As will be shown, the concentrations of PAHs in Bay waters and sediments suggest that this source is not of any significance.

3.2 Organics in Bay waters

As already discussed, the soluble concentrations of hydrophobic organics are likely to be extremely low in Bay waters. Indeed, where they are detected, it is highly probable that they are associated with colloids as well as being attached to larger suspended particulates.

Previous studies have not found either dissolved or particulate organics to be present in high concentrations in the Bay. Total hydrocarbon concentrations were in the µg/L range and PAHs were at ng/L concentrations in both particulate and dissolved fractions. The particulate association results in the concentrations being quite variable in space and time (Phillips *et al.* 1992).

As part of the PPBES [Task T2], a range of organics were measured in water samples collected from 25 sites in the Bay in August-September 1993 (Good and Gibbs, 1995a). Compounds studied included petroleum hydrocarbons, PAHs, organochlorines, PCBs, chlorinated phenols, organophosphorus insecticides and triazine herbicides. As for metals, the highest concentrations of most organics were found near the known point source areas.

Corio Bay had by far the highest concentrations of hydrocarbons, including total hydrocarbons, oils, n-alkanes and PAHs (Good and Gibbs 1995a).

TPH concentrations in Corio Bay near the oil refinery were as high as 6 µg/L but considerably below the 22 µg/L found in 1980 (Burns and Smith 1982). Naphthalene was the dominant PAH present. Given the present limited database, the objective for total aromatic hydrocarbons set by the EPA in its SEPP is not likely to be exceeded except in Corio Bay and Hobsons Bay near the mouth of the Yarra River.

Only traces of dieldrin could be confirmed in most samples and other organochlorine insecticides were close to or below detection limits of 0.2 ng/L. No PCBs, triazine herbicides, organophosphorus insecticides or chlorinated phenols could be detected. Concentrations of all organic contaminants were well below the ANZECC (1992) water quality guideline values (Table 4.8).

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Table 4.8: Organic toxicants in Bay waters (Good and Gibbs 1995a).

Organic Compound	Bay Concentration (ng/L)	ANZECC Water Quality Guideline (ng/L)
TPH	100-6,000	–
Total PAHs	5-37	300
Total PCBs	<4	4
Total organochlorine insecticides	0.1-0.4	2-10
Triazine herbicides	<2.5	
Organophosphorus insecticides	<1	4-100
Chlorinated phenols	<3	5

3.3 Organics in sediments

As part of the PPBES [Task T7], Good and Gibbs (1995b) carried out the first comprehensive study of organic contaminants (petroleum hydrocarbons, polynuclear aromatic hydrocarbons, organochlorine and organophosphorus insecticides, chlorinated phenols and triazine herbicides) in Bay-wide sediments. They analysed sediments from 42 sites within the Bay (Figure 4.5).

They found that the occurrence of elevated concentrations of organic toxicants was mostly restricted to sediments in nearshore areas immediately adjacent to major stream inputs, although some anomalies were found with the distribution of PAH in surface sediments and DDT residues in one core. WTP was not a significant source because of effective removal in the lagoon system.

Hydrocarbons

Petroleum hydrocarbon concentrations ranged from <10 mg/kg in sediments from the centre of the Bay to 810 mg/kg in sediments from the Yarra River (Site 23, Figure 4.12). The concentrations in Corio and Hobsons Bays sediments were little changed from values during the late 1970s (Table 4.9).

The distribution pattern for total PAHs was similar to that of petroleum hydrocarbons (Figure 4.13). The elevated concentrations at Sites 15-19 may represent deposition of contaminated sediments from the Yarra River catchment carried into the Bay during high flow periods, or aerial deposition of PAHs from Melbourne or industrial activities in the western suburbs.

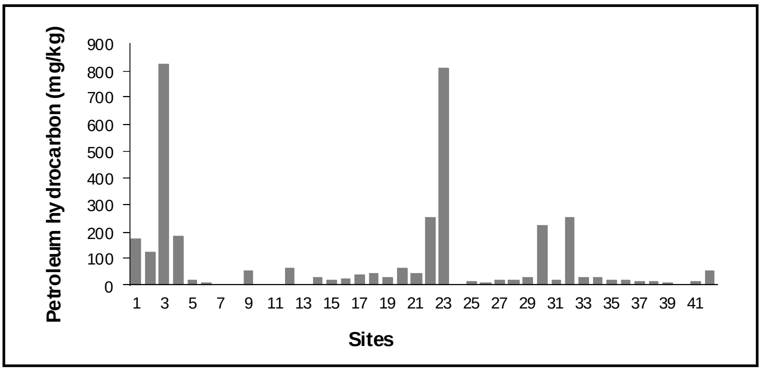

Figure 4.12: Petroleum hydrocarbon concentrations in Bay sediments (mg/kg dry weight) (Good and Gibbs 1995b).

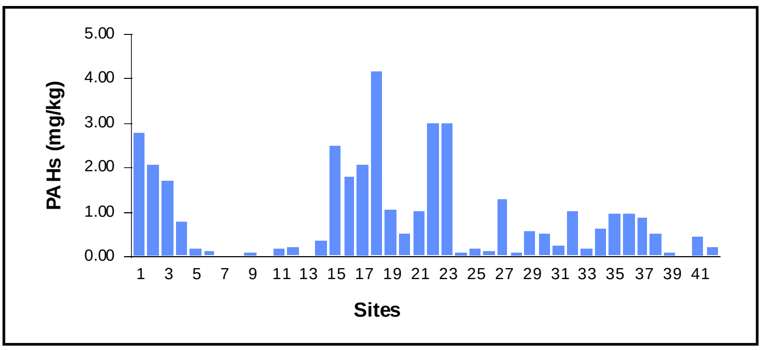

Figure 4.13: Total PAHs in Bay sediments (mg/kg dry weight) (Good and Gibbs 1995b).

A comparison of benzo[a]pyrene concentrations with previous data (obtained using different methods) from a limited number of sites suggests that PAH concentrations may be lower now than in the 1980s (Good and Gibbs 1995b).

Organochlorine compounds

Of the 10 organochlorine pesticides analysed for in sediments (hexachlorobenzene, a-benzene hexachloride, b-benzene hexachloride, heptachlor, heptachlor epoxide, aldrin, dieldrin, DDE, DDD and DDT) only dieldrin and DDT (plus derivatives) were found at more than one site.

Significant DDT concentrations were only found in sediments from Hobsons Bay and the Yarra mouth and ranged from 20 to 1100 ng/g. The sediment quality guideline is 7 ng/g.

The few data available for concentrations of polychlorinated biphenyls (PCBs) in sediments from the more contaminated areas of the Bay (Corio Bay and Hobsons Bay near the mouth of the Yarra River) indicate that they have not changed appreciably since measurements were first made in the late 1970s. Total PCBs in Corio Bay sediments were from 100 to 300 ng/g and in the same range in Hobsons Bay (Good and Gibbs 1995b).

The concentrations of chlorinated phenols were below the limit of detection (3 ng/g approx.) as were organophosphate insecticides (<20 ng/g) and triazine herbicides (<100 ng/g).

Interest in the concentrations of dioxins (PCDDs) and dibenzofurans (PCDFs) in the Bay followed concerns that these compounds were being discharged from the Nufarm chemical plant via the WTP. Measurements of their concentrations in WTP effluent showed less than 0.08 ng/L of the more toxic 2,3,7,8 tetrachlorodibenzodioxin (TCDD). Sediment concentrations ranged from 9 to 62,000 pg/g dry wt for total PCDDs and 0.2 to 1,700 pg/g for PCDFs (Bremner *et al.* 1990).

Table 4.9: Petroleum hydrocarbon concentrations in Bay sediments (mg/kg dry weight).

Period	1970s	1980s	1990s[d]
Corio Bay (>1 km offshore)	29 – 1,520[a]		170 – 180
		78 – 810[b]	
		21 – 86[c]	
Corio Bay (<1 km offshore	105 – 1,520[a]		120 – 820
including stream and drains)		21 – 730[c]	
Hobsons Bay (including	62 – 950[a]		250 – 810
Yarra River mouth)		72 – 310[b]	
WTP (offshore)		3 – 5[b]	<10 – 20
PPB (>1 km offshore)	6 – 120[a]		<10 – 40
PPB (<1 km offshore			
including streams and drains)		50 – 250	

a. *Burns and Smith (1982) – sampled during 1976, and 1977.*
b. *Murray (1987) – sampled during 1984.*
c. *Fabris* et al. *(1992c) – sampled during 1988*
d. *Good and Gibbs (1995b) – sampled during 1993.*

CHAPTER 4

Congener profile patterns indicated that the major source of these compounds was run-off from the Yarra/Maribyrnong catchment, but the Western Treatment Plant at Werribee and the petroleum industry were also sources (Bremner *et al.* 1990). The concentration of 2,3,7,8-TCDD was less than 0.3 pg/g in Bay sediments and 1.2 to 4.3 pg/g in sediments from the estuaries.

The figures quoted above do not support the rather sweeping conclusion of Ruchel (1995) that the Bay is "extensively contaminated ... with organochlorine compounds".

The data obtained from the studies of Bay sediments indicate that the concentrations of these compounds do not represent an environmental threat.

Shipping and dredging

It has been calculated that the material dredged from the Port of Melbourne contains TPH levels 21 times greater than that in an equivalent volume of Bay sediment (Fabris *et al.* 1995). Port of Geelong dredge spoil content ratios are similar.

Dredging operations for the Port of Geelong, Point Lillias and the East Coast Armaments Complex are likely to lead to increased concentrations of petroleum hydrocarbons in the sediments of the spoil grounds, with the Inner Harbour spoil grounds being more affected than the Outer spoil grounds.

The effects, if any, of these transfers on the water quality and biota of the Bay (in the case of the Geelong regional developments) will be monitored by a comprehensive program which includes provision for ameliorative measures to be taken in the event that environmental problems are detected (PGA 1993).

Dumping of sediments from the Port of Melbourne on spoil grounds in the Bay resulted in localised increases in concentrations of PAHs and TPHs, although these were still well below the values measured in the ports or at sites closer to point sources such as the Yarra River or inputs into Corio Bay.

3.4 Organics in biota

Although low concentrations of organics were measured in waters and sediments, if bioconcentration factors are high it is possible that fish, molluscs and other biota might contain concentrations that exceed guideline concentrations.

For example, fish readily accumulate petroleum and organochlorine hydrocarbons from water, sediment and food through their gills and digestive systems and store them in fatty tissues and organs.

The accumulation of chlorinated hydrocarbons is most critical because they are not readily metabolised and can therefore accumulate to concentrations that cause toxic effects, or reduce an organism's resistance to parasites and diseases, or its ability to reproduce.

Other compounds, such as PAHs, may not be directly toxic but may be metabolised to more toxic compounds that induce tumours and lesions in fish.

Consumers of contaminated fish may be adversely affected (Phillips and Rainbow 1994). The effects may range from flavour impairment to health hazards through the ingestion of toxic, carcinogenic or disease-initiating substances.

Toxic effects of acute PCB ingestion by humans include acneform lesions, liver damage and immunosuppresssion.

Petroleum hydrocarbons

There are no official maximum permitted concentration health limits for petroleum hydrocarbons in marine organisms. Data obtained for hydrocarbons in mussels over the past 25 years lie in the range 14 to 700 mg/kg wet weight in samples from Corio and Hobsons Bays and lower in samples from other sites in the Bay (Fabris *et al*. 1992b).

Petroleum hydrocarbons were found in all sand flathead fillet and liver samples collected during a 1990 survey (Table 4.10; Nicholson *et al.* 1991). The highest mean concentrations were found in the fillets of fish from the Geelong Arm and the lowest from fish within Corio Bay and the eastern side of the Bay. Concentrations in the liver samples were less variable.

Polycyclic aromatic hydrocarbons

Polycyclic aromatic hydrocarbons (PAHs) have been implicated as causative agents for many fish diseases. All fillets and liver samples analysed by Nicholson *et al.* (1991) contained very low or undetectable concentrations of PAHs and their metabolic products.

Total PAHs detected in mussels from Corio Bay ranged from 0.03 to 1 ng/g wet weight. (Fabris *et al*. 1992b).

Organochlorine pesticides

Chlorinated pesticides are persistent contaminants with undesirable biological effects that are readily accumulated by marine organisms. For this reason some previously

Table 4.10: Mean concentrations (mg/kg wet weight) of organic compounds in the liver and muscle tissue of sand flathead in PPB (Nicholson *et al.* 1991, site locations in Fabris *et al.* 1992a).

Site	TPH	PCB	DDT	Lindane	Dieldrin	Heptachlor
16 (fillet)	<0.1	0.014	0.003	<0.001	0.004	<0.007
(liver)	250	0.350	0.250	0.022	0.023	0.046
19 (fillet)	1.8	0.014	0.049	nd	0.002	<0.001
(liver)	86	0.350	0.140	<0.001	0.009	nd
24 (fillet)	0.1	0.031	0.009	<0.003	0.009	0.006
(liver)	380	0.710	0.360	0.005	0.052	0.009
25 (fillet)	23	0.009	0.005	<0.002	0.002	<0.004
(liver)	160	0.580	0.350	0.041	0.230	0.059
26 (fillet)	0.1	0.009	0.012	nd	<0.001	nd
(liver)	140	0.117	0.306	nd	0.010	nd
29 (fillet)	1.0	0.019	0.017	<0.002	nd	<0.001
(liver)	130	0.350	0.610	0.006	0.210	0.008
30 (fillet)	7.3	0.0004	0.004	nd	<0.001	nd
(liver)	75	0.107	0.086	nd	0.008	nd
33 (fillet)	11	0.047	0.033	nd	<0.002	nd
(liver)	18	0.310	0.220	nd	0.008	nd
34 (fillet)	0.1	0.008	0.006	nd	0.002	nd
37 (fillet)	3.4	0.009	0.004	<0.001	<0.002	<0.001
(liver)	70	0.099	0.039	0.006	0.020	0.002

common pesticides, such as DDT and dieldrin, have been banned from use in Victoria.

Some sand flathead sampled during 1990 were found to contain traces of lindane and heptachlor in both the fillets and livers, but traces of dieldrin were found in fish from all but one site (Nicholson *et al.* 1991). DDT (and derivatives) was also widespread in the fish samples (Table 4.10).

Dieldrin, heptachlor and DDT were in all cases below the maximum residue limit (MRL, Table 4.11) in all fillet and liver samples analysed, except one liver sample from Corio Bay (Site 25) and one liver sample from the Central Bay (Site 29) (Table 4.10).

The concentrations of organochlorine pesticides determined in fish samples collected during 1994 were all below 0.1 µg/g (Fabris 1995)

Concentrations of individual organochlorine and organophosphate pesticides in mussels from Corio Bay sampled during 1987 were mostly below 0.02 µg/g (wet weight), although many were detected at all sites (Fabris *et al.* 1992b).

Chlorophenols and dioxins

As with sediments, the analysis of biota for chlorophenols and dioxins was prompted by the concern over Nufarm discharges to WTP in 1991. At that time only the Werribee area of the Bay was sampled for both fish and mussels.

Traces of chlorophenols, including dichloro- and pentachlorophenol as well as PCDDs and PCDFs, were detected in both fish species and in mussels from a reference location in Balcombe Bay (EPA 1991).

Concentrations of the less toxic (non-2,3,7,8-TCDD and TCDF) groups in mussels from the WTP were more than one order of magnitude above those in the reference mussels

The concentrations of these compounds in Bay fish were at the lower end of the range of international data (EPA 1991). Qualitative analyses of the data for the various congeners indicate that the sources of the PCDF and TCDF in the Werribee area were of industrial origin.

The highest concentration of the more toxic 2,3,7,8-TCDD found in fish and mussel samples was a value of 0.45 pg/g in flathead muscle (3.25 pg/g in liver) and 0.68 pg/g in mussels. These are more than an order of magnitude below the 25 pg/g limit set by the US FDA as the dioxin limit of concern for human consumption.

Anecdotal evidence of tainted fish in the Bay appears from time to time. During the course of the sand flathead study, one sample of Australian salmon (*Arripis trutta*) with a 'kerosene taste' was found to contain 30 µg/g wet weight of phenol (Nicholson *et al.* 1991).

Polychlorinated biphenyls

Mussels were used to record the first comprehensive surveys of PCBs in the Bay from 1980 to 1985 (Phillips *et al*. 1992). These studies showed that PCB contamination was widespread and that the highest concentrations in mussels were associated with industrialised and urbanised localities in the Bay's catchment (Corio and Hobsons Bays).

Subsequent surveys during the late 1980s indicated that PCB contamination in mussels from Corio Bay had declined (Phillips *et al.* 1992).

Mussels from Corio Bay sampled during 1987 contained PCBs at 1-4 ng/g wet weight (Fabris *et al.* 1992b). In all cases these concentrations are less than one hundredth of the MRL (Table 4.11).

A recent survey of the WTP effluent using translocated mussels deployed for up to 12 weeks offshore of Werribee and at a Clifton Springs reference site, found PCBs, OCs and PAHs were accumulated at concentrations below the most conservative criteria for the protection of mussels and human health (van Maanen 1995).

A comparison with previous data from the WTP site suggested a marked improvement over the past decade.

There is some persistence of PCBs in the Bay, since they were detected in fish from all 10 widely spread sites sampled during 1990 (Nicholson *et al*. 1991).

At all sites, however, PCB levels in fillets were below the MRL of 0.5 mg/kg, while only fish livers exceeded the MRL for PCBs in samples from Corio Bay (0.71 mg/kg) and the Geelong Arm (0.58 mg/kg).

Table 4.11: Maximum Residue Limits for organochlorine pesticides and PCBs in seafood (Food Standards Code 1995).

Compound	Food type	Maximum Residue Limit mg/kg (wet weight)
Aldrin and Dieldrin	Marine fish	0.1
	Molluscs	0.1
BHC (except Lindane)	Marine fish	0.01
	Molluscs	0.01
Chlordane	Marine fish	0.05
	Molluscs	0.05
DDT	Marine fish	1
	Molluscs	1
Heptachlor	Marine fish	0.05
	Molluscs	0.05
Lindane	Marine fish	1
	Molluscs	1
Polychlorinated biphenyls	Fish	0.5

4. Fish health

Measurement of bioaccumulation determines whether seafood is safe to eat but will not necessarily indicate whether contaminant substances exert sublethal effects on biota.

The presence of visible anatomical abnormalities in fish could be indicative of exposure to doses of toxic substances that cause sublethal effects. However, the absence of obvious abnormalities or skin lesions in fish does not preclude toxicant-induced (anthropogenic or natural) sublethal effects.

During a major fish kill in the Bay during 1984 (Gibbs *et al.* 1986), flathead showed no external abnormalities but did show severe visceral haemosiderosis.

More sophisticated approaches (biochemical or physiological indicators) are needed to expose such less obvious sublethal effects (Phillips and Rainbow 1994).

The design of this Study did not include any such measures of pollution stress, so an assessment of whether the concentrations of toxicants in the biota from the Bay pose a concern to the health of the biota could not be made.

Given the low concentrations detected, reference to overseas data suggests that such effects, if any, might be limited to only a few key areas.

Methods for assessing sub-acute response are relatively new and, in many cases, are still being developed. While in the past fish health studies have been carried out in response to visible effects such as lesions, in future investigations of the Bay it would be appropriate to apply currently available techniques for evaluating sub-acute toxicant effects in the most contaminated areas of the Bay.

5. Conclusions

Port Phillip Bay receives inputs of toxicants from a range of sources, the most significant being the many rivers, creeks and streams that carry the drainage from agricultural and urban runoff, as well as some industrial discharges.

The Yarra River contributes more than half of the input of most metal toxicants and is a likely source of a lot of the organics.

The input from the Western Treatment Plant contributes a smaller and more manageable source of toxicants, of which metals are probably more significant than organics.

Other inputs are relatively insignificant although poor location of bores has prevented an accurate evaluation of groundwater inputs from aquifers near contaminated sites in the western suburbs as a potential source of toxicants.

Despite what appear to be high annual loads, particularly from rivers and streams, the concentrations are not excessive and sedimentation and transport of inputs has resulted in Bay waters that are well within accepted water quality guideline values for both heavy metal and organic toxicants. Their concentrations are at the low end of the ranges for estuarine and nearshore waters in developed countries world-wide and in many cases approach those in unimpacted coastal waters.

Highest concentrations of most toxicants were found close to the input sources, particularly in Corio Bay and Hobsons Bay.

Particulate matter entering via the input streams, or depositing as a result of mixing of these fresh waters with the saline waters of the Bay, will carry with it many contaminants bound to iron oxides and natural organic matter.

Sediment deposition occurs near the inputs under low flow conditions, although during storm events these sediments, as well as the less contaminated material washed into the rivers, will be transported further and deposited closer to the centre of the Bay.

The sediments of the Bay represent a sink for many of the toxicants entering via rivers and creeks and depth profiles show that elevated metal concentrations are present to considerable depths, which is not unexpected since contaminated sediments have been entering the Bay for more than 100 years.

Nevertheless, for the most part the concentrations of contaminants in the sediments were below what are currently accepted sediment quality guidelines.

There were exceptions for some heavy metals in or close to inputs at Rippleside Beach, Cowies Creek, Corio Bay, Yarra River and Kananook Creek, as well as in port areas. Again such higher concentrations would be typical of most harbours world-wide, whereas the sediments of the Bay generally, and especially the clay/silt sediments in the Bay centre, have quite low toxicant contents.

Fish, mussels and other biota integrate toxicant loads which may result in sub-acute impacts to these organisms or to human consumers. Localised high concentrations were found in species collected near the point sources, but all were below maximum permitted concentrations for human consumption.

There were minor exceptions for some organics in the livers of sand flathead, because these species preferentially bioaccumulate in lipid tissue. However, the more important fish muscle was considered safe to eat.

No measures of toxicant effects on fish health were investigated apart from a visual examination for lesions, which were absent. Future studies would be advised to consider studies of sub-acute affects of toxicants.

Contents

1. Introduction

The ecosystem of Port Phillip Bay, like all healthy marine ecosystems, is diverse and complex. It includes organisms which range in size from microscopic (the floating algae or phytoplankton and small grazing animals – the zooplankton) through to large organisms such as seagrasses, fish, seals and dolphins. Animals such as seabirds and waders range over large distances and occasionally use the Bay during migration or at certain times of the year.

There is much variability in the distribution and abundance of organisms both in space and time in response to weather and climate variability. It was, therefore, logistically impossible to study everything in the entire Bay ecosystem; it is too large, there are too many organisms and there are a myriad of factors that can affect the Bay's ecology on a wide range of spatial and temporal scales.

Because of this the Port Phillip Bay Environmental Study (PPBES) focused its ecological program on the concerns of management agencies and community groups. These concerns were based around the likely impact on the Bay's ecology of past, present and future inputs of nutrients and toxicants and the effects of human activities.

Important issues included the need to maintain a rich biodiversity of organisms including healthy fish, the ability to predict toxic or nuisance algal blooms and whether the ecosystem of the Bay had changed for the better or worse over time.

The ecological tasks of the Study were designed to describe the present status of the Bay ecosystem over the period of the Study. Any change in the state of the Bay could be determined by comparing conditions now with those found earlier, particularly during the Phase One Study of 1969 to 1971.

The results of the Phase One Study indicated that the Bay was in relatively good shape at that time (see Chapter 1).

In general, chlorophyll *a* concentrations in the water (used to estimate phytoplankton abundance) were low compared with similar embayments and estuaries in Australia and the world, although elevated concentrations of phytoplankton were sometimes associated with proximity to nutrient inputs.

The distribution and abundance of macroalgae and seagrass appeared to be mainly determined by physical factors – such as the degree of exposure to winds and currents, the stability of the substrate and the availability of light underwater – rather than by nutrient levels.

Surveys of bottom-living (benthic) animals revealed a high diversity with a strong correlation between the distribution and abundance of animals and sediment type.

Noticeable effects on benthos from WTP effluent discharge are largely restricted to within 300 m from an outlet.

Trawl surveys of fish within the Bay suggested that fish distribution also corresponded with sediment type, although catches varied from place to place and with season. Proximity to major input sources did not appear to adversely affect fish abundance.

This was the base line which the Port Phillip Bay Environmental Study could use to assess changes over time and the present ecological health of the Bay. A more complete summary has already been given in Chapter 1.

2. What did we do and why?

The Port Phillip Bay Environmental Study design translated ecological concerns into three broad scientific objectives:

- to determine the general status ('environmental health') of the Bay;
- to determine whether the status of the Bay is changing over time; and
- to assess the impacts of nutrients and toxicants on the Bay's ecology.

In order to address these objectives a series of tasks were designed ranging from desk-top studies that collated and reviewed existing local ecological information, workshops of scientists and user/community groups to obtain additional knowledge and the views of local experts, through to a series of field tasks that specifically addressed knowledge gaps regarding the Bay's biota.

Indicators of ecosystem health can be chosen at the sub-organism level (e.g. tissue nitrogen content of macroalgae), at individual level (e.g. reduced egg production capacity), at population level (e.g. changes in abundance or total biomass) or at the community/assemblage level (e.g. changes in biodiversity).

Successful indicators should be causally linked to nutrient or toxicant loads or other influences and need to be easily monitored. In addition, changes in indicators due to natural biological variability (at various time and space scales) need to be unambiguously separated from potential nutrient/toxicant effects.

Most of the indicators chosen in the PPBES were at the population and assemblage levels.

The Study focused on groups of organisms that were either of direct concern to management agencies and the community, such as fish, or those that would act as indicators of change within the Bay, such as the type of benthic infauna (animals living in the surface layers of the seabed).

Desk-top reviews covered a broader range of organisms to provide a more complete picture of information available on the ecology of the Bay.

Areas covered by the specialist reviews included phytoplankton (Wood and Beardall 1992), benthic flora (Light and Woelkerling 1992), benthic fauna (Poore 1992, Wilson *et al.* 1993, Bird 1994), fisheries (Nicholson 1992) and water column trophodynamics (Crawford *et al.* 1992, Holloway and Jenkins 1993).

A later review examined empirical evidence world-wide linking water and sediment quality to benthic community change (Nielsen and Jernakoff 1996).

The Study also considered earlier publications such as an appraisal of the effects of the Western Treatment Plant on Port Phillip Bay (Newell, 1990), the identification and use of indicators for Victorian coastal and marine environments (Colman *et al.* 1991) and a report to the Victorian EPA on biological monitoring in Port Phillip Bay and its relationship to inputs of nutrients and toxicants (Quinn and Keough 1993).

The field tasks were designed to provide information on the current ecological status of the Bay and generally used traditional sampling methods.

Other tasks used cutting-edge technology and, although some of the techniques were still developmental, they offered the potential to provide significant benefits to the Study.

One of these techniques was the use of a multispectral airborne scanner to map submerged habitats.

Another used underwater video film to address problems of describing benthic habitats at various spatial scales. The video tapes provide a data set that can be revisited for additional information in the future and a baseline from which to assess further change.

Both the remote sensing and video surveys provided Bay-wide information on the distribution of benthic habitats. The results of these studies will be discussed later.

Other field tasks were also designed to build on previous data sets of the Bay and to provide information on the current distribution and abundance of organisms.

Field tasks included studies of phytoplankton [Task N2.2], large benthic plants [Task N2.3], microphytobenthos [Task N2.4], benthic infauna and epifauna [Task E5.2] and fish distribution and productivity [Task E8].

An evaluation of an existing five-year database on the Bay's phytoplankton was also commissioned [Task E10.4].

Access was also obtained from the Victorian Fisheries Research Institute (VFRI) to a comparative study of fish trawl surveys between 1969 and 1990.

3. What did we find?

The PPBES provided no evidence to suggest a significant effect of toxicants on marine biota. Heavy metal concentrations within the biota studied (fish, mussels and macroalgae) were well below those laid down in the Australian Food Standards Code (Fabris 1995).

There is even some evidence (from sand flathead and mussels) to suggest that heavy metal contamination of the Bay and its biota has in fact decreased over the past 20 years (see Chapter 4).

Given the current levels of toxicants within the Bay, no studies were carried out examining sublethal effects of toxicants on the biota.

The factor of primary concern to the Bay's ecological health is, therefore, the effect of increased nutrient loadings and the possibility of eutrophication.

3.1 General world-wide patterns of eutrophication

Clear patterns of community change in relation to eutrophication can be identified and have been qualitatively described in coastal bays and estuaries throughout the world.

For primary producers, there is a general shift from large, slow-growing marine plants (macroalgae and seagrasses) towards ephemeral algae and phytoplankton which are better able to assimilate higher nutrient loadings (Nielsen and Jernakoff 1996).

An increased production of free-floating, ephemeral macrophytes, epiphytes and phytoplankton can increase water turbidity with subsequent negative impacts on marine flora living on the seabed.

As bays become more eutrophic, therefore, we tend to see a loss of seagrasses and the development of blooms of phytoplankton in more turbid water (Duarte 1995).

As eutrophication proceeds, increased sedimentation of organic material may occur which promotes deposit-feeders and increases benthic respiration leading, in turn, to greatly reduced oxygen levels in the sediments (hypoxic conditions).

The reduced oxygen levels cause stress in the benthic organisms and, as oxygen levels decline further to a point where virtually no oxygen is left (anoxic conditions), animals on and in the sediments die.

Thus, for secondary producers, an overall reduction in species diversity and abundance is a typical response to eutrophication.

There is a general loss of larger size classes of fauna and a shift towards small opportunists generally dominated by deposit-feeding polychaetes (Pearson and Rosenberg 1978, Gray 1992).

Gray (1992) summarised these stages as:

1. An initial growth response and increase of primary and secondary producers;
2. A change in species composition: for primary producers generally towards ephemeral, opportunistic species and for fauna from large, slow-growing, long-lived species with low production: biomass (P:B) ratios towards smaller, fast-growing, short-lived species with higher P:B ratios;
3. A loss of species and dominance by a few opportunistic species;
4 Hypoxia, anoxia and mass death of the benthos, leaving just bacterial mats and perhaps a few hardy species.

Advanced eutrophication is therefore characterised by a trend towards a loss of species diversity, excessive production by phytoplankton or free-floating macroalgae and a reduction in secondary production and diversity (Figure 5.1).

For benthic flora such as seagrass (although loss has been attributed in the international literature to the effects of eutrophication (e.g. Nienhuis 1983, Walker and McComb 1992)), there is no direct link between nutrient loadings and seagrass decline (Nielson and Jernakoff 1996).

However, studies demonstrate that light levels in the water column and seagrass abundance are highly correlated. Seagrasses decline rapidly under low light regimes and are slow to recover, even months after light levels are restored.

When trying to predict the likely effects of eutrophication on seagrass meadows, measuring relationships between nutrients and light attenuation (either caused by declining water quality or by increased abundance of epiphytic algae growing on and shading the seagrasses) may be more important than direct observations of nutrient effects on the seagrasses themselves.

Large blooms of floating macroalgae, typically green algae such *Ulva*, *Cladophora* and

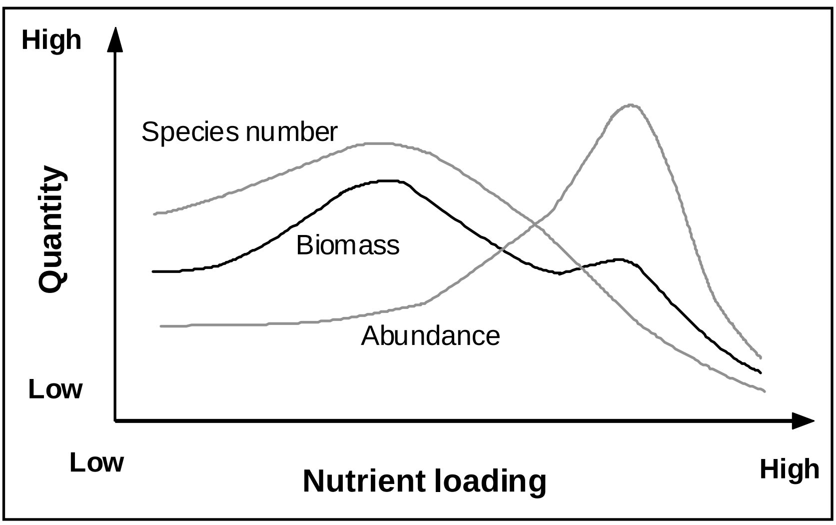

Figure 5.1: Trends in species number, biomass and abundance of marine biota in response to increasing levels of eutrophication (adapted from Gray 1992).

Enteromorpha, are a commonly reported symptom of eutrophication.

Microphytobenthos have also been reported to change from a diatom-dominated assemblage to one dominated by cyanobacteria. These phenomena have not been observed in Port Phillip Bay.

Temporary increases in fish abundance, growth rate, biomass and productivity have been reported in the scientific literature from areas experiencing eutrophication.

Correlation between nutrient loads and fish abundance, biomass and production are difficult to quantify owing to complex links between these parameters. However, changes in community size-structure and species composition have been related to eutrophication.

This is usually a result of changes in the quality of benthic habitats or periods of oxygen depletion in the water column.

The effects of anoxia on fish are direct and relatively easily measured so that declines in fish biomass have been directly linked to low water column oxygen concentrations.

Fish production may initially rise as eutrophication proceeds, but will eventually fall rapidly if anoxia ensues. This pattern also occurs for benthic invertebrates.

3.2 Broad comparison of Port Phillip Bay with other areas

Comparison of the concentrations of nutrients and chlorophyll in the waters of the Bay with other estuaries and embayments near to major cities (Boynton *et al.* 1982, Monbet 1992) shows that there is a relative dearth of inorganic N in the water (about 0.5 μM) and very low concentrations of chlorophyll. Inorganic N:P ratios are very low (see Chapter 3).

The Bay rates as oligotrophic to mesotrophic when compared to the more highly affected northern temperate estuaries in Europe and North America reinforcing the view that water quality in the Bay is good by world standards.

Comparisons with data from the literature (e.g. Boynton *et al.* 1982, Monbet 1992) show that Port Phillip Bay has relatively few problems with phytoplankton blooms that can cause taint or contaminate seafood even though occasional blooms of noxious species have occurred.

Benthic invertebrate communities in the Bay are abundant and diverse compared with other coastal ecosystems (Nixon *et al.* 1986).

There are abundant macrofauna in the Bay and the mean biomass (25 g dw/m^2) compares very well to the highest levels observed in coastal sediments around the world (Nixon 1988).

Studies of nutrient, toxicant and isotope profiles in sediment cores taken during the PPBES have indicated active and deep bioturbation by benthic invertebrates to 30-50 cm depth.

Not only do such organisms mix the sediments and enhance decomposition of organic matter but irrigation of sediments by invertebrates can also stimulate nitrification and subsequent denitrification (Pelegri *et al.* 1994).

A review of bioturbation in Port Phillip Bay has been given by Bird (1994).

It is difficult to compare the state of seagrasses in the Bay with other areas because different seagrass species are affected to different degrees by environmental and anthropogenic impacts.

Unlike most other embayments in Australia and overseas, the main seagrasses in the Bay are *Heterozostera tasmanica* and *Zostera muelleri.*

Western Port has a similar combination of species as Port Phillip Bay but in Western Port large areas of seagrass were lost during the 1970s and 1980s (Hamdorf and Kirkman 1995, Stephens 1996).

The extent of seagrass change in Port Phillip Bay is still uncertain but large healthy meadows still exist and are of prime importance within the Bay as fish feeding and shelter areas.

Port Phillip Bay has the most productive bay and inlet commercial fishery in Victoria with more than 60 species of finfish being recorded from commercial catches (Hall and MacDonald 1986).

Finfish production per unit area may be lower in comparison with other Australian locations such as Sydney Estuary, Jervis Bay, Botany Bay, Tuggerah Lakes and Cockburn Sound. However, including shellfish makes Port Phillip Bay better than those locations in terms of total fishery production (Gwyther 1990).

3.3 Specific patterns within the Bay

3.3.1 Phytoplankton

The distribution of phytoplankton in the Bay is complex in both time and space with approximately half of the large variability in numbers of cells and chlorophyll concentration due to differences between sampling sites (Magro *et al.* 1996).

This variability is dependent upon a multitude of factors including runoff from the surrounding catchment (especially large rainfall events), light, temperature, nutrient availability and grazing by zooplankton.

A decrease in phytoplankton biomass in 1994 coincided with drought and a consequent decrease in nutrient loads from the Yarra River and the eastern creeks and drains (Longmore *et al.* 1996).

Interannual variability in climate over southern and eastern Australia is very much an effect of El Nino Southern Oscillation (ENSO) events. Previously recorded outbreaks of algal blooms in 1975, 1987 and 1993-94 coincided with unusual climatological conditions of warm, calm, sunny winters. At least 1987 and 1994 were ENSO years.

The marked interannual variability in algal biomass in the Bay leads to the conclusion that the phytoplankton community of the Bay is highly sensitive to the volume and quality of the catchment runoff.

The nutrient loads to the Bay are split about equally between the WTP and the Yarra River plus the major creeks and drains. At present the load is about 3,000 t N/yr from each source, with an additional 1,000 t N/yr from the atmosphere (Table 3.1).

The time course of the loading is, however, quite different in each case. The WTP load is relatively constant over time with a dominant seasonal component (Murray 1994) whereas the Yarra load is highly sporadic showing great seasonal and interannual variability. (see Chapters 1 and 3).

In most Australian river systems 80-90% of the loads discharge in less than 10% of the time (Harris 1994a). The main storm loads from the Yarra discharge during summer whereas the discharge loads from the WTP peak during winter when temperatures are low and biological activity is reduced.

The growth of large algal blooms in waters of the north-east corner of the Bay following rain events (Grant and Gayler 1982, Longmore *et al.* 1996, Magro *et al.* 1996) indicate that transient storm induced loads have a more noticeable effect on the ecology of the Bay than do the more constant background loadings from the WTP.

Routine monitoring of phytoplankton by VFRI (Arnott *et al.* 1996) shows a trend of decreasing species diversity in an anticlockwise direction with highest values off Blairgowrie and lowest values off Clifton Springs. Mean chlorophyll *a* is highest off Werribee, Williamstown and Beaumaris, decreasing gradually down the Bay to a minimum off Blairgowrie (see Plate 4).

Despite large spatial and temporal variability, however, there is no conclusive evidence to suggest that mean phytoplankton abundance in the Bay has changed over time since the Phase One surveys (Longmore *et al.* 1996, Magro *et al.* 1996, see also Chapter 7).

The phytoplankton population inhabiting the Bay consists of approximately 205 species or taxa which include 76 diatom and 52 dinoflagellate groups.

The diatoms *Skeletonema costatum* and *Chaetoceros* spp. are currently the most numerically dominant species. The highest concentrations of *Skeletonema* are found in summer and autumn particularly at Williamstown and Sandringham while *Chaetoceros* spp. shows no significant seasonal or spatial variation. *Skeletonema* has decreased in abundance from 1990 to 1994 – a probable result of reduced nutrient loads during the drought.

It is possible to compare the species composition found during Phase One and the current surveys. The most dominant group of phytoplankton within the Bay during the Phase One Study, the diatoms, and the most dominant species at that time, *Skeletonema costatum*, are still the most dominant in Port Phillip Bay (Table 5.1).

Table 5.1: Most abundant species of phytoplankton in Port Phillip Bay 1969-1971 and 1990-1995.

Phase One surveys 1969-1971	Current 1990-1995
Skeletonema costatum	*Skeletonema costatum*
Asterionella japonica	*Chaetoceros* spp.
Thallasiosira sp.	*Plagioselmis prolonga*
Nitzschia sp. Ref . *N. seriata*	*Pseudonitzschia pseudodelicatissima*
Nitzschia closterium	*Pseudonitzschia pungens*
Leptocylindrus minimus	*Leptocylindrus minimus*
Chaetoceros spp.	*Chrysochromulina* spp.
Cryptomonas spp.	*Pyraminimonas* spp.
Dinoflagellate sp1	*Gymnodinium* sp.
Dinoflagellate sp2	*Katodinium* spp.

However, some of the most common species from the more recent surveys of 1990-1995 (*Pyraminimonas* spp., *Plagioselmis prolonga*, *Pseudonitzschia pseudodelicatissima*, *Katodinium* spp.) were not recorded in the Phase One work.

Phytoplankton species consist of a variety of different sized cells. These are generally divided into three size categories: the microplankton >20 µm, the nanoplankton 2-20 µm and the picoplankton <2 µm.

In healthy waters, phytoplankton assemblages tend to be dominated by smaller sized cells which are most susceptible to zooplankton grazing, but in eutrophic waters the larger phytoplankton species are more abundant. Therefore it was important to determine the current size distribution of phytoplankton within Port Phillip Bay. This was addressed in Task N2.2.

The average contribution to biomass (as chlorophyll *a*) of the size fractions was 28%, 40% and 32% for micro, nano and picoplankton respectively. All three size fractions show higher mean annual concentrations of chlorophyll *a* nearer to the Yarra and WTP.

Briefly, coastal stations showed a trend towards higher production than mid-Bay stations. Productivity of the microplankton was highly variable ranging from 0 to 60% of total production.

Nanoplankton were more stable seasonally and ranged between 20% and 60% of production while the picoplankton were very stable making up 20% to 40% of the daily productivity (Figure 5.2).

Thus, the stable productivity of nano and picoplankton was supplemented by more variable periods of production by microplankton which bloomed in response to factors including light and inputs of nutrients. The largest microplankton (diatom) blooms occurred during summer along the north-eastern shore which is the part of the Bay most influenced by the Yarra outflow.

The surveillance of phytoplankton populations in Port Phillip Bay is important as an 'early warning' system for eutrophication. It also alerts authorities to the occurrence of toxic or nuisance species that have impacts

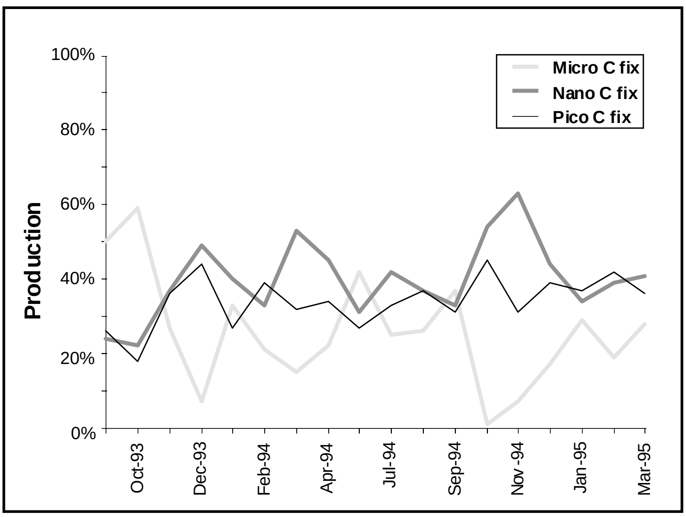

Figure 5.2: Bay-wide production (mean carbon fixation) percentage contribution by micro, nano and picoplankton (Beardall *et al.* 1996).

such as causing a bitter taste in shellfish (Arnott, 1990).

In summary, based on the present patterns of phytoplankton distribution, abundance and composition, Port Phillip Bay is in a relatively healthy state, showing no obvious signs of the onset of eutrophication and no evidence of long-term deterioration in water quality.

3.3.2 Seagrass

Eight species of seagrasses have been recorded from Port Phillip Bay and 95% are found in waters less than 5 m deep. The distribution is dynamic in space and time resulting from the interaction of a particular species with physical factors such as season, light and wave energy.

Prior to the current PPBES, seagrass distribution and abundance at a Bay-wide scale was documented by Willis (1966), Hope Black (1971), King *et al.* (1971), Spencer (1972) and Bulthuis (1981).

More detailed field and aerial photography studies of variable quality have been conducted at a number of other locations (e.g. Swan Bay, Bellarine Peninsula, Corio Bay, Werribee, Altona Bay) usually covering small areas or in response to specific issues (Light and Woelkerling 1992).

The major areas of the dominant seagrass in the Bay, *Heterozostera tasmanica*, occur along the southern edge of Corio Bay and the Geelong Arm as well as in Swan Bay, Prince George Bank and east of Portarlington.

On a Bay-wide scale, Bulthuis (1981) found that, in general, no major changes in seagrass distribution had occurred between 1957-1963 and 1978-1981. However, local declines in seagrass distribution had occurred around Point Wilson and the north shore of Corio Bay. Some local increases in cover occurred around Point Henry, west of Portarlington and between Rye and Rosebud.

Significant loss of seagrass along the south side of Geelong Arm in the mid-1980s was reported by Brown and Davies (1991). More recently a 25% increase in seagrass off Avalon Beach was recorded between 1990 and 1994 (Marine Science and Ecology Ltd 1995).

Additional information on specific studies of seagrass at specific locations can be found in the review by Light and Woelkerling (1992).

In summary, seagrass meadows still occur within the Bay and, in general, their number and extent have declined over the past 30 years. However, there are also areas within the Bay where seagrass meadows have increased over the same period. The distribution and health of seagrass meadows should be monitored owing to their sensitivity to environmental impacts and their value as nursery areas and habitat for marine organisms.

3.3.3 Benthic macroalgae

Variations in species composition and abundance of benthic macroalgae are evident throughout Port Phillip Bay and there are no conspicuous trends from available historical data (Light and Woelkerling 1992).

Several studies have been conducted, particularly near Werribee, and the distribution and abundance of macroalgae appears to be influenced by factors such as substratum type, wave action, currents, water depth and to a lesser extent proximity to effluent outlets (Brown *et al.* 1980).

As part of the PPBES, macroalgal distribution, composition and seasonality was measured quarterly at five sites in the Bay over a 12-month period [Task N2.3].

The greatest biomass of macroalgae was encountered at a depth of 8 m off Altona and Wedge Point and this consisted largely of red algae genera including *Botryocladia*, *Jeanerettia* and *Polysiphonia*.

Although these red algal assemblages were not attached to the seabed, their composition is distinct between depths, location and time of sampling (Chidgey and Edmunds 1996).

The other species which was common but

in lower proportions was the green alga *Caulerpa*. Previous studies offshore from Werribee have also reported large quantities of red macroalgae (e.g. *Jeanerettia* and *Botryocladia*) on the seabed. The habitat off Kirk Point is similar to that at Altona and Wedge Point although the biomass is lower and composed principally of *Caulerpa*, filamentous red algae and some *Ulva*. The macroalgal assemblages near Brighton and Aspendale consisted largely of *Caulerpa* attached to or associated with the filter-feeding sea squirt *Pyura*.

Large quantities of benthic drift (unattached) algae are present in the Bay but their distribution and abundance is difficult to quantify. Benthic drift algae were also recorded in Swan Bay during video surveys [Task G2.3]. The large amount of drifting red algae was one of the reasons the airborne CASI surveys (see below) produced ambiguous results. It is difficult to 'ground truth' beds of drifting weed which confuse spectral signals and are capable of covering other benthic community types.

Spencer (1970) reported that large quantities of drift *Ulva* were found near the Werribee outfalls and large quantities of *Botryocladia* were also common. Although the green alga, *Ulva*, is an indicator of nutrient enrichment, it was only recorded in relatively small amounts in this Study at the 4 m deep sites at Wedge Point, Altona and Kirk Point. Brown *et al.* (1980) reported significant biomass of *Ulva* on reefs in water shallower than 2 m off Werribee. The lack of significant beds of *Ulva* in the areas studied by Chidgey and Edmunds (1996) indicates that red algae species displace *Ulva* at depths greater than 2 m. This is an indication that water quality in the Werribee area has improved since the 1970s.

In summary, the soft sediments of the Bay between 4 m and 10 m are a significant habitat for macroalgae and there is an extensive band of macroalgae up to 5 km wide from Werribee to Altona (Chidgey and Edmunds 1996). Macroalgal assemblages are variable over a year although red algae tend to dominate the northern and north-western regions of the Bay and green algae are predominant in the north-eastern areas.

Macroalgal assemblages in the north-western parts of the Bay can blanket the sediments, thereby excluding microphytobenthos (see section 3.3.4).

Novel techniques of mapping benthic vegetation included the use of video surveys at selected locations in the Bay [Task G2.3]. These provide an historical record of sites that can be revisited accurately in the future using differential GPS. Results from the video surveys showed a similar pattern to those obtained from the macroalgal survey [Task N2.3].

The highest covers of macroalgae were generally evident in the north-western and western areas of the Bay and the algae tended to occur in clumps of varying sizes. The asterisk-shaped design of the video transects enabled maps of macroalgal cover to be interpolated using kriging techniques (Plate 5) and estimates of the reliability of the resulting maps could be generated using coefficients of variation (Plate 6).

It would be possible to revisit these sites using differential GPS and to stratify additional sampling based on cover and error estimates produced from the current surveys to monitor benthic community patterns in the future.

The PPBES recognised the need to provide a spatially accurate and detailed map of the current distribution and abundance of macrophytes at a Bay-wide scale. There were significant technical problems in mapping seagrass and other vegetation using traditional techniques such as diving or even aerial photography.

Studies of existing aerial photography data sets of the Bay indicated that it was sometimes very difficult to distinguish between different types of benthic vegetation. Also, an analysis of existing data sets indicated that the spatial inaccuracy in the location of seagrass meadows, for instance, could be in the order of 500 m. This makes accurate detection of changes in seagrass distribution over time very difficult.

Novel techniques that addressed the issues of spatial resolution, accuracy and the ability to distinguish between different types of submerged vegetation were required. A pilot study in Task G2.1 determined that remote sensing of benthic vegetation using airborne multispectral scanners was capable of addressing these issues. Task G2.2 was subsequently commissioned to produce habitat maps of the Bay showing the current distribution of macrophytes to a depth of at least 10 m using the Compact Airborne Spectrographic Imager (CASI).

CASI flights over the Bay were conducted in 1994. Problems with atmospheric and in-water optical models and variable environmental conditions within the Bay precluded the precise identification of features mapped in the pilot study. However, although benthic features could not be reliably identified from the survey, classes based on spectral characteristics were generated.

A series of spectral class maps were produced showing significant patterns that may correspond to benthic features. Although these maps should not be considered as 'benthic habitat' maps or used as a base line on which to assess future changes in benthic vegetation, they nevertheless offer the future potential to examine the information using improved atmospheric and in-water algorithms to describe the composition and distribution of the benthic vegetation.

3.3.4 Microphytobenthos

These organisms were studied in Task N2.4 by Monash University researchers (Beardall and Light 1996). Of 65 sites sampled within Port Phillip Bay, 55 were found to have microphytobenthos (MPB) present (Figure 5.3). High concentrations of functional chlorophyll *a* (> 2.5 µg/g sediment) were found in the top 10 mm of sediment only in Corio

CHAPTER 5

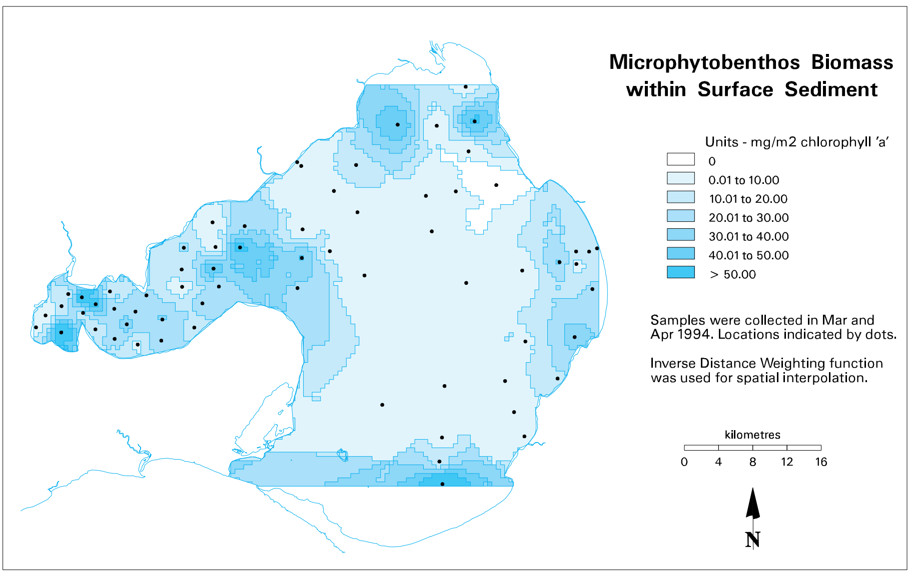

Figure 5.3: The distribution of microphytobenthos in surface sediment in Port Phillip Bay (Beardall and Light 1996).

Bay, in a zone north of Indented Head and offshore from Carrum. Concentrations were generally low deeper than 10 mm.

MPB biomass (measured as chlorophyll *a*) varied from >10 to <50 mg/m^2 across the Bay. Differences between sampling sites at scales of tens of kilometres accounted for more than 90% of the variation but there was no consistent seasonal pattern.

In general, there was greater biomass of MPB in the sediment at shallow and intermediate depth sites compared with the deep sites suggesting that light may limit growth of these organisms in the Bay.

Although Axelrad *et al.* (1979) obtained some data on MPB offshore from Werribee and Portarlington, it was not possible from the limited information they presented to determine whether there have been changes in MPB populations since that time. However, Task N2.4 has disclosed the importance of MPB in primary production in the Bay.

Estimates of total chlorophyll *a* in MPB Bay-wide amount to 49 t, equivalent to 2,434 t of carbon. This biomass turns over 40-50 times a year at a mean P:R ratio of 0.3. This is of the same order as the biomass and turnover of phytoplankton in the Bay (Beardall *et al.* 1996, see also Chapter 6).

The ratio of chlorophyll *a* to *c* indicated that there was little evidence of cyanobacteria in the MPB populations in the Bay. These populations were dominated by diatoms which indicates that the Bay is not experiencing the effects of severe eutrophication in which cyanobacteria are the dominant MPB components.

3.3.5 Benthic invertebrates

Bay-wide and local surveys of the benthic invertebrates in Port Phillip Bay have revealed that the Bay supports a benthos exceptionally rich in species compared to similar ecosystems in other parts of the world (Poore 1992, Wilson *et al.* 1993). This in itself is an indication of a healthy benthic ecosystem in the Bay.

We have already mentioned (Chapter 3) that these organisms play a major role in the functioning of the Bay sediments and are a critical component in maintaining the water quality of the Bay. This role is discussed further in Chapters 6 and 7.

The diversity of the benthic community is primarily correlated with the nature of the sediment of the Bay (Poore *et al.* 1975, Poore and Rainer 1979, Parry *et al.* 1995). It ranges from species-poor communities on muddy environments to species-rich communities on sandy sediments. Benthic molluscs contribute a large fraction to the biomass of the Bay-wide fauna. Suspension-feeding taxa process a significant fraction of primary planktonic production and deposit feeders have been calculated to process a volume equivalent of the top 13 mm of Bay sediments every year (Wilson *et al.* 1993).

Data to assess changes in the benthic invertebrates of Port Phillip Bay are available from a number of sources including the Phase One data set (1969-1973) which sampled 86 stations in a regular grid, the Phase Two survey (1973-1975) which sampled three stations at quarterly intervals over three years, the Scallop Dredging Effects (SDE) survey at St Leonards, Portarlington and Dromana (1991-1992) and the current Port Phillip Bay Environmental Study sampling 14 stations at six monthly intervals over 18 months [Task E5.2] (Wilson *et al.* 1996).

Bay-wide changes in the benthic community could only be assessed by comparing the Phase One and Task E5.2 surveys because the Phase Two and SDE surveys were conducted at specific locations or on limited taxa.

The Phase One survey found that there was no clear indication of an effect on benthic community structure of effluent from the WTP except for a selection of euryhaline taxa within about 300 m from the outlets.

The Phase One surveys also found that

molluscs comprised 36% of the total macrobenthic biomass.

Suspension feeders (49% of the total biomass) were the major trophic group (irrespective of taxa) in terms of biomass yet they constituted only 18% by number.

Crustaceans and molluscs were the most abundant groups with densities of more than 1,100 individuals/m^2. The most diverse groups were crustaceans (240 species) and polychaetes (238 species).

The current Port Phillip Bay survey found that only three of the 10 most abundant species in 1994-96 were among the 10 most abundant during the Phase One Study (Table 5.2).

Three of the six most abundant species now (not counting *Sabella spallanzanii*) are new to the Bay and may be introduced or be native species undergoing dramatic growth. Introductions of exotic species of invertebrates have had a major impact on the species composition of benthic organisms in the Bay – greater, in fact, than any change in water quality that can be detected.

Several taxa found during the Phase One and Two surveys were rare or absent in the current survey. These taxa include *Nassarius nigellus* (gastropod), *Birubius mayamayi* (amphipod), *Owenia* sp. (polychaete), *Montacuta semiradiata* (bivalve), *Polyonyx transversus* (decapod), *Melliteryx acupuncta* (bivalve) and *Chaetopterus* sp. (polychaete).

In common with previous studies, it was demonstrated that the major influences on spatial distribution of benthic invertebrates in Port Phillip Bay are sedimentary characteristics i.e. mud or sand. All other variations in species abundance are minor by comparison, including those resulting from interannual change, the introduction of exotic species and the effects of physical disturbance such as scallop dredging.

The magnitude of the taxonomic separation between communities was not as great in 1994-1995 as determined in the Phase One survey indicating that there are now more common species occurring across both sediment types. Some of these common species are exotics, others are not. This indicates that there has been an increase in species that are common on a range of sediment types and a relative reduction in numbers of species with more specific sediment preferences. These

Table 5.2. The 10 most abundant benthic invertebrates.
M = mollusc, C = crustacean and P = polychaete (Wilson *et al.* 1996).

Phase One surveys	Current Port Phillip Bay Study
Theora lubrica (M)	*Corbula* (*Varicorbula*) cf. *gibba* (M)
Neocallichirus limosus (C)	*Lumbrineris* sp. (P)
Notospisula trigonella (M)	*Theora lubrica* (M)
Dimorphostylis cottoni (C)	*Dimorphostylis cottoni* (C)
Rhinoecetes sp. (C)	*Nephtys inornata* (P)
Euphilomedes sp. (C)	*Euchone* sp. (P)
Phoronis pallida (Phoronida)	*Leitoscoloplos bifurcatus* (P)
Lumbrineris sp. (P)	*Magelona* cf. *dakini* (P)
Chioneryx cardioides (M)	*Byblis mildura* (C)
Nuculoma obliqua (M)	*Phyllochaetopterus* sp. (P)

trends were readily apparent at the 'family' level of taxonomic resolution.

Changes to the trophic structure of the benthic invertebrates of Port Phillip Bay are also evident (see Table 5.3). There has been a significant increase in the abundance of suspension feeders and a corresponding decrease in the abundance of deposit feeders.

While an increase in the abundance of suspension feeders can, in part, be attributed to introduced species such as *Euchone* and *Sabella,* reasons for the decline in deposit feeders such as *Echinocardium* are not readily apparent. It is possible these trophic shifts may reflect an improved nutrient status of the Bay towards lower concentrations as they are certainly not what would be expected from a water body experiencing increases in nutrient loading.

As well, the shift could be the result of the spread of exotic suspension-feeding species which have no predators locally.

However, there is no evidence from the zoobenthos of a deterioration in water quality over the 25 year period between Phase One and the current study.

3.3.6 Fish

Major commercial fish species (many of which are also taken by recreational fishers) include flathead, snapper, King George whiting, southern anchovy, pilchard, calamari, mussels, scallops and abalone. Detailed descriptions of these fisheries are provided by Nicholson (1992).

Information obtained from replicate trawls at 22 depth-stratified stations from 1990 to 1994 (Parry *et al.* 1995) showed that fish distribution in the Bay is correlated with benthic prey types which in turn are correlated with sediment type [Task E8].

Based on fish distribution, four regions of the Bay were identified. These were the shallow sites (<7 m deep) which corresponded to 'sand' described by Beasley (1966), intermediate depth sites (12-17 m) corresponding to intermediate sediments (described by Beasley as silty sand and clayey sand) and the deep region (22 m) classified by Beasley as clay with some sand and silt. Corio Bay was identified as a separate area.

Parry *et al.* (1995) found that although there were significant interannual differences in fish abundance between 1990 and 1994 there were no trends in time apart from an increase in the abundance of little rock whiting.

Changes in fish abundance over a longer time period were assessed using data from 14 trawl stations across the Bay from the periods 1972-1975 and 1990-1991. Although other data from other sampling times were available, different sampling methods and seasons

Table 5.3. Percent abundances of common trophic groups (Wilson *et al.* 1996).

Trophic Group	Port Phillip Bay Phase One and Two	Current Port Phillip Bay Study
Deposit	73	58
Herbivore / Grazer	1	<0.1
Predator	6	10
Scavenger	1	2
Suspension	19	30

precluded their use in the comparison (Hobday *et al.* 1996).

These authors found significant decreases in the catches of many fish species although a few increased in abundance (Table 5.4). These changes are difficult to ascribe to a particular cause although Hobday *et al.* (1996) suggest that fish abundance in the Bay can be affected by fishing pressure, habitat loss, reduced competition and the establishment of exotic species. Of course, the percentage changes in Table 5.4 conceal a wide difference in actual abundance of species. Sand flathead and a few other species still dominate the fish community.

Many species have shown very large and statistically significant decreases in abundance. Species that have declined by at least 80% include cobbler, elephant shark, common gurnard perch and King George whiting. Toothy flathead are now relatively rare in the Bay with a 99.9% decline in abundance from numbers caught in the 1970s.

The most likely cause for the decline of the heavily fished species such as sand flathead, yank flathead, toothy flathead and common gurnard perch is increased fishing pressure within the Bay (Hobday *et al.* 1996).

For other species such as King George whiting, cobbler, greenback flounder, six-spined leatherjacket, mosaic leatherjacket and long-finned pike, the decrease may be due to habitat changes resulting from a possible decline of seagrass in the Geelong Arm during the early 1970s and a further decline in 1985 (Axelrad, VFRI unpublished data, cited in Hobday *et al.* 1996).

While the declines in seagrass during the early 1970s occurred shortly after the onset of scallop dredging in the area, Hobday *et al.* (1996) conclude that neither the 1970s nor the 1985 decline in seagrass in the area have been well documented and it is not possible to conclusively demonstrate what the cause of the decline was.

The reasons for the very large increases in

Table 5.4. Percent change in fish abundance for species with sufficient statistical power to detect significant changes between 1972-1975 and 1990-1991.

⇧ denotes an increase, ⇩ denotes a decrease and ? denotes no change/uncertain.

Table modified from Table 7 of Hobday *et al.* (1996).

Species	Change	Species	Change
Sand flathead	47.2 ⇩	Elephant shark	82.9 ⇩
Globefish	17.3 ⇩	Common gurnard perch	80.7 ⇩
Stingarees	>100.0 ⇧	School whiting	- ?
Eagle ray	>100.0 ⇧	Red mullet	30.8 ⇧
Yank flathead	33.6 ⇩	King George whiting	94.4 ⇩
Toothy flathead	99.9 ⇩	Six-spined leatherjacket	75.0 ⇩
Spiny gurnard	73.5 ⇩	Sandy sprat	- ?
Banjo ray	63.4 ⇩	Little rock whiting	- ?
Cobbler	85.5 ⇩	Mosaic leatherjacket	- ?
Greenback flounder	28.2 ⇩	Long-finned pike	- ?

abundance of eastern shovelnose stingaree and sparsely spotted stingaree are uncertain although they may be linked with a reduction in populations of species that have the same or similar diets, such as sand flathead. The reason for increased abundance of other species such as the red mullet is unclear because there is minimal overlap between its diet and the diet of species that have declined.

Of the exotic or introduced species, the oriental goby has been present in the Bay since the 1970s. In the Geelong Arm there has been an increase in the abundance of little rock whiting and globefish possibly due to habitat changes resulting from the abundance of exotic species (Hobday *et al.* 1996). Two introduced invertebrate species, the bivalve *Corbula* and the majid crab *Pyromaia tuberculata*, now make up more than 50% of the diet of globefish populations. The little rock whiting is now usually found in association with populations of *Sabella spallanzanii*. The impact, if any, of these introduced species has yet to be determined.

For some species there appears to be no significantly detectable change in numbers within the Bay. For species such as snapper and silver trevally there was insufficient statistical power to detect objectively whether changes have occurred (see Hobday *et al.* 1996 for a detailed report). However, it appears that the recruitment of snapper over the past two years has been good.

For other fish, such as the sandy sprat (see Table 5.4), changes were insignificant. There is also evidence of increased numbers of fish eggs and larvae, at least in the Geelong Arm, in recent years according to icthyoplankton research conducted for the Fisheries Research and Development Corporation.

Although the abundance of several pelagic species, measured from commercial catches, has also changed in the Bay between the 1970s and 1990-1991, it is difficult to determine causes. For example, the decline in barracouta reflects widespread trends in southern Australia (although their abundance has increased in 1995) and large, long-term fluctuations are believed to occur for other pelagic species (Hobday *et al.* 1996).

In conclusion, it seems that significant changes to fish populations in the Bay did occur between 1972-1975 and 1990 but not since then. The cause of these changes cannot be determined from the available data. However, likely causes include increased fishing pressure and habitat changes. Long-term natural fluctuations in fish abundance due to ocean climate cycles are also likely to have contributed to the observed changes.

Finally, there appears to be no evidence, based on available data, that fish populations have declined in the Bay in response to current or past levels of nutrient and toxicant inputs.

3.3.7 Birds and mammals

A formal workshop on marine birds and mammals was convened during the PPBES to assess the state of knowledge on these biota with respect to the Study objectives. This workshop considered the extent of information available as well as logistic, community, management and scientific issues.

Because the migratory habits of many of the bird species meant that their population numbers around the Bay could be affected by factors other than nutrients and toxicants within the Bay, they were not selected for study within the PPBES.

Similarly, marine mammals within the Bay were considered but excluded from study in preference for indicators that were more easily monitored and for which appropriate historical data sets were available.

4. What does it mean?

Based on the available information, Port Phillip Bay is in a relatively healthy condition.

The present nutrient levels in the Bay have not caused Bay-wide ecological impacts like those associated with eutrophication in European and Northern American estuaries, coastal embayments and seas. The Bay appears to have acceptable conditions with respect to nutrients and toxicants.

Phytoplankton abundance is low by world standards. Although phytoplankton species composition has apparently changed since the Phase One survey, the most abundant species then, *Skeletonema costatum*, is still the most abundant.

Diatoms continue to dominate the system and are influenced by silicate and nitrate inputs from the Yarra, especially in summer.

Physical factors such catchment runoff seem to play a major role in affecting phytoplankton distributions.

The distribution of microphytobenthos, like that of phytoplankton, shows significant variation in abundance between locations. MPB have high productivity and may be responsible for enhancing oxygen levels and recycling of nutrients within the sediments.

The physical environment, especially light levels within the water column, is a major determinant of the distribution of MPB, which consist predominantly of diatoms. This is indicative of a healthy environment compared with MPB assemblages in eutrophic environments which are generally dominated by cyanobacteria.

Detailed knowledge of changes in the distribution and abundance of seagrasses is not available. Suspected declines in seagrass since the Phase One Study cannot be linked to nutrient and toxicant levels within the Bay. There is increasing evidence elsewhere in the world linking seagrass abundance with light levels even though light penetration in Bay waters seems generally good.

Rather than nutrient loads, the distribution and health of seagrass beds in the Bay may be affected more by mechanical disruption (e.g. dredging) and by resuspension of bottom sediments resulting in smothering and/or reduction in light levels.

Large blooms of opportunistic green macroalgae such as *Ulva* are characteristic of eutrophic environments. Such growths are not found within Port Phillip Bay on a large scale indicating that the Bay is in a healthy state.

Another feature of eutrophic systems can be the occurrence of large beds of drift algae but, although beds of red drift algae do occur in Port Phillip Bay, this seems most likely a normal pattern and not the result of excessive nutrient loadings.

There is currently little, if any, evidence linking nutrients to any changes to, or impacts on, the benthic invertebrates within the Bay. However, there has been a change in the species composition and relative abundance of benthic invertebrates since the Phase One survey.

New species have appeared and species common during Phase One sampling have disappeared or declined in relative abundance.

A characteristic of environments that are becoming more eutrophic is an increase in deposit feeders at the expense of suspension feeders, but in the Bay the reverse has occurred. Unfortunately the deposit feeders that have declined include bioturbators and bioirrigators which significantly enhance nitrification and denitrification and hence the escape of nitrogen from the Bay (Chapter 3).

In addition, the increased relative abundance of suspension feeders is largely the result of introduced organisms such as *Euchone*

CHAPTER 5

and *Sabella*. Their exact impact on the Bay environment is not known.

They have the potential to increase ammonia concentrations in the Bay due to their excretion, and there is the potential for them to displace other species.

Sabella spallanzanii do not appear to have any natural predators within the Bay and continue to expand their range (Figure 5.4). Changes to their distribution and abundance should be monitored carefully. If they displace the bioturbating animals their impact on the Bay's nitrogen cycling process could be significant.

Fish populations within the Bay have changed significantly since the 1970s. The relative abundance of many species have declined, a few have increased, and some have not changed significantly.

Nutrient and toxicant levels within the Bay do not appear to be related to these declines. Exotic prey species are now common items in the diets of some fish species while other exotics such as *Sabella spallanzanii* apparently provide habitat, at least in the case of the little rock whiting.

In summary, the ecology of Port Phillip Bay has changed since the Phase One survey but the Bay is still healthy.

Some of these changes may be due to natural fluctuations in biological populations, others are in response to the introduction of exotic species in the Bay and still others may be the result of habitat modification and fishing practices.

Currently, on a Bay-wide scale, there is no evidence of significant ecological effects from present nutrient and toxicant loads. However, the potential impact of changes in the trophic structure of the benthic infaunal community is of concern because of the potential to alter how the Bay deals with the present and future nutrient loads.

Any potential changes that may affect the balance between water/sediment nutrient exchange needs to be carefully monitored.

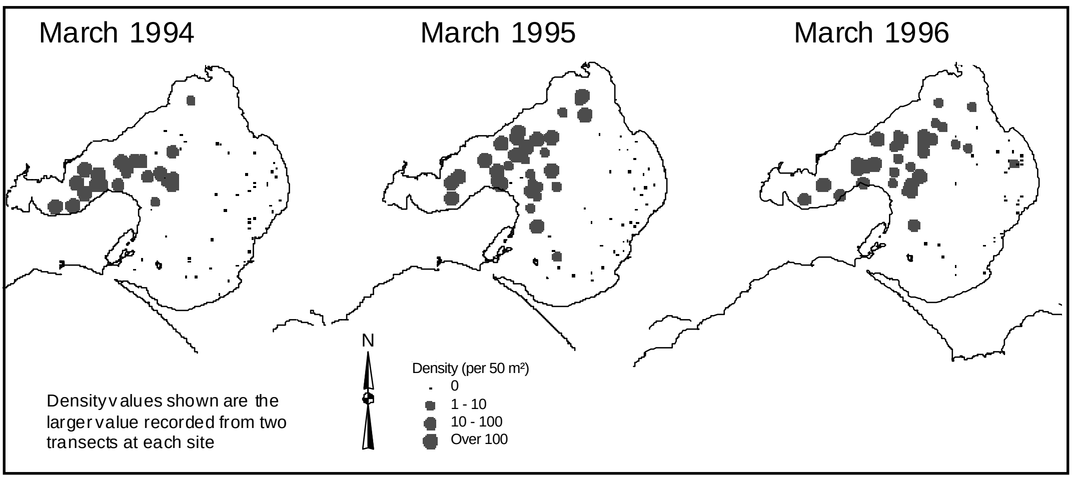

Figure 5.4: The distribution of *Sabella spallanzanii* within Port Phillip Bay (courtesy of the Victorian Fisheries Research Institute).

5. Where do we go from here?

The PPBES has identified several issues crucial to the continued well-being of the Bay. Long-term time series of data are required to distinguish between anthropogenic influences and natural fluctuations in the distribution and abundance of biota.

This necessitates the development of a long-term monitoring program which is practical in terms of the indicators to be monitored and the time and space scales at which monitoring takes place. These issues will be discussed in Chapter 7.

Contents

1. Introduction

1.1 Objectives

The Port Phillip Bay Environmental Study has deliberately adopted a systems approach in developing the scientific framework needed to address the environmental management of the Bay.

The models described in this chapter play two key roles within this approach. They provide a conceptual basis for synthesis and integration of the data sets and process knowledge provided by the diverse field surveys and experiments which make up the major part of the Study. They also provide a basis for connecting past and future management actions to the Bay's environmental health, allowing the development and analysis of management strategies and scenarios. In this role, models serve as a focus for the interface between environmental studies and management.

During the Study's design phase, CSIRO undertook extensive consultation with managers and user agencies so as to identify key management issues and concerns. This process was continued in the early phase of the modelling Task through a modelling workshop in 1992, attended by representatives of user agencies and scientists with key expertise.

The workshop was based on the Adaptive Environmental Assessment and Management principles described by Holling (1978). The development of a conceptual design for models of Port Phillip Bay was considered as a series of choices (e.g. the level of component and process resolution, the selection of state variables, the formulation of process sub-models, and the time and space scales to be resolved).

These choices were assessed in terms of the fundamental requirement that models allow us to link management actions to system outputs of value or concern to users. The management actions of primary concern to the managers are those affecting the loadings of sediments, nutrients and toxicants at the Bay boundaries, either through major point sources such as the Werribee Western Treatment Plant (WTP), the Yarra River and major creeks, or through distributed inputs (minor creeks and drains, groundwater, shipping and atmospheric deposition). The impacts of dredging are also of interest.

It is of particular importance for the modelling tasks that the Study was not charged with addressing processes in catchments which control loadings to the Bay. This means that management actions in the catchments can be addressed by the model only by making assumptions about their effects on loads at the Bay edge.

It is desirable in the long term that the models developed for the Bay be linked with catchment models, to provide an 'end-to-end' management support system. (In practice, as described below, the Study has been forced to develop simple empirical catchment models in order to interpolate the generally sparse loading data.)

Models have the advantage (or disadvantage, depending on your point of view) that they force us to be specific and generally quantitative about both hypotheses and objectives. We cannot develop an ecosystem model of the Bay which has 'environmental health of the Bay' as an output variable. We need to encourage managers to specify environmental health in terms of properties which we can at least potentially predict as output variables.

We can distinguish two classes of these output or indicator variables. One class consists of properties of direct concern to managers or the public. The modelling workshop identified the following list of variables of this type:

- Water column dissolved oxygen, turbidity, foams, oil slicks;
- Sediment grain size, organic content, anoxia;
- Toxic or nuisance algal blooms;

- Valued benthic communities such as seagrass, wetlands, reefs;
- Valued species such as mammals, birds, and general biodiversity;
- Abundance of commercial and recreational finfish and shellfish;
- Contamination of seafoods.

It was recognised at the time that not all these indicators might be amenable to modelling.

The second class of indicator variables consists of water column and sediment properties which, while not necessarily directly valued or even noticed by the public, reflect changes in system state which can affect valued components.

These are generally water quality and sediment quality criteria, and include nutrient and toxicant concentrations, phytoplankton biomass, and bioirrigation levels. The distinction between the two classes of indicators corresponds roughly to the distinction between Environmental Quality Objectives and Environmental Quality Standards (Orr *et al.* 1990).

It has been recognised for some time that it would be desirable to manage for objectives based on the variables of direct concern. However, it is easier for regulatory purposes to manage on the basis of water and sediment quality criteria, and this is the current situation for Port Phillip Bay.

As a result of discussions at the workshop, it was decided that the 'integrated' models developed in the Study would consist of nutrient cycling models, involving tight coupling among physics and nutrient processes, and toxicant cycling models, involving tight coupling among physics and toxicant processes, with more empirical models linking the outputs of these models (water and sediment quality) to ecological indicator variables. (Some ecological indicator variables such as turbidity, dissolved oxygen, and seagrass distribution are addressed in the nutrient cycling model.)

After considering both the key management issues, and the intrinsic dynamics of Bay processes, the workshop identified two sets of time and space scales for models to address. Managers were concerned about the long-term impacts of nutrient and toxicant loads on the Bay, and asked the Study to address loading scenarios extending out as far as 50 years.

The flushing time of the Bay is about one year, but there is the potential for sediment and benthic components to respond to changes in loadings on much longer time scales. Models extending over these time frames need to address the broad spatial gradients within the Bay.

Managers are also concerned about dispersal and fate of loadings near point inputs, especially the Werribee WTP, over short time and space scales. To provide the flexibility to cope with this range of time and space scales, it was decided to base the integrated models on a physical transport model with flexible spatial resolution. This transport model is connected to the hydrodynamic model via a particle-tracking module, in a manner described in more detail in Chapter 2.

It should be noted that, primarily because of delays in commissioning the physical tasks, the completion date for the modelling tasks was set back to December 1996. This chapter therefore represents an interim rather than final report for the integrated modelling.

This has two principal consequences for the reader.

First, modelling effort to date has focused almost entirely on nutrient cycling, because of the generally reassuring results of the toxicant surveys, and the fact that accurate models of sediment transport, which have been most delayed in the physics tasks, are seen as critical for toxicant models. This chapter deals entirely with nutrient cycling and impacts.

Second, model calibration is not yet complete, and the application of models to management strategy evaluation and monitoring design is planned to proceed through a series of workshops in the second half of 1996. The results reported here should therefore be regarded as preliminary only.

1.2 Previous modelling

Previous attempts at modelling nutrient distribution in Port Phillip Bay have been relatively limited.

As part of the Phase One Study (MMBW/FWD 1973), a numerical linked-node circulation model was developed, and used to simulate the distribution of phosphate in the Bay, treating it as a passive tracer. The Phase One Report envisaged that models of the kind presented here would be developed as part of a Phase Two Study, but this did not occur.

However, an integrated model for Port Phillip Bay based on a 2-D depth-averaged finite difference circulation model was subsequently developed by Williams (1978) in his PhD thesis research. This model represented cycling of N and P in the water column through phytoplankton, zooplankton and DON components.

While the model was certainly 'state-of-the-art' at the time of its development, analysis of the model was limited, and the study appears to have been neglected subsequently by managers and the scientific community.

As explained in more detail below, the models we have developed for Port Phillip Bay draw on a rich international scientific literature describing the cycling of nutrients through planktonic ecosystems in both marine and freshwater systems.

There have been major advances in the modelling of both offshore and coastal marine ecosystems in the last decade, often driven by international research programs designed to address concern about global biogeochemical cycles (e.g. Evans and Fasham, 1993), and eutrophication of coastal seas and estuaries (e.g. the North Sea / European Regional Seas Modelling Program, Radford 1993).

Awareness of the potential value of system models for coastal studies and coastal management has also increased in Australia in the past decade. The Perth Coastal Waters Study used a model of nutrient distribution and fate, and impacts on seagrasses, to address nutrient assimilative capacity of coastal waters near Perth (Lord and Hillman 1995). Modelling is also under way or under consideration as part of major studies of eutrophication near other Australian population centres.

2. Bay nutrient cycling: an overview

2.1 A summary of the nitrogen cycle in Port Phillip Bay

This section describes briefly the pools and fluxes of nutrients on a Bay-wide, annual basis, as deduced from observations during the Study field program. This picture ignores much of the information at finer spatial and temporal scales collected during the study, and other information on longer-term interannual variation, but it provides a necessary introduction to the key pools and fluxes, and provides a set of constraints which any prognostic model must meet.

The section focuses on the nitrogen budget, because nitrogen is the key limiting nutrient for primary producers in Port Phillip Bay. The N:P ratio in loadings to the Bay is about 5:1 by weight, compared to the Redfield requirement of about 7:1 by weight (16:1 by atoms).

While P starts out in slight excess, the subsequent loss of N to denitrification in the Bay, compared with the more conservative behaviour of phosphate, means that P is in considerable excess in Bay waters.

Typical phosphate concentrations are about 2 µM, and the average water column pool size of about 1,500 t, with a flushing time of about one year, implies an annual export to Bass Strait which roughly matches the annual load. Thus, although phosphate undoubtedly participates in biogeochemical cycles in the Bay, it is present in such excess that it can be treated almost as a conservative tracer.

As noted in Chapter 3, silicate is seasonally depleted in spring to a point where it limits diatom growth, which may have implications for both algal bloom composition and water column recycling of nitrogen. This will be discussed further below.

Much of the field activity in the Study has been aimed at quantifying and understanding different components of the nitrogen cycle in the Bay. The external load of nitrogen is estimated to be about 7,700 t/yr, of which about 5,000 t is dissolved inorganic nitrogen (DIN). The remainder is organic nitrogen, which we would like to separate into dissolved and particulate fractions. Unfortunately, the measurements in input streams and drains do not allow this.

An annual load of 5,000 t of DIN, distributed throughout the volume of the Bay, would produce a mean concentration of 14 µM. If nitrogen behaved like phosphate, and was removed principally by flushing on annual time scales, we would expect to observe ammonia and nitrate concentrations of this order.

However, the observed mean DIN pool is only about 380 t (concentration about 1 µM), because DIN in the water column is rapidly taken up by primary producers (primarily phytoplankton, but also benthic macroalgae). Nitrogen does not accumulate long-term in phytoplankton: the estimated pool of phytoplankton nitrogen is only about 200 t N. In fact, the total pool of particulate nitrogen in the water column is only about 630 t N.

The phytoplankton N pool may be small, but it is turning over rapidly. Primary production studies in the Bay, extrapolated Bay-wide, provided estimates of carbon fixation equivalent at Redfield ratios to a nitrogen uptake of about 40,000 t N/yr (Beardall *et al.* 1996). This estimate is more than double earlier estimates in the Bay, and means that the phytoplankton community is maintaining an average growth rate of about 0.55/d.

One might wonder how a phytoplankton demand of 40,000 t DIN/yr is to be supported on the basis of an annual input of only 5,000 t DIN. The answer is that, on average, phytoplankton growth is matched by losses to grazing and/or sinking, and that nitrogen taken up by phytoplankton is generally remineralized to DIN, either directly in the water column, or indirectly after sedimentation.

The external load of organic matter (about 2,700 t N) presumably also joins this nitrogen cycle, although an unknown fraction of this (PON or DON) may be highly refractory. Even if all of this matter joined the recycling loop, production levels of this order would require that each atom of N is recycled through phytoplankton about 5.5 times in the Bay, before being lost. This recycling efficiency is comparable to efficiencies observed in some oligotrophic oceanic systems.

There is a sizeable pool of dissolved organic nitrogen (DON) in the water column, amounting to about 3,500 t N Bay-wide. However, the mean concentration of DON in Bay waters, about 10 µM, is only about 3 µM higher than concentrations observed in Bass Strait water at the Heads. It is well known that there is a background level of highly refractory dissolved organic matter in oceanic water. It is probable that a significant proportion of the DON in the Bay is refractory DON which is mixed into the Bay with ocean water but takes little part in nitrogen cycling inside the Bay. The excess of about 3 µM implies an export of DON from the Bay to Bass Strait of about 1,000 t N/yr.

We do not have direct estimates of the production of DIN through recycling in the water column, nor of the net sedimentary flux of organic matter to the sediments. However, the benthic chamber experiments provided direct estimates of the dark oxygen consumption rates by the sediment.

While these experiments were not intended to provide the spatial and temporal coverage needed for Bay-wide annual fluxes, attempts to do this extrapolation as rigorously as possible have produced estimates of oxygen consumption that are equivalent, at Redfield ratios, to DIN production rates of 30,000 t N/yr. (Nicholson *et al.* 1996). The benthic chambers also measured directly the DIN efflux from the sediment to the overlying water column: this amounted to only about 8,100 t N/yr.

The difference between N production calculated from oxygen consumption, and N flux from the sediments, has been interpreted as loss of fixed nitrogen to N_2 gas via denitrification. This interpretation poses a significant problem for the Bay N budget, as the loss of N to denitrification estimated in this way amounts to 22,000 t N/yr, or three times the known external loading. Either the estimates of external loadings, or the chamber estimates, would have to be in error by a factor of three to account for this discrepancy.

An alternative solution to this problem may come from another benthic component, the microphytobenthos (diatoms living and growing in the top 1 to 2 cm of sediment). In many of the chamber deployments, it was noted that there was a marked difference between oxygen consumption rates in dark and light chambers, with oxygen often increasing initially in deployments in daylight hours. However, the chamber experiments were not designed to quantify oxygen production during daylight hours, and the Bay-wide oxygen consumption estimates are based on dark consumption rates only.

A separate Task was commissioned by the Study to look at primary production by the microphytobenthos. This Task measured microphytobenthos biomass and photosynthetic parameters (P_m and I_k), and used these, along with data on surface irradiance and light attenuation, to calculate *in situ* production rates over the Bay. These estimates suggest the microphytobenthos are surprisingly active.

The estimated Bay-wide primary production is equivalent at Redfield ratios to about 20,000 t N/yr, based on a microphytobenthos pool containing on average 400 t N, and therefore turning over about 50 times a year

(Beardall and Light, 1996). (Note that the Bay-wide microphytobenthos biomass is twice the phytoplankton biomass, and it is turning over four times slower, due presumably to light limitation.)

The microphytobenthos production estimates represent gross photosynthesis. The chamber dark respiration rates presumably include dark respiration by microphytobenthos, as well as microbial and infauna respiration involving breakdown of organic matter.

Since diatoms are quite capable of storing and conserving nitrogen over periods of 24 hours, that part of the dark oxygen consumption representing microphytobenthos respiration may not represent any release of DIN. Moreover, as microphytobenthos take up nutrient from sediment porewaters, and their fate is presumably to be grazed and recycled within the sediment, the consumption and remineralization within the sediment of organic matter produced by microphytobenthos presumably also looks like a closed loop as far as the nitrogen fluxes seen by benthic chambers are concerned. (This implies that microphytobenthos are efficiently trapping DIN produced in the sediment before it can escape to the overlying water column.)

Under this interpretation, it is only the net sediment oxygen consumption (dark respiration minus microphytobenthos gross production) which represents remineralization of organic matter exported to the sediment from the water column. It is the N equivalent of this net oxygen consumption which should be compared with DIN release from the sediments.

Taken at face value, the two independent estimates from benthic chambers and microphytobenthos studies imply that this net oxygen consumption is only equivalent to 10,000 t N/yr, and the loss to denitrification is only 2,000 t N/yr.

Unfortunately, the process of scaling up both benthic chamber fluxes and microphytobenthos photosynthesis is such that the difference (net respiration) involves significant uncertainty, probably of order ±10,000 t N/yr. The simplest approach to reconciling these estimates is to assume that loss to denitrification is of the order of 6,700 t N/yr (i.e. sufficient to balance the N budget, allowing for DON export to Bass Strait), and that the net sediment respiration is therefore of the order of 14,800 t N/yr (assuming the chamber estimate of DIN release from the sediments is correct). This implies that the microphytobenthos production estimates and the benthic chamber gross respiration estimates contain a combined error of about 4,700 t N/yr, which is well within the possible extrapolation errors.

The net sediment respiration must equal the export of organic matter from the water column to the sediment, assuming no net long-term burial of organic N. For these high chamber fluxes, it follows that about 35% of the organic nitrogen produced in the water column is exported to the sediments, and that about 45% of this is ultimately loss to denitrification. However, nitrogen is cycled multiple times within the sediment and on each pass through remineralization, on average about 20% of the DIN produced is lost to denitrification, about 25% escapes to the overlying water column, and about 55% is captured and recycled within the sediment by microphytobenthos.

Nicholson *et al.* (1996) present an alternative set of lower estimates of Bay-wide sediment fluxes, excluding the Great Sands and the area shallower than 5 m. These estimates provide dark respiration rates of 18,600 t N/yr, DIN fluxes of 3,600 t N/yr, and apparent denitrification rates of 15,000 t N/yr, about twice the required rate. With these flux estimates, microphytobenthos production of 8,300 t N/yr is required to balance the N budget overall, and this means that net sediment respiration is reduced to 10,300 t N/yr.

These lower chamber flux estimates exclude the Bay sediment shallower than 5 m. This area is an important contributor to microphytobenthos production: estimated microphytobenthos production deeper than 5 m is only 11,000 t N/yr, and again the correction needed to balance the budget (2,700 t N/yr) is well within the extrapolation error.

For this low flux budget, about 25% of water column production is exported to the sediment, and about 65% ultimately lost to denitrification. On each pass through the remineralization loop, about 35% is denitrified, 45% taken up by microphytobenthos, and 20% effluxed to the water column.

These calculations assume there is currently no net storage of organic nitrogen in the sediments. The pool of organic N in the sediments is extremely large, about 147,000 t N in the top 20 cm alone, with high concentrations extending well below 20 cm.

Based on tracers in sediment cores, bioturbation rates in Bay sediments are thought to be sufficient to mix sediments to a depth of 30 cm on time scales of about 20 years, and long-term sediment accumulation rates in Port Phillip Bay are thought to be very low (about 2 m in 6,000 years, or 0.03 mm/yr) (Chapter 2). Under these conditions, it is impossible to tell directly from N profiles whether this organic N is of recent (20th century) origin, implying a high N accumulation rate, or of ancient origin (accumulated over 1,000 years or more).

If the pool of organic N in the sediments has accumulated over 1,000 years, a burial rate of about 200 t of highly refractory organic N/yr is indicated. This is relatively insignificant in terms of the current N budget.

On the other hand, if a significant fraction has accumulated during the past 100 years of increasing N loads from the catchment, this could imply a much higher current burial rate, of around 2,000 t N/yr or more.

The accuracy of our estimates of other terms in the N budget is not sufficient to distinguish among these alternatives.

The hypothesis of recent accumulation must be of concern to managers, as it implies that the sediment organic N reservoir is not in steady-state balance with current loads, and that there could be a substantial reservoir of semi-labile organic N in the sediment.

In the discussion given above, we have ignored the contributions of macroalgae and seagrasses. A preliminary analysis based on earlier studies suggested that their likely contribution to primary production and N cycling in the Bay is small, although they could be locally important. However, earlier studies had focused on local areas, especially around Werribee, and did not lend themselves to Bay-wide extrapolation.

In this Study, a field task was commissioned to look at seasonal cycles in biomass and nitrogen content of macroalgae along five representative offshore transects along the western and eastern shores (Chidgey and Edmunds 1996). The results, extrapolated along depth contours between transects, suggest that N content of macroalgae varied from about 200 t N in December 1995 to 1,000 t N in March 1996. The uncertainty in these estimates is quite high because of the variation among quadrats within sites and the need to extrapolate over long distances between transects. The major contribution to these estimated pools came from filamentous drift algae between 5 and 10 m in the north-west of the Bay, around Wedge Point and Altona Bay. The turnover rates of these algae are not well known, but the seasonal variation in pool size suggests that their contribution to N cycling, although probably small compared with phytoplankton and microphytobenthos, is not negligible.

2.2 Nitrogen cycling and export efficiencies

The important relationships in the nutrient budget just presented can be captured by defining two key efficiencies. We define f_W as the water column export efficiency: that is, the proportion of the primary production in the water column which reaches the sediment as organic matter. This is analogous to the f-number used by biological oceanographers for open ocean ecosystems. We also define f^*_D as the proportion of the organic flux reaching the sediment which is lost to denitrification. Then, under certain simplifying assumptions, steady-state mass balance requires that:

Water Column Primary Production = (Catchment Load of DIN) / (f_W . f^*_D)

This equation is important because it relates the catchment load to primary production, which is, in turn, closely related to phytoplankton biomass, water quality and the degree of eutrophication. It says that primary production is amplified over the input load by an amount which increases as export efficiency and denitrification efficiency decrease. For Port Phillip Bay currently, this amounts to an amplification of about five to six times.

The simplifying assumptions are that catchment loads of organic N, export of N to Bass Strait, and burial of N, have all been neglected. It is possible to modify this equation to include all three effects, but the corrections are relatively minor for Port Phillip Bay in its present state.

Port Phillip Bay differs from some other embayments in that the flushing time is long, and so denitrification rather than export to Bass Strait is currently the principal process for removing N. However, if conditions in the Bay changed so that the denitrification efficiency f^*_D decreased towards zero, this equation predicts a rapid increase in primary production. As we discuss below, a more complete analysis shows that, as f^*_D approaches zero, levels of DIN or organic matter will increase to the point where flushing replaces denitrification as the principal N loss. At these levels, the Bay would be severely eutrophied.

In discussing the nitrogen budget, it became clear that microphytobenthos can increase the effective denitrification efficiency of the sediment by intercepting DIN which would otherwise escape from the sediment, and converting it into organic matter where it re-enters the sediment microbial loop. We can quantify this effect by defining two further fractions: f_{MPB} is the proportion of DIN released by respiration which is captured by microphytobenthos, and f_D is the proportion which is denitrified. Then the effective denitrification efficiency f^*_D is given by:

$f^*_D = f_D / (1 - f_{MPB})$

Recall that for Port Phillip Bay as a whole, estimates of f_{MPB} range from 45 to 55%, so that microphytobenthos typically double effective denitrification efficiency.

3. Simple process-based models

The equations presented in the previous section are really no more than restatements of conservation of mass in terms of empirical efficiencies. We cannot assume that these efficiencies will remain constant. In order to explain or predict the system response to changes in external loads or other environmental factors, we need to develop a model incorporating the biogeochemical processes controlling these efficiencies.

We would also like this model to explain the observed relationships among fluxes and pools in the Bay. For example, in the face of large DIN fluxes from both external loads and internal recycling, why should the water column pools of DIN and phytoplankton N remain so low and relatively constant, not only from year to year, but also apparently in the face of large changes in external loads over the past 50 years.

Ultimately, we must consider and model the water column and sediment components of the Bay as a single integrated system. However, it helps to understand the response of this more complicated system if we first consider the water column and sediment components separately. This separation is facilitated by the fact that the key components in the water column (DIN and phytoplankton) turn over rapidly (time scales of hours to days), whereas the sediment detrital pools turn over more slowly. Thus, to a limited extent, we can think of the planktonic ecosystem as adjusting to the total nutrient loads imposed by the net effect of nutrient cycling in the sediments.

Box 6.1: A simple model of planktonic nutrient cycling.

$$\mathrm{d}\,\mathbf{P}/\mathrm{dt} = \mu_I\,.\,\mathbf{P}\,.\,\mathbf{N}/(K_N+\mathbf{N}) - g_M\,.\,\mathbf{Z}\,.\,\mathbf{P}/(K_P+\mathbf{P}) - m_P\,.\,\mathbf{P} - \mathbf{P}/\tau$$

$$\mathrm{d}\,\mathbf{Z}/\mathrm{dt} = a\,.\,g_M\,.\,\mathbf{Z}\,.\,\mathbf{P}/(K_P+\mathbf{P}) - m_Z\,.\,\mathbf{Z} - \mathbf{Z}/\tau$$

$$\mathrm{d}\,\mathbf{N}/\mathrm{dt} = \mathbf{F_I} - \mu_I\,.\,\mathbf{P}\,.\,\mathbf{N}/(K_N+\mathbf{N}) + (1-f_W)\,.\,(\,(1-a)\,.\,g_M\,.\,\mathbf{Z}\,.\,\mathbf{P}/(K_P+\mathbf{P}) + m_P\,.\,\mathbf{P} + m_Z\,.\,\mathbf{Z}) - \mathbf{N}/\tau.$$

Here, the state variables **N**, **P**, **Z** represent the concentration of dissolved inorganic nitrogen, phytoplankton and zooplankton respectively, all in units of nitrogen (mg N m^{-3}). Each equation prescribes the rate of change of one of the state variables as a function of the values of all state variables. Each term on the right-hand side represents an ecological or physiological growth or loss term.

For the **P** equation, the first term represents phytoplankton growth, which depends on DIN concentration according to a Monod law. The second term represents ingestion by grazing zooplankton, which also increases with phytoplankton concentration according to a saturating (Holling Type-II) functional response. The third term represents direct phytoplankton mortality, primarily due to sinking.

For the **Z** equation, the first term represents growth, equal to ingestion multiplied by a growth efficiency a to allow for assimilation and respiration losses. The second term represents zooplankton mortality.

For the **N** equation, the first term $\mathbf{F_I}$ represents the total DIN load to the water column. The next term represents uptake by phytoplankton, and the third term recycling of organic N back to DIN. In all three equations, the last term represents losses to Bass Strait, and τ represents the flushing time of the Bay.

3.1 The response of simple plankton models to increasing nutrient loads

The simplest dynamical plankton models which can address the relationship between load and phytoplankton biomass have three state variables, namely dissolved inorganic nitrogen, phytoplankton and zooplankton, and are often referred to as **NPZ** models. This level of description is needed because phytoplankton biomass determines nutrient uptake rates, and zooplankton grazing determines phytoplankton losses and nutrient recycling rates. An example of an **NPZ** model is given in Box 6.1 (opposite).

It is instructive to consider how this simple model responds to a constant nutrient load $\mathbf{F}_\mathrm{I}$. In fact, the model displays two kinds of qualitative behaviour. At low nutrient loads, it approaches a steady-state solution, with **N**, **P**, and **Z** constant. However, as the load increases past a threshold level, this steady-state solution becomes unstable, and limit cycle solutions, with fluctuating levels of **N**, **P** and **Z**, appear (Figure 6.1). This phenomenon has been known for a long time as the 'paradox of destabilization by enrichment' (Rosenweig, 1971). It arises from the fact that the phytoplankton-zooplankton interaction in this model is fundamentally unstable: it is stabilized at low nutrient loads by nutrient limitation, but once nutrients accumulate in the water column, the underlying instability takes over.

Figure 6.2 shows the mean and maximum values of phytoplankton biomass predicted by the model as a function of load. At very low loads, the phytoplankton biomass is low and increases with load. In this range, the load is too low to support both zooplankton and phytoplankton mortality, and zooplankton disappear. Once the load is sufficient to support zooplankton, the predicted phytoplankton concentration is then held constant by zooplankton grazing, while the load increases by a factor of 2.5. After the instability threshold is crossed, there is a rapid jump in mean and especially in maximum phytoplankton biomass. Above a loading of about 3 mg N/m^3, both mean and maximum biomass tend to increase linearly with further increases in loading.

The instability occurs when the steady-state phytoplankton biomass, set by zooplankton

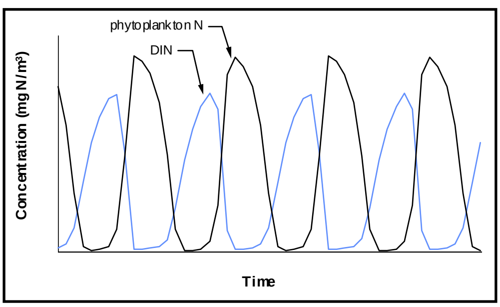

Figure 6.1: Plot of the concentration of DIN and phytoplankton N vs time, as predicted by the NPZ model in response to a constant load exceeding the threshold load by 20%.

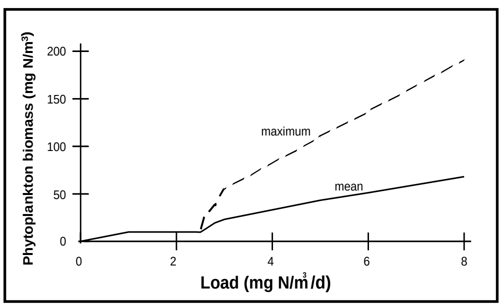

Figure 6.2: Change in mean and maximum phytoplankton biomass predicted by the NPZ model in response to different nitrogen loads.

CHAPTER 6

grazing and mortality, can no longer cope with the total nutrient load, even growing at maximum growth rate.

The critical loading is consequently determined by a small number of key system parameters or properties: the steady-state phytoplankton biomass (set by zooplankton parameters), the maximum (light-limited) growth rate of the phytoplankton, and the degree to which water-column recycling amplifies the nutrient loading.

The behaviour of this simple **NPZ** model has potentially important implications for managers.

It suggests that planktonic ecosystems may at first show little evidence of increasing eutrophication as loads increase, with phytoplankton remaining low and constant. A further small increase in load may then trigger a switch into a different kind of behaviour, with periodic phytoplankton blooms.

Environmental managers of a coastal system are more likely to be concerned with peak phytoplankton bloom densities than with average concentrations, as peak densities determine turbidity, the risk of anoxia, and the probability of toxic or nuisance algal blooms.

From a manager's viewpoint, the rapid increase in peak concentrations above the instability threshold could represent a significant deterioration in system health. Such a manager might regard the threshold loading as one measure of assimilative capacity.

How confident can we be that real systems will show this kind of behaviour? Are there alternative model formulations which display different behaviours?

The answer to the second question is definitely yes, and the alternative formulations revolve around a problem known by plankton modellers as the problem of trophic closure.

In the simple NPZ model just presented, the zooplankton mortality is described by a constant per capita loss term, m_Z. It might seem reasonable instead to have a term in the equation explicitly representing predation by carnivores on herbivorous zooplankton, analogous to the grazing term in the phytoplankton equation.

Of course, this raises the spectre of requiring an additional equation for carnivorous zooplankton, and then for fish, and so on. This is the problem of 'trophic closure'.

Most biogeochemical / plankton modellers have agreed to stop at herbivorous zooplankton, but there is some debate about the form of the loss term which should be used to try to represent the effects of higher trophic levels.

A constant per capita loss term suggests that there is a constant stock of predators, whose per capita consumption increases linearly with zooplankton abundance.

Others have argued that the concentration of predators should also increase linearly with zooplankton abundance, so that the loss term actually increases like the square of zooplankton abundance (so-called quadratic mortality). A chief protagonist for this point of view has been Steele (Steele and Henderson 1992).

This is not just an esoteric argument, because these different forms of trophic closure produce two key differences in both the steady-state and dynamical solutions of NPZ models.

First, a quadratic loss term tends to counteract the intrinsic instability of the phytoplankton-zooplankton interaction, so that steady-state solutions do not become unstable at high nutrient loads.

Second, with quadratic mortality, the steady-state phytoplankton concentration increases continuously and smoothly with nutrient loading (Figure 6.3). There is no range of loadings across which phytoplankton biomass remains constant.

On the other hand, the increase in phytoplankton biomass does accelerate above a loading of about 3 to 4 mg N/m^3, and the predicted mean phytoplankton biomass at high loadings is quite similar in both models. The quadratic mortality term effectively imposes a fixed upper limit on zooplankton biomass, and on zooplankton-mediated export of organic matter. Once this limit is reached, export is dominated by direct phytoplankton mortality.

In fact, this is true in both models: in the linear mortality model, zooplankton are also responsible for a diminishing proportion of organic matter export from the water column at high loadings. The fact that export is dominated by direct phytoplankton losses (i.e. sinking) at high loadings means that the phytoplankton loss rate m_P is a critical parameter. In fact, the predicted phytoplankton biomass becomes inversely proportional to this loss rate at high loadings.

While there are similarities in the two models, the difference in behaviour and especially in predicted peak phytoplankton concentrations is still marked. Can we distinguish between these models easily on the basis of observations? For Port Phillip Bay itself, we do not have a long enough time series of observations to make an unequivocal assessment. Most of the observations have been made in the past 30 years, a period of relatively stable nutrient loadings. Over this period, the data suggest relatively little change in mean chlorophyll levels in the Bay.

Phytoplankton levels in Port Phillip Bay averaging around 1 to 2 mg Chl *a*/m³ are considered mesotrophic. If we look at other eutrophic estuaries and lakes, we see chlorophyll levels more than 10 times those observed in Port Phillip Bay. If we look at very oligotrophic lakes or open ocean pelagic systems, we see plankton ecosystems (including zooplankton) with chlorophyll *a* levels which are more than a factor of 10 lower than those observed in Port Phillip Bay. At least for these systems which are adapted to very different N loads, phytoplankton biomass increases with nutrient load over more than two orders of magnitude. This is more consistent with the quadratic mortality model.

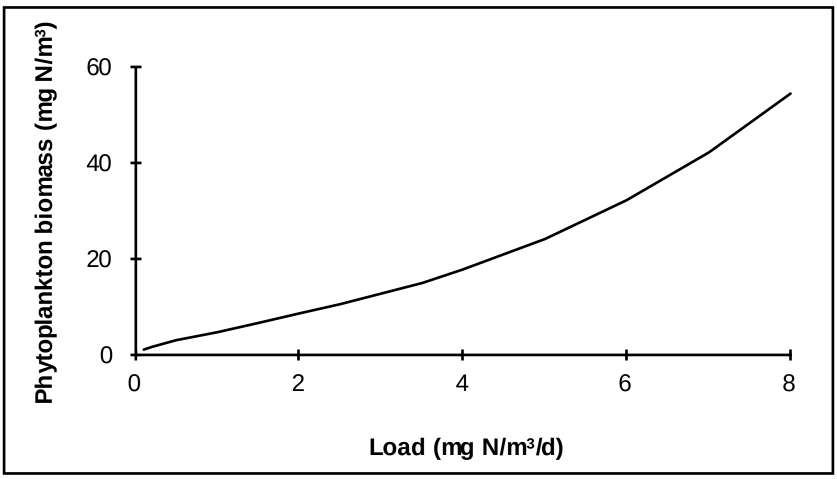

Figure 6.3: Change in phytoplankton biomass predicted by an NPZ model with quadratic zooplankton mortality, in response to different nitrogen loads.

3.2 Nutrient loads and plankton community composition

A new consensus or paradigm has started to emerge among aquatic ecologists about the response of planktonic communities to changes in nutrient loading.

Very oligotrophic communities are typically dominated by tiny phytoplankton cells (picophytoplankton and nanophytoflagellates), which compete effectively for low nutrient concentrations.

These cells are actively grazed by small zooplankton (zooflagellates and ciliates), which are similar in size to many phytoplankton and have maximum growth rates which may, in some cases, exceed those of their food.

In these oligotrophic communities, phytoplankton and zooplankton form a tightly coupled system which is very efficient at retaining nitrogen, and may turn over quite rapidly despite low external loadings.

As nitrogen loadings increase, and especially as DIN levels increase in the water column, the phytoplankton community starts to be dominated by larger bloom organisms such as diatoms or dinoflagellates.

These often have higher nutrient-saturated growth rates, although they compete less effectively at very low nutrient concentrations. They are also probably less subject to grazing losses and, therefore, have higher net growth rates when nutrients are saturated.

The lower grazing pressure may be primarily a matter of mismatch in time and space. The larger grazers (mesozooplankton) typically have longer life history cycles (months to years) and are not able to respond numerically to rapid unpredictable changes in food availability (although their life history strategies are often attuned to regular phenomena such as spring blooms in temperate waters). Still, the evidence suggests that many diatom and dinoflagellate blooms are terminated by senescence and sinking following nutrient depletion, rather than by grazing.

The picophytoplankton communities do not disappear during phytoplankton blooms. However, they typically do not increase very much either, despite the increase in nutrients. Instead, a variety of time series studies in both coastal and ocean waters (Harris *et al.* 1991) show relatively constant levels of picophytoplankton.

The observations by Beardall *et al.* (1996) in this Study support this directly: overall, coefficients of variation for picophytoplankton were only 0.44, compared with 1.27 for microphytoplankton. It appears plausible that this lack of variation in pico-phytoplankton is due to their tight coupling with grazers; that is, these small cell communities do constitute an example of the kind of grazing control predicted by the linear mortality models above.

This paradigm about plankton communities seems to offer an alternative resolution of the choice offered by the trophic closure models. It suggests that we should divide our phytoplankton and zooplankton variables each into at least two size classes: small and large.

The small phytoplankton should be characterized by lower maximum growth rates but high affinity for DIN; the large phytoplankton by higher maximum growth rates and lower affinity for DIN.

The small phytoplankton should be grazed by small zooplankton with high maximum grazing and growth rates (of order 1 to 2/d), and the large phytoplankton by zooplankton with quite low maximum growth rates (of the order 0.01 to 0.1/d).

Given the stability of the small community, a quadratic mortality term seems most appropriate for small zooplankton.

It is not clear which is more appropriate for large zooplankton. However, it seems preferable that, at least during blooms, the direct phytoplankton loss term m_P should dominate over grazing losses for the large size class.

In Port Phillip Bay, we would expect losses due to sinking to dominate m_P, and sinking rates should be zero for small phytoplankton and high for large phytoplankton.

One should always be wary of including more variables in a model in a search for more realism, unless the observations justify it.

The phytoplankton field studies carried out by Beardall *et al.* (1996) as part of the Study did measure phytoplankton biomass, growth rates and grazing rates in three size fractions: 0.2 to 2 μm, 2 to 20 μm and >20 μm. These correspond to the standard picophytoplankton, nanophytoplankton and microphytoplankton categories used by biological oceanographers.

It is reasonable to assign the picophytoplankton to the small (non-bloom) model category, and the microphytoplankton to the large (bloom) category. However, the nanophytoplankton represent something of a dilemma, as they almost certainly include a mixture of non-bloom and bloom species.

As our process understanding does not really justify modelling three phytoplankton size fractions, it seems better to assume that there is some background component of the nanofraction which is functionally non-bloom, and a fluctuating component which is functionally bloom.

3.3 The processes controlling export efficiency

This section so far has treated the plankton community in isolation, but of course it is tightly coupled to the benthic community. In the simple dynamical models presented above, the loading term $\mathbf{F}_I$ included nutrient fluxes from the sediment, and the organic matter export to the sediment was controlled by a single parameter, the export efficiency f_W. However, f_W is really a system property which is likely to change under changes in nutrient loading. In order to predict these changes, we need to consider the fate of organic matter in more detail.

Direct phytoplankton mortality is assumed to be dominated by sinking, and the sinking rate is assumed to be zero for small cells, and of order 1 to 2 m/d for large cells. Sinking represents a transfer of the associated organic matter to the sediment with close to 100% efficiency (the model does allow resuspension of viable cells in shallow water).

The difference between zooplankton ingestion and growth constitutes losses in assimilation and to respiration. That proportion of food not assimilated is excreted as fecal pellets, or lost in messy feeding, and joins the pool of labile detritus in the water column. In the model, labile detritus has a relatively high sinking rate of about 5 m/d. The loss due to respiration is excreted directly as DIN. Nitrogen associated with the zooplankton mortality term is assumed to enter the labile detritus pool.

From this discussion, we can see that the effective export efficiency f_W is indeed a dynamical variable. It depends not only on phytoplankton and zooplankton parameters and relative growth and assimilation efficiencies, but also on the balance between growth and mortality, and on the balance between small and large cells. For the small phytoplankton – zooplankton community, the export efficiency is low. For the large phytoplankton – zooplankton community, export efficiency should be high, both through direct sinking of phytoplankton, and through the efficient transfer of large faecal pellets and corpses to the sediment.

In dealing with the fate and recycling of organic material in the water column, we need to consider two new pools of organic nitrogen: labile detritus, produced in the water column, and dissolved organic nitrogen.

The labile detritus is assumed to sink rapidly, and to be broken down fairly rapidly (on time scales of days) by microbial decomposition to produce ammonium, although some proportion is assumed to be resistant to breakdown and to enter a long-lived refractory detritus pool.

In the full integrated model, suspended detritus (both refractory and labile) is potentially exchanged rather rapidly with the surface sediment through physical resuspension and sedimentation, so that these pools cannot be considered in isolation, even temporarily, from sediment processes.

The model currently does not include labile DON, and represents only a semi-labile or long-lived DON component, with a typical lifetime of about 200 days. The model implicitly treats labile DON as if it is instantly converted into ammonium. DON is produced by the breakdown of both labile and refractory detritus in the water column and in the sediment.

Other recent biogeochemical/plankton models (e.g. Fasham *et al.* 1990) have explicitly represented bacteria as a state variable, and dealt explicitly with the processes controlling bacterial biomass and remineralization of organic matter. In common with many other modellers, we have chosen to represent these processes implicitly. We do not think this affects the conclusions, and the Study did not include measurements of microbial biomass or activity.

One of the key difficulties in dealing with both particulate detritus and dissolved organic matter is that these are complex mixtures of compounds having very different sources and breakdown rates, yet standard measurements deliver only bulk estimates.

The Study obtained direct measurements of total PON and DON pools in the water column. Our estimates of phytoplankton N come only from applying typical N:Chl *a* ratios.

If we assume that the heterotrophic organic N pool is comparable in size to the phytoplankton pool, it appears that the autotroph, heterotroph and detrital pools each contribute approximately one-third of the 600 t of total particulate N in the water column.

The DON pool in the water column is much larger (about 3,500 t), although up to 2,500 t of this may be highly refractory material of marine origin. One might reasonably expect the DON load to the Bay to include both labile and refractory DON.

Measured DON concentrations are calculated as the difference between total dissolved (kjeldahl) nitrogen, and ammonia, and potentially include highly labile forms such as urea.

Recent studies (Carlson *et al.* 1994) suggest that dissolved organic matter in open ocean waters consists of three parts: a background level of highly refractory material, a small highly labile component which turns over on time scales of days, and a seasonal component which turns over on time scales of months and reaches peak concentrations of the same order as the background.

In a coastal system such as Port Phillip Bay, we might expect the sediment to be of equal or greater importance than the water column in producing this semi-labile DON. In Port Phillip Bay, concentrations of DON in sediment porewaters are about 10 times water column concentrations, suggesting a net flux from sediments to the overlying water column.

3.4 The fate of detrital organic matter in the sediment

Detrital organic matter produced in the water column and settling to the surface of the sediment supports a rich and diverse community of benthic infauna, and an epibenthic community of filter-feeders, as well as a bacterial community.

As explained in Chapter 3, from a biogeochemical point of view, there are a number of key chemical transformations which this organic matter, and its associated nitrogen, can undergo. These include aerobic and anaerobic respiration, nitrification and denitrification. These reactions, together with physical transport, determine the fraction of nitrogen returned to the water column as ammonia or nitrate, lost to denitrification, or buried as organic N in the sediments. It is the concentration of oxygen which determines the relative reaction rates, and the ultimate fate of the organic nitrogen.

There are a number of models describing these reactions in a simplified form (Blackburn and Blackburn 1993, Omori *et al.* 1994). Following Omori *et al.* we defined five key state variables: the concentrations of labile detritus, oxygen, ammonia, nitrate and sulphide. The model also specifies oxygen dependencies and threshold oxygen concentrations for aerobic and anaerobic respiration, nitrification and denitrification.

Denitrification can also be driven directly by the presence of sulphide, but sulphide oxidation proceeds very rapidly in the presence of non-zero concentrations of sulphide and oxygen.

It is the effect of the biota on the physical structure and transport processes in sediments which makes modelling sediment biogeochemistry particularly difficult. Standard one-dimensional sediment models compute vertical profiles of porewater concentrations by combining the local reaction models with a diffusive transport model.

In Port Phillip Bay, the boundary conditions at the surface of the sediment prescribe high O_2, low NO_3 and NH_4 and zero sulphide. Reaction-diffusion models predict a rapid depletion of oxygen with depth, an accumulation of ammonium with depth, and a near-surface layer of nitrate production. The depth of oxygen penetration may be very small. In a series of careful measurements and model calculations, Burke (1995) concluded that oxygen demand is typically so high that oxygen penetrates only about 1 to 2 mm into Port Phillip Bay sediments.

Against this smooth diffusive background, we need to consider two sources of spatial heterogeneity. The distribution of organic matter and bacteria around individual sediment grains is likely to create strong variation in oxygen demand and oxygen concentrations on sub-millimetre space scales. This microscale patchiness is likely to have significant impacts on non-linear reactions with oxygen thresholds. In particular, reactions which might in theory be expected not to co-occur in the same depth layer, or at a given mean layer oxygen concentration, may co-occur. Models try to mimic the effects of this microstructure by shifting or blurring oxygen thresholds.

Heterogeneity is particularly important for denitrification, which apparently involves a series of processes with incompatible oxygen requirements. Respiration proceeds most rapidly if it is aerobic, nitrification requires oxygen, while denitrification requires an anoxic environment.

In a simple 1-D steady-state diffusive model, ammonia produced in or below the thin oxygenated layer must be nitrified within the oxygenated layer, and the resultant nitrate must then diffuse down into the anoxic layer to be denitrified, all without diffusing upward into the overlying water column. It seems likely that this process might be facilitated in the presence of microscale structure in oxygen concentrations, or in an environment with temporal fluctuations in oxygen levels.

Aside from microscale heterogeneity, burrowing infauna impose macroscale structure on the sediments. In Port Phillip Bay, the infauna community appears to be particularly well-developed and important, and has two important effects on sediment biogeochemistry.

One is to redistribute particulate material in the vertical (bioturbation). The ^{210}Pb and TBT results from Port Phillip Bay suggest that burrowing infauna mix the top 30 cm of sediment on time scales of 20 years. These results suggest a maximum mixing time for the top 1 cm of about 1 week: in practice, it is likely to be much shorter, as burrowing activity will be more con-centrated in surface sediments.

Bioturbation, combined with the temporal pattern of surface arrival and the vertical distribution of breakdown rates, determines the vertical profiles of both labile and refractory organic matter in the sediment. The observations show that total PON is almost uniformly distributed over the top 20 to 30 cm throughout most of the Bay. However, the labile fraction is almost certainly a very small percentage of total PON, and is probably concentrated near the surface.

The second effect of infauna is the accelerated movement of porewater by pumping within burrows (bioirrigation). Measurements of radon in benthic chambers in Port Phillip Bay suggest that bioirrigation is much more important than diffusion in terms of the exchange between bulk porewater and the overlying water column.

However, its importance for individual solutes will depend on their vertical distribution and reaction processes. For example, it appears that about half of the water-sediment oxygen flux is accounted for by diffusion within the top few mm, and about half by bioirrigation. For most solutes, changes in concentration within burrows may be small compared with changes between burrows and the surrounding porewater. This means that steep diffusion gradients across the surface extend to steep gradients surrounding burrows. One can think of the burrow system as a great expansion of the sediment-water interface, analagous to a lung.

In fact, there is evidence from other sediment studies that the large burrows, such as those of the callianassids in Port Phillip Bay, are just the trunks of an extensive system of finer burrows, so that the lung analogy may be a good one. While some 1-D models have tried to represent the effect of bioirrigation as a non-local transport process, it is not clear that this is an appropriate solution.

Given these difficulties, the predictions of sediment biogeochemical process models

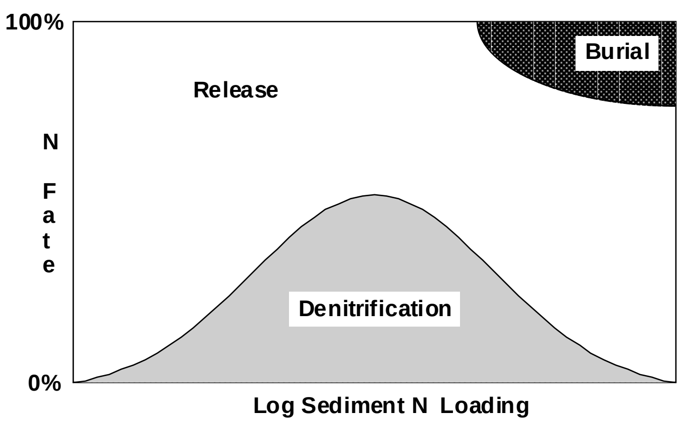

Figure 6.4: Changes in the fate of remineralised nitrogen with increasing sediment organic load.

must be treated with some caution. However, they do agree, at least qualitatively, on the effect of increasing organic nitrogen load to the sediment on changes in the fate of organic nitrogen.

This pattern, shown in Figure 6.4, is easily understood. At very low organic nitrogen loads, oxygen consumption is low, the oxygenated zone is deep, and most organic matter is remineralized in the upper part of the oxygenated zone. Most of the resulting ammonia is converted to nitrate, but very little of this nitrate encounters low enough oxygen levels for denitrification to occur. Denitrification efficiency is therefore low, and inorganic nitrogen is recycled to the water column in the form of nitrate with high efficiency.

At very high organic loads, the oxygenated zone becomes extremely thin, and the bottom water itself may become anoxic. There is little or no chance for ammonia released by remineralisation to be nitrified, and almost all is returned to the water column as ammonia. Denitrification cannot occur, because nitrate is not produced. Because anaerobic metabolism tends to proceed at slow rates, the organic N load may exceed the metabolic capacity, resulting in the burial of potentially labile organic N in the sediment.

It is at intermediate organic N loads, where there is a strong interface between oxygenated and anoxic sediment zones, that denitrification efficiencies are maximum.

While this qualitative pattern is quite robust, the absolute efficiencies, and the quantitative dependence on organic N load, depend on the specific assumptions and parameters in the model. One feature common to models with relatively simple layer geometries is that they do not predict maximum denitrification efficiencies greater than 50%. There is again a relatively simple explanation for this. Given that a nitrate molecule has been produced through nitrification in an oxic zone, it may diffuse in one of two directions relative to the local oxygen gradient. It can diffuse towards higher oxygen levels, in which case it is likely to remain as nitrate and escape to the overlying water column, or it can diffuse towards lower oxygen concentrations, in which case it will probably be denitrified.

This simple random walk model suggests it has no better than a 50% chance of denitrification, whether the local gradient is vertical, in the near surface layer, or horizontal, around burrow walls.

One can develop hypothetical mechanisms which could allow higher denitrification rates, but these are as yet speculative. (An exception is the effect of microphytobenthos on nitrogen cycling in sediments, discussed in Section 2.1.)

The qualitative changes in denitrification efficiency with organic load predicted by these models are in agreement with observations. In fact, the benthic chamber results from Port Phillip Bay are also consistent with this pattern. They show a decline in apparent denitrification efficiency with increased load, with minimum denitrification efficiencies in Hobsons Bay (Figure 6.5). (In Figure 6.5, the denitrification rate is not corrected for microphytobenthic production.) Figure 6.5 shows little evidence of low denitrification efficiencies at very low loads, as predicted in Figure 6.4. This is hardly surprising, given that the depth of the oxygenated zone is already very small throughout most of Port Phillip Bay.

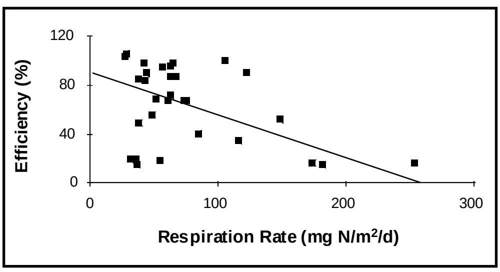

Figure 6.5: Apparent denitrification efficiency (%) vs sediment respiration rate from sediment chamber deployments in the Bay.

3.5 Processes at the sediment – water interface

Aside from the transport processes already discussed, there are several potentially important biological processes at the sediment – water interface. These involve benthic primary producers, and epibenthic grazers (filter feeders).

The benthic primary producers can be divided into three functionally distinct groups.

Seagrasses are rooted in the sediment, and obtain their nutrient from sediment porewater. They generally have high light requirements, and are found in shallow water (<5 m) in Port Phillip Bay.

Seagrasses appear to be disadvantaged by high nutrient concentrations, perhaps because these encourage growth of epiphytic algae on seagrass leaves (see Chapter 5).

Macroalgae may be attached to hard substrate, or unattached (Port Phillip Bay contains extensive beds of filamentous drift algae: Chidgey and Edmunds 1996). They can have lower light requirements than seagrass (some species are found down to 10 m or deeper), and obtain their nutrients from the water column.

They do not appear to compete well at very low nutrient levels, but have a considerable capacity for 'luxury' uptake and storage of nutrients during periods of elevated concentration.

Grazers on macroalgae and seagrasses are not represented explicitly, and these components are assigned low, constant per capita loss rates, so that they build up a high biomass which turns over slowly.

The microphytobenthos have already figured largely in the discussions of sediment nitrogen and oxygen budgets.

They consist primarily of benthic diatoms, and are not strictly speaking epibenthic, as they are distributed through the top 2 cm of sediments.

Thanks to the study by Beardall and Light (1996), we have good information on their distribution, abundance, and productivity in the Bay. Based on that study, maximum photosynthetic rates and growth rates for microphytobenthos are comparable to those of phytoplankton, and microphytobenthos are strongly shade-adapted especially in deeper water, with I_k values below 20 µmoles/m^2/s.

These values suggest the potential for significant oxygen production by microphytobenthos even at 20 m depth in the Bay centre, a conclusion supported by benthic chamber results.

Microphytobenthos are exposed to elevated concentrations of nutrients in sediment porewater and, being diatoms, require silicate as well as nitrogen and phosphorus.

Again, grazers on microphytobenthos are not represented explicitly in the model. However, as grazers on microphytobenthos may include meiofauna with a rapid numerical response, they are assigned a quadratic mortality loss term.

The microphytobenthos are clearly major players in the biogeochemical and ecological cycles in the Bay, with an estimated N pool size about twice that of phytoplankton, and an estimated primary production about half that of phytoplankton. They are particularly important because of their link to denitrification efficiency in sediments.

The macroalgae are also potentially important. The N pool tied up in macroalgae is larger than that in phytoplankton, but they are expected to turn over much more slowly.

The seagrass are spatially restricted to a relatively small area of the Bay. They are potentially locally important (like microphytobenthos, they may trap DIN released in the sediment and recycle it back as organic matter), but are unlikely to have a major impact on Bay-wide nitrogen budgets.

On the other hand, both macroalgae and seagrasses are seen as important environmental indicators for Bay managers, and we have therefore retained these components within the integrated model.

The benthic filter feeders consist of a diverse set of molluscs, polychaete worms, and crustaceans which filter living and dead organic matter from bottom water. This functional group can be particularly important in shallow coastal environments, and is thought to play an important role in Port Phillip Bay.

Estimated biomass and clearance rates at the beginning of the Study (Wilson *et al.* 1993) suggested that benthic filter feeders may sweep the water column every 16 days.

In the model, this group is given a relatively low maximum ingestion rate and clearance rate per unit biomass compared with zooplankton, but feeds on both living matter and labile detritus, with a lower assimilation efficiency on detritus. Filter feeder mortality is assumed to be quadratic.

The benthic filter feeders may effectively increase the recycling efficiency of the water column, by ingesting particles settling to the bottom, and excreting DIN directly into the water column.

3.6 The response of the combined water-column and sediment system

We now have a qualitative understanding of the likely response of the plankton system to changes in its DIN load, which represents the sum of the external load from the catchment and the recycled flux from the sediment. We also have a qualitative understanding of the likely response of the sediment system to changes in external loads of organic N, which in Port Phillip Bay comes primarily from phytoplankton production in the water column. In practice, of course, these are tightly coupled systems. What happens when we put them together?

It is possible to deal with this at a very simple level using some of the recycling efficiency concepts which have already been introduced. We have defined the denitrification efficiency f_D for the sediment so that the denitrification flux equals f_D times the total sediment respiration rate, F_R. Sediment process models suggest that the denitrification efficiency should decrease as sediment respiration increases, and in fact the sediment chamber studies in the Bay suggest that denitrification efficiency decreases roughly linearly with increasing sediment respiration rate:

$$f_D = f^M_D \, . \, (1 - F_R / F^0_R).$$

The chamber data (Figure 6.5) show that denitrification efficiency approaches zero at a respiration rate F^0_R equivalent to 250 mg N/m^2/d, or on a Bay-wide basis, to a flux of 170,000 t N/yr The value to assign to f^M_D is not so clear, because we need to correct the chamber denitrification rates for the effects of microphytobenthos production. Attempts to balance the Bay-wide budget led to Bay-wide average estimates of f_D ranging from 0.2 to 0.35, or values of f^M_D ranging from about 0.25 to 0.4.

The denitrification flux is given by $f_D \, . \, F_R$, or $f^M_D \cdot F_R \cdot (1 - F_R / F^0_R)$. According to this expression, as the sediment respiration rate increases, the denitrification flux will increase initially, but then reach a maximum when the respiration rate is just half F^0_R. Further increases in respiration rate will lead to decreasing denitrification flux, until it finally drops to zero when the sediment respiration rate equals F^0_R. We can think of the maximum rate, given by $(0.25 \cdot f^M_D \cdot F^0_R)$, as an intrinsic denitrification capacity. For the parameter values above, **this capacity corresponds to about 11,000 to 17,000 t N/yr Bay-wide**.

Once the catchment load exceeds the denitrification capacity, the surplus must be balanced by export out of the Bay. This means that levels of particulate or inorganic nitrogen in the water column must start to build up so that export out of the Bay matches the excess load.

However, as levels of particulate N build up, the flux to the sediment due to sinking will also increase, as will the sediment respiration rate.

We might expect the ratio of flux to Bass Strait over sediment respiration rate to match the ratio of flushing rate to loss rates due to sinking. For example, if N was primarily exported as phytoplankton which sank at 1 m/d in a water column with an average depth of 14 m, and the flushing time was one year, the ratio would be about 14/365, or 0.04. This suggests that we can approximately (at least for high loadings and biomass) treat the export to Bass Strait as a constant fraction f_B of the sediment respiration rate. We then have:

Catchment load = denitrification flux + export to Bass Strait

$$= f^M_D \cdot F_R \cdot (1 - F_R / F^0_R). + f_B \cdot F_R.$$

This relationship is plotted in Figure 6.6, for $f^M_D = 0.4$, and $f_B = 0.04$. We see that there are three parts to this curve. The lower solid portion corresponds to the current regime in the Bay (the arrow indicates the approximate current state). In this regime, denitrification is the dominant loss term for nitrogen. As catchment load is increased, the net sediment respiration must increase to support increased denitrification. However, denitrification efficiency is dropping, and so the sediment respiration increases ever faster, until finally it reaches a point where the total denitrification flux reaches the Bay's maximum capacity.

The dashed part of the curve represents a theoretical state where catchment loads are low, but denitrification efficiency is also low, and sediment respiration is very high. However, this dashed portion is unstable. Any real system pushed beyond the maximum

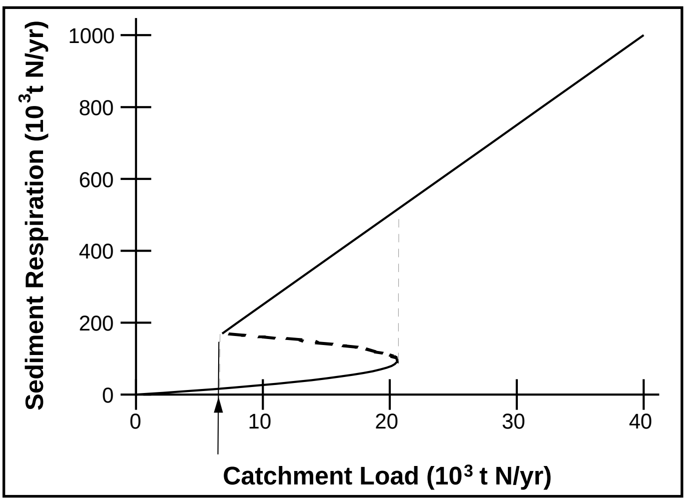

Figure 6.6: The steady-state relationship between catchment load and sediment respiration rate, based on sediment chamber results.

denitrification capacity will instead jump to the upper solid curve. This is a curve in which denitrification is shut down completely, and catchment loads to the Bay are instead exported entirely to Bass Strait. This involves an order of magnitude increase in sediment respiration, which in turn implies an order of magnitude increase in phytoplankton biomass and production, involving Bay-wide chlorophyll levels in the range 10-20 mg Chl a/m^3. At this point, the Bay would have switched to a highly eutrophic state, with severe impacts on benthic macrophytes and infauna.

Now, consider what happens if managers decide that the highly eutrophic situation on the upper curve is unacceptable, and start to decrease the catchment load.

In a steady-state analysis, denitrification will remain switched off as the load decreases, because the sediment respiration still exceeds F^0_R. Denitrification will not begin to recover until the catchment load drops below the minimum value on the upper solid curve. At that point, the system will drop back to the lower solid curve, with low sediment respiration and high denitrification efficiency.

In fact, in tests with simple models, we have found that the dynamical response depends on the water column model formulation. Models with quadratic zooplankton mortality have a stable steady-state corresponding to the upper (eutrophic) curve, and display the predicted steady-state hysteresis, remaining trapped on that curve as loads are decreased. Models with linear zooplankton mortality undergo large amplitude limit cycles on the upper curve. Once loads are decreased below the denitrification capacity, these models jump back to the lower curve, and do not display hysteresis.

While the arguments underlying the qualitative pattern of behaviour shown in Figure 6.6 seem reasonably robust, rather wide confidence limits should be placed on the key parameters, particularly the upper and lower transition loads. The denitrification capacity is calculated by extrapolating an empirical relationship, based on spatial variation in denitrification efficiencies within the Bay, to Bay-wide fluxes. The capacity in Figure 6.6 uses the high estimate of f^M_D of 0.4, corresponding to minimum estimates of sediment respiration. The low estimate of 0.25 would yield a capacity of about 14,000 t N/yr.

Aside from the obvious problems of spatial extrapolation, there are other potential feedbacks in such large-scale Bay-wide changes. At present, dissolved oxygen levels in bottom water are generally high, and are not observed to fall below about 70% saturation (Mickelson 1990). However, a three or four-fold increase in sediment respiration could lead to bottom water anoxia during periods of prolonged stratification, with catastrophic impacts on benthic fauna.

Aside from the implications for conservation, biodiversity, fisheries and recreation in the Bay, this could produce important and long-lasting changes in bioirrigation rates. Denitrification rates would not only drop immediately to near-zero, but might be expected to remain low until benthic fauna recover, if the role of burrows and bioirrigation is as important as we think.

We argued earlier that the effective denitrification efficiency in the sediments is significantly increased (by about a factor of two) by internal nutrient recycling by microphytobenthos. We have corrected for this effect in f^M_D, but have assumed that the high oxygen consumption rates observed in Hobsons Bay, corresponding to near-zero denitrification efficiencies, can be used to estimate F^0_R without correcting for microphytobenthos production.

In the Bay currently, light is a key limiting factor for microphytobenthos. If phytoplankton or detritus concentrations increased so as to significantly increase attenuation coefficients and decrease bottom light levels, enhancement of denitrification by microphytobenthos would diminish. This feedback is addressed in the integrated model results, presented below.

CHAPTER 6

4. An integrated process model of nitrogen cycling

4.1 Model processes

The preceding sections have introduced the key variables and processes involved in nitrogen cycling in Port Phillip Bay. These have been combined to produce an integrated model, consisting of water column, sediment, and epibenthic modules, linked to the transport model described in Chapter 2.

The water column variables are ammonia, nitrate, small and large phytoplankton, small and large zooplankton, labile and refractory detritus, and dissolved organic matter.

The epibenthic variables are seagrass, macroalgae, microphytobenthos, and filter feeders.

Because this is essentially a nitrogen cycle model, all living and organic components are by default measured in terms of nitrogen content.

Note that DIN has been separated into nitrate and ammonia. This was done primarily to provide an additional means of calibration, as nitrate and ammonia have different principal sources, and different roles in the sediment cycle.

Given the uncertainties and difficulties involved in process-based sediment models, the integrated model in its current form does not attempt to model sediment processes in detail, but uses the empirical relationship between denitrification efficiency and oxygen consumption derived from benthic chambers in Port Phillip Bay (Figure 6.5).

Labile and refractory sediment is allowed to accumulate in the upper sediment layer, and breaks down at a fixed (temperature-dependent) rate.

The empirical relationship then determines the proportion of inorganic nitrogen produced as nitrate or ammonia, or lost to denitrification. A fixed proportion of the labile and refractory sediment is converted to semi-labile DON.

As well as representing the nitrogen cycle in detail, the model also carries phosphate and oxygen, and calculates the impact of biogeochemical transformations on their concentrations assuming fixed Redfield ratios for carbon to nitrogen, oxygen to nitrogen, and nitrogen to phosphate.

There is recent evidence that carbon to nitrogen ratios can vary from Redfield in coastal waters under nitrogen limitation, but this is likely to be a relatively small effect.

In order to compare predicted phytoplankton concentrations with observations, fixed C:Chl *a* and N: Chl *a* ratios of 50:1 and 7:1 are assumed.

Given the importance of diatoms in Port Phillip Bay, and the significance of silicate limitation in some seasons, it would be highly desirable to model silicate as well.

This has not yet been done, primarily because silicate has not been routinely measured in the principal rivers and drains, and so the silicate loads are not well known.

We plan to carry out a trial implementation of silicate dynamics during the remainder of the modelling task.

To allow for the effects of changes in water column components on light attenuation, the attenuation coefficient is modelled as a sum of contributions from a background component, chlorophyll, DON, detritus and inorganic suspended sediment, each with its own specific absorption coefficient.

All rate parameters in the model are assumed to be temperature dependent, and are assigned a default Q_{10} value of 2.

Like all such dynamical models, this model prescribes the local rate of change of its state variables based on their current values.

It has been implemented in a modular way and integrated with the transport model described in Chapter 2, to provide a model capable of predicting the evolution of state variable fields with (in principle) arbitrary spatial and temporal resolution.

4.2 Temporal forcing of Port Phillip Bay

Temporal variation in the Bay is driven by environmental forcing from a variety of sources. Variation in circulation and mixing is prescribed by the hydrodynamic and transport models. There is also variation in incident solar radiation, prescribed from historical records, and in temperature (again prescribed by the hydrodynamic model). Both variables in Port Phillip Bay follow a regular seasonal cycle with some short-term fluctuations.

As noted earlier, all biological rate processes in the model are assumed to be temperature dependent, with a default Q_{10} of 2. The seasonal variation in light intensity affects primary production levels where these are light limited.

The most important temporal variation is that associated with changes in loading of nutrients from the key point sources: WTP, the Yarra, and the major creeks. Of these, only the WTP is monitored with sufficient frequency and consistency to allow us to prescribe inputs into the Bay directly.

The data sets available for the Yarra-Marribyrnong system, and the major east coast creeks and drains, consist of continuous runoff records but only intermittent measurements of loads or concentrations.

An early Study Task estimated annual loads from these sources (NSR 1993) but, of course, annual loads are of little use in driving a dynamical model.

Sokolov (1996) made a particularly valuable contribution to the Study by developing simple semi-empirical catchment models and fitting them to the available load and runoff data for these sources. Once the model parameters were estimated for each source, he was able to calculate daily loads over the entire period for which continuous runoff records were available. These predicted loads provide our best current estimates of annual loads and their interannual as well as daily variation, for the principal sources.

The Study faced a further complication in the Yarra River, where the gauging and monitoring stations are well upstream of the mouth of the estuary. As the estuary dynamics were not part of the Study, we needed to find a way to reliably extrapolate from loads into the estuary to loads into Port Phillip Bay. This was finally accomplished by instituting a high-frequency monitoring program at the Westgate Bridge for a period of six months. Sokolov has subsequently used these data to calibrate an empirical model to link loads at the Bridge to freshwater loads (Sokolov 1996).

This model is complicated by the fact that it must deal with a two-layer estuarine circulation, with outflow of dissolved constituents to the Bay in the surface layer, and net inflow of particulates from the Bay in the bottom layer.

The temporal patterns in the principal nutrient sources have already been discussed in Chapter 3. The nitrogen load from the WTP is dominated by a regular seasonal cycle in ammonia inputs, with a maximum in winter. The load from other catchments is highly variable with strong interannual variation in both annual load and timing of peak loads. In the Study period, 1993 was a year of unusually high runoff, and 1994 a year of unusually low runoff (Figure 6.7).

At present, we have available calibrated hydrodynamic model output to drive the transport model only for the years 1993 and 1994. We also have observed (calculated) daily loads for that period. We have therefore run the integrated model over the 1993-1994 period. This overlaps with about 19 months of the Bay-wide nutrient surveys, and a substantial portion of the other nutrient tasks.

We do not have the information needed to accurately define initial conditions at the beginning of 1993. While some model variables such as water column DIN can be expected to relax rapidly to values determined by current loadings, other variables such as sediment detrital N can take several years to equilibrate with a given loading pattern.

To avoid artifacts arising from initial conditions, we have run the models out for up to 20 years by repeating the 1993-1994 forcing until they approach a quasi-steady two-year cycle. Results are presented for the last two years, unless otherwise specified.

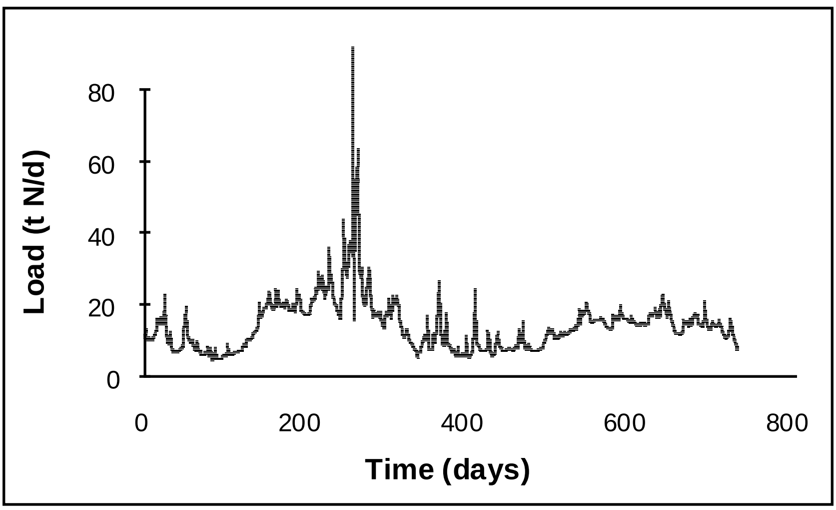

Figure 6.7: Estimated N load to the Bay for 1993-1994.

4.3 The response of a single compartment model of the Bay to temporal forcing

There are striking spatial gradients and patterns in nutrient dynamics in Port Phillip Bay (Chapter 3), and the full integrated model has been implemented with a spatial resolution designed to address these. However, analysis and understanding of the full spatial model is a complex task.

In order to improve understanding, we present first results from the integrated model, implemented as a single spatial compartment. This one-box model represents an end-to-end dynamic test of the budget, cycling and dynamical concepts we have discussed in the preceding sections.

We present first summary annual budgets of the key pools and fluxes at the current loadings, for versions of the models with linear or quadratic zooplankton mortality (Table 6.1). There is little in these summaries to distinguish the models.

With the default parameter set, both models predict annual totals for primary production, sediment respiration, microphytobenthos production and efflux from sediments which are more or less in accord with the summary budget based on observations presented in Section 2 (the models have been run with f^M_D set to 0.25, and not surprisingly produce budgets consistent with the 'high flux' budget in Section 2).

Both models predict that the majority of the N load is lost to denitrification, with a significant contribution (about 900 t/yr) from DON export. (The DON values predicted by the model represent values in excess of background Bass Strait concentrations, and correspond to excess concentrations of about

Table 6.1. Summary nitrogen budgets for the Bay-wide models with linear mortality and quadratic mortality, in response to estimated loadings to the Bay in 1993 and 1994.

Model	Linear Mortality		Quadratic Mortality	
Year	1993	1994	1993	1994
Flux (t N/yr)				
Catchment Load	7,700	6,000	7,700	6,000
Primary Production	33,800	30,500	37,400	34,800
Recycling (water column)	21,000	19,000	24,800	23,500
Efflux from sediments	7,600	6,800	7,400	6,600
Microphytobenthos	16,800	16,500	16,900	16,300
Sediment respiration	32,600	31,000	32,200	30,600
Denitrification	6,300	6,000	6,200	6,000
Pool size (t N)				
DIN	64	52	93	85
Phyto-N	275	270	200	180
DON	870	890	870	890

3 µM.) The predicted phytoplankton N pool is about 200 to 300 t N, equivalent to a chlorophyll level of about 1 to 1.5 mg Chl *a*/m^3. The DIN pool is somewhat low, and corresponds to a concentration of about 0.2 µM.

Both models tend to diminish rather than amplify the interannual differences in N-loads to the Bay. While the difference in load between 1993 and 1994 is about 20%, the differences in most fluxes are closer to 10%. This may be partly due to the fact that there is a significant runoff event in late 1993 (Figure 6.7), whose effects are reflected in Bay fluxes in 1994. The model does include semi-refractory pools of PON and DON, with turnover times of one year and 200 days respectively. Together with the long Bay flushing time, these pools tend to dampen the Bay's response to changes in load from year to year.

For the linear mortality case, plots of model variables and fluxes versus time (Figure 6.8) show two dominant time scales: a seasonal cycle, and short period oscillations. The short period oscillations in phytoplankton, DIN and primary producton indicate that at these high production rates, the system is poised around the threshold of the phytoplankton – zooplankton instability discussed in Section 3.1.

The amplitude of this cycle is seasonally modulated, and increases during the high production period in spring and early summer in both years. There is an interesting phase shift in the seasonal cycle among pools and fluxes, with peak DIN building up in late

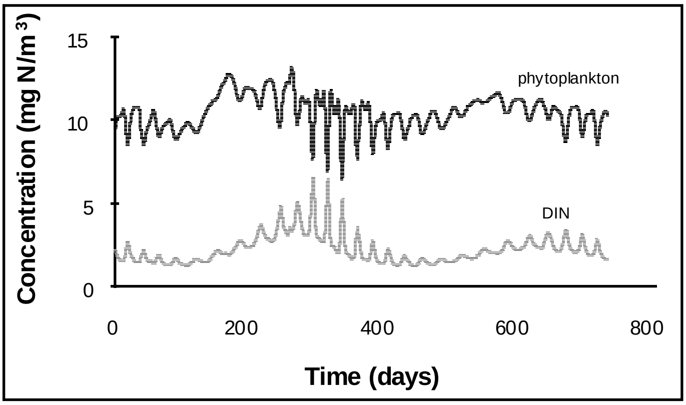

Figure 6.8a: Concentrations of phytoplankton and DIN vs time, predicted by the linear mortality model for 1993-1994.

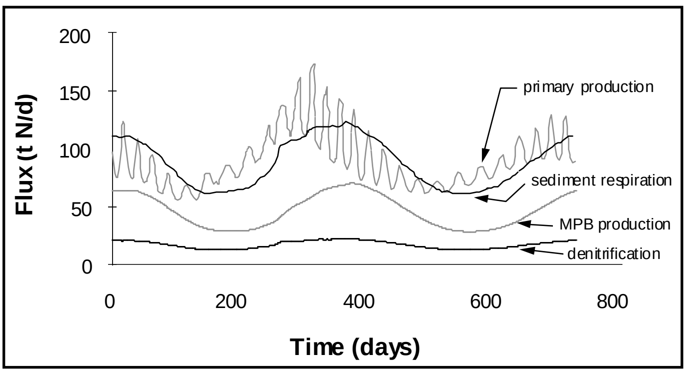

Fig 6.8b: Fluxes of primary production, sediment respiration, microphytobenthic production and denitrification for the linear mortality model, 1993-1994.

winter and early spring (probably in response to the Werribee input), but peak primary production occurs in spring and early summer, peak microphytobenthic production in midsummer, and peak sediment respiration in late summer. It appears that sediment respiration is driven primarily by the seasonal cycle in temperature. Peak denitrification rates tend to coincide with peak microphytobenthic production.

In comparable plots for the quadratic mortality model (Figure 6.9), there are no short period oscillations in DIN, phytoplankton N and primary production. The quadratic mortality model does show a pronounced seasonal cycle in phytoplankton biomass, peaking in spring and early summer. Both these differences in behaviour between the two models are entirely consistent with the earlier analysis of simple models.

The increased production and sediment respiration associated with the increased load in late 1993 continue into early 1994. This explains why the models do not predict a large difference in annual fluxes between 1993 and 1994. There is a marked interannual variation in primary production in Figure 6.8 and 6.9, but this occurs in years from May to May, rather than calendar years.

While the annual N budgets predicted by the linear and quadratic mortality models are quite similar at current loads, the models differ significantly in their response to increased loads. Both models were run to steady biennial cycles with repeated 1993/1994 forcing, for a range of different loadings, artificially imposed by multiplying all N and P loadings from point sources (but not the atmospheric N load) by factors ranging from 0.5 to 10. The responses of the key pools and fluxes, averaged over the last two years of simulation, are shown in Figure 6.10 and Figure 6.11.

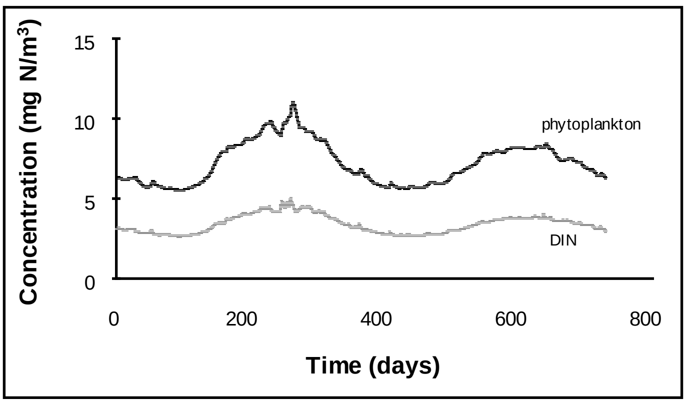

Figure 6.9a: As in Figure 6.8a, but for quadratic mortality.

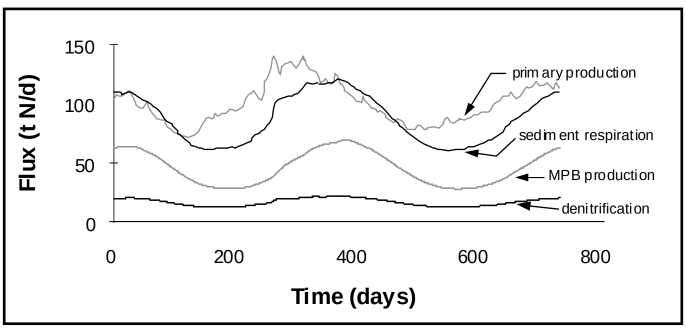

Figure 6.9b: As in Figure 6.8b, but for quadratic mortality.

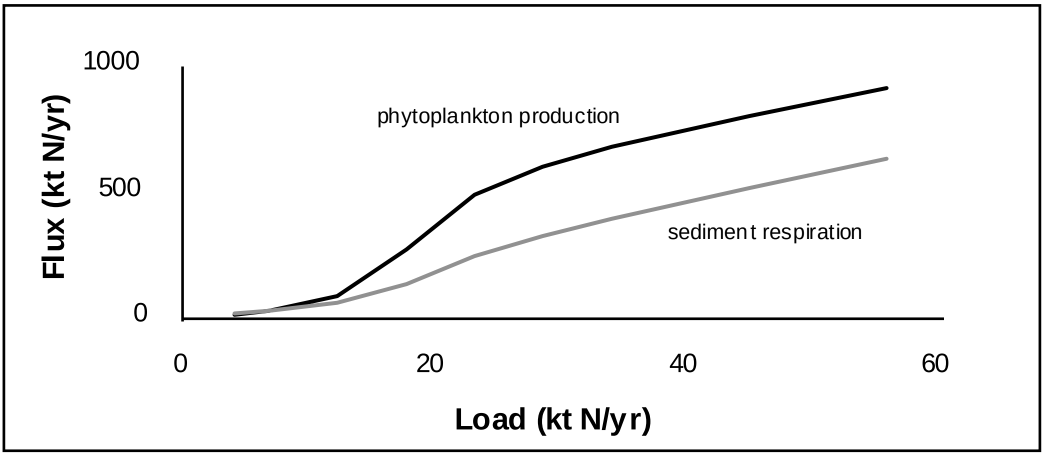

Fig 6.10a: Annual N fluxes (kt N/yr) predicted by the linear mortality model vs load.

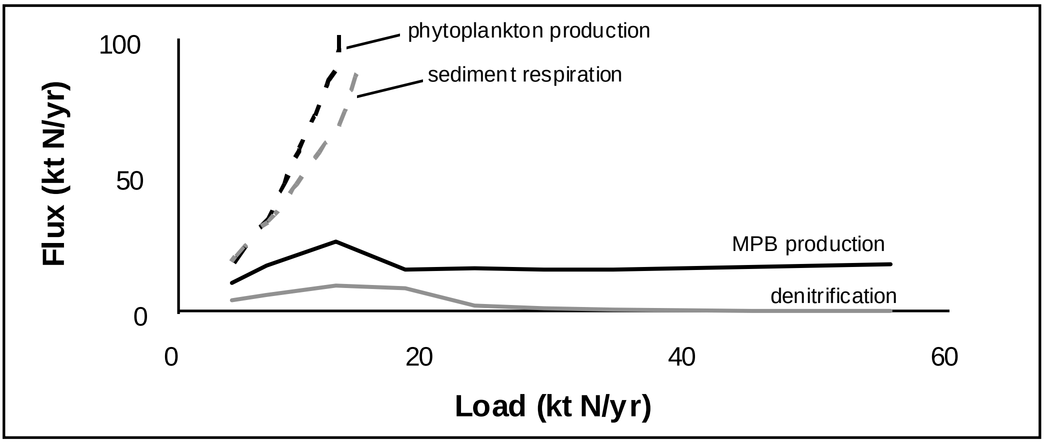

Figure 6.10b: Same as Figure 6.10a, except that the vertical scale has been expanded.

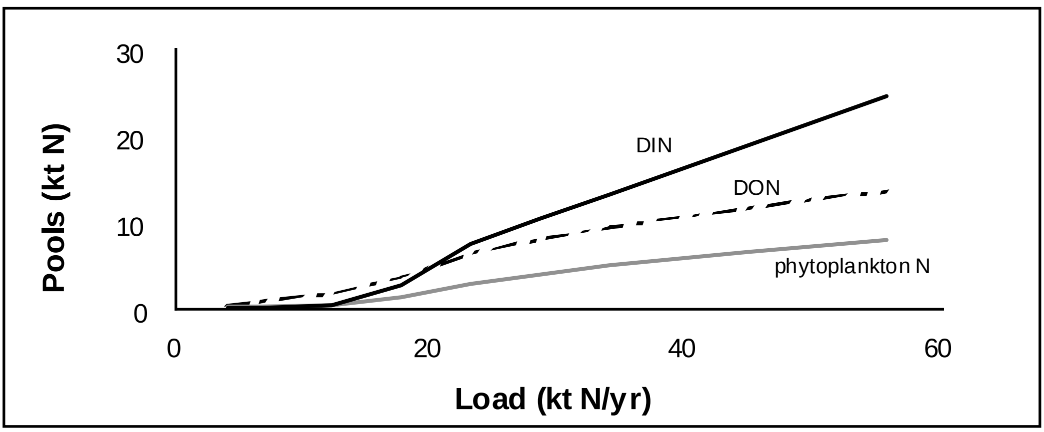

Figure 6.10c: Mean annual pools (in kt N) predicted by the linear mortality model vs load.

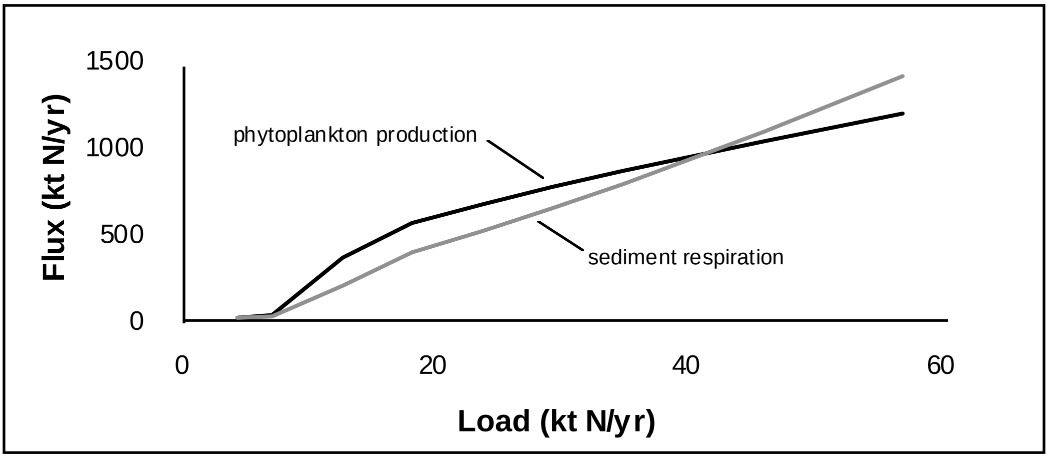

Figure 6.11a: As in Figure 6.10a, but for the quadratic mortality model.

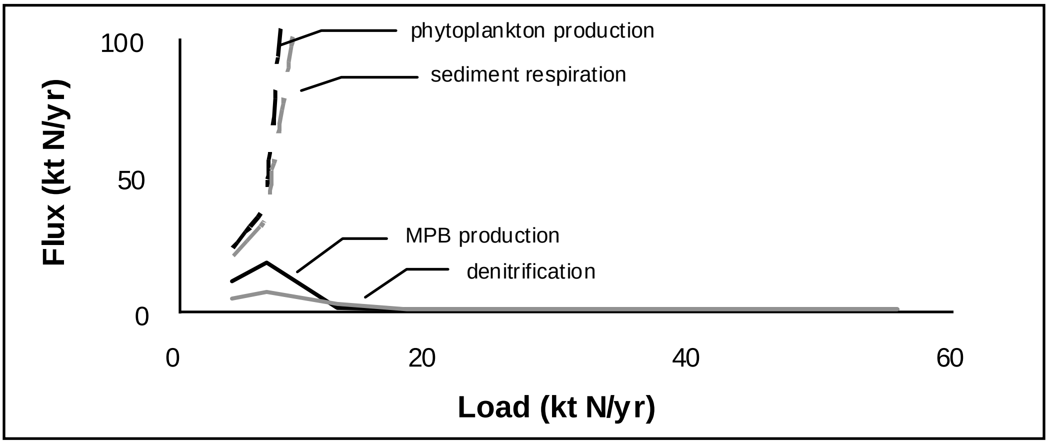

Figure 6.11b: Same as Figure 6.10b, but for the quadratic mortality model.

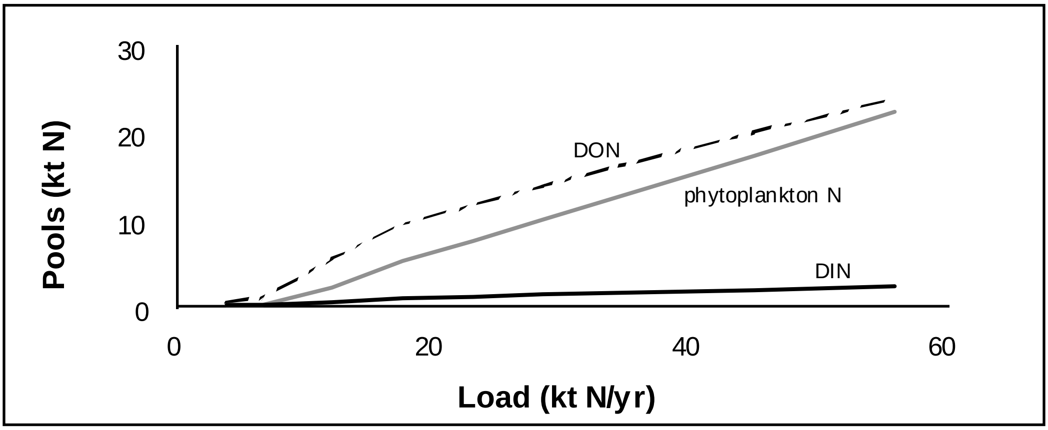

Figure 6.11c: Same as Figure 6.10c, but for the quadratic mortality model.

CHAPTER 6

For the linear mortality model, the primary production and sediment respiration accelerate as the load increases to two, three and four times current loads. The increase slows as the load is increased further. The water column pools of DIN, phytoplankton N and DON also increase rapidly over the same range of loads.

In fact, over this range, the concentration of DIN (as ammonium) increases to more than 10 µM, so that it is non-limiting for phytoplankton growth, and contributes significantly, along with DON, to the export of N from the Bay.

Denitrification peaks at loads between two and three times current loads, and then declines to very low values. High N loads to the Bay are predominantly balanced by export of DIN, DON and to a lesser extent phytoplankton N, rather than denitrification.

The qualitative analysis in Section 3 also predicted that denitrification would peak in this range of loadings, and that export of water column N would then become the dominant loss term. However, that analysis ignored the possible production and export of DON, and assumed that phytoplankton N rather than DIN would accumulate in the water column and form the predominant export medium.

Why does the linear mortality model predict that DIN rather than phytoplankton N should accumulate at high loadings? Under a three times loading, phytoplankton undergo a series of blooms which deplete DIN, and are then terminated by zooplankton blooms (Figure 6.12). The peak phytoplankton concentrations are very high, corresponding to more than 40 mg Chl *a*/m^3. However, the mean phytoplankton levels are lower than mean DIN levels because of zooplankton grazing.

The quadratic mortality model predicts an even more dramatic increase in phytoplankton production and sediment respiration over the range of two to three times loadings (Figure 6.11a, b). In fact, it predicts that denitrification should approach zero at about twice current loadings. Moreover, as loadings increase further, DIN levels increase but remain relatively low, whereas phytoplankton N increases rapidly (Figure 6.11c), and export is dominated by phytoplankton N and DON in roughly equal proportions.

The quadratic mortality term stabilizes the phytoplankton - zooplankton interaction, and eliminates the 'bloom and bust' cycles of the linear mortality model (Figure 6.13). It also imposes an upper limit on zooplankton biomass, so that zooplankton become less and

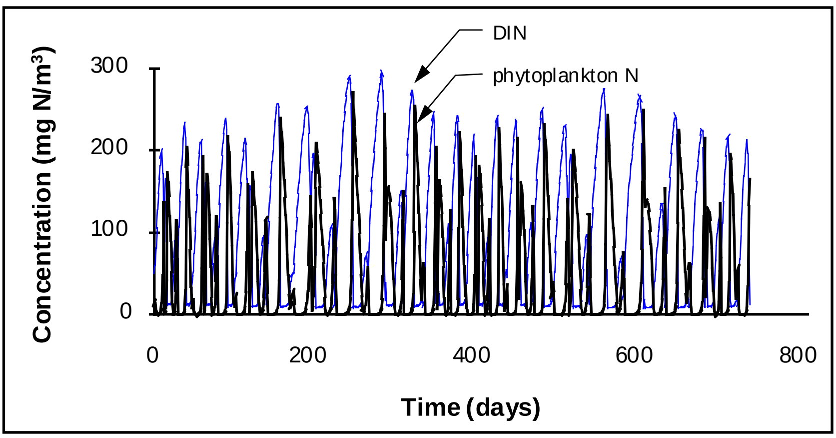

Figure 6.12: Concentrations of phytoplankton N and DIN predicted by the linear mortality model at three times current loading.

less important as N loads and production increase.

At three times loading, phytoplankton N has reached concentrations equivalent to about 30 mg Chl *a*/m^3, and shows only a small seasonal variation. The DIN pool does increase with load, albeit slowly (Figure 6.11c), and DIN concentrations at three times loading reach about 3 μM (Figure 6.13), which does not limit phytoplankton growth significantly.

At these loadings, phytoplankton growth in this model is light limited due to self-shading. The model accounts for the increase in attenuation coefficient with increased chlorophyll concentration, and at these high chlorophyll levels, mean water column light intensities become low enough to limit phytoplankton growth.

High attenuation also results in light intensities on the bottom which are low enough to stop microphytobenthic production. In fact, light attenuation has increased even at two times loading to the point where microphytobenthic production is essentially zero. This explains the rapid decrease in denitrification efficiency in Figure 6.11b, which occurs at lower loads than those suggested by the earlier qualitative analysis.

As microphytobenthic production is reduced due to light attenuation, the effect of microphytobenthos in increasing the effective denitrification rate in the sediment is lost. This positive feedback through decreased light, microphytobenthos production and denitrification, and increased ammonium recycling, accelerates the transition to a eutrophic state. It is a plausible mechanism, but its effect may be exaggerated in the one-box model, which assumes a constant depth of 14 m.

This analysis of a single-compartment version of the integrated process model has generally supported the conclusions based on qualitative arguments in earlier chapters. It suggests an assimilative capacity for the Bay between about 11,000 and 20,000 t N/yr, and suggests that light attenuation and self-shading may significantly affect both the assimilative capacity, and the response of the Bay to loadings exceeding the capacity.

The discussion in Section 3.6 about the ecological impacts of loadings exceeding the assimilative capacity still apply. The chlorophyll and DIN levels in Figure 6.12 and 6.13 are high enough for the Bay to be described as severely eutrophic. At these levels, major impacts on benthic communities, and a high risk of bottom water anoxia, would be expected.

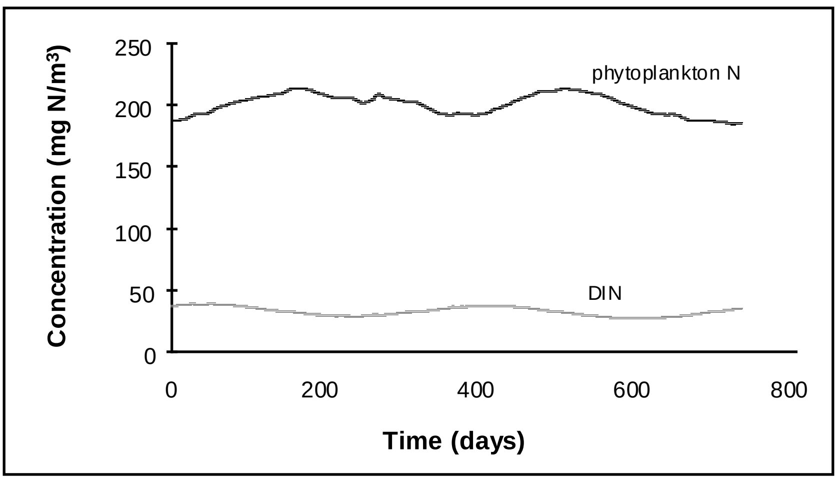

Figure 6.13: As in Figure 6.12, but for the quadratic mortality model.

4.4 Spatial variation

Throughout most of this chapter, we have implicitly or explicitly treated Port Phillip Bay as if it was a single well-mixed compartment. However, both the Phase One study and this Study have identified recurring and significant spatial gradients in many important properties within the Bay.

The high-resolution underway measurements made by Longmore *et al.* (1996) have characterised this variation more comprehensively than ever before. Analysis of this under- way data shows that while significant variation does exist at fine spatial scales (hundreds of metres), and intermediate scales (kilometres), the most important and persistent spatial contrasts are those identified in the Phase One Study, between a coastal strip about 5 km wide around the Bay edge, the bulk of the Bay centre, and the area over the Great Sands involving exchange with Bass Strait.

There are some obvious reasons for these gradients. One is proximity to inputs: most of the nutrient load to Port Phillip Bay occurs from a few important point sources, and almost all (except atmospheric deposition) occurs into the coastal strip. Exchange across the Great Sands with the waters of Bass Strait creates another set of spatial gradients.

A second obvious cause is water depth itself. This is especially important for benthic plants: seagrasses may only see sufficient light for net growth down to about 5 m, and even microphytobenthos become significantly light limited in the Bay centre. Water column depth, along with wave exposure, also controls bottom stress and resuspension, which affects both sediment composition and water column turbidity.

These spatial gradients are obviously important locally, and vital to the survival of key indicator communities such as seagrasses. However, it is not clear a priori how important they are to the trophic balance and biogeochemical cycles of the Bay as a whole. How different would we expect the behaviour of a horizontally well-mixed Bay to be from the real Bay, and how differently will a spatially-resolved model behave from the single compartment model just described?

The answer will, in general, depend on the degree of nonlinearity of the processes controlling nitrogen cycling in the Bay. As we have already seen, many of the underlying processes are non-linear, and the behaviour of the resulting water column and sediment models can be strongly non-linear. Smooth steady-state solutions can give way to limit cycles. The proportion of sediment nitrogen recycled may change strongly with throughput. A breakdown of the denitrification balance can lead to a sudden increase in DIN and in phytoplankton concentrations.

On the other hand, and somewhat counter-intuitively, inclusion of spatial variation may act to dampen out some of this non-linear or threshold behaviour. If present loads lead to local coastal blooms, then increasing nutrient loads may simply lead to a rather smooth increase in bloom intensity and extent, without any drastic shift in Bay-wide behaviour.

While the system implications of spatial variation are scientifically interesting and important for long-term management, the model must also make predictions with sufficient spatial resolution to address management concerns. Users and managers are not really interested in whether the Bay-wide pool of phytoplankton contains 200 t N, but they are concerned about whether typical bloom concentrations of phytoplankton have unacceptable local effects on water clarity, dissolved oxygen, or benthic communities.

At this stage of model development, two levels of spatial resolution have been defined. A 'fine' box structure containing 59 compartments has been defined by following depth contours at 5 m intervals, and imposing additional boundaries to resolve major inputs and gradients as necessary (Figure 2.20).

A 'coarse' box structure has been defined by aggregating fine boxes to approximate the compartments defined in the Phase One Study. Originally six coarse boxes were defined, but this was increased to eight to resolve the strong east-west gradients at the northern end of the Bay, and to resolve the strong mixing gradients across the Great Sands.

These are both quite coarse levels of spatial resolution, but at this stage of model development, we have sought to keep run times low to allow maximum experimentation and analysis.

The particle tracking approach used to link hydrodynamic and transport models was chosen because it allows almost arbitrary refinement of this spatial resolution in the future where this is judged necessary.

In fact, there is good reason to think that the 59 box resolution may be close to adequate, at least for water column processes.

As already noted, persistent spatial differences in water column concentrations among coarse boxes dominate those among fine boxes within coarse boxes.

Hydrodynamical modelling and particle tracking indicate that exchange of volume elements among neighbouring fine boxes over a one-day time step is generally extensive and at times complete. The biogeochemical model is currently advanced on a nominal one-day time step, although the integration procedure uses an adaptive time step to avoid errors when the equations become stiff. One would arguably have to use a much shorter time step to match a finer spatial resolution.

As we have argued that the coarse box structure resolves the major spatial patterns, it might appear that an eight-box model would suffice. However, given the smearing associated with any numerical transport model, it makes good sense to retain a model with one level of resolution better than one is trying to simulate.

4.5 Results from the spatially resolved 59-box model

As noted above, we have run the model with observed or calculated forcing over the 1993-1994 period. Results are presented for the last two years of a 20-year run, unless otherwise specified.

The summary Bay-wide annual budgets and average pool sizes for the model under current loadings are shown in Table 6.2.

The model predicts a level of primary production which is arguably too high, primarily because recycling of N through the sediments is too high.

The model predicts an annual efflux of 12,000 to 15,000 t DIN from the sediment, which is 1.5 to two times maximum estimated fluxes based on sediment chambers. This, in turn, depends primarily on the relative proportions of DIN denitrified, assimilated by microphytobenthos or escaping to the overlying water column.

Bay-wide, the model is predicting too little microphytobenthos uptake and denitrification as a fraction of sediment respiration. Sediment respiration is too high, by the same proportions as primary production, and for the same reasons.

The predicted Bay-wide pool sizes for DIN, Phyto-N and DON are approximately correct. Again, the model is predicting an annual export of about 1,000 t DON/yr.

In terms of Bay-wide fluxes and pools, the results of the 59-box model are not markedly different to those of the one-box model. However, this does not mean that the spatial contrasts in the 59-box model are small.

In Figure 6.14a, the phytoplankton and DIN concentrations have been plotted for four fine boxes, corresponding to the regu-

Table 6.2. Summary nitrogen budgets for the 59-box models with linear mortality and quadratic mortality, in response to estimated loadings to the Bay in 1993 and 1994.

Model	Linear Mortality		Quadratic Mortality	
Year	1993	1994	1993	1994
Flux (t N/yr)				
Catchment Load	7,700	6,000	7,700	6,000
Primary Production	41,900	40,500	45,800	43,400
Recycling (water column)	26,100	25,100	25,600	24,900
Efflux from sediments	12,400	11,400	15,500	14,100
Microphytobenthos	16,800	16,500	17,100	17,100
Sediment respiration.	35,900	34,800	38,800	37,500
Denitrification	6,300	6,200	6,200	6,100
Pool size (t N)				
DIN	304	259	252	252
Phyto-N	190	180	160	150
DON	960	990	1,050	1,090

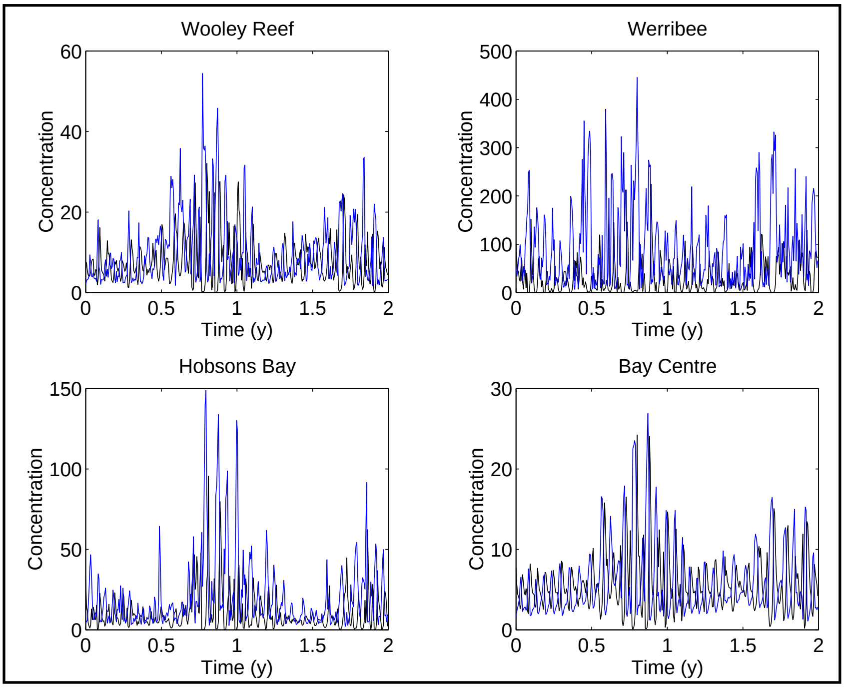

Figure 6.14a: Concentrations (mg N/m³) of phytoplankton N (black line) and DIN (blue line) vs time, predicted by the 59-box model with linear mortality for current loading over the 1993-1994 period. Note the different scales.

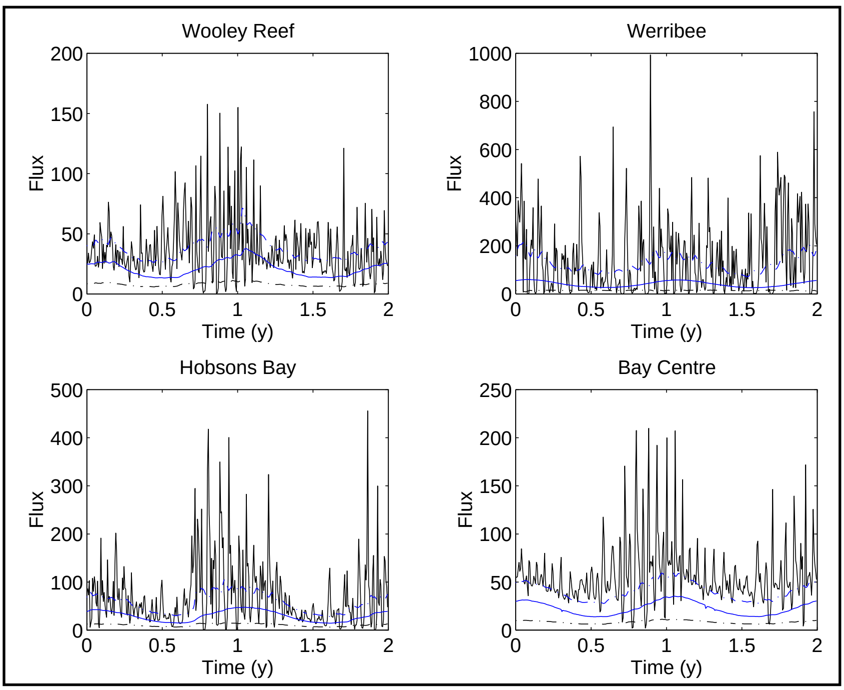

Figure 6.14b: Fluxes (mg N/m²/d) due to primary production (solid black), sediment respiration (dashed blue), microphytobenthic production (solid blue) and denitrification (dashed black), predicted by the 59-box model with linear mortality for current loading over the 1993-1994 period. Note the different scales.

CHAPTER 6

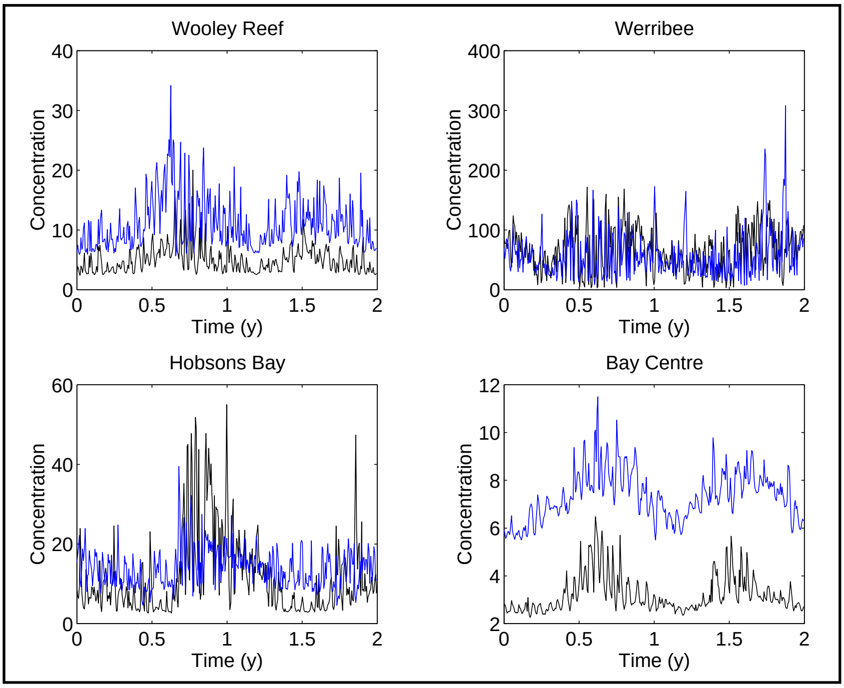

Figure 6.15a: Concentrations (mg N/m^3) of phytoplankton N (black line) and DIN (blue line) vs time, predicted by the 59-box model with quadratic mortality for current loading over the 1993-1994 period. Note the different scales.

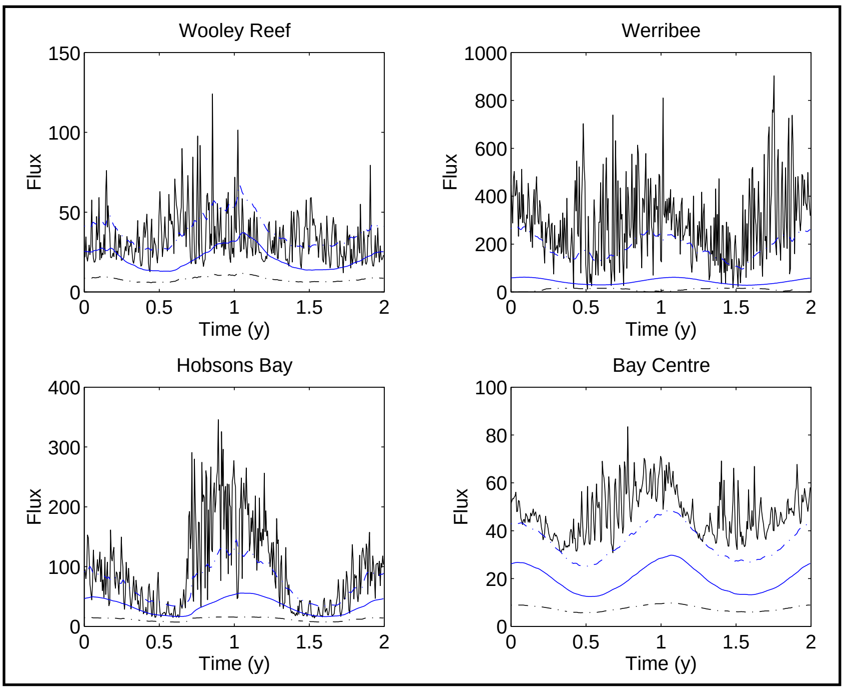

Figure 6.15b: Fluxes (mg N/m^2/d) due to primary production (solid black), sediment respiration (dashed blue), microphytobenthic production (solid blue) and denitrification (dashed black), predicted by the 59-box model with quadratic mortality for current loading over the 1993-1994 period. Note the different scales.

lar sampling sites off Werribee, at Wooley Reef, in Hobsons Bay and in the Bay centre.

Figure 6.14b shows similar plots for fluxes (primary production, sediment respiration, microphytobenthos production and denitrification). Note that the concentrations in these and subsequent figures are in mg N/m^3, and the fluxes are areal fluxes (mg N/m^2/d).

As expected, the results for the Bay Centre most resemble those predicted for the one-box model, although the amplitude of oscillations in phytoplankton and zooplankton is greater in the 59-box model, even in the Bay centre. Results for Wooley Reef are quite similar to those in the Bay Centre, with the amplitude of peaks in phytoplankton and ammonia increased slightly.

Werribee and Hobsons Bay provide the most interesting contrast.

Off Werribee, we see a build-up in DIN levels in late winter and early spring to values as high as 20 µM, associated with the winter input of ammonia. This leads to phytoplankton blooms, but generally these are of lower intensity than one might expect from the DIN levels, reaching about 70 mg N/m^3, or about 10 mg Chl *a*/m^3.

In Hobsons Bay, we also see increases in DIN and phytoplankton N in late spring and early summer, especially associated with the high Yarra runoff events in late 1993. Here, DIN values reach about 10 µM, and chlorophyll values about 10 mg Chl *a*/m^3.

The fluxes show a strong seasonal signal in primary production, sediment respiration, microphytobenthos production and to a lesser extent denitrification. The fact that sediment respiration tracks primary production quite closely in terms of both the seasonal cycle, and differences among sites, is quite consistent with the empirical correlation discovered in Task N2.2 (Nicholson *et al.* 1996, see also Figure 3.10) between measured column primary production and sediment respiration in benthic chambers.

The interannual variation in primary production and sediment respiration is especially strong in Hobsons Bay, and to a lesser extent at Wooley's Reef, but weaker in the Bay Centre and off Werribee.

The comparable results for the quadratic mortality model are shown in Figure 6.15. This model predicts rather lower phytoplankton concentrations in the Bay centre (about 3.5 mg N/m^3 or 0.5 mg Chl *a*/m^3), and higher concentrations of DIN (about 0.5 µM).

Again, Wooley Reef shows a rather similar cycle, but mean and maximum values are somewhat higher.

Off Werribee, we see a build-up of mean DIN and phytoplankton in winter, with peak values reaching about 10 µM DIN, and 100 mg Phyto-N/m^3 or 15 mg Chl *a*/m^3.

In Hobsons Bay, prolonged phytoplankton blooms at concentrations of about 5 mg Chl a/m^3 are predicted in late 1993, following the high Yarra runoff event. The seasonal and interannual patterns of fluxes for the quadratic model are quite similar to those of the linear model, although there is less variance in primary production in the quadratic model at all sites.

The rapid fluctuations in DIN and chlorophyll predicted by the 59-box quadratic mortality model (Figure 6.15) make an interesting contrast with the smooth cycles predicted by the equivalent one-box model (Figure 6.9).

There appear to be several reasons for this. One is simply the local variability caused by changes in advection and mixing of the broad-scale gradients established by inputs. A second reason is that day-to-day changes in loads from point sources have a greater local impact when they are not immediately mixed into the entire Bay.

The third reason is rather more interesting dynamically. In the local boxes off sources such as Werribee, advection is continually bringing water from elsewhere in the Bay into contact with newly input nutrients. Even if the plankton ecosystem was in some kind of local equilibrium in the source water, this will be immediately disrupted by the addition of extra nutrient. This leads to blooms, primarily of microphytoplankton, which temporarily escape grazing control. In the vicinity of inputs, one can argue that the spatially resolved model is rarely or never close to steady-state.

The mean summer and winter distributions of key water column pools predicted by the quadratic mortality model under the current loading are shown in Plate 7 (note that all variables are presented on logarithmic scales.)

In summer, the model predicts high concentrations of DIN and Phyto-N in the areas immediately adjacent to Werribee and the Yarra, with elevated concentrations along the western shore, and to a lesser extent down the eastern shore. Lower values are predicted in winter, particularly in Hobsons Bay. Lowest values are predicted in the Bay centre and the Great Sands. The predicted distributions of DON and phosphate both show highest values along the western shore and in Corio Bay, moderate values in the Bay centre, and a strong gradient across the Great Sands to Bass Strait. These variables show little seasonal variation.

Comparable plots for the major N fluxes are shown in Plate 8. Primary production and sediment respiration are both maximum off Werribee and towards Corio Bay. Elevated values occur in Hobsons Bay in summer, but not in winter. Predicted microphytobenthos production is high in the shallow regions around the Bay edge, and lower in the Bay centre, with lowest values over the Great Sands. It is significantly higher in summer than in winter.

Denitrification rates are highest along the western and eastern shores, but moderate levels are predicted to occur in the Bay centre, especially in summer. The predicted sediment respiration rates are higher in summer than in winter throughout the Bay, and are elevated

Table 6.3: Summary nitrogen budgets for the 59-box models with linear mortality and quadratic mortality, in response to doubled and trebled loadings to the Bay in 1993-1994.

Model	Linear Mortality		Quadratic Mortality	
Load	2 X	3 X	2 X	3 X
Flux (t N/yr)				
Catchment Load	12,300	17,700	12,300	17,700
Primary Production	97,200	189,800	152,600	393,200
Recycling (water column)	58,500	102,100	76,000	165,000
Efflux from sediments	36,600	80,500	68,800	216,600
Microphytobenthos	22,400	21,100	26,100	7,800
Sediment respiration.	68,300	110,300	103,000	225,900
Denitrification	9,200	8,800	8,100	1,500
Pool size (t N)				
DIN	1,200	3,500	450	680
Phyto-N	450	1,200	630	3,300
DON	1,900	3,100	2,900	6,300

along both the western shore, and the eastern shore. The very high values predicted immediately adjacent to Werribee, and in Hobsons Bay in summer, are high enough to almost shut down denitrification.

While the introduction of spatial variation has to some extent reduced the contrast between the linear mortality and quadratic mortality at current loadings, they continue to differ significantly in their response to increased loadings.

Table 6.3 presents average fluxes and pools predicted by both models at two and three times current loading. Both models predict a serious deterioration in water quality over this range of loadings.

The linear mortality model predicts an increase in the mean DIN pool to levels of 1,200 and 3,500 t, equivalent to concentrations of 3 and 10 µM ammonia, and an increase in mean phytoplankton N to 450 and 1,200 t, equivalent to 2.5 and 7 mg Chl *a*/m^3.

The quadratic mortality model predicts an increase in the mean DIN pool to 450 and 680 t N, equivalent to 1.2 and 1.9 µM ammonia, but an increase in phytoplankton N to 630 and 3300 t, equivalent to 3.5 and 18 mg Chl *a*/m^3.

As in the case of the one-box model, the quadratic model predicts a much more rapid increase in phytoplankton biomass, primary production and sediment respiration than the linear model.

The high chlorophyll levels predicted by the quadratic model lead to strong light attenuation, and much lower microphytobenthos production.

The higher sediment respiration in the quadratic model also results in very low denitrification rates at three times loadings.

Because of spatial variation in depth and bottom light levels, and in sediment respiration, microphytobenthic production and denitrification do not drop to zero, but they do become almost negligible in the Bay-wide N balance.

In the linear mortality model at trebled loadings, all sites are characterised by the kind of extreme fluctuations in DIN and phytoplankton predicted by the one-box model (Figure 6.12). The amplitude of these cycles does vary across the Bay. Amplitudes in the Bay centre and at Wooley's Reef are similar to those in the one-box model, while amplitudes in Hobsons Bay and off Werribee are two to four times higher.

The DIN and chlorophyll concentrations predicted by the quadratic mortality model at three times loading are shown in Figure 6.16a. In the Bay centre, DIN remains quite low (about 1.5 µM), but phytoplankton N is high, equivalent to about 15 mg Chl*a*/m^3 year round. Werribee shows large fluctuations in chlorophyll on short time scales, with typical values around 60 mg Chl *a*/m^3. Hobsons Bay continues to show a clear seasonal cycle in phytoplankton biomass in response to Yarra inputs, with a summer maximum equivalent to around 40 mg Chl *a*/m^3.

All four sites show a strong seasonal cycle in primary production in response to the seasonal variation in light (Figure 6.16b). Microphytobenthic production is essentially shut down due to lack of light in the Bay Centre and at Werribee. Some microphytobenthic production occurs during the low chlorophyll season in Hobsons Bay, and below the shallow water column at Wooley's Reef.

The summer distribution of key pools predicted by the quadratic model at trebled loading is shown in Plate 7. Both DIN and phytoplankton N are very high along the western and eastern shores, and no longer show concentrated local maxima around the sources. Values are now also elevated in the Bay centre, and the pattern starts to resemble that of phosphate and DON, with strong gradients across the Great Sands indicating the importance of export of phytoplankton N and DIN to Bass Strait.

The spatial patterns of DON and phosphate are qualitatively similar to those predicted for

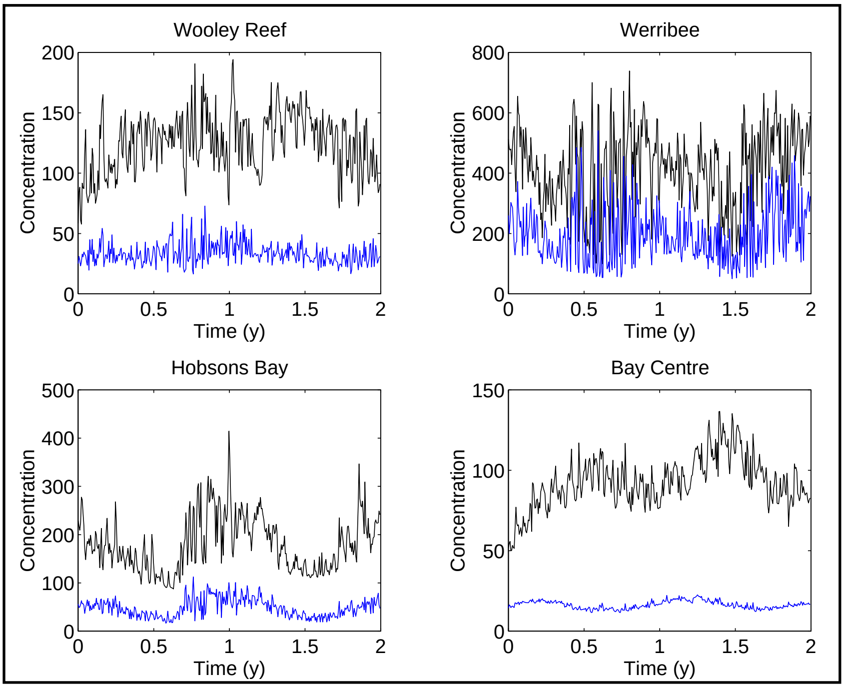

Figure 6.16a: Concentrations (mg N/m^3) of phytoplankton N (black line) and DIN (blue line) vs time, predicted by the 59-box model with quadratic mortality for trebled loading over the 1993-1994 period. Note the different scales.

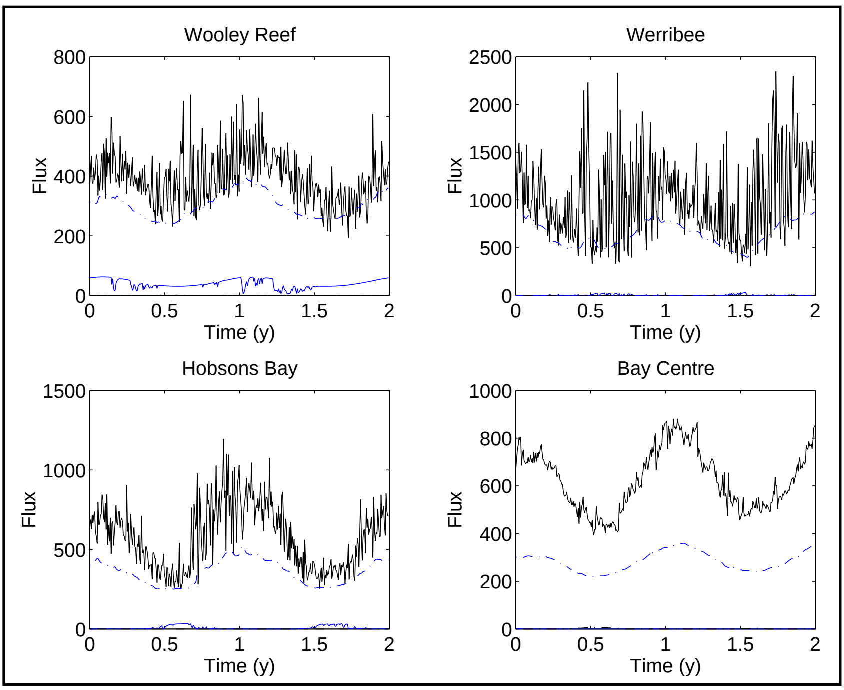

Figure 6.16b: Fluxes (mg N/m^2/d) due to primary production (solid black), sediment respiration (dashed blue), microphytobenthic production (solid blue) and denitrification (dashed black), predicted by the 59-box model with quadratic mortality for trebled loading over the 1993-1994 period. Note the different scales.

one-times loading, with values increased throughout commensurate with increased load, and in the case of DON, increased production.

The summer distribution of fluxes predicted by the quadratic mortality model at trebled loading is shown in Plate 8. Primary production is elevated throughout the Bay, and shows relatively little spatial variation. Microphytobenthic production is shut down completely below the 10 m contour. Sediment respiration rates are very high throughout the Bay, with local maxima still predicted off Werribee and in Hobsons Bay. Denitrification is shut down everywhere except in the Great Sands.

At the chlorophyll levels predicted by either model under three-times loading, the Bay would generally be regarded as highly eutrophic.

The high sediment respiration rates might be expected to lead to depletion of oxygen in bottom water under calm stratified conditions. The high light attenuation would be expected to inhibit macrophyte and seagrass growth, except in very shallow water.

5. Conclusions

In this chapter, we have followed a path from a nitrogen budget for Port Phillip Bay based on observations, through simple empirical models based on recycling efficiencies (analogous to Vollenweider models, see Chapter 7), to simple conceptual process models, to a one-box integrated process model with highly simplified physics, to the spatially-resolved integrated model just discussed.

This approach may have seemed unduly prolonged: after all, we could have simply given a brief description of the integrated model, and plunged into an analysis of its predictions.

We believe that the approach adopted here is essential if modelling is to serve its dual objectives: to increase scientific understanding and synthesis, and to provide managers with both predictions and some feeling for the uncertainty in those predictions.

The full integrated, spatially-resolved process model is a relatively complicated non-linear system, with a large number of state variables, and an even larger number of parameters. The fact that the results from this model are broadly consistent with the predictions of the one-box model, which are in turn directly linked to qualitative predictions based on simple cycling and process models, gives us considerable assurance that the full model is behaving 'reasonably'.

At the same time, it should be stressed again that analysis and calibration of the full model is still in a preliminary stage: most of the parameters are still assigned either default litterature values, or estimates based on preliminary analyses of field data. Indeed, estimates of some key fluxes in the Bay-wide nitrogen budget, which have significantly affected our understanding of the nitrogen cycle, have only become available in the last month,.

The analyses of model responses to increased loadings presented here are intended to convey, in a semi-quantitative way, the implications of our current understanding of the Bay for changes in N loading.

The specific numerical values should be treated with caution. As model analysis, and the comparison of model predictions with observations, proceed, we expect to be able to make more quantitative assessments of prediction errors and uncertainties. Workshops involving local management agencies will then be held to allow an interactive assessment of model predictions for prescribed management scenarios.

What are the key lessons from the model development and analysis to date? As stressed repeatedly in this report, the Study's principal contribution has come from a proper recognition of the role of the sediments in the Bay.

The measurement of high denitrification rates in Bay sediments filled an embarrassing gap in the Bay's N budget.

The measurement of high microphytobenthic production in Bay sediments is not only interesting in terms of benthic ecosystems: the simple recycling arguments presented in Section 2 suggest that microphytobenthos may double the effective denitrification efficiency in the sediments, by taking up inorganic nitrogen from porewater and cycling it back through the denitrification loop.

However, the role of the plankton in the Bay nitrogen cycle should not be overlooked.

The Study found that water column primary production in the Bay is significantly higher than earlier (sparse) data suggested, and that the relatively low phytoplankton biomass in the Bay is currently turning over very rapidly.

The high primary production is supported by a moderate recycling efficiency in the water column, and a moderate recycling efficiency in sediments.

The high phytoplankton turnover implies a high mortality rate, which, at least for the small flagellates, is most likely due to microzooplankton grazing.

The relative contributions of mesozooplankton grazing, filter feeder grazing, and sinking, to the mortality of large phytoplankon (predominantly diatoms) is still unclear.

We have paid considerable attention in the preceding sections to the problem of trophic closure, and the effects of alternative formulations of zooplankton mortality.

A particularly interesting result is that the striking differences between these formulations in simple one-box models close to equilibrium are substantially diminished in the 59-box model when more realistic spatial and temporal variation in forcing establishes local disequilibrium.

However, the two models differ substantially in their predicted responses to increased loadings, at least for current parameter sets. The extreme, regular oscillations predicted by the linear mortality model under high loadings do seem rather 'pathological' and the behaviour of the quadratic model at high loadings more realistic.

The Bay's response to increased loadings is dominated by positive feedbacks involving denitrification. Both sediment process models and empirical evidence from chambers suggest that denitrification efficiencies should eventually diminish as sediment loads increase. This feedback essentially fixes a maximum denitrification capacity for the Bay.

Once loads exceed this capacity, nitrogen must accumulate in the Bay until loss through flushing to Bass Strait matches inputs. This can only occur at extremely high water column concentrations of PON, DON or DIN. By any account, the Bay would be considered eutrophic once this occurs.

The data and model results suggest that this capacity is probably somewhere in the range from two to three times current loadings.

Aside from the denitrification process itself, several other feedbacks potentially affect this capacity.

If the microphytobenthos do increase denitrification efficiency in the way the observations suggest, then reductions in their production due to shading by phytoplankton can act as a significant positive feedback, accelerating the slide into eutrophication. Results from the quadratic mortality model suggest this feedback may be important.

The other potential feedback is depletion of oxygen in bottom waters. The model will have the capability to address oxygen depletion, although this has not yet been fully implemented.

Some of the key uncertainties in both the Study outcomes and the model predictions are associated with the composition and fate of organic detritus and DON. We noted earlier the large pool of organic N of uncertain age in the sediments. The versions of the model presented here do not include a very refractory detrital N pool, and therefore do not bury large amounts of organic N in sediments.

If the organic N in the sediments is truly ancient, then the error involved in ignoring its accumulation is relatively trivial, at least for current loadings. On the other hand, if a substantial fraction is recent, then the model should be expanded to include a more refractory N pool. This would have important implications for the Bay's transient response to changes in loadings.

These alternative assumptions will be explored carefully in future model development.

We have focused in this Chapter on the response of the N-cycle to changes in loading, with only brief references to the likely ecological consequences of these changes. We do plan to examine the impacts on important ecosystem attributes such as seagrasses and macrophytes, benthic fauna, and toxic algal blooms, in more detail.

However, the predicted behaviour of the N-cycle in some ways simplifies the analysis of ecological impacts. The models developed here all predict that the Bay is likely to make a sudden transition from the current mesotrophic state to a highly eutrophic state once the assimilative capacity is exceeded.

As discussed in earlier sections, the ecological consequences of this transition would be severe, and almost certainly unacceptable to the Bay managers and the public.

The management implications are discussed further in Chapter 7.

Port Phillip Bay, like all natural ecosystems, contains many levels of nested complexity, and any mathematical or conceptual model is inevitably a caricature of the real system. We believe that the models presented here have captured some of the key processes and feedbacks in the Bay, but there will inevitably be gaps in our understanding, and surprises in the future behaviour of the Bay.

It is important that models of the kind developed here are not seen as a finished product, but as a conceptual framework which can grow along with future knowledge and understanding. We have already had discussions with management agencies to ensure that this happens.

Both nationally and internationally, natural resource researchers and managers are trying to move towards integrated, adaptive or experimental management of natural ecosystems for multiple use. This means integrating monitoring and research design, data analysis and assimilation, model development and management strategy evaluation in an ongoing adaptive process. At present, this is more an attractive concept than a reality. Making it happen in Port Phillip Bay poses an exciting challenge for both researchers and managers.

Contents

1 Highlights and achievements

1.1 Field work and monitoring

A number of scientific advances have been made during the execution of this Study, thanks largely to the high level of marine science expertise in Victoria. The details of these advances may be found in the individual chapters of this report, but it is desirable to mention a few highlights here.

Through new field work and modern data handling we have brought together a consolidated databank of historical and current information.

The master data archive is an unprecedented source of information about the past and present state of the Bay. These data are the basis of the statistical and integrated models of the Bay which allow us to make predictions about future states and the results of possible management actions. The database should be maintained and updated.

The toxicology work has shown that there has been a long-term decline in the levels of most toxicants in the Bay.

Toxicants originating from freshwater inputs drop out rapidly on entering the seawater and so are restricted largely to the mouths of creeks and drains. There is, therefore, not a widespread toxicant problem in the Bay, but areas of human contact along the edge require attention.

We have pioneered the combined use of under way monitoring of water quality and GIS databases to provide high resolution maps of water quality (Longmore *et al.* 1996 and Chapter 3). It has been possible to rapidly visualise and study the spatial and temporal scales of variability in the patterns of inflows, water quality and phytoplankton biomass.

The GIS database has assisted the modelling activity by allowing detailed analysis of the most important scales of temporal and spatial variability – a prerequisite for an adequate description of pattern and process and of the pools of major elements.

An accompanying product of the Study will be a CD-ROM disk with a number of graphical and map-based outputs from the master database and the model.

We have made great progress in understanding the basic features of the nitrogen cycle in the Bay, a previous critical gap in our knowledge of the Bay ecosystem. We have described the major pathways of nitrogen (N) in the Bay and have been able to quantify the major pools and fluxes. We have shown that because of the rapid turnover of N in the planktonic and benthic pools, flushing is not an important process. What nitrogen is put into the Bay stays in the Bay and is assimilated there.

We have shown that the sediments are the primary determinant of the water quality in the Bay. The sediments are the least-known but most important component of the ecosystem. We have carried out a large number of benthic chamber deployments at various times during the year and have used these chambers to study the fluxes of elements between the sediments and the water column.

This research dramatically altered our view of the functioning of the Bay and caused us to do "mid-course corrections" to a number of tasks.

It allowed us to close the N budget for the Bay and brought to light the key role of the benthos in determining the water quality.

The PPBES has been a significant stimulus to the Victorian marine science community and has significantly lifted local capabilities and expertise. The combined expertise of

MAFRI (formerly VFRI) and AGSO in nutrient chemistry is probably the best available in Australia.

We have been able to consolidate a considerable amount of data on the benthic macroinvertebrate communities of the Bay and use these to examine changes over time.

We were very fortunate because extensive data existed from the Phase One study. We had an excellent baseline from which to compare long-term changes.

We have shown that benthic biodiversity is critical to the long-term health of the Bay and that little known unicellular plant organisms like the microphytobenthos (MPB) are a very important group.

The vitally important role of biodiversity in maintaining the health of the Bay places great emphasis on the conservation and protection of the Bay and its resources as well as the management of nutrient and toxicant loads.

Some existing exotic (introduced) species pose a significant risk to the sustainable management of the Bay, so we have flagged the risks associated with the present exotics as well highlighting the dangers of allowing further introductions of exotic organisms into the Bay.

1.2 The PPBES model

Most previous studies in the Bay have been largely descriptive and have resulted in much monitoring data but little process understanding.

A process model, which quantitatively describes the major fluxes of energy and materials, provides a means of turning data into understanding, of explaining why the Bay is the way it is and of predicting the outcomes of management options.

Models also provide a means of interpolating between data which are often sparse in space and time and a way to calculate pools and fluxes. Models can provide a guide to the most sensitive parts of the system, to critical data sets and missing information.

Calibrated and validated models of ecosystem function also provide a powerful means of project design and management in that they synthesise and integrate data from disparate sources and provide graphical outputs and maps.

We have deliberately chosen a dynamic modelling approach. There are a number of reasons for this, not the least of which is the fact that the Bay is subject to much climate variability and extreme events. The model must be able to simulate the effects of these events on the Bay's long-term health.

Empirical models link such things as ecosystem processes and nutrient pools but require steady state assumptions and time averaged properties. The assumption of steady state is largely violated by the Australian environment.

Perhaps the best-known examples of these models are the mass balance and empirical modelling approaches of Vollenweider (1968, 1975, 1976), who modelled the response of lakes to phosphorus enrichment.

Vollenweider's models have some significant limiting assumptions and, like all empirical models, should not be applied under conditions which violate these assumptions (see Harris 1994b). They should not, for example, be applied to N-limited coastal waters (although this is frequently done).

Hybrid mass balance/empirical models (like Vollenweider's) are, in fact, a subset of a broader class of dynamic model (Thomann 1977).

However, in this Study we have used *some* empirical relationships to help us with the estimation of the critical N load to the Bay.

We have used a measured relationship between benthic fluxes and denitrification efficiencies to estimate the point at which anoxia and zero denitrification efficiency might occur.

We recognise that we are extrapolating from a set of measurements taken over a two-year period at various points in the Bay to a longer term set of conclusions about the behaviour of the entire Bay over time. Nevertheless, a combination of dynamic modelling and empirical response functions can provide robust insights into system behaviour.

The PPBES model requires some sophisticated dynamic behaviour to model the temporal response of the Bay ecosystem to strongly pulsed flows.

Time scales vary from hours to years as the storm loads enter the Bay, are transported by 3-D physical processes, stimulate phytoplankton growth, settle to the bottom and are reworked by macrobenthic organisms and microbiological processes.

The growth rates of the relevant organisms also vary from hours to years. While the bacterial components might be expected to respond quickly to changes in load, the microbial processes associated with the structure of the sediment and the activities of the macrobenthos (denitrification) will respond as the biomass of macrobenthos changes. This is presumed to take months to years.

The problem with any dynamic ecosystem model is that some aggregation of pattern and process is essential. It is impossible to model all the complexities of physiologies, population interactions, age structures and processes at the full range of spatial and temporal scales. We are therefore required to approximate. The important question is: can we still make useful predictions with acceptable risk?

Studies have now been done on the effects of aggregation error and some rules for model design have been produced (Turner and O'Neill 1995). Costanza and Sklar (1985) analysed 87 mathematical models of freshwater systems and showed that complex models were less accurate (they said little about much) than simple models which tended to be more accurate (they said much about little). The preferred model structure was a compromise between the two.

The dynamic modelling approach explicitly assumes that it is possible to aggregate the complexities of system function in a way which gives not only a concise description of the most important aspects of the system, but also allows reasonably precise predictions of future states to be made.

Despite the acknowledged complexities of ecosystem dynamics (and the possibility of chaotic properties), aggregation of pools and fluxes to the level of biomass and fluxes of materials does, it seems, allow predictions to be made with some confidence at this level (Harris 1994b, 1996).

The PPBES approach is to estimate system function at this level, to model the Bay and to simulate ecosystem dynamics, while at the same time checking both measurements and models against empirical data sets from as many other coastal ecosystems as possible.

A similar approach has recently been used by others (Asknes *et al.* 1995, Gurney *et al.* 1995).

This approach assumes that, to an acceptable extent, differences in biogeography, species identities and community structures do not invalidate such comparisons. The problem of scaling from species to ecosystems is not a trivial one (Lawton and Jones 1995) and the possibility of a changed ecosystem state influencing species composition directly in unpredictable ways (and vice versa) is acknowledged.

The chosen modelling framework is, however, capable of representing these effects if they can be estimated. This problem is given more point in the Bay by the presence of introduced, exotic species. In particular, the Bay now has a number of introduced species (e.g. *Sabella*) which, in abundance, may well have a significant impact on system function.

The state-of-the-art in ecology is such that perfect predictions of the impacts of management actions cannot be given with zero risk. Risk assessment is a part of any management scenario and the PPBES model is but one part of the information managers require.

We do believe, however, that the model is producing realistic representations of system function and that the response of the model to altered loads is both as accurate and precise as can presently be achieved. Similar approaches have recently been used in Denmark (Malmgren-Hansen and Warren 1992), Holland (Nauta *et al.* 1992), Scotland (Gurney *et al.* 1995) and the North Sea (Baretta-Bekker 1995).

Rather than produce an over-parameterised model which probably can never be validated, the Study has chosen to produce a more parsimonious model in which careful attention has been paid to estimating the required terms (Gurney *et al.* 1995).

There has been a complete marriage of the field work and the modelling throughout the Study. In addition, estimates have been made of the risks involved in not knowing key parameters with sufficient accuracy, so that the overall reliability of the model output is known in some detail.

2. The state of the Bay

We are now in a position to distil the results presented in the various Technical Reports and in the other chapters of this Final Report. What, therefore, has the Study found and what can we say about the present state of the Bay?

We, the Technical Group, have concluded that toxicants are not at present a threat to the Bay. We have discovered a set of important and previously unsuspected processes in the sediments by which the Bay assimilates the nutrient load and we have shown that the present nutrient load to the Bay, while having measurable effects, is not immediately dangerous. We have been able to define the critical nutrient load, beyond which irreversible damage to the Bay would occur.

Not foreseen in the Study Design was the discovery that the introduction of a number of exotic or alien species of organisms into the Bay has already had an observable effect on its ecology. We have concluded that maintenance of the benthic habitat is essential.

These conclusions have major implications for the future management of the Bay. Certainly emphasis must be placed on the future management and control of nutrient and toxicant loads. In addition it is vital that the Bay ecosystem, its habitats and resources be conserved and protected if the Bay is to be managed sustainably for future generations.

While the Study placed considerable emphasis on nutrient and toxicant loads, we have concluded that conservation of biodiversity in the Bay is equally important for sustainable management.

The PPBES was similar in its aims and objectives to a number of other coastal ecology studies around the world. Comparison of the concentrations of nutrients and chlorophyll in the waters of the Bay with other estuaries and embayments near major cities (Boynton *et al.* 1982) shows that water quality is good and that there is a relative dearth of inorganic N (about 0.5 µM) in the water of the Bay. Nitrogen to phosphorus ratios are very low by world standards.

Comparison of the Bay with other micro- and macro-tidal estuaries (Monbet 1992) shows relatively low levels of both N and chlorophyll in the Bay. The Bay rates as oligotrophic to mesotrophic compared with the more highly affected temperate estuaries in Europe and North America. This reinforces the view that water quality in the Bay is good by world standards. It was a surprise to discover that it has been maintained that way by benthic processes.

Port Phillip Bay differs from well-studied systems like Chesapeake Bay in being strongly N-limited at all times rather than intermittently N and P limited (D'Elia *et al.* 1992) and in the fact that surficial sediment oxygen concentrations never fall to zero (Kemp *et al.* 1992).

The internal load resulting from ammonia fluxes is lower (Kemp and Boynton 1984) and denitrification rates are higher (Harding *et al.* 1992).

Overall, Port Phillip Bay rates as much less eutrophic than Chesapeake Bay (Hakanson 1994) and the management challenge is to keep it that way.

Comparisons with data from the literature (e.g. Boynton *et al.* 1982, Seitzinger 1988, Monbet 1992) shows that Port Phillip Bay has low nutrient loadings, high denitrification rates, low N:P ratios in the water column, low phytoplankton biomass (as chlorophyll *a*), high benthic fauna abundance and biodiversity, few algal bloom problems and a healthy population of marine mammals (dolphins and seals).

The Bay is a significant conservation area. Nutrient loadings must be maintained at levels well below the critical level to ensure the continued 'health' of the Bay ecosystem. Despite being largely ignored in the past, particular attention must be paid to sediment processes and the health of the benthic communities to ensure the continued sustainable use of the Bay.

Light penetration and water clarity in the Bay are still good, algal biomass is low except in one or two places around the margins, seagrasses are still present, and the benthic community is still dominated by a rich diversity of deposit feeders.

Anoxia on the sediment surface does not occur – in fact the lowest recorded oxygen concentration in bottom waters was more than 60% saturation (Mickelson 1990, Longmore *et al.* 1996).

We have not seen a marked increase in polychaete populations characteristic of eutrophication.

It is true that we have seen recent increases in the populations of exotic suspension feeders (*Sabella*) and there are beds of red drift algae around the Bay. These are signs of the early stages of nutrient enrichment. However, anecdotal evidence (and some documented) suggests that drift algae have always been present.

The *Sabella* invasion is characteristic of exotic species exploiting a new environment rather than a response to eutrophication. We have certainly not yet reached the critical nutrient load and we can see no evidence of long term deterioration in benthic communities.

The biggest threat to the Bay at present may be the introduction of exotic species rather than the nutrient load.

Species like *Sabella* can change the way the entire Bay functions and the introduction of organisms such as the Pacific Sea Star could seriously affect the populations of valued molluscs.

We have found that, at present, toxicants are not a threat to the health of the Bay or to those using the Bay for recreation or fishing.

Overall, toxicant concentrations rarely exceed guideline levels and some are decreasing with time (see Chapter 4).

This is not to say that toxicant loads should not be controlled (as now) and eliminated wherever possible, but measurements of toxicant concentrations (both organic and inorganic) have been made in the water, sediments and biota and, with the exception of a few samples from the mouths of creeks and drains, all are at or below guideline levels.

There is no indication of significant biomagnification in valued species such as molluscs and fish. Few lesioned organisms have ever been found and, while there is an historical cadmium problem in Corio Bay, toxicant levels appear to be declining as a result of stricter effluent controls.

Studies of low-level, chronic effects of toxicants have not been done, but any assessment based on the present information would not indicate an acute problem.

3. Processes in the Bay – water quality, the benthos and denitrification

We now know that a primary concern for the health of the Bay should be nutrient loadings. The present loadings to the Bay, while having an influence on water quality and the sustainable use of the Bay ecosystem, are not threatening.

The management problems of the Bay therefore revolve around potential eutrophication and the impact of future nutrient loadings on the Bay ecosystem. Is this a cause for concern and is the present nutrient load sustainable into the future?

There are a number of ways of addressing this question. Above all, it must be remembered that the residence time of water in the Bay is sufficiently long (about one year, Chapter 2) that all the important interactions between loads and the ecological response occur in the Bay – flushing is not an important process.

As the Study progressed, more and more attention became focused on the benthic macrofauna and microflora as major determinants of the water quality in the system.

Over the years, much work had been carried out on the water quality of the Bay but little attention had been given to the important role of the benthic communities of bacteria, plants and animals. This feature of the Bay's ecosystem was not well studied in Phases One and Two because those studies were essentially descriptive and did not attempt to provide functional descriptions of key processes.

Newell's (1990) work had identified an anomaly in the nitrogen budget for the Bay. Our understanding of the anomaly was dramatically improved when work in PPBES using benthic chambers showed that much of the benthic N flux resulting from decomposition of organic matter reaching the sediments could not be accounted for. The most likely explanation was denitrification.

Studies in Phase Two (Bauld and Millis 1977) had demonstrated nitrification and denitrification in the sediments off-shore from Werribee, but activity decreased sharply 200 m off-shore so that these processes were thought to be limited to the area affected by the suspended organic load from the WTP.

Later work by Gorfine *et al.* (1996) and Longmore *et al.* (1994) demonstrated nutrient fluxes across the sediment-water interface throughout the Bay, but the presence of denitrification could not have been detected with the techniques they used.

PPBES work has now demonstrated denitrification directly and indirectly. Denitrification is a significant process which strips N from the system. Effective losses of 70-90% of N_2 input place the Bay at the high end of the range of such observations world-wide (Seitzinger 1988) and provide an explanation of the low N:P ratio in the Bay water.

Estimates of the biomass of benthic macrofauna showed that there are abundant macrofauna in the Bay and the biomass (25 g dw/m^2) compares very well with the highest levels observed in coastal sediments around the world (Nixon 1988).

Studies of nutrient, toxicant and isotope profiles in sediment cores further showed active and deep bioturbation by invertebrates to depths of 30-50 cm.

These organisms mix the sediments and enhance decomposition and, by providing an irrigated heterogenous structure with adjacent oxic and anoxic microzones, enhance denitrification (Aller 1988, Hendriksen and Kemp 1988, Kristensen 1988, Pelegri *et al.* 1994).

We now also realise (Chapter 6) that the microphytobenthos (MPB) are playing an important role in the N cycle of the Bay by taking up and recycling N released from the sediments by decomposition. This has the effect of making it appear as though denitrification rates are higher than they really are. Actual denitrification efficiencies in the Bay are probably of the order of 40-60% – on a par with other sites around the world (Seitzinger 1988).

Because they live on the bottom of the Bay, MPB depend on clear water and excellent light penetration and thrive in shallow, oligotrophic to mesotrophic embayments.

Diatoms are the most abundant type of phytoplankton in the water column of the Bay (Magro *et al.* 1996) (see Plate 4) and most of these eventually sink to the bottom and are decomposed in and on the sediments. There they provide food for deposit feeders and play an important role in the recycling of N.

The Study has also discovered an important silica cycle whereby Si taken up by the diatoms in the water column is regenerated at the sediment surface and fluxed back into the water (Chapter 3).

The concentrations of all the major elements (C, N, P, O, Si) in the water column are controlled by sediment exchanges as well as inflows, outflows and advection within the Bay. This is why a dynamic model of the Bay is required (Chapter 6).

It is not possible to infer what is going on in the Bay simply by observing what happens to the concentrations of nutrients and chlorophyll in the water. Fluxes between pools of elements in the water column and in the sediments are very large and nutrients may turn over many times. For example the N pool in the phytoplankton turns over about 150 times a year. In order to manage the system effectively it is necessary to understand the dynamics of the Bay ecosystem and its response to changing climate and nutrient loads.

The constant recycling of nutrients through the sediment/water interface and the combined effects of bacteria, MPB and macroinvertebrates result in the water quality that we currently experience and enjoy. **It is the hidden sedimentary components of the Bay ecosystem which must be protected if the water quality of the Bay is to be maintained**.

4. Other ecosystem components

In addition to the invertebrates burrowing in the sediments, seagrasses and other benthic plants are important components of the Bay ecosystem. Seagrasses provide important nursery habitat for fish and prawns and also prevent erosion of sand.

Anecdotal evidence claims loss of seagrasses in the Bay during the 1960s and 1970s, but the studies of Bulthuis (1981) showed mainly fluctuations about a mean. Regeneration has occurred in some areas.

Benthic communities have changed since the Phase One studies, but the biggest effects have been the introduction of exotic or alien species (Chapter 5). There are now a number of exotic species of clams (*Corbula*), crabs (*Pyromaia*) and polychaete worms (*Sabella*) which are abundant in the Bay (Wilson *et al.* 1996). Besides the introduction of new benthic invertebrate species there is little evidence for deleterious long-term changes in these communities. Whatever effects are present they are not a result of changes in water quality.

While future introductions could be controlled by improved handling and port regulations, those present now do not lend themselves to simple management solutions. A close watch needs to be kept on the spread of these exotic organisms because some have the potential to adversely affect the present ecological balance (Walters 1996).

The recent introduction of the fanworm, *Sabella spallanzanii*, has the capacity to change the basic functioning of the Bay ecosystem. This tube-building polychaete worm anchors itself to any firm substrate and grows vertically to 30 cm or more. By intercepting suspended food particles, this organism can impede the flux of food and oxygen to the rest of the benthic community. It may also crowd out or smother other benthic macrofauna. It is not known to have any local predators. At present, *Sabella* is largely restricted to Corio Bay and the Bay's western and northern parts.

The PPBES model predicts a distribution of suspension feeders rather like the present distribution of *Sabella* so it is to be hoped that the species will not spread throughout the Bay. Nevertheless, the situation needs to be closely monitored.

Fish populations have oscillated over the years but, while the biomass has not changed substantially, there have been some changes in the species composition since 1969 (Hobday *et al.* 1996). Any conclusions drawn about changes in fish populations must be tempered by the uncertainty associated with the nature of the data. Variations in sampling methods and small sample sizes mean there is considerable uncertainty about these conclusions. What changes there are can probably be ascribed to recreational and commercial fishing pressure, habitat alterations, natural population variations and the introduction of exotic species rather than to water quality changes.

Fish species caught on hooks have declined and those associated with seagrass beds showed marked declines in the early 1970s. Some species, particularly those associated with *Sabella* beds, have increased in abundance.

Large changes in barracouta (snoek) populations have been noted in recent years but we have just seen two of the best years for recruitment of snapper in a long while.

As in other parts of the world (Cushing 1982) fish populations in south-east Australia undergo long-term (10-25 yr) fluctuations driven by climate and El Nino Southern Oscillation (ENSO) events (Harris *et al.* 1988, Thresher 1994). Scallops, in particular, are notorious for large, irregular changes in population size associated with climate variability (Coleman *et al.* 1996).

For many species in the Bay, the 'signal' of eutrophication is not visible beneath the natural 'noise' induced by climate variability, habitat loss and natural variations in recruitment.

5. Is the Bay stable at present?

First, we can examine historical trends in water quality data. This might give a clue as to whether or not the water quality of the Bay is declining with time. Water quality data (in the form of nutrient concentrations) has been collected since the first CSIRO studies in 1947.

We might have expected both P and N levels in the Bay water to increase with time as the catchment was progressively cleared, as the City of Melbourne grew and as the sewage loadings increased.

P levels in the water have risen but nitrate levels have remained relatively constant over time (Figure 7.1). The N:P ratio in the water is very low and N is the key limiting element for biological processes in the Bay. There is no indication of any increase in N or chlorophyll concentrations in the water over the last five years. Total inorganic N levels in the Bay water are so low that Bass Strait is, if anything, a small source of NO_3 from the incoming tide even if DON is exported.

Unfortunately we only have records of algal biomass in the Bay (as chlorophyll *a* concentrations) from 1970 onwards. The dominant feature of these records is year to year variation without any consistent trend over the whole Bay. This statement is supported by Brown (1989), Saunders and Goudey (1990) and Longmore (1992).

The marked interannual variability in the data points to natural climate forcing rather than anthropogenic influences.

The decrease in algal biomass in 1994 (Longmore *et al.* 1996) should be noted; this was coincident with the drought and the decrease in nutrient loads from the Yarra River and from the eastern creeks and drains. Magro *et al.* (1996) noted a similar decrease

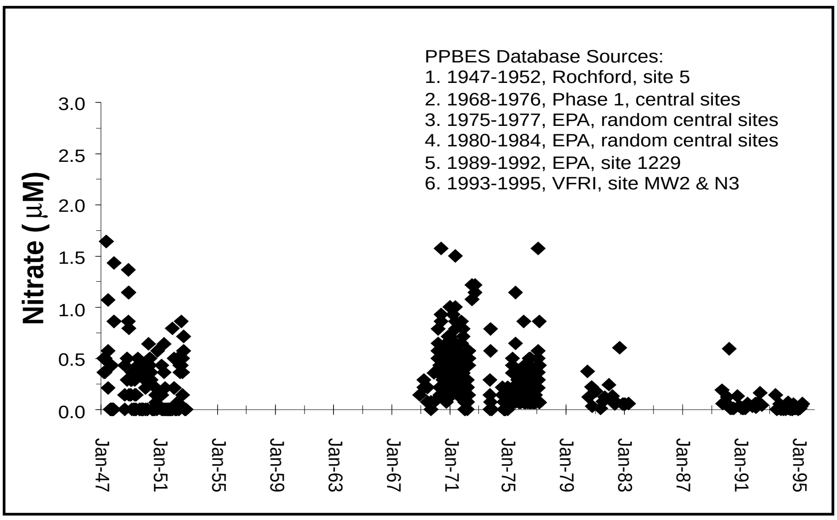

Figure 7.1: Nitrate concentrations at sites within the central area of the Bay.

CHAPTER 7

in the frequency and magnitude of algal occurrence around the Bay during this period.

Interannual variability in climate over southern and eastern Australia is very much an effect of ENSO events. Harris *et al.* (1988) noted a strong effect of changing wind stress on inland and coastal ecosystems in Tasmania caused by this phenomenon. Simpson *et al.* (1993) have shown a marked effect of ENSO events on rainfall, runoff and streamflow in south-east Australia. It has also been shown that ENSO influences rainfall, runoff and aquatic ecology as far north as south-east Queensland (Harris and Baxter 1996).

The most recent outbreaks of algal blooms (*Rhizosolenia cf. chunnii*) causing bitter taste in Port Phillip Bay mussels in 1975, 1987 and 1993-1994, have been associated with unusual climatological conditions in warm, calm, sunny winters; 1987 and 1994 were ENSO years.

The marked interannual variability in algal biomass in the Bay, driven almost certainly by variability in climate and freshwater inputs from the Yarra River, leads to the inevitable conclusion that the health of the Bay is highly sensitive to the volume and quality of the catchment runoff.

The nutrient loads to the Bay are largely split between the WTP and the Yarra River plus the major creeks and drains. At present, the load is about 3,000 t N/yr from the former and 3,000 t N/yr from the latter. The atmosphere contributes about 1,000 t N/yr to the Bay (Chapter 3). The time course of the loading is, however, quite different in each case. The WTP load is more constant over time with a seasonal component (Murray 1994), whereas the Yarra load is highly sporadic, showing seasonal and interannual variability.

While storm runoff and catchment land use are an important determinant of the long-term health of the Bay there is not such a simple relationship between river flow and algal biomass as there is in Chesapeake Bay (Harding *et al.* 1992).

This is partly due to the sporadic nature of the Yarra River inflows. As in most other Australian river systems, 80-90% of the load arrives in less than 10% of the time (Harris 1994a).

The main storm loads from the Yarra arrive in summer whereas the WTP inputs peak during winter when temperatures are low and biological activity is reduced.

By stimulating large algal blooms in coastal waters around the north-east corner of the Bay (Grant and Gayler 1982, Longmore *et al.* 1996, Magro *et al.* 1996), transient storm-induced loads seem to have a greater impact on the ecology of the Bay that do the more constant, background WTP inputs.

At present, the storm loads to the Bay are not sufficient to cause long-term detriment to the health of the Bay, but the situation needs to be watched carefully.

While the Bay as a whole can tolerate the present situation, there is a clear warning that in localised areas efforts need to be made to reduce impact. Two things can be done.

First, and most importantly, efforts need to be made to reduce the storm loadings from the Yarra catchment and from urban areas on the eastern side of the Bay. Depending on the size of the flood, N loads from the Yarra can have a widespread effect on the Bay. Both the PPBES model and the monitoring data indicate that storm loads from the Yarra River and the other creeks and drains are the major perturbation to the Bay ecosystem, particularly during summer.

Second, efforts could be made to improve the denitrification efficiencies of the WTP. Nitrogen loads from the WTP are relatively low in summer because denitrification at the plant is effective when the weather is warm. Nitrogen loads are higher in winter (Murray 1994) and ways should be sought to reduce the winter N loads. However, the impact of the WTP load on the Bay is not widespread, being mainly restricted to the north-west shore of the Bay.

6. What is the critical nutrient load?

The PPBES model predicts that there are two major points of instability in systems like the Bay (see Chapter 6). One lies in the planktonic interactions in the water column and the other lies in the sediments.

As nutrient loads are increased phytoplankton growth rates eventually exceed the ability of the zooplankton to graze the populations down and irregular blooms may result. Ungrazed blooms sink to the bottom and decay, using up oxygen in the process. If the rate of oxygen demand exceeds the rate of reoxygenation from the air, anoxia will result. Once anoxia occurs the system will become self fertilising (Kemp *et al.* 1990). Anoxia at the bottom of the water column and on the sediment surface will lead to the elimination of most benthic macrofauna, the cessation of denitrification and the recycling of ammonia to stimulate further phytoplankton growth. Phosphorus is abundant in the Bay waters and empirical relationships from other coastal waters show that if anoxia became widespread the chlorophyll levels could rise 10 to 20 times before the system becomes P, instead of N, limited (Vollenweider *et al.* 1992, Giovanardi and Tromellini 1992). The waters of the Bay would become visibly green, a state quite unacceptable by any criteria of water quality or ecosystem health.

The PPBES model predicts that if the Bay is treated as one or two compartments or 'boxes', the transition to instability and anoxia will be swift and virtually irretrievable.

The critical load at which instability is reached can be estimated with some precision but there is a marked hysteresis effect whereby, in order to recover the situation, the external load must be drastically reduced before oxygenated conditions can recur on the bottom (see Chapter 6, Section 3.6).

How will the Bay respond? This is where the sheer size and scale of the Bay becomes important: the Bay is not a simple one or two box system. With an area of almost 2,000 km^2 and a volume of about 26 km^3 the Bay is large, wide and shallow. The large surface area of the Bay, coupled with its shallowness and low freshwater input means that wind mixing is effective, stratification rarely occurs and oxygen can always reach the bottom.

Nutrients entering the edges of the Bay will be assimilated close to the shore. The time scales of algal growth and physical mixing are such that the blooms will usually occur before nutrients can be mixed offshore. This restricts the main impact of the nutrient load to nearshore waters. As the blooms are mixed offshore, their impact diminishes rapidly. The concept of a critical load is therefore blurred by the sheer size of the Bay.

This is not to say that nutrient loads above the threshold could not eventually cause the Bay to go bright green and induce anoxia, but the effect will not be immediate all across the Bay. Increased nutrient loads will first have an effect around the edges of the Bay – indeed, even in the presettlement era strong Yarra River inflows would most likely have stimulated algal blooms in Hobsons Bay and down the eastern shore as now.

The algal monitoring data (Arnott *et al.* 1996, Magro *et al.* 1996) show changes in algal biomass in a clockwise direction around the Bay beginning in the Geelong Arm. This is a direct result of the predominantly clockwise circulation in the northern part of the Bay under the influence of westerly winds. The Yarra River plume tends to hug the north-east shore of the Bay as it flows to the south (see Frontispiece). The expressed result of the nutrient loads is therefore a function of the geography of the Bay, the position of the inputs and the wind-driven circulation.

So, how do we know if we are approaching the critical load? We have already seen that chlorophyll concentrations in the water are at present not rising over time. However, the most sensitive indicator of the trophic state of the Bay is not the water quality but the characteristics of the nutrient fluxes from the sediments (see Chapters 3 and 6).

In the main Bay, the sediment nitrogen fluxes are dominated by denitrification – this is the desirable state. Overall, the Bay loading is less than the critical level. However, in a few places, notably in Hobsons Bay off the Yarra River mouth, the character changes and the return N flux from the sediments is in the form of ammonia. This is not desirable and is an indication that in one or two marginal sites close to major inputs the threshold has been exceeded.

Both the PPBES model and the benthic chamber data indicate that the relationship between N loads and denitrification efficiencies should be 'hump shaped'. That is, denitrification efficiencies show a maximum at intermediate N loads (see Chapter 6, Figure 6.4).

The biomass of benthic macrofauna increases as water column production increases and organic deposition increases (Kemp and Boynton 1992) – but only up to a point. Benthic communities become depauperate, burrowing species disappear and bioturbation ceases if the sediment surface is too heavily loaded with decomposing organic material so that anoxia results (Chapter 5, Pearson and Rosenberg 1978, Giblin *et al.* 1995, Levinton 1995). Under these circumstances, N is buried rather than denitrified and there is a significant ammonia flux (internal load) to the water column. Also, there are major changes to the bacterial flora of the sediments with changes in the sulphur chemistry (Giblin *et al.* 1995). The critical load is therefore best expressed in terms of changes in the rates of sedimentary denitrification.

Based on actual data on sedimentary fluxes and denitrification rates from the Bay, the PPBES model (Chapter 6) estimates that the critical catchment load at which sedimentary anoxia would begin is somewhere between double and treble present loadings, say 17,000 t N/yr or 8.5 g/m^2/y. This is the irreversible point at which permanent eutrophication of the Bay would occur.

While it is difficult to make precise comparisons between coastal ecosystems, according to Lukatelich (1990) seagrass losses began to occur in Cockburn Sound at an areal load of 3.6 gN/m^2/yr and phytoplankton blooms broke out at a load of 7.2 g N/m^2/yr.

As is indicated in Chapter 6, the relationship between N load and water quality is highly non linear and while it may seem that there is a wide margin for error between the present load (about 7,000 t N/y) and the load at the irretrievable point, eutrophication and the deterioration of water quality is a continuing phenomenon. There is evidence to support this argument.

Both the model (Chapter 6) and data from overseas experience with N reductions (Hinga *et al.* 1995) indicate a non-linear response of coastal embayments to reductions in N load. At high loads the relationship between changes in N and changes in algal biomass is about unity. At lower loads (in the range found in the Bay) the proportionality constant is higher (about two to five times) so that greater reductions of N are necessary for a given reduction in algae. Denitrification plays a more important role in influencing water quality in oligotrophic systems.

Hence the critical load of 17,000 t N/yr may be an overestimate in the longer term. Sedimentary processes, such as feedbacks between widespread hypoxia and reduced benthic diversity, may intervene to steepen the relationship between benthic fluxes and denitrification efficiency and so progressively reduce the critical load over a period of years.

The field observations indicate that the Bay is presently on top of or slightly to the right hand side of the 'hump' (Chapter 6, Figure 6.4) i.e. a further increase in total N loads to the Bay will lead to a further reduction in denitrification efficiencies. This is a Bay-wide phenomenon, so this situation needs to be watched with care.

Clearly it is desirable to maintain the absolute maximum denitrification efficiency in the Bay. This will ensure the long-term health and stability of the Bay and provide some cushion against unforeseen extreme events.

We recommend for long-term sustainability a precautionary reduction in total N loads of the order of 1,000 t N/yr – particularly the loads arising from the Yarra River and from the major creeks and drains in the urban area.

The benefits of such a strategy are clearly seen in the recent 1994 drought. During the latter part of the Study inflows from the Yarra and from creeks and drains were greatly reduced – leaving only the WTP inflows. During this time the Bay became markedly more oligotrophic and chlorophyll concentrations dropped to the lowest levels seen for years (Longmore *et al.* 1996). The abundance of algae around the margins of the Bay also dropped markedly (Magro *et al.* 1996). This is a clear indication that reduced storm inflows and urban runoff can have a marked effect on the primary production and hence water quality of the Bay.

A reduction of the total N load to the order of 6,500 t N/yr would rapidly show sustained improvements in water quality in the Bay. In summary, N loads to the Bay should probably never exceed present levels and should ideally be reduced. They should certainly never be allowed to reach double present loadings.

Finally, we believe that the future of the Bay must be seen as an integral part of the entire regional development plan for the Melbourne region. The Bay is but one part of the whole urban water cycle for the city and the future planning of water supply and sewage disposal options must include the health of the Bay. Integrated catchment management must be implemented.

The present Study was confined solely to processes and events occurring in the waters of the Bay. Nutrient and toxicant loads were estimated from actual measurements and from empirical models of loads at the mouths of rivers, creeks and drains. We believe that, in the longer term, the key to successful management of the Bay will be to link the Bay to the city and to its catchments via a set of process based water supply and catchment models which will drive inputs to the Bay model. In this way the impact of various water supply, sewage and catchment management options can be assessed objectively.

7. Recommendations

Toxicants

Recommendation 1. Although toxicants are at low levels in the Bay at present, inputs should continue to be managed and ideally reduced. As a precaution one should monitor toxicants in valued ecosystem components every five years.

Recommendation 2. Although the impact of toxicants is largely restricted to the mouths of creeks and drains these are areas of human contact. It would seem wise to develop local catchment management strategies to reduce toxicant inputs to these waterways.

Recommendation 3. Investigate the possibility of long-term chronic effects of low-level toxicants on the biota of the Bay.

Ecology

Recommendation 1. The benthos are a vitally important component of the Bay ecosystem. To protect the biodiversity of the benthos and its key role in ecosystem function, habitat destruction must be reduced to a minimum. The effects of fishing, dredging and coastal engineering on seagrasses and the benthos must be minimised and should be closely monitored. The disposal of dredged spoil must be confined to as small an area as possible.

Recommendation 2. There is insufficient historical monitoring data to fully document any variability of seagrass beds and reefs over the years. Changes in the extent of seagrass beds and reefs must be closely monitored on a regular basis. Community involvement in these activities should be encouraged by the use of volunteer naturalists.

Recommendation 3. Protection and conservation of the Bay ecosystem and its habitats are essential for the sustainable health of the Bay. Wherever practicable habitat restoration should be attempted – even to the extent of establishing artificial reefs. Community adoption of and involvement in these programs is desirable.

Recommendation 4. The introduction and spread of exotic species is a major issue for the sustainable use of the Bay. We endorse moves at National and State levels to control such introductions. In conjunction with these programs, steps should be taken to minimise further introductions to the Bay. Monitoring around port areas needs to be initiated and maintained. The role and function of exotic species in the overall ecology of the Bay must be assessed.

Recommendation 5. To protect valued resources, commercial and recreational fishing should be managed so as to ensure continued sustainable exploitation. Fish habitats must be protected and where possible restored. To conserve the resources of the Bay management agencies should consider the establishment of protected areas.

Nutrients

Recommendation 1. To ensure the sustainable health of the Bay for future generations (and allow for climate variability) a target reduction in overall load of 1,000 t N/yr should be adopted. This target is consistent with ongoing stability and protection of the Bay ecosystem.

Recommendation 2. The impact of storm loads and urban runoff in the Yarra River and the major creeks and drains must be reduced. Control of storm loads is more important than control of nutrient loads during base flow conditions. Wherever possible Total Suspended Solids and N loads to the Bay from these sources should be reduced by catchment remediation and reduction in storm overflows. **Strategies to reduce N loads to the Bay should give this recommendation highest priority.**

Recommendation 3. Improve the denitrification efficiency of the WTP where practicable. This is especially important in winter when denitrification efficiencies in the WTP are low. If denitrification efficiencies in winter could be increased to a similar level to those in summer then ammonia loads from the WTP could be decreased significantly. Cost effective ways to achieve this should be investigated.

Recommendation 4. A clear distinction must be made between environmental quality around the edges of the Bay and the overall health of the entire system. Local impacts around the mouths of creeks and drains are higher than in the centre of the Bay. In addition to catchment remediation, good planning and sensitive foreshore development is essential. It is important to protect the aesthetic and conservation values of the coastal and intertidal zones by a combination of public education, litter control, beach restoration and minimisation of physical impacts.

Recommendation 5. An ongoing monitoring program should be established. The monitoring program should be designed to monitor the ongoing health of the Bay, to provide early warning of unforeseen impacts (e.g. climate variability and exotic species) and to measure performance on a year-to-year basis and ensure compliance with these targets. It should include water quality. Loading targets for the Bay should be agreed and a timeline set for achievements of these goals. This is the responsibility of Government. Sediment fluxes, benthic biodiversity and other sensitive indicators of Bay function should be monitored to ensure compliance with management goals and sustainable ecosystem health.

General

Recommendation 1. The integrated model and its predictive capabilities should be used on an ongoing basis to assess trends in the data and monitor performance in improving the trophic state of the Bay (this will be a massive ecosystem experiment – good performance criteria will be needed). Publicly report the annual performance and implement the necessary changes in terms of changes in management policies and water quality criteria.

Recommendation 2. If the reduction in N loads is achieved, SEPP chlorophyll criteria may be reset.

Recommendation 3. The Bay and its catchments must be seen as an integrated whole. An integrated catchment management strategy is required. In any sustainable future, environmental science must support the development of future regional water management strategies and must underpin the reconciliation of economic, social and environmental imperatives. Models of the Bay must be integrated into catchment models and models of the entire water cycle so that long term sustainable management of the Bay becomes possible.

8. An ongoing monitoring program – suggested activities

We suggest that an ongoing monitoring program should consist of the following (full details will be provided in a more detailed Technical Report):

1. Monitor and report annual loads of TSS, C, N, Si and P to the Bay. This will require some sampling of storm events to calibrate the relationships between flow and load. Keep the present empirical catchment loading models up to date.
2. Continue the fortnightly network of coastal stations (EPA, Arnott or 700 series) to provide ongoing monitoring data to calibrate/validate the model, to monitor chlorophyll *a*, water quality data and toxic algal blooms (species composition). Data should be collected under way between stations and two Bay centre stations (MW1, MW2) plus The Rip should be included in the cruise track. Data are to be fed to a GIS system and used for compliance monitoring purposes. As an indicator of change, monitor and report Bay-wide pools of major nutrients on a quarterly basis.
3. Monitor underwater light levels, temperature, oxygen and fluorescence profiles and nutrient concentrations at a small number of fixed sites around the Bay. Pay careful attention to changes and trends in bottom water oxygen concentrations. Use moored technologies wherever possible to obtain a complete data set over a range of time scales and reduce manpower requirements. One mooring plus a meteorological buoy should be installed in the Bay centre – other moorings should use existing fixed piles and other structures.
4. Use benthic chambers to monitor sediment fluxes of C, N, Si, P, O, denitrification, decomposition and irrigation rates. Sampling design to be established on the basis of existing knowledge of spatial and temporal variability of benthic fluxes in the Bay. A mid-summer transect from the Bay centre to Hobsons Bay plus stations in Corio Bay and off Werribee would suffice.
5. Collect sediment cores from central, Corio Bay and Hobsons Bay stations on a periodic basis and examine cores for depth of bioturbation, pore water profiles of nutrients and sulphide concentrations, metals and toxicants. Continue analyses of lipid biomarkers to detect changes in major depositional processes.
6. Monitor benthic communities annually. Family level identifications may be sufficient for detection of major changes in community structure but identification to species level is necessary if exotic introductions are suspected. Compare results to previous data to detect changes over time. A full community analysis is required every five years. Monitor biomass of major floral and faunal groups including macroalgae and microphytobenthos on a shallow to deep transect annually. Monitor seagrass extent and density every five years. Redo the benthic habitat maps every five years. Community groups and volunteer divers could be used to greatly assist with these tasks.

7. Continue the present MAFRI fish survey design. The present boat ramp surveys and creel census of recreational fishers should continue. Monitor populations of dolphins, seals and penguins annually using volunteers and community groups.
8. As now, monitor and manage toxicant loads to the Bay. As an early warning system monitor dissolved toxicant concentrations in the Bay centre annually. Monitor toxicants at five-year intervals in sediments and ecosystem components. Look for signs of biomagnification and test for low-level chronic effects on ecosystem components. Sites are to be chosen in Corio Bay, Hobsons Bay, Werribee and the Bay centre. A risk analysis should be carried out every five years.
9. Monitor closely any dredging activities in the Bay, paying particular attention to increases in suspended solids loads and toxicant levels adjacent to dredging operations and in spoil grounds. Monitor dredged spoil disposal for transport of exotic species.

9. Some research issues arising from PPBES

1. The accurate measurement and monitoring of nutrient loads arising from the atmosphere and the many rivers, creeks and drains entering the Bay. This is important for long-term management. The Study relied on empirical models of loads entering the margins of the Bay. We believe that in the long term an integrated catchment management program will be required linking catchment models with the models of the Bay.
2. Given the interannual climate variability which has been observed and the presence of long-term (>5 yr) trends in the water quality data, the impact of extreme events (floods and droughts) on the ecology and stability of the Bay must be studied. Long time series observations of loads, water quality and the ecosystem response are required.
3. The prediction of the impact of introduced exotic species on system function and of the impacts of system function on the abundance of individual species in the ecosystem.
4. Given the size and complexity of the Bay, it has proved to be difficult to obtain adequate benthic habitat maps. Drifting weed is a particular problem. Given the importance of conservation, habitat preservation and biodiversity in the Bay this is an important task.
5. The light requirements of seagrasses, macroalgae and microphytobenthos need to be studied as well as the interaction of suspended solids and water clarity on the growth rates of these important groups.
6. More work is required to fully document the reasons for the seasonal and interannual variability in fish populations to separate natural from anthropogenic influences.
7. The question of sedimentation rates needs to be solved. In the light of deep bioturbation by benthic macroinvertebrates this is a difficult task. In addition the sources and sinks of organic N in the Bay need to be studied. High rates of deposition and remineralisation of organic N in the sediments may significantly affect the calculations of the N budget for the Bay.
8. Benthic chamber experiments should carry out longer term light/dark incubations. This would give valuable measurements of microphytobenthos activity on the sediment surface and complete the sediment N system understanding.
9. Improved knowledge of the wind fields over the Bay would improve the 3-D hydrodynamic modelling.
10. Monitor closely any dredging activities in the Bay, paying particular attention to increases in suspended solid loads and toxicant levels adjacent to dredging operations and in spoil grounds. Monitor dredged spoil disposal for transport of exotic species.

Now the major report is complete and our conclusions presented, the Study will be wound down. The final Technical Reports from the various Tasks will be published giving the full scientific results of the Study. A series of management scenario workshops will be held and the model will be used to analyse and develop a number of possible strategies. Data from the Study will be made available and a manual will be produced detailing in full the proposed ongoing monitoring program. We hope to publish a CD-ROM disc of model results and other graphic outputs from the Study.

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Glossary

Abiotic: Non-living.

Absorption: The taking up of substances onto a surface.

Algae: A large group of non-flowering plants, many microscopic, containing chlorophyll. Most algae are aquatic.

Ambient: The prevailing environmental situation. Sometimes called ‘background’.

Ammonia (ammonium): Compound consisting of a single nitrogen atom coupled with three or four hydrogen atoms. It is a nitrogen source for algae.

Amplitude: Half the range (or height) of a wave or tide.

Anoxic: Devoid of oxygen.

ANZECC: Australian and New Zealand Environment and Conservation Council.

Anthropogenic: Changes resulting from human activities.

Aqua Regia: A mixture of nitric and hydrochloric acids.

Assimilation rate: A measure of photosynthetic activity.

Autotroph: An organism which can grow by photosynthesis.

Bacteria: A large group of single cell or filament-like microscopic organisms lacking chlorophyll *a* and well defined cell nuclei. Cells multiply by simple fission.

Bathymetry: Depth characteristics of a water body.

Bathypelagic: Usually applied to fish living at mid depth or all through the water column.

Benthic: Belonging to the sea floor.

Benthos: Organisms living on or in association with the sea floor.

Bioaccumulation: The concentration of substances (especially toxicants) in the tissues of plants and animals.

Biochemical: Chemical reactions occurring in living organisms.

Biocide: A chemical which kills animals and plants (also pesticide).

Biodiversity: Measure of the number of species inhabiting a given area.

Bioirrigation: Water movement through the sea bed resulting principally from the feeding activities of benthic suspension-feeding invertebrates.

Biomarkers: Complex organic molecules characteristic of particular plants, algae and bacteria.

Biomass: The living weight of animal or plant populations or communities.

Biota: All living organisms of a region.

Bivalve: A type of mollusc possessing a hinged two-valve shell (e.g. scallops and mussels)

Bloom: Microalgae occurring in dense numbers in a water body.

BP: Before present

Calibration: Use of measured field data to set up a mathematical model.

Catchment: The area of land from which run-off from rain enters a waterway.

Chlorophyll: The green pigments of plants which capture and use the energy from the sun to drive the photosynthesis process.

Chronic: Over a long time. Opposite of ‘acute’.

Ciliate: A class of single celled animals with hair-like appendages used for locomotion.

Colloid: A form of solution between true dissolution and suspension. Consists of particles 10^{-7} to 10^{-9} m in size.

Contaminant: A substance out of place (also pollutants).

Coriolis: The force due to the earth’s rotation which moves air and water currents to the left in the southern hemisphere and right in the northern hemisphere.

Crustacean: An animal living in water which has jointed limbs and a hard outer surface (e.g. crabs).

Cycling/recycling: The movement of an element (like nitrogen) through various forms, living and non-living back to its starting form.

Demersal: Used for fish which live on or near the sea floor.

Denitrification: The conversion of bound nitrogen to elemental (gaseous) form.

Density current: Movement of lighter (less dense) water over denser water.

Deposit feeder: An animal which eats organic matter on or in the sea bed.

Detritus: Non-living organic matter (e.g. dead algae).

Diagenic: Used to describe chemical processes occurring within the sea floor sediments.

Diatom: A variety of microalgae which have siliceous skeletons.

Diffusivity: A measure of the rate at which properties mix or spread out in all three dimensions
Ecology: The relationship of living things to their environment.
Ecosystem: A community of plants or animals or both.
Eddy: A rotating or whirling movement of air or water.
Effluent: An outflow, usually sewage or wastewater.
Efflux: A flow of material outwards.
Elutriate: A water extract of soluble material from sediments.
Epibenthic: Living on the surface of the sea bed.
Estuary: The zone where a river mixes with the sea.
ETP: Eastern Treatment Plant (Carrum).
Euryhaline: Able to live in both fresh and salt water.
Eutrophic: Having an unnaturally high content of algae due to excess nutrients.
Eustatic: Rise or fall of sea level, usually due to melting or formation of ice sheets (e.g. polar ice caps or glaciers).
Fauna: All kinds of animals.
Flagellate: A single-celled organism with a whip-like appendage used for locomotion.
Filter feeder: An animal which gains its food by filtering organic particles from water.
Floc: A fluffy mass of material suspended in water.
Flora: All kinds of plants.
Fluorometer: An instrument for measuring fluorescence (e.g. of chlorophyll).
Flushing: The rate at which a lake or bay exchanges its water content.
Flux: Flow of material.
Food chain: The sequence of consumption of plants by animals and those animals by other animals and so on.
Food web: A complex of food chains.
Geochemical: Relating to earth chemistry.
Geology: The study of earth processes.
Geomorphology: The study of land forms.
Gravimetric: Measuring mass or weight.
Grazing: The eating of plants by animals.
Groundwater: That part of rainfall which seeps into the ground and moves slowly in a horizontal direction.
Gyre: A large eddy.
Habitat: The place where a plant or animal lives.
Haemosiderosis: The presence of breakdown products of blood in the gut of fish. A pathological condition often due to toxic substances.
Haline: Salty.
Heavy metals: A general term for cadmium, chromium, copper, iron, mercury, nickel, manganese, lead, zinc, arsenic and selenium.
Heterotroph: An organism which cannot photosynthesise. They live on organic matter already formed by autotrophs.
Hydrocarbons: Compounds of hydrogen and carbon such as petroleum.
Hydrodynamic: Related to movement of water.
Hydrophilic: Having affinity for water.
Hydrophobic: Having no affinity for water.
Ice Age: Those stages in geological time when temperatures on earth fell so low that the polar ice-caps expanded, glaciers formed in river valleys and sea level fell. Also called glacial periods or epochs.
Icthyoplankton: Fish eggs and larvae floating in water bodies.
Indicator species: Animals or plants which show up some effect e.g. pollution or loss of habitat.
Infauna: Animals living within the sediment on the sea bed.
Influx: Inward movement.
Inputs: Substances entering a water-body.
Interfacial shear: The tendency for two fluids flowing over each other to mix (e.g. freshwater flowing into the sea).
Interstitial: Relating to interstices between grains in sediment.
Invertebrate: Animals without backbones.

Ionic: Having an electric charge due to loss or gain of electrons.
Labile: Easily decomposed.
Macrofauna: Large animals.
Macroinvertebrates: The larger invertebrate animals (i.e. macroscopic).
Macrophyte (macroalga): A seaweed.
Macrotidal: Having large tidal excursions.
Melbourne Eddy: A circulating wind pattern about 30 to 60 km in diameter over Melbourne and the Bay.
Mesotrophic: A water body which has a moderate algal population.
Metabolism: The build up and break down of living matter.
Meteorology: The study of climate and weather.
Microalgae: Single-celled plants. Generally less than 1/10th millimetre in length or diameter, usually much smaller.
Microbial loop: A food chain composed of the smallest organisms – bacteria, ciliates and microzooplankton.
Microphytobenthos (MPB): Single-celled algae which live in and on the sea floor. They are mostly diatoms.
Microtidal: Having a small tidal rise and fall.
Microzooplankton: The smallest zooplankton, those less than 1/5th of a millimetre in size.
Mineralisation: The conversion of organic matter to inorganic.
Mixing coefficients: Measures of the rate of mixing of two or more water bodies.
Mixing processes: The ways in which water bodies mix e.g. shearing, wave action or wind stirring.
Model: A mathematical expression of a natural system.
Molecule: The smallest particle of a substance capable of independent existence.
Mollusc: An invertebrate animal with a shell (e.g. mussel).
Monitoring: Continuous measurement.
Nanoplankton: Those micro-algae between 2 and 20 thousandths of a millimetre in size.
Neoplasm: A tumour.
NFR: Non filterable residue (suspended particles in water).
NH&MRC: National Health and Medical Research Council.
Nitrate: The NO_3 anion.
Nitrification: Formation of nitrate from reduced forms of nitrogen, such as nitrite and ammonia.
Nitrite: The NO_2 anion.
Nutrients: Substances required for the growth of plants (like fertiliser).
Oceanography: The study of oceans.
Oligotrophic: Term describing water bodies which have low nutrient and algal levels.
Optical backscatter: One means of measuring particulate matter in water.
Orbital: Motion of a wave. A water particle follows a circular orbit as a wave passes.
Organism: A living entity of any size, plant or animal.
Organochlorines: Complex organic molecules with chlorine atoms attached (e.g. many pesticides).
Oxic: Having oxygen present.
PAR: Photosynthetically Active Radiation. The spectrum of light required by plants for photosynthesis.
Particulates: Particles suspended in water.
Pathogen: An organism which can produce disease.
Phase: A measure of any stage in the passage of a wave from crest to trough and back to crest.
Photolytic: Dissolving or decomposing under the influence of light (usually sunlight). Also photochemical.
Photosynthesis: The transformation of carbon dioxide and water to organic matter and oxygen by means of light energy.
Phototroph: An organism which can photosynthesise.
Phylum: A term used in the classification of animals and plants.
Phytoplankton: Microalgae which live in the water column.
Picoplankton: Organisms less than two microns (µm) in size.
Plume: In oceanography a term applied to a recognisable outflow into a receiving water body (e.g. Yarra plume in the Bay).
Pollutant: A substance in excess or not belonging.
Polychaete: General term for a class of segmented worms with several seta (bristles) per segment. Very widespread in the marine environment.

PPBES: Port Phillip Bay Environmental Study.

Producer: An organism which can create living matter out of inorganic or inanimate matter.

Productivity: The magnitude of a producer's activity.

Refractory: Resisting decomposition.

Residence time: The nominal time spent by a substance in a water body subject to tidal exchange or river flushing.

Respiration: The taking up of oxygen by living things.

Salinity: The salt content of seawater.

Sea breeze: A movement of air from seaward to landward during the day when the land becomes warmer than the sea (opposite to land breeze).

Sea level: Some agreed mark from which tidal and other excursions can be measured. Also applied loosely to the sea surface.

Seagrass: A group of flowering plants which live rooted in the sea floor.

Sediment: Any solid material which sinks to the bottom.

Senescence: Aging and dying.

Sequestration: Locking up. In the context of oceanography usually applied to elements becoming immobilised in sea floor sediment.

Sewage: Strictly speaking household waste but loosely applied to any waste sent to a treatment plant.

Sewage Treatment Plant (STP): A place where human and industrial wastes are treated before disposal to land or water. Primary treatment means removal of solids only. Secondary treatment means breakdown of organic matter and trapping of most toxicants. Tertiary treatment means removal of almost everything in the sewage and discharge of only the water. The WTP treats effluent to the secondary stage using a lagoon process and irrigation.

Stoichiometry: A term in chemistry which describes a balance of elements in reactions.

Stormwater: Run-off during storms.

Stratification: Layering (usually due to temperature or salinity differences).

Substrate: A surface on which organisms live or a substance serving a biochemical reaction (e.g. Bay sediments and MPB).

Suspended matter: See particulates.

Suspension feeder: An animal which lives by filtering particles from the water.

Taxa (taxon): General term for a class, order, species etc. of animals or plants.

Teratogenic: Causing deformity.

Terrestrial: Of the land.

Tidal prism: The volume of water contained in the flood/ebb tidal excursion.

Tides: The movements of water in the ocean in response to gravitational pull of the moon and sun. As the earth turns the tides perform a daily cycle. Because the moon revolves around the earth the tides also undergo a lunar monthly cycle. The revolving of the earth around the sun imposes an annual cycle. All show up as high and low water levels.

Topography: Mapping of land features.

Toxic: Poisonous.

Toxicant: A poison.

Trophic: Related to food chains and food webs.

Trophodynamics: The processes in food chains. Their rate and magnitude.

Turbidity: Cloudiness caused by sediment suspended in water.

USEPA: United States Environment Protection Agency.

Validation: Checking the truth as in seeing that models simulate natural processes accurately.

Vertebrates: Animals with backbones.

Wastewater: Water that has been used and discarded. Not strictly the same as sewage.

WTP: Western Treatment Plant (Werribee).

Zoobenthos: Animals living on the sea floor.

Zooplankton: Small animals living in the water column, usually drifting with the water.

Technical Reports

Water column trophodynamics in Port Phillip Bay by C. Crawford, G. Jenkins and G. Edgar. Technical Report No. 1.

Soft-bottom macrobenthos of Port Phillip Bay: a literature review by G.C.B. Poore. Technical Report No. 2.

Literature review of the physics of Port Phillip Bay by K.P. Black and S. Mourtikas. Technical Report No 3.

Literature and information review of groundwater input of nutrients and toxicants to Port Phillip Bay by C.J. Otto. Technical Report No. 4.

Literature review of the physical and chemical atmospheric inputs to Port Phillip Bay by T. Beer, F. Carnovale, C. Carvalho and M.E. Cope. Technical Report No. 5.

Literature and information review of the benthic flora of Port Phillip Bay, Victoria, Australia by B.R. Light and Wm J. Woekerling. Technical Report No. 6.

The present state of knowledge on fish stocks and commercial and recreational fisheries in Port Phillip Bay, Victoria, and their interactions with nutrients and toxicants by G.J. Nicholson. Technical Report No. 7.

Phytoplankton ecology of Port Phillip Bay, Victoria by M. Wood and J. Beardall. Technical Report No. 8.

Port Phillip Bay Environmental Study: status review by CSIRO Port Phillip Bay Environmental Study Technical Group members. Technical Report No. 9.

The role of suspension-feeding and deposit feeding benthic macroinvertebrates in nutrient cycling in Port Phillip Bay by R.S. Wilson, B.F. Cohen and G.C.B. Poore. Technical Report No. 10.

The role of zooplankton feeding in nitrogen and carbon cycling in Port Phillip Bay by M. Holloway and G. Jenkins. Technical Report No. 11.

Surface inputs into Port Phillip Bay (excluding the Yarra River and WTC) by NSR Environmental Consultants Pty Ltd. Technical Report No. 12.

Groundwater nutrient and toxicant inputs to Port Phillip Bay by HydroTechnology Pty Ltd. Technical Report No. 13.

The effects of bioturbation in Port Phillip Bay by F.L. Bird. Technical Report No. 14.

Western Treatment Plant outputs to Port Phillip Bay by A.G. Murray. Technical Report No. 15.

Biomass, productivity and nutrient requirements of microphytobenthos by J. Beardall and B. Light. Technical Report No. 16.

The distributions and concentrations of biomarkers for sewage, primary producers and bacteria in the major sediment types of Port Phillip Bay by T. O'Leary, R. Leeming, P.D. Nichols and J.K. Volkman. Technical Report No. 17.

Characterisation of toxicants in waters from Port Phillip Bay: metals by G.J. Fabris and C.A. Monahan. Technical Report No. 18.

HF radar measurements of surface currents in southern Port Phillip Bay by A. Prytz and M.L. Heron. Technical Report No. 19.

Impact of shipping and dredging on toxicants in Port Phillip Bay by G.J. Fabris, C.A. Monahan, G.F. Werner and T. Theodoropoulos. Technical Report No. 20.

The distribution, abundance and diets of demersal fish in Port Phillip Bay by G.D. Parry, D.K. Hobday, D.R. Currie, R.A. Officer and A.S. Gason. Technical Report No. 21.

Toxicants in aquatic biota from Port Phillip Bay: data analysis by G.J. Fabris. Technical Report No. 22

Sediment characteristics in Port Phillip Bay by P. Greilach, Z. You, K. Black and R. Gorman. Technical Report No. 23.

Nutrient status of the waters of Port Phillip Bay by A.R. Longmore, R.A. Cowdell and R. Flint. Technical Report No. 24.

A review of the impact of sediment and water quality on benthic communities: evidence of empirical links by J. Nielsen and P. Jernakoff. Technical Report No. 25.

Nutrient status of the sediments of Port Phillip Bay by G.J. Nicholson, A.R. Longmore and R.A. Cowdell. Technical Report No. 26

Algal blooms in Port Phillip Bay from March 1990 to February 1995: temporal and spatial distribution and dominant species by K.L. Magro, G.H. Arnott and D.R.A. Hill. Technical Report No. 27.

Field measurements of suspended sediment concentrations in Port Phillip Bay by Z.J. You, P.R. Greilach, J. Turnbull, D.N. Hatton and K. Black. Technical Report No. 28.

Monitoring Port Phillip Bay benthos, 1969-95 by R.S. Wilson, S. Heislers and G.C.B. Poore. Technical Report No. 29.

Microphytobenthos in Port Phillip Bay: distribution and primary productivity by J. Beardall and B. Light. Technical Report No. 30.

Microzooplankton grazing on phytoplankton in Port Phillip Bay by G. Beattie, A.M. Redden and R. Royle. Technical Report No. 31

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Mixing, flushing and CTD surveys of the waters of Port Phillip Bay by C. Pattariatchi *et al* (final title and authors to be decided). Technical Report No. 34.

Phytoplankton productivity in Port Phillip Bay: seasonal and spatial distributions by J. Beardall, S. Roberts and R. Royle. Technical Report No. 35.

Food webs of demersal fish in Port Phillip Bay by R.A. Officer *et al.* (final title and authors to be decided). Technical Report No. 36.

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A transport model of Port Phillip Bay by S.J. Walker. Technical Report No. 39.

The composition and distribution of phytoplankton in Port Phillip Bay from March 1990 to February 1995 by G.H. Arnott, D.R.A. Hill, D.J. Reilly, A.S. Gason, K.L. Magro and A. G. Coots. Technical Report No. 40.

Study of sedimentation rates in Port Phillip Bay using lead-210 by J.D. Smith. Technical Report No. 41.

Spatial interpolation of underway nutrient data by Q. Shao and D.R. Fox, Technical Report No. 42.

Monitoring Report by G.P. Harris. Technical Report No. 43.

Port Phillip Bay integrated model: final report by J.S. Parslow and A.G. Murray. Technical Report No. 44.

Data storage and use of GIS for the Port Phillip Bay Environmental Study by R.P. Molloy. Technical Report No. 45.

Units, symbols and abbreviations

Prefixes

pico	p	10^{-12}
nano	n	10^{-9}
micro	µ	10^{-6}
milli	m	10^{-3}
kilo	k	10^{3}
mega	M	10^{6}
giga	G	10^{9}

Length

micrometre/micron/µm/µ	10^{-6} metres
millimetre/mm	10^{-3} metres
centimetre/cm	10^{-2} metres
m	metre
km	kilometre

Volume

microlitre	µL	10^{-6} litres
millilitre	mL	10^{-3} litres
litre	L	
kilolitre	KL	10^{3} litres
megalitre	ML	10^{6} litres
gigalitre	GL	10^{9} litres

Weight

picogram	pg	10^{-12} grams
nanogram	ng	10^{-9} grams
microgram	µg	10^{-6} grams
milligram	mg	10^{-3} grams
gram	g	
kilogram	kg	10^{3} grams
tonne	t	10^{6} grams

Time

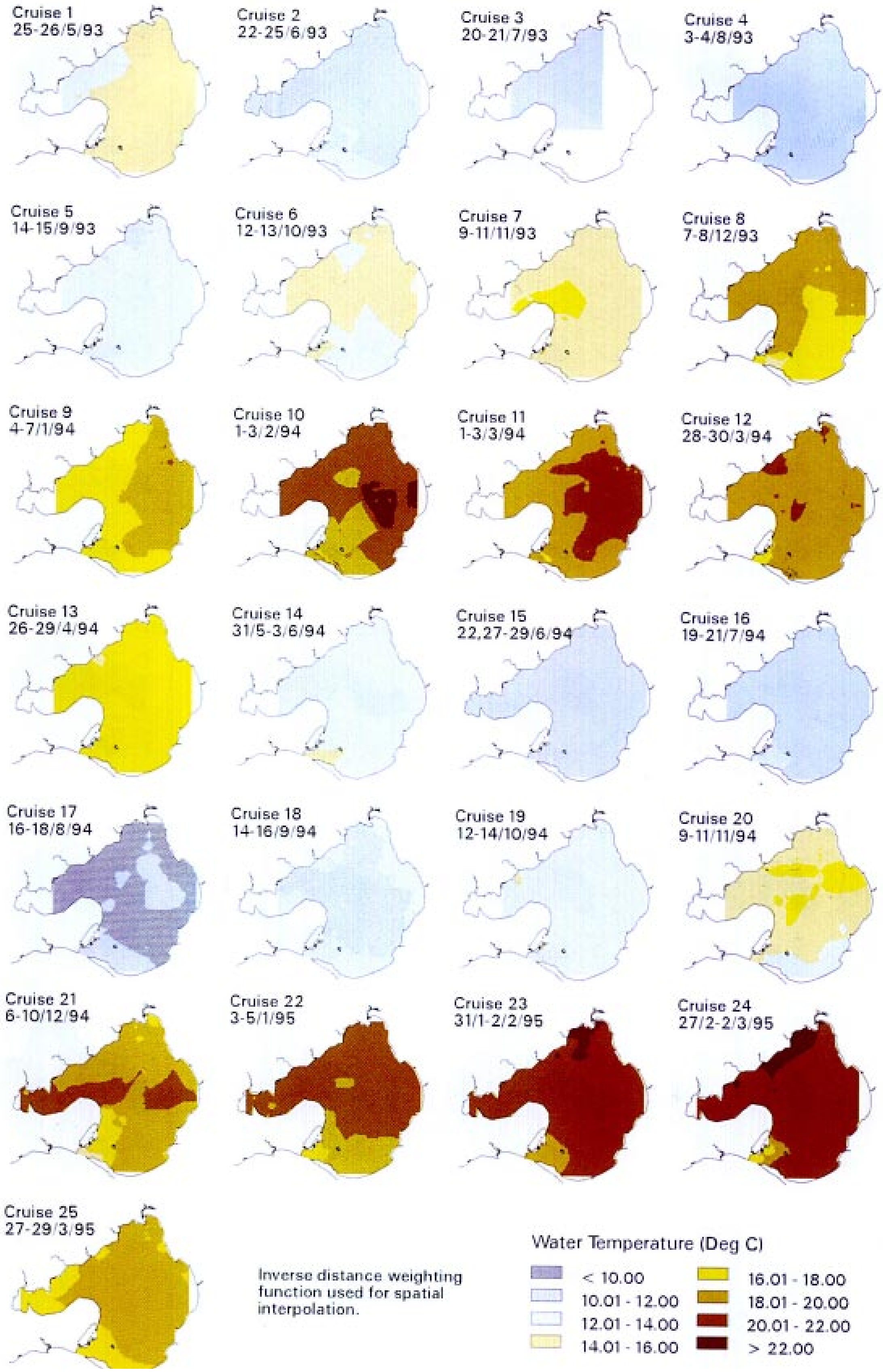
second	s
minute	min
hour	h
day	d
year	yr

Chemical

micromolar	µ mol	10^{-6} moles
millimolar	m mol	10^{-3} moles
molar	-M	(- number prefix)
carbon	C	
nitrogen	N	
phophorus	P	
silica	Si	
DIN	dissolved inorganic nitrogen	
DON	dissolved organic nitrogen	
PN	particulate nitrogen	
TN	total nitrogen	
TKN	total kjeldahl nitrogen (archaic)	
PON	particulate organic nitrogen	
DIP	dissolved inorganic phosphorus	
DOP	dissolved organic phosphorus	
PP	particulate phosphorus	
TP	total phosphorus	

Miscellaneous

hectare	ha (area)	
Pascal	Pa (pressure)	
hecta Pascal	hPa	10^{2} Pa
ppm	parts per million (*m*g/L or mg/m³)	
ww	wet weight	
dw	dry weight	
afdw	ash free dry weight	
m/s	metres per second	
PSU	Practical Salinity Units	
°C	degrees Celsius temperature	
W	Watts	
e	base of natural logarithms (ln)	
Pm	Light saturated photosynthetic rate	
I_K	Light intensity where photosynthetic rate begins to show light saturation	
Q_{10}	Change of reaction rate for each 10°C rise	



Inverse distance weighting function used for spatial interpolation.

Plate 1

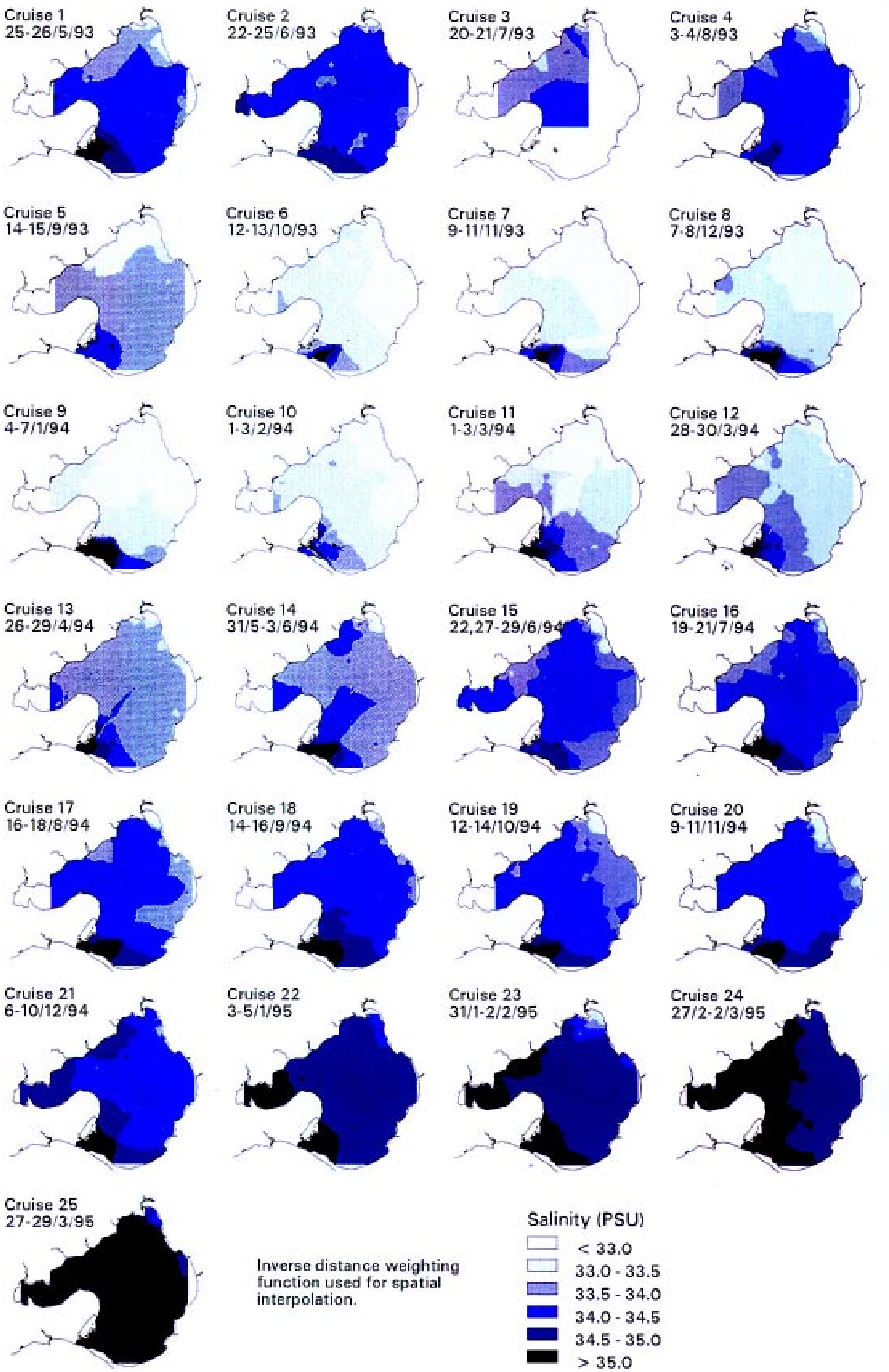
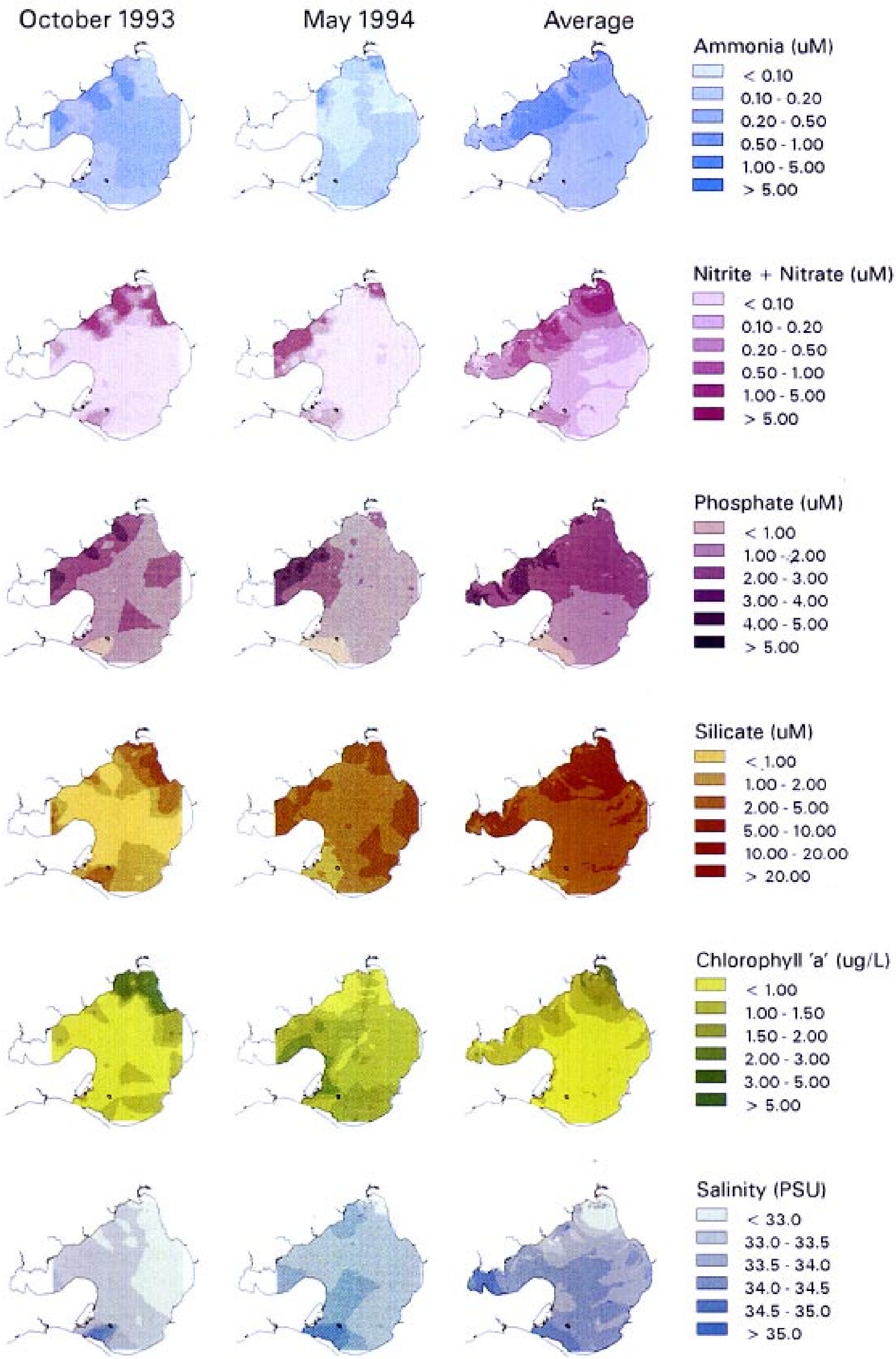
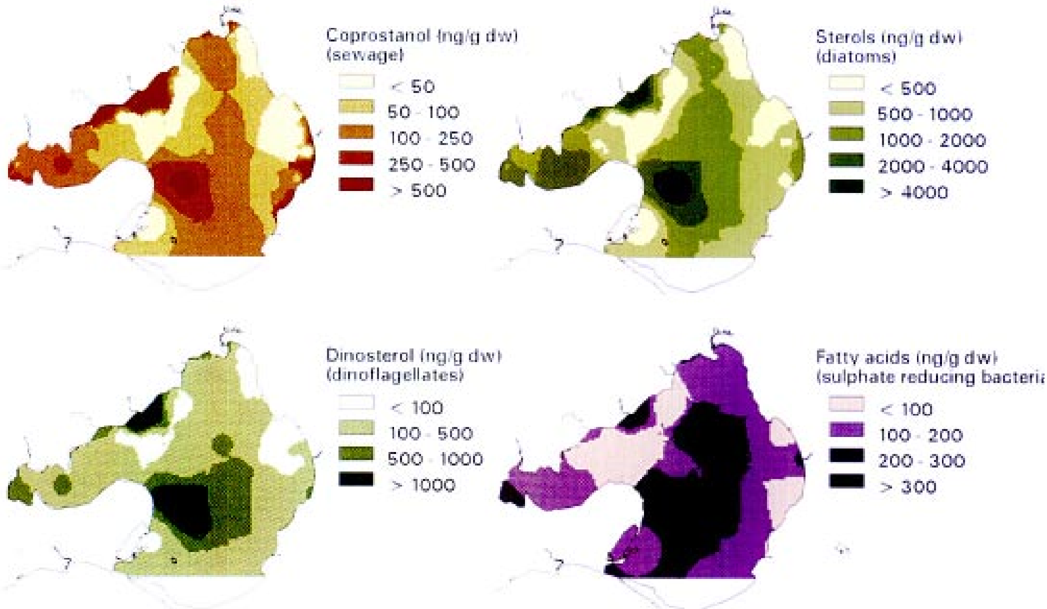

Cruise 1
25-26/5/93
Cruise 2
22-25/6/93
Cruise 3
20-21/7/93
Cruise 4
3-4/8/93
Cruise 5
14-15/9/93
Cruise 6
12-13/10/93
Cruise 7
9-11/11/93
Cruise 8
7-8/12/93
Cruise 9
4-7/1/94
Cruise 10
1-3/2/94
Cruise 11
1-3/3/94
Cruise 12
28-30/3/94
Cruise 13
26-29/4/94
Cruise 14
31/5-3/6/94
Cruise 15
22,27-29/6/94
Cruise 16
19-21/7/94
Cruise 17
16-18/8/94
Cruise 18
14-16/9/94
Cruise 19
12-14/10/94
Cruise 20
9-11/11/94
Cruise 21
6-10/12/94
Cruise 22
3-5/1/95
Cruise 23
31/1-2/2/95
Cruise 24
27/2-2/3/95
Cruise 25
27-29/3/95
Inverse distance weighting function used for spatial interpolation.
Salinity (PSU)
< 33.0
33.0 - 33.5
33.5 - 34.0
34.0 - 34.5
34.5 - 35.0
> 35.0

Plate 2

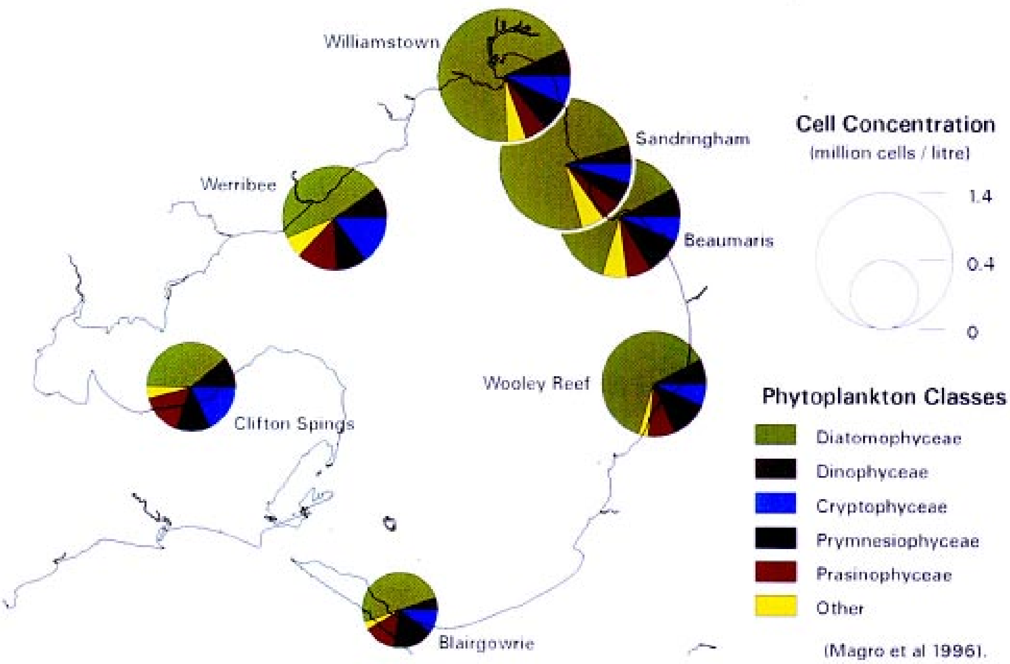

October 1993
May 1994
Average
Ammonia (uM)
< 0.10
0.10 - 0.20
0.20 - 0.50
0.50 - 1.00
1.00 - 5.00
> 5.00
Nitrite + Nitrate (uM)
< 0.10
0.10 - 0.20
0.20 - 0.50
0.50 - 1.00
1.00 - 5.00
> 5.00
Phosphate (uM)
< 1.00
1.00 - 2.00
2.00 - 3.00
3.00 - 4.00
4.00 - 5.00
> 5.00
Silicate (uM)
< 1.00
1.00 - 2.00
2.00 - 5.00
5.00 - 10.00
10.00 - 20.00
> 20.00
Chlorophyll 'a' (ug/L)
< 1.00
1.00 - 1.50
1.50 - 2.00
2.00 - 3.00
3.00 - 5.00
> 5.00
Salinity (PSU)
< 33.0
33.0 - 33.5
33.5 - 34.0
34.0 - 34.5
34.5 - 35.0
> 35.0

Distribution of Biomarkers within Surface Sediments

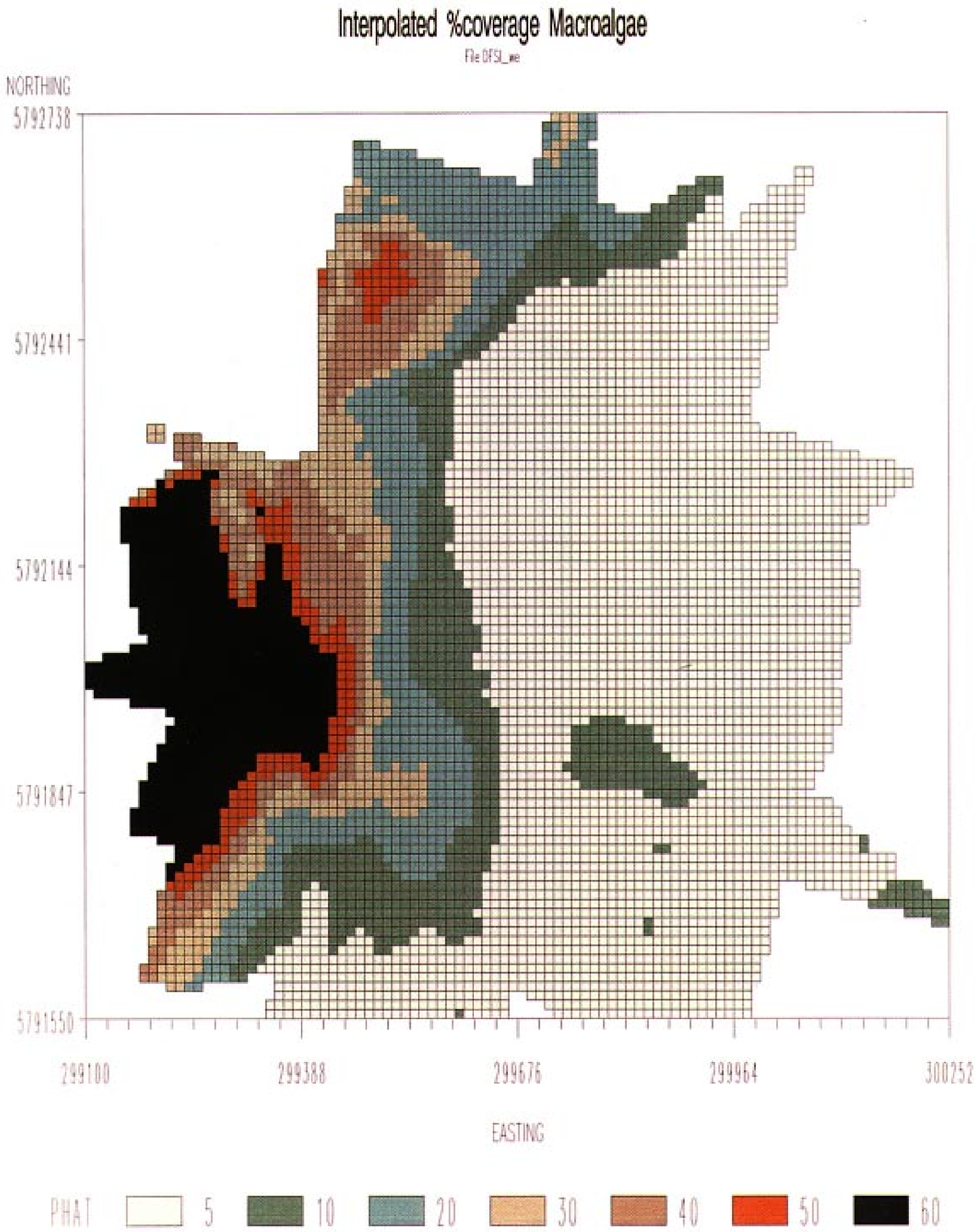


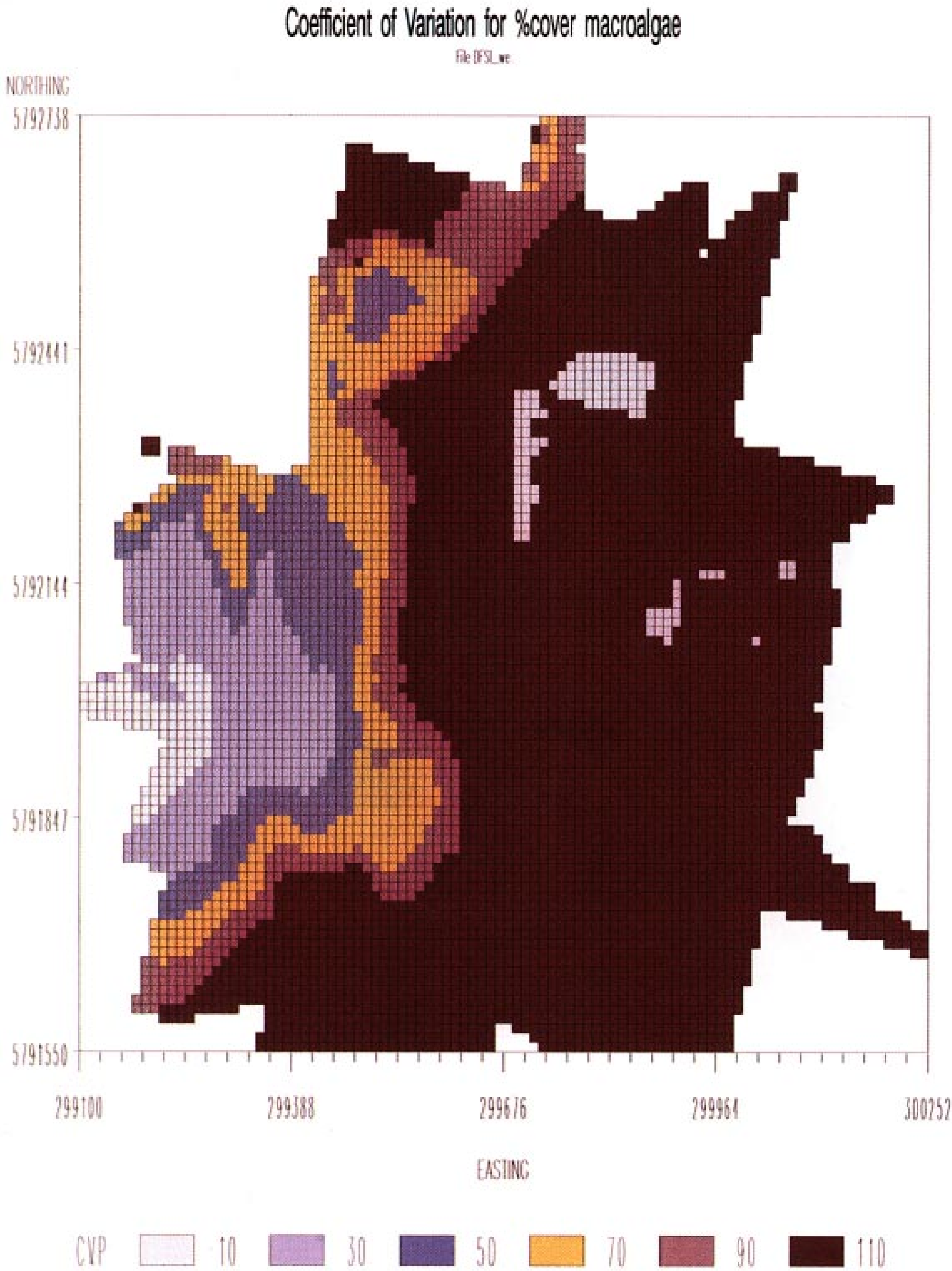
Forty-three sediment samples were collected in June, 1993 (same sites as Task T7, see Chapter 4) and analysed for indicator compounds (biomarkers). Spatial distributions are shown above (O'Leary et al 1994).

Average Phytoplankton Composition (1990 - 1994)

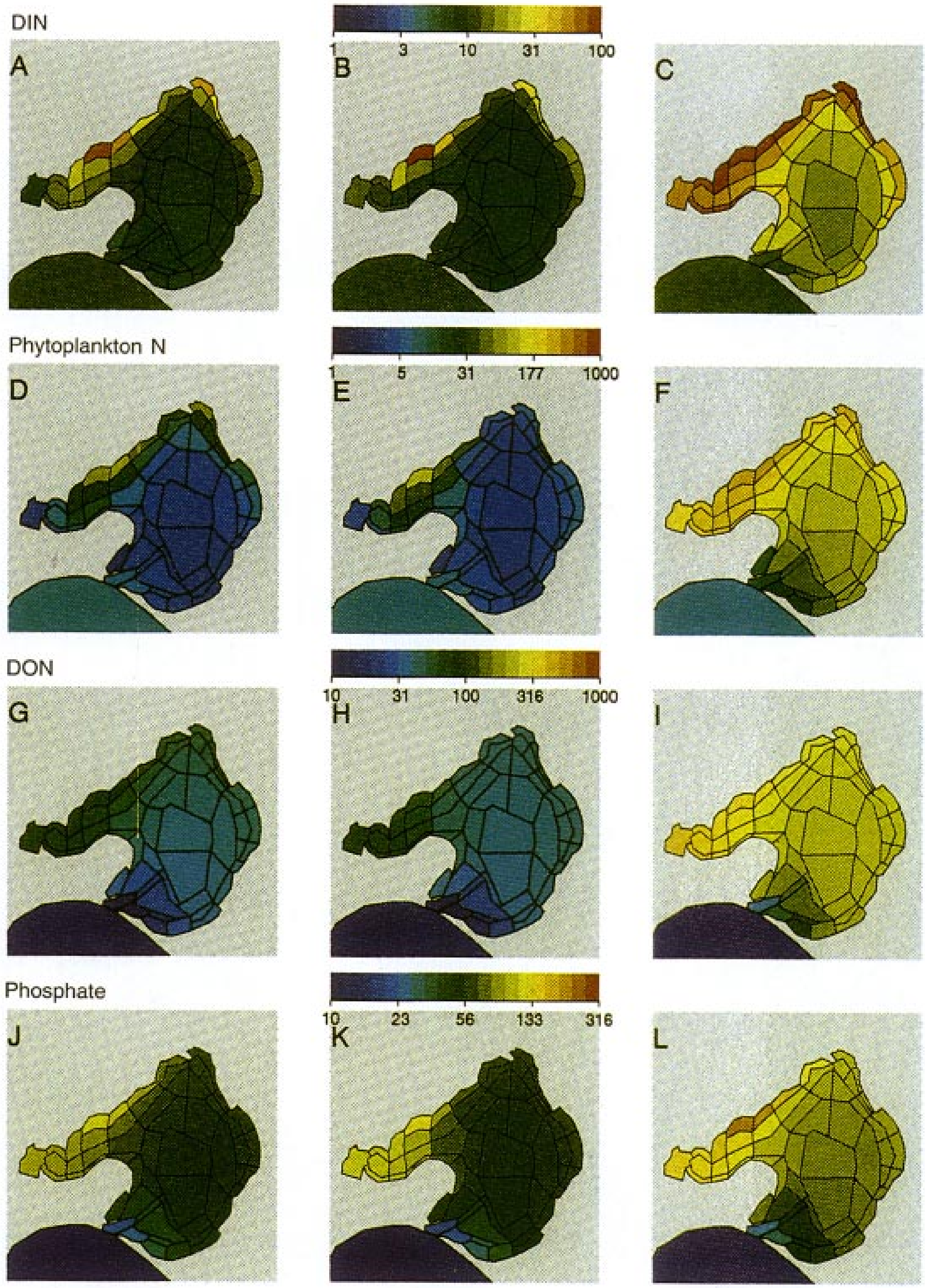


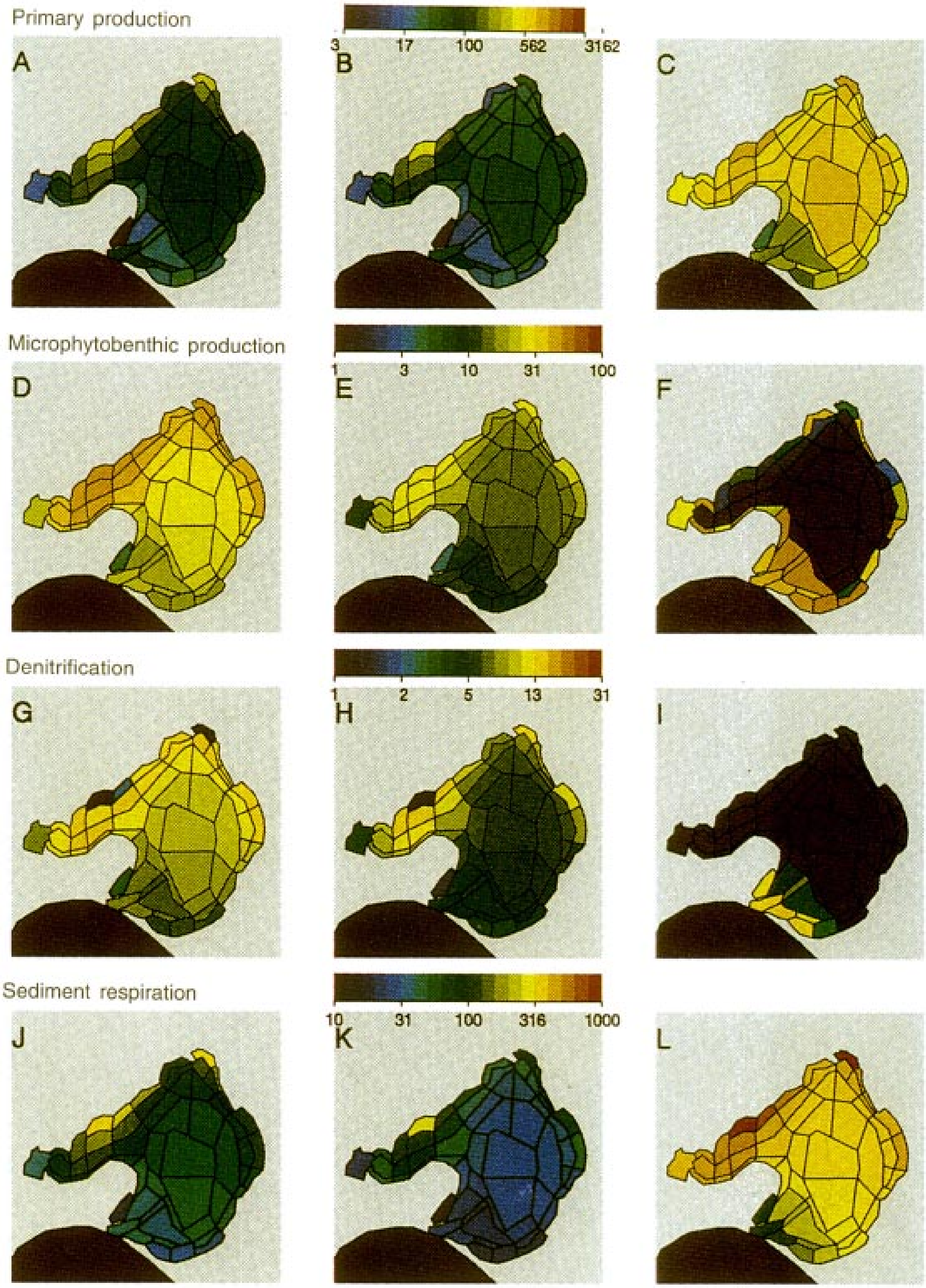
(Magro et al 1996).


Interpolated %coverage Macroalgae
File OFSI_ma
NORTHING
5792738
5792441
5792144
5791847
5791550
299100
299388
299676
299964
300252
EASTING
PHAT
5
10
20
30
40
50
60



Coefficient of Variation for %cover macroalgae
NORTHING
5792738
5792441
5792144
5791847
5791550
299100
299388
299676
299964
300252
EASTING
CVP
10
30
50
70
90
110


DIN
1 3 10 31 100
A
B
C
Phytoplankton N
1 5 31 177 1000
D
E
F
DON
10 31 100 316 1000
G
H
I
Phosphate
10 23 56 133 316
J
K
L



Primary production
3
17
100
562
3162
A
B
C
Microphytobenthic production
1
3
10
31
100
D
E
F
Denitrification
1
2
5
13
31
G
H
I
Sediment respiration
10
31
100
316
1000
J
K
L

Plate 8