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Port Phillip Bay Environmental Study: Status Review

Edited by

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FOREWORD

In July 1991 Melbourne Water, on behalf of a Study Management Committee (representing Melbourne Water, the then Department of Conservation and Environment, the Environment Protection Authority and the Port of Melbourne Authority), commissioned CSIRO to prepare a Study Design for a major environmental study of Port Phillip Bay.

The Study objectives were designed to address deficiencies in the existing knowledge of the extent, fate and effects of input loads of nutrients and toxicants to Port Phillip Bay.

It is anticipated that the results of the ensuing studies will provide the basis for informed decisions concerning environmental management of the Bay for at least the next 20 years.

Fundamental to the evaluation of objectives for study tasks was the assessment of the current state of knowledge of processes such as the transport, distribution and mixing of discharged materials, and knowledge of the status of inputs, waters, sediments and biota.

This report presents review papers by CSIRO Port Phillip Bay Technical Group members and - in Chapter 3 - scientific colleagues from other organizations with appropriate expert knowledge. Much of the information on which this collection of papers has been based is contained in commissioned specialist reviews (see page 9) which have been published as other papers in this technical report series.

I thank each of the Technical Group members, Andy Longmore (Department of Conservation and Natural Resources), Tony Chiffings (Melbourne Water) and numerous other contributors for their contributions to this report.

Over the next two years a large data-base will be accumulated offering the prospect of predictive models of the Bay's condition. I look forward to continued cooperation and help from all those who appreciate and respect this magnificent natural asset.

BRIAN NEWELL
Project Manager
<u>Port Phillip Bay Environmental Study</u>

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Chapter 1. Introduction

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1.1 Introduction

In order to accommodate the short leadtimes for commencement of the Study, the strategy of the Project Design was - irrespective of the validity and extent of existing databases - to identify the scientific information needed to address the Study objectives and provide detailed descriptions of the scientific tasks needed to obtain that information. The Study plan allowed for the immediate commencement of tasks addressing the status of nutrients and toxicants in the Bay - tasks required independent of any historical information.

In this report we address the validity and extent of existing databases and provide a summary of existing knowledge and hypotheses on the Bay. The review assesses the available information describing the status, fate and distribution of nutrients and toxicants in Port Phillip Bay as required by the Study Project Design (CSIRO, 1992). The associated reviews of the physics and ecology of the Bay, and the discussion of integrated modelling, provide the essential framework for study and assessment of nutrient and toxicant inputs and fate. This report should be read in conjunction with the Project Design for the Port Phillip Bay Environmental Study (CSIRO, 1992).

This report is not a literature review in the traditional sense but is aimed at identifying and summarising valid data in the context of the aims and purpose of the Project Design for the Port Phillip Bay Environmental Study. Information available in the public domain has been reviewed and considered along with unpublished studies made available to the Study. The report also draws upon a recent bibliography compiled by the Victorian Institute of Marine Sciences for Melbourne Water (VIMS, 1992a). A number of specialist reports provide source material for the review; these were commissioned by the Study from authors and agencies with extensive sectoral knowledge on the Bay.

This report identifies knowledge gaps and research needs and assigns priorities for the scientific tasks to be carried out during the 4-year Study. These will provide the scientific information essential for the development and implementation of immediate and long-term management options by the agencies responsible for the Bay. The recommendations on the scientific tasks contained here will be subject to peer review and comment, directly and in a specific workshop during November 1992. These task recommendations may supersede the scope, outputs and priorities earlier described for implementation in the Study Project Design, subject to approval by the Study management committee of Victorian agencies (Melbourne Water, Environmental Protection Authority, Department of Conservation and Natural Resources, Port of Melbourne Authority).

1.2 Port Phillip Bay

1.2.1 Background

The Melbourne metropolitan area (population 2.6 million in 1990) extends across a number of river catchments (e.g., Yarra, Maribyrnong, Werribee Rivers) from the northern and eastern shores of Port Phillip Bay (Figure 1.1). The Bay is a large but shallow marine embayment, which has restricted exchange with the open ocean. The Bay has a surface area of approximately 1,950 km^2 and about half of its area is less than 14 metres deep; the central basin has a maximum depth of 22 metres. Exchange with the open ocean is thought to be limited, and an average Residence Time of one year is frequently quoted (MMBW/FWD, 1973).

Port Phillip Bay represents a major asset to the people of Melbourne, other regional cities and towns, the State in general, and Australia. The Bay provides for a range of port, industrial and recreational usage including fisheries, tourism, mariculture, transport and shipping. It also receives a variety of urban and industrial wastes.

The Bay has received waste products from the city of Melbourne since the latters' establishment in 1835 but a sewerage system was not commissioned until 1897. This led to some amelioration of direct sewage and industrial waste discharge effects (Chiffings *et al.*, 1992) although as recently as 1970 the Senate Select Committee on Water Pollution documented a poor situation in and around Melbourne in terms of ambient water quality. Progressive action over the last 20 years has seen considerable improvements resulting from State initiatives (e.g., establishment of the Victorian EPA) and Commonwealth assistance through the national sewerage program. Presently some 98% of the estimated resident population (2.6 million people) are connected to the sewerage system (MMBW, 1990a).

The Werribee Treatment Complex (WTC) - established in 1897 - collected and treated most of Melbournes' sewerage until 1975. The Complex discharged final secondary treated effluent to the Bay.

The commissioning of the South East Purification Plant at Carrum in 1975 reduced the effluent load discharged to the Bay from the WTC. The quantities of nitrogen released to the Bay from the WTC declined to approximately 4000 tonnes y^{-1} representing between 55 and 70 percent of the total nitrogen discharged to the Bay. The fate and impacts of such nutrient additions to the Bay remain largely unexplained.

Industrial effluents containing heavy metals and organic compounds formerly provided significant discharges into the Bay from a variety of discharge points, especially in the Corio Bay sector. However, integration of industrial effluent into sewers and a program of licensing to control volume and quality of point discharges has improved the quality of industrial discharges to the catchment and to the Bay itself. Urban runoff to the Bay and the development of relevant management options remains an issue of increasing concern to catchment managers (Collett, 1992). Agencies responsible for the Bay need to develop a range of options to effectively manage the inputs, to ensure that the multiple and beneficial uses of the Bay environment and catchments are sustained in the interests of the community.

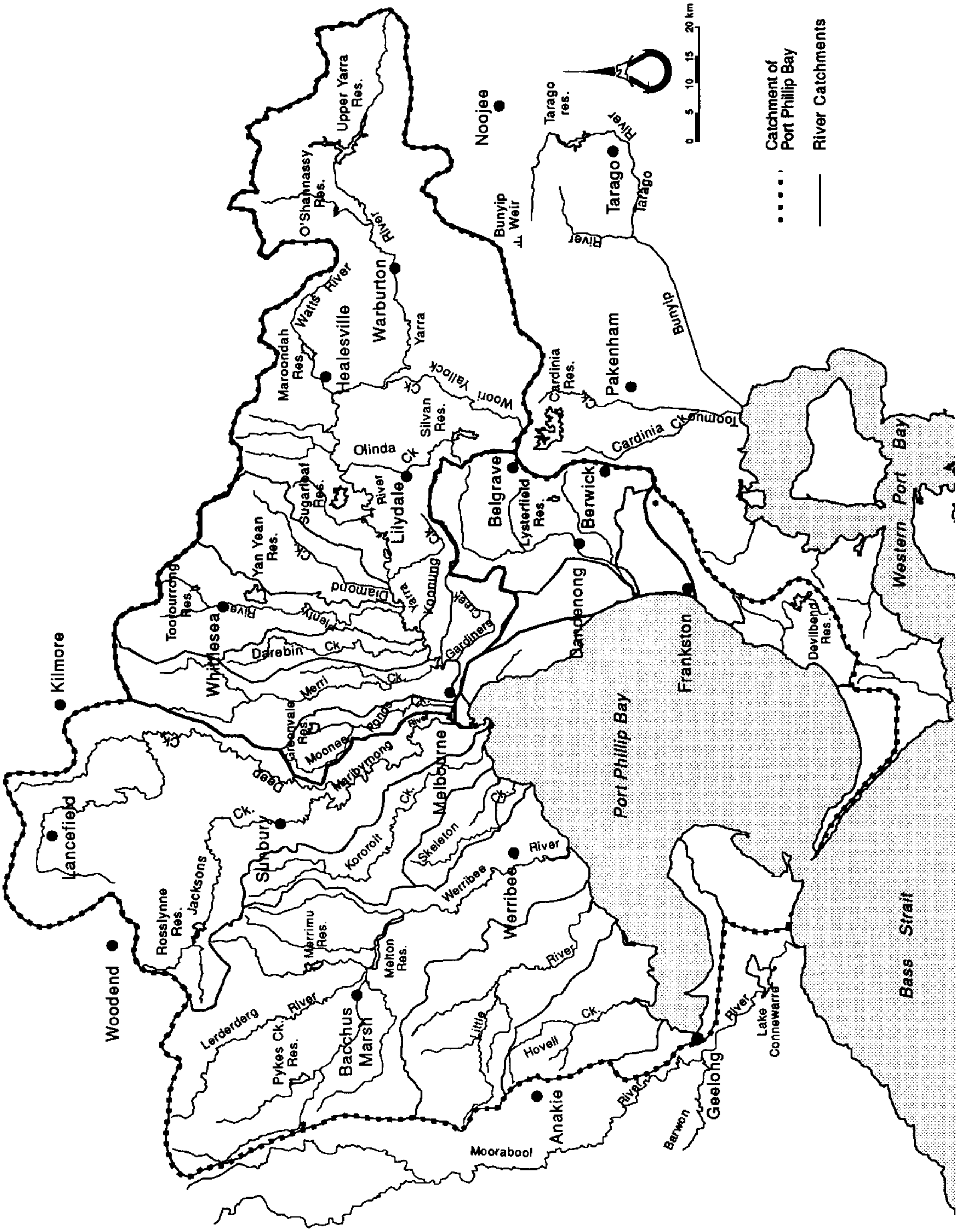

Figure 1.1 Port Phillip Bay Catchment Boundary

Nutrients (mainly nitrogen and phosphorus) and a variety of toxic materials discharged to the Bay have potential to affect the environmental "health" of the Bay (and its socioeconomic values) and to influence the opportunities for a range of recreational and commercial activities. Earlier studies provided Bay managers with the initial scientific knowledge with which to develop management plans to maximise the multiple and beneficial uses of the Bay environment and its catchment. However, changes in land use patterns, population and industrialisation over the last 20 years (see Bremner *et al.*, 1989; Collett, 1992) all necessitate a new and comprehensive evaluation of the status of water, sediments and biota in the Bay. This will provide managers with critical information on which to base the best possible management plans to ensure that the community receives maximum benefits from the Bay well into the next century.

1.2.2 **Environmental management of the Bay**

Melbourne now has extensive services for domestic and industrial waste management, which have been designed to minimise pollution and to protect the health and well-being of its residents, as well as the quality of its waterways and the Bay.

Several government agencies have responsibility for the management of the Bay. The Department of Conservation and Natural Resources (DCNR) has overall responsibility for managing the natural resources, including flora and fauna, fisheries and coastal land. The Environment Protection Authority (EPA) is responsible for protecting Bay water quality by setting environmental standards, monitoring status, and controlling discharge in order to ensure that the natural environment and beneficial uses are protected. Melbourne Water and several smaller water authorities manage water supply, drainage and sewerage systems.

The Port of Melbourne and Port of Geelong Authorities are responsible in their respective port areas and coastal waters for dredging, navigation aids and prevention of oil pollution.

Local governmental authorities regulate land usage in the catchments, as well as waste disposal and litter control. The Bay-side councils have additional responsibility over coastal development.

Victorian governments have, over the years, developed a strategic approach to planning and administration of the State and the coastal zone, with several major strategies and policies which address the Bay and its catchment (CSIRO, 1992). In addition to discharge licensing, major instruments controlling and limiting pollution have been the State Environment Protection Policies (SEPP) developed and administered by the EPA. The first SEPP for Port Phillip Bay was declared in 1975; a revised SEPP was released by the EPA for review in August 1991 (EPA, 1991a). Key elements in the SEPP include beneficial uses to be protected, water quality criteria to define long-term objectives and a specified attainment program. The prescribed environmental parameters were drawn from the data and recommendations of the major Port Phillip Bay Phase 1 Study (1968-71)(MMBW/FWD, 1973) and subsequent monitoring and evaluation projects.

1.2.3 **Environmental studies of the Bay**

Previous studies of the Bay started about 100 years ago and their history is summarised in the Study Project Design (CSIRO, 1992). Taxonomic and descriptive studies of the Bay and its marine communities have been carried out from time to time since 1890 (MacPherson and Lynch, 1966). Work on nutrient and physical factors between 1947 and 1952 (Rochford, 1966) provided the first "systems" overview of the Bay, but the first investigation of significance, undertaken by the National Museum and the Fisheries and Wildlife Department, was the 5-year project commenced in 1957 to evaluate macrofloral and faunal distribution and populations of the Bay.

The first fully integrated and systematic study of the Bay was carried out from 1968-71. This was the Port Phillip Bay Phase 1 Study (MMBW/FWD, 1973). The joint work by the Board of Works (now Melbourne Water) and the then Fisheries and Wildlife Department aimed at providing a comprehensive environmental study of the Bay and its tributary region in order to determine the relationship between prevailing inputs and conditions in the Bay. In addition the study sought to collect baseline data in the vicinity of the proposed outfall (near Patterson River) for the to-be-constructed South-Eastern sewerage system at Carrum. Although the decision in 1969 to direct the Carrum outfall into Bass Strait obviated the latter requirement, the Government recognised the long-term need for and value of information about the Bay, particularly in relation to the impacts of increasing urbanisation. The Study concluded that the Bay, as a whole, was relatively unpolluted but despite extensive mixing near outfalls, some ambient nutrients occurred in the Bay at higher concentrations than in Bass Strait so that a potential existed for enhanced biological productivity in the Bay. This signal research work, which provided much of the basis for current management and options for the Bay, integrated results from water chemistry, physics and biological communities with information on inputs, land use and population around 1970. The segmented approach to the Bay for management was an outcome of the Phase 1 study and the subsequent management policies.

A second Phase of the Port Phillip Bay Study (1975-80) addressed some questions raised in Phase 1, focussing on the effects of the WTC discharges to the Werribee segment of the Bay. The Phase 2 study provided valuable insights into nutrient transformation by benthic microbial communities (Bauld and Millis, 1977) and established the tenet that the Bay is nitrogen limited (Axelrad *et al.,* 1981). The Phase 2 work was loosely integrated and the only overview is an unpublished manuscript (Kelly *et al.*, 1987).

Recently Newell (1990) reviewed data available on the effects of WTC nutrient discharges on the ecology of the Bay and suggested most input nitrogen is exported to Bass Strait as dissolved organic nitrogen, and that physical processes rather than nutrient availability control primary production.

A series of studies have been undertaken by the EPA and the Marine Science Laboratories (DCNR) with regard to heavy metals in the Bay, but work on the status or impact of organic toxicants has been limited (Coller *et al.*, 1991).

Scientific work since the Phase 1 Study has lacked the systematic overview and coordinated development of knowledge and understanding needed for comprehending the effects of nutrient and toxicant discharges on the Bay, thus allowing the development of long-term management options. A major component of the work since Phase 1 has been water quality monitoring to fulfil the statutory responsibilities of the EPA, and until recently included very little work on the Bay's biological characteristics or its sediments. The results from the latter programs have been reviewed by Longmore (1992) and are not conclusive with regard to assessing the effects on the Bay of changes in nutrient loads.

1.3. **Port Phillip Bay Environmental Study (1992-96)**

Since the Phase 1 Study considerable changes have taken place in the use and development of the Port Phillip Bay catchment which influence the quality and loads of point and diffuse inputs. The lack of comprehensive and focussed scientific studies on the Bay-wide effects of inputs prevents a systematic assessment of the status of the Bay, especially with regard to nutrients and toxicants. In addition, the eutrophic poise and trend with time of the Bay, as indicated by chlorophyll *a* concentrations in the water column, have been the subject of some uncertainty (Brown, 1989; Saunders and Goudey, 1990). Global advances in aquatic ecology over the last two decades have provided a better understanding of key processes, especially the role of sediments. There is also now sophisticated ability to describe and model the physics of receiving waters. These factors integrated with nutrient and toxicant information will enhance the scientific foundation for the development of tools to assist managers in the development of options, their implementation and the monitoring of the effectiveness of applied management plans.

The current Study is aimed at using a systems approach to provide the scientific information on Port Phillip Bay which is fundamental to addressing the management issues that have been identified by the consortium group of Victorian agencies responsible for the Bay. The summary of key management issues (Table 1.1) reflects the strategy development and public responsibilities of the agencies, and in turn incorporates many of the community concerns raised in recent years. These issues have significant socioeconomic implications. For example, were discharge of treated effluent from WTC to be incompatible with the community demand for water quality in the Bay, alternative sewerage disposal options could cost the community up to $1 billion (GHD, 1992). Accelerated eutrophication resulting from nutrient enrichment or widespread toxic algal blooms would devastate Bay-side property values and tourism/recreation. These issues emphasise the need for comprehensive, long-term strategies for the management of the Bay. As natural resource usage in the catchment intensifies, with its consequent increases in environmental, social and economic pressures, it becomes imperative that management strategies are based on current information and the best possible risk assessment techniques.

While a great deal of data have been collected on the Bay and its catchment since the Phase 1 study, there has been a lack of focus in scientific activities from which to develop the comprehensive understanding essential to achieving environmental quality objectives. There is a need to assess the present database and further develop it in order to determine the current environmental "health" of the Bay. The process should derive tools for use in developing predictions about the ability of the Bay to cope with projected inputs in the long-term (e.g., time horizons of the order of 50 years are required for considering management options for WTC).

Table 1.1 Summary of key management issues to be addressed by the Study

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i) Nitrogen load to Port Phillip Bay:
- Werribee Treatment Complex, other MW treatment plants and other sewerage authorities;
- Urban and agricultural runoff;
- Atmospheric

ii) Bay capacity to handle incoming nutrient loads:
- Light and nutrient availability;
- Primary (plant) production and trophic status;
- Mixing and exchange with Bass Strait.

iii) Dispersion of Werribee Treatment Complex effluent:
- Werribee mixing zone and segment boundary;
- Nutrient levels
- Toxicants

iv) Dispersion of Yarra River and Mordialloc Creek.

v) Toxicants in Port Phillip Bay:
- Contamination of sediments and biota;
- Implications for long-term environmental health of the Bay.

vi) Impacts of shipping:
- Dredging and spoil disposal
- Antifouling paints.

These issues and elements of the Study are contained in the Objectives (Table 1.2) and discussed in detail in the Project Design (CSIRO, 1992).

Table 1.2 Objectives : Port Phillip Bay Environmental Study, 1992-6

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 Werribee Treatment Complex and major freshwater inputs such as the Yarra River, Werribee River and Mordialloc Creek.

Toxicants

iii) To determine the 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 Werribee Treatment Complex 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 Study is not an academic research program but is aimed at developing the "big picture", building on existing data and - from identification of additional scientific work needs - carrying out key tasks to fill the knowledge gaps. From this base the Study will provide the relevant scientific framework for strategic planning and development of management options and plans whose effectiveness can be evaluated through monitoring programs and adaptive management. The process of implementing the Project Design for the Study requires consultation between the scientific community and Bay managers in order to maintain a focussed direction and purpose, to make efficient and coordinated use of resources, and to combine existing skills and experience in the assessment and evaluation of results. The need to remove duplication of research effort is obvious.

This Status Review and Study Update acknowledges existing databases and other research activities. This is to ensure that the Study only pursues those tasks described in the Project Design which are essential to complement results of existing work and databases in order to achieve the Objectives.

This Report provides a series of reviews which identify the current scientific knowledge, understanding and hypotheses' for the key areas of physics, nutrients, toxicants and ecology of the Bay. The principles, components and management implications of integrated models for the Bay are discussed. Recommendations are given for the tasks (their scope and priority) which should be implemented in the Study.

The preparation of this Report has been assisted by the commissioned specialist reviews (Table 1.3) and the bibliography recently compiled by the Victorian Institute of Marine Sciences (VIMS, 1992a). Scientists from Melbourne Water, EPA, DCNR, Victorian universities and private sector organisations have contributed information and comment to the Study Management Group in its development of these reviews.

Table 1.3 Commissioned specialist reviews on Port Phillip Bay

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i) **Literature Review of the Physics of Port Phillip Bay**
K.P. Black and S. Mourtikas, VIMS

ii) **Literature Review of Physical and Chemical Atmospheric Inputs to Port Phillip Bay**
1. **Meteorological Aspects**. T. Beer, CSIRO Division of Atmospheric Research
2. **Aeolian and Atmospheric Inputs of Carbon Nitrogen and Phosphorus.** F. Carnovale, C. Carvalho and M.E. Cope, EPA Victoria
3. **Aeolian and Atmospheric Inputs of Toxicants.** F. Carnovale, C. Carvalho and M.E. Cope, EPA Victoria

iii) **Literature and Information Review of Groundwater Input of Nutrients and Toxicants to Port Phillip Bay**
Claus J. Otto, CSIRO Division of Water Resources.

iv) **Soft Bottom Macrobenthos of Port Phillip Bay : A Literature Review**
Gary C.B. Poore, Museum of Victoria

v) **The Present State of Knowledge on Commercial and Recreational Fisheries in Port Phillip Bay, and their Interactions with Nutrients and Toxicants**
Geoff Nicholson, Marine Science Laboratories, Victoria

vi) **Literature and Information Review of the Benthic Flora of Port Phillip Bay, Victoria, Australia**
Brett R. Light and W.J. Woelkerling, Melbourne Water and Latrobe University, Victoria

vii) **Phytoplankton Ecology of Port Phillip Bay, Victoria**
Mark Wood and John Beardall, Monash University, Victoria

viii) **Water Column Trophodynamics in Port Phillip Bay**
Christine Crawford, Greg Jenkins and Graham Edgar, VIMS

Chapter 2: Review of physical processes

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2.1 Introduction

This section describes the dominant physical processes that affect the waters of Port Phillip Bay and modelling techniques that may be used to simulate these processes. It is based primarily on the reviews of Beer *et al.* (1992), Black and Mourtikas (1992) and Otto (1992).

2.2 Physical Inputs

2.2.1 Atmospheric Inputs

2.2.1.1 Introduction

Physical atmospheric inputs have been reviewed by Beer *et al.* (1992) who indicated that there are currently 21 operational automatic weather stations in the vicinity of Port Phillip Bay (10 operated by the Bureau of Meteorology, 8 by the Environment Protection Authority of Victoria (EPAV) and 3 by the Port of Melbourne Authority). All of these stations record wind speed and direction and some of the following: air temperature, evaporation, rainfall and solar radiation. The sampling and averaging periods for these measurements cover a wide range (ten minutes to one hour) but, at some stations, observations may be recorded more frequently if required. It is expected that these stations will continue to operate during the Port Phillip Bay Environmental Study.

In addition, CSIRO and Melbourne Water operate an anemometer and water temperature sensor on the Hovell Pile tide gauge (near the southern coastline of the Bay). At present, only the anemometer is operational.

Since these stations are all located around the periphery of the Bay, it may be useful to install a further station near the centre of the Bay during the forthcoming field program (see also Sections 2.2.1.2, 2.2.1.4 and 2.2.1.5).

2.2.1.2 Wind speed and direction

As noted in Section 2.2.1.1, all of the 21 operational automatic weather stations record wind speed and direction. Some stations (i.e. those operated by EPAV) compute full vector averages of the instantaneously measured values, while others (e.g. Port of Melbourne Authority (PMA)) record speed averaged over ten minutes together with the instantaneous direction. For modelling purposes, full vector averaging is preferable.

The synoptic winds are variable. Beer *et al.* (1992) stated that, for Melbourne, which is `indicative of the synoptic pattern over the whole Bay', `in the mornings, northerlies prevail for most of the year except in summer when southerlies are the prevailing winds; in the afternoons southerlies predominate except in winter when the main winds are northerly'. However, Black and Mourtikas (1992) stated that `westerlies prevail in winter and south or south-westerlies in summer'. Inspection of the climatological wind roses for 15 stations around Port Phillip Bay (Beer *et al.* (1992), pp.4,5) indicate that the winds are quite variable in both space and time over the Bay, giving rise to the differing interpretations of the synoptic patterns that are noted above. Beer *et al.* (1992) noted that significant features of the synoptic winds are the passage of cold fronts and the occurrence of summer northerlies.

Sea breezes over the Bay are the result of both the temperature contrast between the land and the Bay (the `Bay breeze') and between the land and Bass Strait (the `ocean breeze'). They give rise to onshore and offshore winds, of magnitude typically 5 m s^{-1}, in a direction that is perpendicular to the `generating' coastline.

The wind is further modified by the Melbourne Eddy - a circulation pattern caused by the blocking of easterly and northeasterly winds by the Australian Alps - under stably stratified conditions. The eddy weakens during the day due to morning heating.

In summary the winds over Port Phillip Bay have significant spatial and temporal variations which are associated with a number of different processes: the large-scale synoptic pressure patterns (including those related to cold fronts and summer northerlies), sea breezes (both `Bay' and `ocean') and the Melbourne Eddy. Since the water circulation in the Bay is very dependent on the wind stress (see Section 2.3.1.2.2), any hydrodynamic modelling should include (at least in a statistical sense) all these features of the atmospheric forcing. The present distribution of automatic weather stations is probably adequate although, as noted in Section 2.2.1.1, the installation of a further station near the centre of the Bay would provide additional valuable information concerning the spatial structure of the winds. The sampling and averaging periods currently in use is also adequate, although those stations using full vector averaging would be preferred.

2.2.1.3 Rainfall

Rainfall is recorded at five automatic weather stations around Port Phillip Bay. Unfortunately these are clustered around the northern part of the Bay and near Geelong, so that the rainfall in the southeastern part of the Bay is not, at present, automatically recorded.

The rainfall increases substantially in a southeastward direction across Port Phillip Bay.
The mean annual rainfall figures at Laverton (to the northwest of the Bay) and Mornington (to the southeast of the Bay) are 566 mm and 735 mm respectively. For Melbourne, there is little difference between summer and winter rainfalls.

The catchments of the Maribyrnong and Werribee Rivers are in a marked rain shadow compared with that of the Yarra River, which therefore dominates the river flow to the Bay (Beer *et al.*, 1992).

If we assume a typical rainfall over the Bay (which has a surface area of 1.95 x 10^9 m^2 (MMBW/FWD, 1973)) of 650 mm y^{-1}, then the average freshwater input due to rain is 40.2 m^3 s^{-1}. This is comparable with other components of the water budget (see Section 2.2.3), so that the effect of rainfall over Port Phillip Bay must be considered in any water or buoyancy budget of the Bay.

The rainfall also shows a wide temporal variability. At Laverton, a single storm event produced a rainfall of 105 mm in 2 hours, equivalent to a flow rate of 28,000 m^3 s^{-1} if integrated over the surface area of the Bay.

For the purpose of evaluating the water or buoyancy budget of the Bay, it is possible to rely entirely on historical rainfall data. However, if these budgets are required for the duration of the field program, it may be necessary to install one or more automatic rain recorders in the region to the south-east of the Bay. In addition, consideration should be given to the installation of automatic rain recorders on beacons and/or on a meteorological buoy in the Bay, and to the possible use of weather radar images for the estimation of rainfall over the Bay.

2.2.1.4 Evaporation

Evaporation is recorded at only two automatic weather stations, which are in the region to the north of Port Phillip Bay.

Historical data show that the annual evaporation rate measured at Werribee, Melbourne and Aspendale is typically 1200 mm (Beer *et al.*, 1992) which is equivalent to a rate of loss of freshwater integrated over the surface area of the Bay (1.95 x 10^9 m^2) of 74 m^3 s^{-1}. This is almost as large as the sum of the other components of the water budget (91.0 m^3 s^{-1}, see Section 2.2.3), suggesting that the total input of freshwater is only about 17 m^3 s^{-1}.

However, estimations of the evaporation rate made over land, either by direct methods (e.g. an evaporation pan) or by `bulk' formulae (e.g. the Combination Method, Beer *et al.*, 1992), are generally overestimates of the evaporation rate over a nearby large body of water, since the relative humidity over land is normally lower than over water. The evaporation rate over Port Phillip Bay can only be evaluated accurately if measurements of relative humidity are made over the Bay itself. It is shown in Section 2.2.3 that estimates of the water budget (and hence the buoyancy budget) and flushing time of the Bay are critically dependent on the accurate determination of the evaporation rate.

It is hence concluded that determination of the water and buoyancy budgets of the Bay requires that the evaporation rate (which is a dominant component) be accurately determined. Beer *et al.* (1992) recommended that the Combination Method be used. This requires the installation of an automatic weather station on the Bay waters, recording wind speed, net radiation, air temperature, water temperature and humidity variables (although problems of fouling of the meteorological sensors should be considered). The only existing automatic weather stations on Bay waters (Fawkner Beacon, Point Wilson and South Channel) record only wind speed and direction.

2.2.1.5 Heat

The atmospheric input of heat, which influences the occurrence of density driven currents in Port Phillip Bay, may be estimated in conjunction with the determination of the evaporation rate (see Section 2.2.1.4). The heat flux through the water surface is made up of three components:

1. The net radiation, which is the resultant of short wave radiation from the sun and sky, and long wave radiation from the clouds (downwards) and the sea surface (upwards)
2. The turbulent (or `sensible') heat flux
3. The evaporative heat flux

Term (1) may be measured by a suitable radiometer (none of the existing automatic weather stations have one of these), or by using a combination of a short wave radiometer (which is carried on some of the existing automatic weather stations) and a `bulk' formulation for the long wave radiation, based on conventional meteorological observations.

Term (2) may be obtained from an appropriate `bulk' formulation, based on conventional meteorological observations.

Term (3) may be obtained directly from the evaporation rate, which, as noted in Section 2.2.1.4, would require the installation of an automatic weather station on the Bay waters recording wind speed, net radiation, air temperature, water temperature and humidity variables.

In summary the atmospheric input of heat may be derived from conventional meteorological observations, plus the determination of the evaporation rate based on measurements from an appropriately instrumented automatic weather station on the Bay waters. As noted in Section 2.2.1.4, problems of fouling of the meteorological sensors should be considered.

2.2.1.6 Photosynthetically available radiation

There is no literature dealing specifically with photosynthetically available radiation (PAR, the radiation of wavelength in the range 0.38 to 0.71 micrometres) over Port Phillip Bay (Beer *et al.*, 1992). PAR could be determined either by direct measurement (the preferred method) or by formulations based on the incoming solar radiation. It is recommended that a PAR sensor be installed on the proposed automatic weather station in the waters of the Bay, and that other PAR recorders be operated at a number of locations around the Bay.

2.2.1.7 Particulates

There are no direct measurements of particulate deposition to Port Phillip Bay. However, Beer *et al.* (1992) discussed the measurements of the concentrations of PM10 (particles of diameter less than 10 micrometres) and `total suspended particles' (TSP, particles of diameter less than about 25-50 micrometers) in the atmosphere, and the deposition rate of dust (particles of diameter greater than about 50-100 micrometers), in the Port Phillip Bay Region. Dust normally remains localised near (i.e. within about 100 metres of) its source, and so probably does not contribute a significant quantity of particulates to the Bay.

A rough estimate of the input of particulates to the Bay may be made from the measured particulate concentrations of TSP (typically 42 g m^{-2}, Beer *et al.*, 1992), the settling velocity (greater than 0.005 m s^{-1} for particles of diameter greater than 2 micrometres) and the surface area of the Bay (1.95 x 10^9 m^2), yielding a value somewhat greater than 0.4 kg s^{-1}.

2.2.2 **Water-borne inputs**

2.2.2.1 Introduction

This section reviews the inputs of water, heat, salt and particulates from Werribee Treatment Complex, rivers, land drainage and groundwater. The importance of these inputs are summarised as follows:

Water. This acts to dilute `tracers' such as salt, which are present in `oceanic' waters, allowing us to make estimates of the flushing time of the Bay. Freshwater is also buoyant with respect to salt water and so may drive density currents, as in an estuary.

Heat. The addition of heat reduces the density of water and hence can lead to density-driven currents. However, in this respect, the relative effects of heat and salt are such that 3^{o}C of temperature difference is equivalent to about 1 Practical Salinity Unit (PSU - a unit that is almost equivalent to parts per thousand of salt) of salinity difference. Therefore a freshwater inflow to a body of water of 35 PSU salinity, but the same temperature, is equivalent (in buoyancy terms) to a similar inflow of water of the same salinity, but raised in temperature by about 100^{o} C. The exact temperature of a freshwater inflow to the Bay hence has little influence on the hydrodynamics of the currents. It is hence recommended that only a few measurements of water temperature are required for freshwater inflows. However, it should be noted that where measurements of salinity are required for river inflows, these observations will normally be accompanied by temperature measurements.

Salt. As noted above, salt plays an important part in determining the density of water, and hence its buoyancy. The salinity is really only an important density consideration for river inflows which have a significantly estuarine (i.e. salty) character (whereas rivers that flow over a weir clearly have a negligible salinity compared with Bay waters). The salinity of groundwater is probably dynamically unimportant, due to its relatively small contribution to the total water inflow to the Bay (2%, see Table 2.1).

Particulates. Particulates act as carriers for nutrients and toxicants both at inflow points (i.e. they can transport nutrients and toxicants into the Bay) and within the Bay (i.e. they can pick up dissolved nutrients and toxicants and transport them elsewhere). They can be introduced by Werribee Treatment Complex, by rivers and by land drainage.

2.2.2.2 Werribee Treatment Complex

The inflow to the Werribee Treatment Complex is about 179,000 megalitres (ML) y^{-1} or 5.7 m^3 s^{-1}, using figures for 1988-89 (Bremner *et al.*, 1989). The present outflow of the complex is estimated to be 4.8 m^3 s^{-1}, based on inflow and outflow figures for 1970-71 (MMBW/FWD, 1973, p.65). This outflow passes to the Bay through five outlets (15E, 145W, Little River, Lake Borrie and Murtcaim) (MMBW, undated).

The discharge rate of particulates from Werribee has been monitored in the past on a monthly basis. In 1970-71 the rate was estimated to be 0.20 kg s^{-1} (MMBW/FWD, 1973, p.65).

It is recommended that Task P3 be implemented, in coordination with Task N1.1 and with Melbourne Waters' own observations, as described in the Project Design.

2.2.2.3 Rivers

The total river flow to the Bay is about 44.0 m^3 s^{-1} of which 28.3 m^3 s^{-1} is contributed by the Yarra River where it enters the Bay (22.7 m^3 s^{-1} coming from the Yarra River above the Maribyrnong River and 19.2 m^3 s^{-1} coming from the Yarra River above Merri Creek (MMBW/FWD, 1973, p.59)).

Beckett *et al.* (1982) described observations of water movement and salinity in the Yarra and Maribyrnong estuaries. They showed that:

- Under high flow conditions, the salt wedge in the Yarra extends from the Bay to King Street.

- Under low flow conditions, the salt wedge in the Yarra extends from the Bay to Bridge Road

- The Yarra may be classified as a salt wedge estuary and the Maribyrnong as a partially mixed estuary.

The plume of the Yarra and Maribyrnong estuaries into the Bay has also been investigated by Hinwood *et al.* (1979).

Observations of the plume of the Patterson River have been described by Caldwell Connell Engineers (1978).

Probably the best estimates of river-borne particulate input have been made by Lammerts van Bueren (1983, pp. 122 and 123). From a combination of direct measurements of suspended solids and continuous recording of optical transmittance over a total of 139 days of observations during 1981 and 1982 Lammerts van Bueren estimated a mean particulate flux for the Yarra River, above Merri Creek, of 1.71 kg s^{-1}. An estimate of the total river-borne load to the Bay may be made by scaling the above load by the ratio of the total river water inflow (44.0 m^3 s^{-1}) to the Yarra River inflow above Merri Creek (19.2 m^3 s^{-1}), yielding a value of 3.9 kg s^{-1} .

It is recommended that Task P4 be implemented as described in the Project Design. Site selection for the proposed survey(s) of the Yarra River should take into consideration the findings of Beckett *et al.* (1982).

2.2.2.4 Land drainage

There is little information on land drainage inflows to the Bay (Black and Mourtikas, 1992). EPA (1979) investigated the water, particulate, nutrient and toxicant fluxes in a number of rivers and drains discharging to the Bay, but no attempt was made to estimate total water or particulate fluxes to the Bay. Such inflows are believed to be `comparatively minor' (MMBW/FWD, 1973, p.58) and have been omitted from the water budget of Section 2.2.3. However, it is recommended that Task P5 be implemented, in coordination with Task N1.2, as described in the Project Design.

2.2.2.5 Groundwater

The total groundwater flow to Port Phillip Bay is `greater than 5.5 x 10^7 m^3 y^{-1}' (Otto, 1992) or >1.7 m^3 s^{-1}. We will assume an approximate value of 2 m^3 s^{-1}, which represents a contribution of about 2% of the total water inflow to the Bay (see Table 2.1). Otto (1992) provided a useful table (his Table 1) indicating the known discharge rates and concentrations of salt, nutrients and toxicants for the various aquifer systems adjoining the Bay. Although the groundwater salinities can cover a wide range (0.1 - 40 PSU), it is probable that such inflows are relatively unimportant dynamically due to their small contribution to the total water inflow to the Bay. It was proposed in the Project Design that Task P6 (a study of physical groundwater inputs) should be coordinated with Task N1.4 (a study of nutrient inputs *via* groundwater). It is recommended that, due to the relatively low importance of groundwater flows as regards the physics of the Bay, the P6 component be downgraded accordingly.

2.2.3 **The water budget**

Table 2.1 indicates estimates of the water budget for Port Phillip Bay, based on the figures given in Sections 2.2.1.3, 2.2.1.4, 2.2.2.2, 2.2.2.3 and 2.2.2.5.

Source	**Inflow (m^3 s^{-1})**
Rainfall	40.2
Evaporation	-74 ?
Werribee Treatment Complex	4.8
Total rivers	44.0
(Yarra River at Bay)	(28.3)
(Yarra River above Maribyrnong River)	(22.7)
(Yarra River above Merri Creek)	(19.2)
Land drainage	?
Groundwater	2
Total	**17.0 ?**

Table 2.1: Estimated water budget for Port Phillip Bay

The water budget may be used to estimate the flushing time of the Bay as follows. The salinity distribution in the Bay indicates that the salinity is typically 1 PSU below the `oceanic' value in Bass Strait (MMBW/FWD, 1973). Only in Corio Bay, where there is little river inflow, does the effect of evaporation appear to cause the salinity to be slightly higher (by about 0.5 PSU) than that in the adjacent portion of Port Phillip Bay. The flushing time may be estimated from:

Flushing time =
((volume of Bay)/(total water inflow)) x ((salinity depression)/(oceanic salinity))

which, for a Bay volume of 25 x 10^9 m^3 and an oceanic salinity of 35 PSU, yields a flushing time of 16 months, in good agreement with the following previous estimates:

1. 300-400 days, from the response time of a water quality model (MMBW/FWD, 1973, p. 213)

2. `A little more than one year', from the ratio:
   (total mass excess in the Bay)/(loading rate)
   for a conservative chemical, obtained from the results of a water quality model (MMBW/FWD, 1973, p. 213)

3. 15 months, from the ratio:
   (total mass excess in the Bay)/(loading rate)
   from orthophosphate measurements (Newell, 1990)

4. One year, from the results of a time-dependent single-box model describing the salinity in the Bay (Mickleson, 1990, p. IV-3, see below)

However, it should be noted that a small error in the evaporation rate (say, 10%) leads to a large error (44%) in the total water inflow and a correspondingly large error (44%) in the estimated flushing time. As noted in Section 2.2.1.4, the evaporation rate given in Table 2.1 is subject to some doubt, so that a better determination of this rate is required for water (and buoyancy) budgetting purposes.

Mickleson (1990) described a time-dependent single-box model of the salinity in the Bay. This included the effects of rainfall, evaporation, river inflow and the exchange with Bass Strait (described by a flushing time of one year). The model quite successfully predicted the salinity in the centre of the Bay over a period of 10 months in 1988/89.

2.2.4 **The particulate budget**

The budget of particulate matter in Port Phillip Bay is not well understood. An estimate of the atmospheric input of particulates may be made from Beer *et al.* (1992) (see Section 2.2.1.7). Lammerts van Bueren (1983) provided useful estimates for flux of suspended sediments in the Yarra over a full range of seasonal conditions during 1981 and 1982. However, there appears to be little other comparable information for other rivers or other times. Further, the input of particulates through land drainage is also poorly known (Black and Mourtikas, 1992). The input of particulates to the Bay from coastal erosion was discussed by Black and Mourtikas (1992), but no quantitative estimates of the particulate loads were given. It is hence recommended that a brief review of coastal erosion be conducted with a view to estimating the resultant particulate load to the Bay.

Table 2.2 indicates estimates of the particulate budget for Port Phillip Bay, based on the figures given in Sections 2.2.1.7, 2.2.2.2, 2.2.2.3.

Source	**Inflow (kg s^{-1})**
Atmospheric (estimated)	0.4
Werribee Treatment Complex	0.20 (1970-71)
Yarra River above Merri Creek	1.71 (1981-1982)
All rivers (estimated)	3.9
Land drainage	?
Coastal erosion	?

Table 2.2: Estimated particulate budget for Port Phillip Bay

2.3 **The physical environment of Port Phillip Bay**

2.3.1 **Water movements**

2.3.1.1 Tidal motions

Tidal motions in the Bay are well understood (Black and Mourtikas, 1992). The tides are semi-diurnal, driven predominantly by the `ocean' tide in Bass Strait and exhibit a significant reduction in amplitude and change of phase over the Sands area. Tidal elevation amplitudes range from 0.4 metres at the Heads to 0.2 metres at the north of the Bay. Tidal current amplitudes range from 0.8 m s^{-1} in the Sands region, through 0.05 m s^{-1} in the centre of the Bay, to 0.02 m s^{-1} in the northern part of the Bay (Rosenberg *et al.*, 1992a). A feature of the ebb flow from the Bay to Bass Strait is the development of a tidal jet that efficiently removes water-borne material from the Bay. This jet gave rise to the high exchange efficiency at the Heads (90%) estimated during the Phase 1 Study (MMBW/FWD, 1973). It should be noted, however, that due to restricted exchange of material over the Sands region this high exchange efficiency is not relevant to the overall flushing of the Bay (this is clear from the phosphate distribution observed during the Phase 1 Study, MMBW/FWD, 1973, p.120).

The sites of tide gauges in the Bay region are summarised in Table 2.3. There have been several studies of tidal currents in Port Phillip Bay (as part of the Phase 1 Study, MMBW/FWD, 1973; as part of an environmental assessment at Point Wilson, Caldwell Connell Engineers, 1981; by the Victorian Institute of Marine Sciences, e.g. Rosenberg *et al.*, 1992a; and by the Coastal Investigations Unit, Port of Melbourne Authority, unpublished).

Tidal elevations and currents are also particularly amenable to simulation by numerical modelling techniques; tidal elevations may be predicted well by two-dimensional models, while tidal currents are better simulated by full three-dimensional models (see Section 2.4.3).

It is recommended that, due to our good understanding of tidal motions and the existence of permanent tide gauges at the entrance and within the Bay, measurements of tidal elevation should not form part of Task P1. The measurement of tidal currents will, however, be a by-product of the current meter deployments during Task P1, which would be aimed mainly at investigations of non-tidal motions (see Section 2.3.1.2).

2.3.1.2 Non-tidal motions

2.3.1.2.1 Introduction

Motions in the Bay may be driven by the surface elevation at the entrance (which moves in response to tides (see Section 2.3.1.1), by meteorological forcing (e.g. coastally-trapped waves in Bass Strait, atmospheric pressure and the surface wind stress) and by horizontal density gradients in the water. Black and Mourtikas (1992) indicated, from a regression analysis, that the non-tidal surface elevation at Williamstown can be well explained by a combination of the effects of the surface elevation at the entrance (which may be measured by a suitably sited tide gauge), the surface wind stress and horizontal density gradients (i.e. the direct effect of atmospheric pressure changes is unimportant and only appears to be a driving force by virtue of its effect, through processes acting outside the Bay, on the surface elevation at the entrance).

2.3.1.2.2 Wind-driven currents

Wind-driven currents in the Bay have been both observed by current meters and modelled (Black and Mourtikas, 1992). The magnitude of such currents is typically 0.05 m s^{-1}. In common with the circulation of other comparable semi-enclosed basins they show two main features:

- Horizontal circulations consisting of topographic gyres, in which the current tends to be directed with the wind in shallow water and in opposition to the wind in deeper waters.

- An overturning of the water, in which the current tends to be directed with the wind near the surface and in opposition to the wind near the bottom.

Site	Period	Source
Williamstown (Victoria Dock)	Permanent (2 gauges)	Port of Melbourne Auth.
Williamstown (Breakwater Pier)	Permanent (2 gauges)	Port of Melbourne Auth.
Queenscliff	Permanent	Port of Melbourne Auth.
Hovell Pile	Permanent	Port of Melbourne Auth.
Lorne	Permanent	Port of Melbourne Auth.
Rip Bank	1991	Port of Melbourne Auth.
Point Lonsdale	Permanent	Nat. Tidal Fac. / P.M.A.
West Channel Pile	1973, now permanent	Nat. Tidal Fac. / P.M.A.
Geelong	Permanent	National Tidal Facility
Williamstown	Permanent	National Tidal Facility
Saint Kilda	1975	National Tidal Facility
Altona	1975	National Tidal Facility
Werribee	1976, 1977	National Tidal Facility
Sandringham	1975	National Tidal Facility
Mordialloc	1975 - 1986	National Tidal Facility
Richards Channel	1976, 1977	National Tidal Facility
Point Richards	1975	National Tidal Facility
Wilson Spit	1976, 1977	National Tidal Facility
Saint Leonards	1976, 1977	National Tidal Facility
Mornington	1976, 1977	National Tidal Facility
Barwon Heads	1975 - 1977	National Tidal Facility
Nepean Bank	1983, 1984	National Tidal Facility
West Channel Wedge	1973	National Tidal Facility
Rip Bank	1983, 1984	National Tidal Facility
South Channel	1964, 1965	National Tidal Facility
Rosebud Jetty	1974, 1975	National Tidal Facility
23 temporary deployments	1968	MMBW/FWD, 1973
9 temporary deployments	1978 - 1989	VIMS

Table 2.3: Sites of tide gauges in the Port Phillip Bay region

Hunter and Hearn (1987) derived a criterion, based on the geometry of the basin, for determining the relative importance of the above circulation features. It is recommended that this criterion be applied to Port Phillip Bay to serve as a guide to the type of circulation and dispersion modelling that should be applied in Task P8.

Our understanding of the three-dimensional structure of the wind-driven currents in the Bay is poor (Black and Mourtikas, 1992). It is recommended that current meter observations are made in accordance with Task P1 in the Project Design, and that the resultant data is integrated with previous current observations and three-dimensional hydrodynamic modelling (see Section 2.4.3). The use of alternative methods of current measurement, such as the acoustic doppler current profiler (ADCP) or coastal radar, should be considered.

2.3.1.2.3 Density-driven currents

Density-driven currents have received little attention in Port Phillip Bay. However, horizontal salinity gradients as high as 0.15 PSU km^{-1} (equivalent to a horizontal density gradient of 0.11 kg m^{-3} km^{-1}) are associated with the Yarra plume as it passes along the north-east shore of the Bay (MMBW/FWD, 1973). It may be shown that this has approximately the same forcing effect as a 2 m s^{-1} wind blowing over the Bay. Density-driven currents can therefore not be neglected.

It is therefore recommended that surveys of temperature and salinity be implemented in accordance with the description of Task P1 in the Project Design and that the resultant data be integrated with previous observations.

2.3.2 **Salinity and temperature**

2.3.2.1 Vertical structure

Vertical stratification of the water column has been reported by Rochford (1966), MMBW/FWD (1973), Mickelson (1990) and Longmore *et al*. (1990). Temperature stratification from surface heating during the summer occurs mainly in the centre of the Bay. The difference between the surface and bottom temperature is generally less than 3^{o}C.

Mickelson (1990) described a simple empirical model for the prediction of thermal stratification, based on the ambient air temperature. The model was successful at predicting the times of strong stratification, but not its magnitude. A better way in which to predict the occurrence of thermal stratification is to model the balance of surface heating and water column mixing, thereby yielding the following criterion for the onset of stratification (Simpson and Bowers, 1981):

$$Q > q \qquad (1)$$

$$q = 11{,}200\ u^3/h + 4.87\ W^3/h \qquad (2)$$

where Q (watts m^{-2}) is the rate of surface heating, u (m s^{-1}) is the tidal stream amplitude near the seabed (or, alternatively, a typical wind-driven current if this is stronger), W (m s^{-1}) is the wind speed and h (m) is the water depth. For Port Phillip Bay during the summer heating period, Q is approximately 100 watts m^{-2} (Oberhuber, 1988).

Table 2.4 shows typical values for the Sands region and for the middle and north of the Bay, indicating the potential for thermal stratification in summer in the middle and north of the Bay (and other parts of the Bay with similar values of u and h).

Variable	Sands	Middle	North
h (m)	15	20	15
u (m s^{-1})	0.8 (tidal)	0.05 (tidal/wind-driven)	0.05 (wind-driven)
W (m s^{-1})	5	5	5
q (watts m^{-2})	423	30.5	40.7

Table 2.4: Values used for predicting thermal stratification in Port Phillip Bay

Salinity stratification, which occurs at various times during the year, is associated with freshwater inflow (the main effect being due to the Yarra plume which generally stretches around the north-east of the Bay) and with rainfall. Salinity stratification is weak in the centre of the Bay. In the northern part of the Bay the salinity variation over the depth may be as large as 6 PSU (equivalent, in density terms, to about 17°C), while in the centre it is generally less than 1 PSU (equivalent, in density terms, to about 3°C) (Rochford, 1966).

Vertical turbulent mixing, an important component of any three-dimensional transport model of the Bay, is inhibited (by up to two orders of magnitude) by the presence of vertical density stratification. The vertical stratification in the Bay should therefore be adequately described and understood.

2.3.2.2 Horizontal structure and fronts

The horizontal distribution of salinity and temperature has been described by MMBW/FWD (1973) and Longmore *et al.* (1990).

Horizontal temperature gradients may be generated by differential heating due to the variable bathymetry of the Bay and by temperature difference between the Bay and Bass Strait (the Bay being colder in winter and warmer in summer than Bass Strait). Horizontal temperature gradients may be as large as 0.2°C km^{-1} (Longmore *et al.*, 1990), equivalent to a horizontal density gradient of about 0.05 kg m^{-3} km^{-1}, which, as indicated in Section 2.3.1.2.3, would be associated with appreciable currents.

Horizontal salinity gradients may be generated by freshwater inflow, rainfall or evaporation. As noted in Section 2.3.1.2.3, the Yarra plume is associated with horizontal salinity gradients as high as 0.15 PSU km^{-1} (equivalent to a horizontal density gradient of 0.11 kg m^{-3} km^{-1}), which has approximately the same forcing effect as a 2 m s^{-1} wind blowing over the Bay. The salinity in Corio Bay is generally significantly higher than the salinity in the remainder of Port Phillip Bay (by typically 0.5 PSU, MMBW/FWD, 1973; Longmore *et al*.,1990), due to evaporation dominating the water budget in this region. The associated horizontal salinity gradients appear to be less than those in the Yarra plume (MMBW/FWD, 1973) but still imply a small net flow into Corio Bay.

It is recommended (as in Section 2.3.1.2.3) that surveys of temperature and salinity be implemented in accordance with the description of Task P1 in the Project Design and that the resultant data be integrated with previous observations.

2.3.3 Mixing and flushing

2.3.3.1 Introduction

There have been few quantitative observations of mixing in Port Phillip Bay. There have been several investigations of the mixing of the effluent plume in the immediate vicinity (i.e. within about 4 km) of the Werribee Treatment Complex (e.g. Hickman, 1977; Pearson, 1977, 1978a, 1978b; and Pattiaratchi *et al.*, 1989). The experiments involved water sampling (e.g. Pearson, 1977), dye (Pearson, 1978a), `black liquor' (a fluorescent effluent from the paper pulping industry, discharged from Werribee) (Hickman, 1977) and oceanographic and turbulence measurements (Pattiaratchi *et al*., 1989). The results of these investigations have been summarised by Black and Mourtikas (1992). However, only Pattiaratchi *et al.* (1989) attempted to quantify the observations in terms of mixing coefficients and the results are only applicable to the region close to the Werribee Treatment Complex.

2.3.3.2 Vertical mixing

The only measurements of vertical mixing in the Bay appear to be those by Pearson (1978a) and Pattiaratchi *et al.* (1989), who made observations in the effluent plume from the Werribee Treatment Complex.

Pearson (1978a) concluded that vertical mixing of the plume was mimimal for winds less than 3 m s^{-1}.

Pattiaratchi *et al.* (1989) concluded that complete vertical mixing of the plume was accomplished under wind speeds of greater than 5.5 m s^{-1}. For wind speeds less than this the plume formed multiple horizontal fronts, extending up to 4 km into the Bay. They derived an expression for the vertical mixing time as a function of the wind speed.

2.3.3.3 Horizontal mixing

There have been few direct measurements of horizontal mixing in Port Phillip Bay.

MMBW/FWD (1973, p.265) described a numerical hydrodynamic / water quality model in which horizontal mixing was parameterised by a `beta coefficient' which was adjusted for optimal fit between observations and model predictions. However, it is not clear how the `beta coefficient' (which decreases with increased mixing) relates to the horizontal mixing coefficient.

In the region of the effluent plume from the Werribee Treatment Complex, Pattiaratchi *et al.* (1989) estimated the horizontal mixing coefficient to be about 5 $m^2 s^{-1}$, based on a simple box model and energy dissipation arguments. However they noted that such estimates `would only be accurate to within an order of magnitude'. They also provided an approximate relationship relating the horizontal mixing coefficient to the wind speed.

Caldwell Connell Engineers (1981) estimated the horizontal mixing coefficient under `a moderate cross wind' in the vicinity of Point Wilson to be 0.8 $m^2 s^{-1}$, based on a dye release experiment carried out on one day. Under lighter winds they estimated the mixing coefficient to be 0.1 - 0.2 $m^2 s^{-1}$.

2.3.3.4 Exchange through the Heads

MMBW/FWD (1973, p.96) described estimates of the efficiency of tidal exchange at the Heads based on the number of drogued floats, released into the ebb tidal current on a line between Point Lonsdale and Point Nepean, that returned on the flood tide. Under northerly wind conditions the exchange ratio (i.e. the proportion of drogues lost to Bass Strait during a tidal cycle) was found to be 0.9. Under westerly and southerly winds the exchange ratios were estimated to be 0.75 and 0.5 respectively. However, since these experiments involved the tracking of drogues that were set at a constant depth, they may not be applicable to dissolved and suspended matter that is subject to vertical mixing.

A simple estimation of the exchange ratio may be made by comparing the estimates of the flushing time (about 400 days, see Section 2.2.3) with the ratio of the Bay volume (25 x 10^9 m^3) to the tidal volume (0.94 x 10^9 m^3) (MMBW/FWD, 1973, p.80), which is 27 tidal cycles, or 14 days. These figures indicate an exchange ratio of 14/400 or 0.035. The large discrepancy between this value and the estimates from the drogue studies is the result of the limited exchange over the Sands region (as discussed in Section 2.3.1.1).

2.3.3.5 Summary

Since there is little existing information on vertical and horizontal mixing and on exchange through the Heads, it is recommended that Task P2 be implemented as described in the Project Design, with the exception of the wave measurements (since, as will be described in Section 2.3.4, the wave climate of the Bay is reasonably well understood).

2.3.4 **Wave climate**

2.3.4.1 Surface elevations

Measurements of the surface elevation due to waves have been made in Port Phillip Bay using Waverider buoys (Marine Models Laboratory, 1977, Caldwell Connell Engineers, 1981). Other data sources have been indicated by Black and Mourtikas (1992). However, Black and Mourtikas (1992) indicate that Waverider data tends to underestimate the short period waves (i.e. those of period less tahn about 2 seconds) that may be present in the Bay.

Wilson (1982) tested three empirical models of wind wave generation in Port Phillip Bay. He found that one of them provided reliable forecasts of wave characteristics (height and period). Inputs to this model are wind speed and direction, and the water depth defined at each point on a horizontal computational grid.

Falconer (1972) predicted the waves at the Heads using a model of refraction, shoaling, and bottom friction and percolation. The results were not verified using wave observations.

2.3.4.2 Bed orbital motion

Black and Mourtikas (1992) described an extension of the work of Wilson (1982) to the prediction of sea bed orbital velocities. Although little detail was given, it appears that the technique provides results that compare well with measured orbital velocities. It is hoped that such a method can be used to provide wave data for input to future hydrodynamic and sediment transport models of the Bay (see Section 2.4).

2.3.4.3 Summary

Although there exists a considerable body of wave observations in Port Phillip Bay and that existing wave models yield apparently satisfactory results, Black and Mourtikas (1992) have indicated that further measurements of surface elevations due to waves may be justified, using instruments other than Waverider buoys (although the contention of Black and Mourtikas (1992), that Waverider buoys may be unsuitable in the Bay, is debatable). They also advised that further measurements of sea bed orbital velocity be made.

2.3.5 **Sediments**

2.3.5.1 Introduction

It was shown in Section 2.2.4 that the particulate budget of Port Phillip Bay is poorly understood. It is also known that the transport of nutrients and toxicants is closely related to the transport of particulate matter. A better understanding of the transport of both suspended and bed sediment is therefore required. Further, the transport of nutrients and toxicants *within* the seabed is also important, requiring the determination of sediment porosity.

2.3.5.2 Sea bed morphology and sediment description

Black and Mourtikas (1992) have reviewed a significant body of literature concerning observations of the sea bed in Port Phillip Bay (e.g. Beasley, 1966; Link, 1966; Caldwell Connell Enginers, 1981; Buckley, 1987; VIMS, 1992b). However, they indicated that due to continuing deposition of riverine material and the disturbance of the bed by scallop dredging, further sea bed surveys may be warranted. They also suggested that fall velocity analyses, rather than sieving (as used by Beasley (1966)), should be employed in future studies.

There appear to have been no measurements of sediment porosity.

2.3.5.3 Suspended sediment and turbidity

Measurements of suspended sediment have been made throughout Port Phillip Bay by water sampling and filtering (e.g. MMBW/FWD, 1973, p.134; Longmore *et al.*, 1990), by Secchi Disc (e.g. Longmore *et al*., 1990) and in the vicinity of Portarlington and St Leonards by optical backscatter (Rosenberg *et al.,* 1992b).

Black and Mourtikas (1992) recommended that long-term measurements of suspended sediment be made, in order to cover storm conditions. Such measurements would be made by moored instruments involving either optical backscatter (e.g. Rosenberg *et al.*, 1992b) or optical transmission. However, the Project Design recommended the alternative strategy of vessel-based surveys, some being organised on a reactive basis to coincide with significant events such as high river flows or storms. Consideration should therefore be given to the best choice of experiment type, with due regard to existing data. Consideration should also be given to the use of the reflective signal strength from an acoustic doppler current profiler as an indicator of the suspended sediment concentration.

2.4 **Modelling**

2.4.1 **Introduction**

Black and Mourtikas (1992) summarised models of Port Phillip Bay.

2.4.2 **Simple models**

Simple box models may be used to describe `global' budgets for Port Phillip Bay (e.g. Section 2.2.3, MMBW/FWD, 1973, p.213; Mickleson, 1990, p.IV-3; Newell, 1990). Such models provide valuable insight into the dominant processes that are active in the Bay and provide the basis for the simplest management tools.

2.4.3 **Two-dimensional and three-dimensional hydrodynamic models**

Hydrodynamic models of Port Phillip Bay have been developed by MMBW/FWD (1973), Williams (1978) and by the Victorian Institute of Marine Sciences (e.g. Black *et al.,* 1990; Black, 1992; Black *et al*., 1992). A model of Corio Bay was described by Easton (1978).

The majority of this modelling is two-dimensional, describing only the vertically-averaged current velocity. Such two-dimensional models predict tidal elevations well, tidal currents rather less well and wind-driven currents quite poorly (especially in basins, such as Port Phillip Bay, which have a relatively flat bottom (Hunter and Hearn, 1987)). Two-dimensional models are inappropriate for the simulation of density-driven currents.

The models of the Victorian Institute of Marine Sciences include nesting of smaller grids inside larger ones, but it is unclear whether the interaction between the different grids is uni- or bi-directional (i.e. whether the smaller grids influence the larger ones, as well as the larger grids influencing the smaller ones). The three-dimensional hydrodynamic model described by Black (1992) and Black *et al.* (1992) was uncalibrated and did not involve variable density effects.

There has apparently been no modelling of density-driven currents in Port Phillip Bay.

2.4.4 Dispersion models

Dispersion models of Port Phillip Bay have been developed by MMBW/FWD (1973, p.257), Williams (1978), Black (1992) and Black *et al.* (1992). The model of MMBW/FWD used a differencing technique on a network of triangular elements, while the models of Williams, Black and Black *et al*. used particle-tracking methods. Williams employed particle-tracking in an interesting technique that effectively interfaced the hydrodynamic finite-difference model with a larger scale box model. Only the (uncalibrated) simulations of Black and Black *et al.* resorted to three-dimensional modelling, thereby indicating the importance of the shear diffusion mechanism in horizontal dispersion.

2.4.5 Sediment transport models

There are no published models of sediment transport in Port Phillip Bay.

It should be noted that, in addition to modelling of the transport of both suspended and bed sediment, simulation of the transport of water - and associated nutrients and toxicants - within the bed is required.

2.4.6 Summary

Hydrodynamic modelling of the Bay should be mainly three-dimensional in order that wind- and density-driven currents be adequately modelled.

Dispersion and sediment transport modelling should be three-dimensional in order to fully capture the essentials of the hydrodynamics and to satisfactorily simulate processes such as shear diffusion.

Some form of wave modelling should be incorporated in order to adequately describe bed friction and sediment resuspension.

The models MUST be capable of being readily interfaced with the integrated models of Task G8/G9.

2.5 Knowledge gaps and research needs

- Consideration should be given to the installation of a further automatic weather stations in and around the Bay, to provide further information concerning the winds, rainfall and evaporation rate. The use of weather radar for rainfall estimation should also be considered.

- PAR recorders should be operated at a number of locations around the Bay.

- Further measurements of tidal elevation should not form part of the Study.

- Some further observations of currents, temperature and salinity are required.

- Estimations of mixing and flushing are required.

- Measurements of physical inputs from WTC, other effluent discharges, rivers and land drainage are required.

- The physical effects of groundwater inputs is relatively unimportant.

- Consideration should be given to the best choice of experiment type for the measurement of suspended sediment (e.g. based on moored instruments or vessel surveys; the use of an ADCP).

- Any hydrodynamic modelling should include (at least in a statistical sense) the following features of the atmospheric forcing: the large-scale synoptic pressure patterns (including those related t cold fronts and summer northerlies), sea breezes (both 'Bay' and 'ocean') and the Melbourne Eddy.

- Hydrodynamic modelling of the Bay should be mainly three-dimensional.

- Dispersion and sediment transport modelling should be three-dimensional.

- Some form of wave modelling should be incorporated in the hydrodynamic and sediment transport models.

- The models should be capable of being readilly interfaced with the integrated models to be developed for the Study.

2.6 **Coastal erosion**

It is recommended that a brief review of coastal erosion be conducted with a view to estimating the resultant particulate load to the Bay.

Chapter 3. Nutrients

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3.1 Introduction

Human activities in coastal catchments may greatly influence the load and quality of nutrients to coastal lakes, bays and estuaries. This may lead to eutrophication - a process of nutrient enrichment of water leading to enhanced organic production.

The effects of human activity may be beneficial or detrimental. Enhanced nutrient loads may stimulate both primary and secondary production, resulting in increased biomasses of fish and bird life. On the other hand elevated nutrient loads and retention of nutrients within an embayment can lead to hypertrophication and undesirable effects such as nuisance phytoplankton and toxic algal blooms, pollution of benthic habitats, the degradation of fisheries resources and degraded water quality. Significant socioeconomic decline may occur through depressed tourism and local property values.

Clearly a balance is required between the beneficial effects of nutrient input on the increased productivity of an embayment and the detrimental effects of accelerated eutrophication. Managers continually seek to maintain this balance through manipulation of the trophic status of an embayment by regulation and control of point discharges and human and animal activities in the catchments.

The major nutrients contributing to eutrophication are the chemical forms of nitrogen and phosphorus available to aquatic phototrophs. Under some circumstances other essential nutrients such as silica, iron and micronutrients may have important local effects. The quantities of nitrogen and phosphorus (and other nutrients) which become available to the phototrophic synthesisers depend on the concentrations and loads of inputs, the rates of transformations between dissolved and inorganic and organic particulate forms, the sedimentation rate and total losses such as export to the ocean or atmosphere. The trophic balance of marine ecosystems is a response of the phototrophs and consumers to changes in nutrient availability. Factors which control trophic balance may operate over different scales of time and space in different zones of a marine environment.

Port Phillip Bay, a relatively shallow embayment of 1950 km^2, receives nutrients from a total area of 12,000 km^2 which includes the greater Melbourne area (Figure 3.1.1). Nutrients are input from the atmosphere and a variety of water sources, such as the major river systems and drains, industrial and sewerage discharges (notably Werribee Treatment Complex), agricultural lands and groundwater. A comprehensive study of the Bay some 20 years ago took account of the majority of the inputs at that time (MMBW/FWD, 1973). Since then there have been major changes in catchment activities and development, and the regulation of key industrial point discharges including the Werribee Treatment Complex (WTC). The quality, scale and period of nutrient inputs are in urgent need of reassessment in order to provide managers with information to assess the effects of earlier control measures and to develop further options for management of the Bay and its catchment.

The dynamic processes which determine the fate and distribution of nutrients in the Bay have been major research projects for several scientific studies over the last two decades. The comprehensive Phase 1 Report of the Port Phillip Bay Study (MMBW/FWD, 1973) described nutrient distributions in the water column and produced a water quality model which laid the foundation for the management of the water quality of the Bay. Subsequent work provided valuable information on the recycling and transformation of nutrients in the Werribee Zone. This work, together with that of Phase 1, established the notion that the Bay is nitrogen limited and probably oligotrophic. However, the question remains: Where do the input nutrients go? Atmospheric loss of N_2 due to denitrification has been estimated to be in the order of 10-15% (Bauld and Millis, 1977; Axelrad *et al.*, 1981) and Newell (1990) attributed nutrient balance in the Bay to the export of dissolved organic nitrogen to Bass Strait. These estimates were based on limited scientific information and assumptions, some of which are testable. There is an urgent need for new and comprehensive information which includes the effects of season and climate on nutrient distribution, dynamics and distribution in the water column and sediments and of biomass standing crops. The gathering of this information and subsequent analysis and modelling must be based on sound scientific principles.

This review summarises the past and present distribution of nutrients in the waters and sediments of the major environmental zones of the Bay. We also re-examine the major reviews and publications of the last two decades with particular attention to nutrient budgets and dynamics for the Bay-wide situation. We have also reviewed nutrients in the Werribee Zone and Hobsons Bay as being representative of the situations in which nearshore environments are affected by sewage effluent and catchment run-off respectively. We have also used this review to identify the serious gaps in our knowledge and understanding of nutrient dynamics in Port Phillip Bay. One of the prime objectives of the present study (Phase 3) is to provide the information and scientific principles essential for the effective management of nutrient budgets and dynamic processes in Port Phillip Bay.

3.2 Inputs

3.2.1 Streams and drains, rural and urban runoff

The Port Phillip Bay catchment is characterised by having several major river systems and a number of other much smaller surface natural input sources (Table 3.2.1, Figure 3.1.1). In addition there are more than 300 drains discharging directly to the Bay.

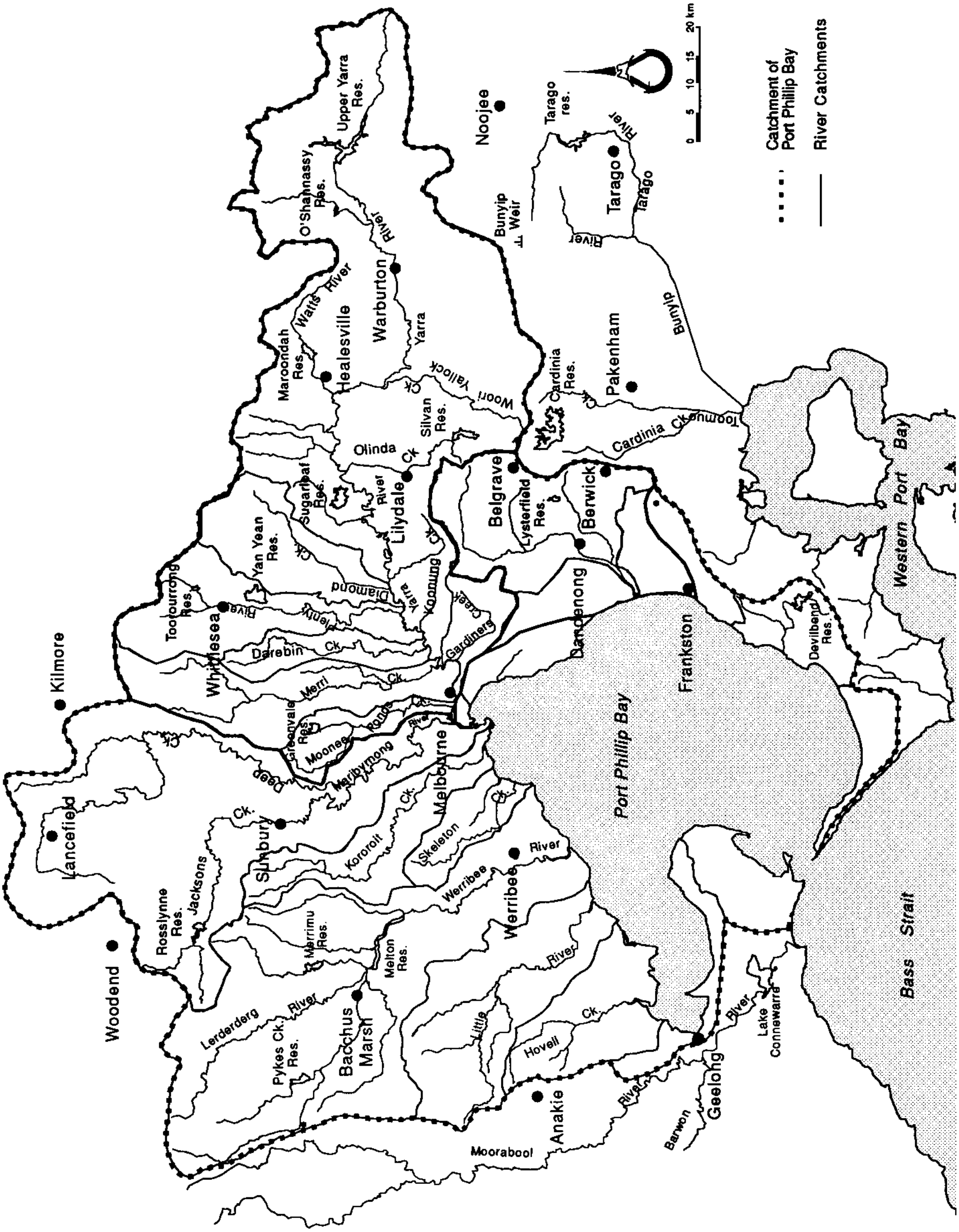

Figure 3.1.1 The Catchment of Port Phillip Bay: (from EPA, 1991a)

The first quantification of nutrient loads to the Bay from surface sources was made during the Phase 1 Study (MMBW/FWD, 1973). Estimates of the average annual loads to the Bay were determined for the Yarra River, for typical tributary areas representing specific land use, sewerage and drainage practices, and for intensively irrigated agricultural areas. From these data units annual loads were calculated for sullage, trade wastes discharged to stormwater drains, urban runoff and agricultural runoff. Unit area loads were then applied to land use to calculate total annual loads for each individual catchment draining to the Bay (MMBW/FWD 1973). Estimates of 7500 tonnes of nitrogen, and 1700 tonnes of phosphorus per year were obtained.

Table 3.2.1 Average annual inflows from streams to Port Phillip Bay (Inflows from river flows from data in MMBW/FWD (1973)).

	CATCHMENT	Inflow (ML) x 10^3	Percent of total flow
1.	South Bellarine	16	1.2
2.	North Bellarine	10	0.7
3.	Geelong	5	0.4
4.	Hovell Ck	25	1.8
5	Little River	43	3.1
6,	Werribee River	157	11.4
7.	Skeleton Ck	13	0.9
8.	Kororoit Ck	36	2.6
9.	Yarra River		
10.	Maribyrnong	176	12.8
11.	Moonee Ponds Ck	21	1.5
12.	Merri Ck	55	4.0
13.	Darebin Ck	22	1.6
14.	Plenty	44	3.2
15	Diamond Ck	41	3.0
16	Gardiners Ck	33	2.4
	Remainder	495	36.0
	Total Yarra Catchment	887	64.5
17.	North-east coast	51	3.7
18	Paterson River	73	5.3
19	Kananook Ck	26	1.9
20	Mt. Martha	21	1.6
21	Nepean Peninsula	11	0.8
	Total	1374	
WTC 1969-70		108	
WTC 1990-91		200	

Blutstein *et al.* (1982) subsequently estimated the nutrient loads to the Bay from data collected between 1979 - 1981. A total of 146 inputs from within 15 input strata were sampled randomly at a frequency of up to 3 times per week for some stations. The loads were averaged over each year of the study for the inputs sampled within each strata. Their estimates of total nitrogen and phosphorus loads to the Bay were very similar to those of the Phase 1 Study some ten years earlier, being 7510 tonnes of nitrogen and 1536 tonnes of phosphorus per year. These figures were achieved in spite of the fact that some 40 per cent of Melbournes sewage (approximately 2000 tonne nitrogen) had been diverted from the Bay catchment with the commissioning of the Carrum Treatment Complex in 1975.

Peck and Leticq (1991) calculated loads to the Bay using the same methodology as the Phase 1 Study but using 1989-90 distributions of land-use. They obtained a total nitrogen load of 7764 tonnes per year. This figure was also very similar to that obtained 20 years earlier.

Over the 20 year period considerable land-use changes have taken place within the catchment (e.g. extensive urban expansion, the termination or treatment of industrial waste discharges or their diversion to the sewerege system). The impact of these changes was reflected in Peck and Leticq's data. Over the period from 1971 to 1991 their figures show a substantial decrease in nitrogen loads resulting from sullage and trade waste discharges to storm water, whereas increases had occured in discharges from regional sewerage authorities, urban run-off and atmospheric sources.

One possible implication of these changes is that, although the load overall may not have changed, the shift to increased nitrogen from urban run off and inputs to eastern streams by local sewerage authorities means an increased load to the northern and eastern sides of the Bay. Increased nitrogen loads to these areas of the Bay occur during winter months when peak riverine and drainage flows occur (Figure 3.2.1.1). This situation is in sharp contrast to the more consistent loads of nutrients delivered throughout the year on the north-western side of the Bay adjacent to the WTC (see below).

3.2.2 **Sediments**

Figure 3.2.2.1 is taken from the Phase 1 Report (MMBW/FWD, 1973, Fig 9-15) and it shows the distribution of sediment types in the Bay. Until recently little was known about the nutrient pool sizes in sediments of the Bay and less about nutrient fluxes at the sediment/water interface. Bauld and Millis (1977) describe several sediment ammonia profiles to 20 cm below the surface at sites in a WTC effluent drain and extending 100 m into the Bay. In the Bay porewaters the ammonia concentrations were between 30 and 50 g N L^{-1}, peaking at around 10 cm. Even at 20-25 cm deep ammonia concentrations were around 20 g N L^{-1}. Newell (1990) reviewed these data and other data (Stanley, 1977) and concluded that the sediments of the Bay were generally low in nutrient and organic content, reflecting a low net productivity in the water column and a slow sedimentation rate. The notion of nutrient-lean sediments had a significant bearing on the Bay-wide nutrient dynamics model which, he suggested, was largely indicative of an oligotrophic environment. However, the more recent work of Gorfine and Abbott (unpublished) and Longmore (unpublished) on the chemical and biological characteristics of nearshore and Central Zone sediments show that the surficial sediments in Corio Bay, Hobsons Bay and the Central Zone have contents of 3% to 5% organic carbon. They also showed that all sediment types efflux inorganic nitrogen to the water column. The following section provides the details of this important work.

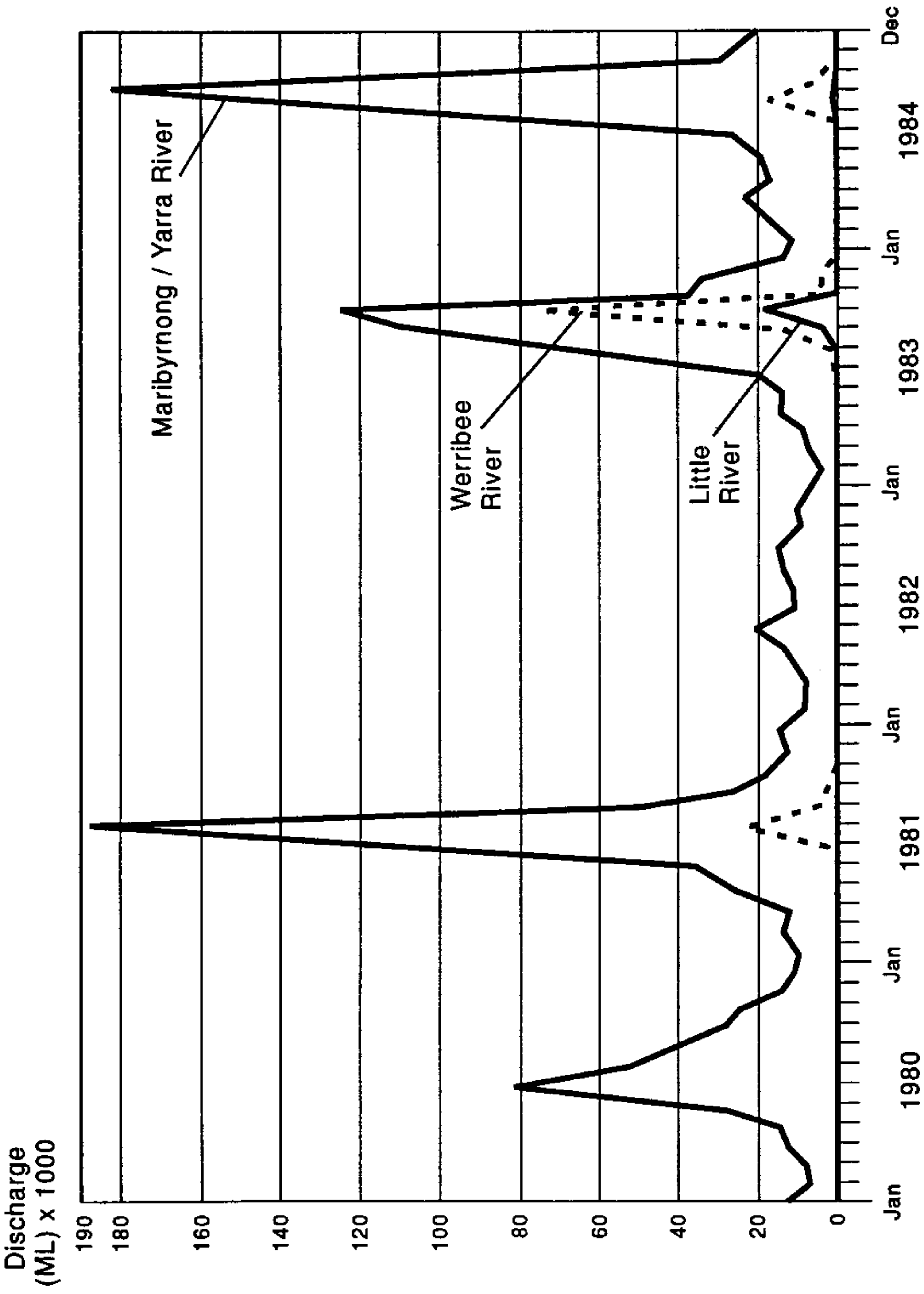

Figure 3.2.1.1 Monthly discharge of major streams to Port Phillip Bay 1980 to 1984 (from Brown and Davies, 1991)

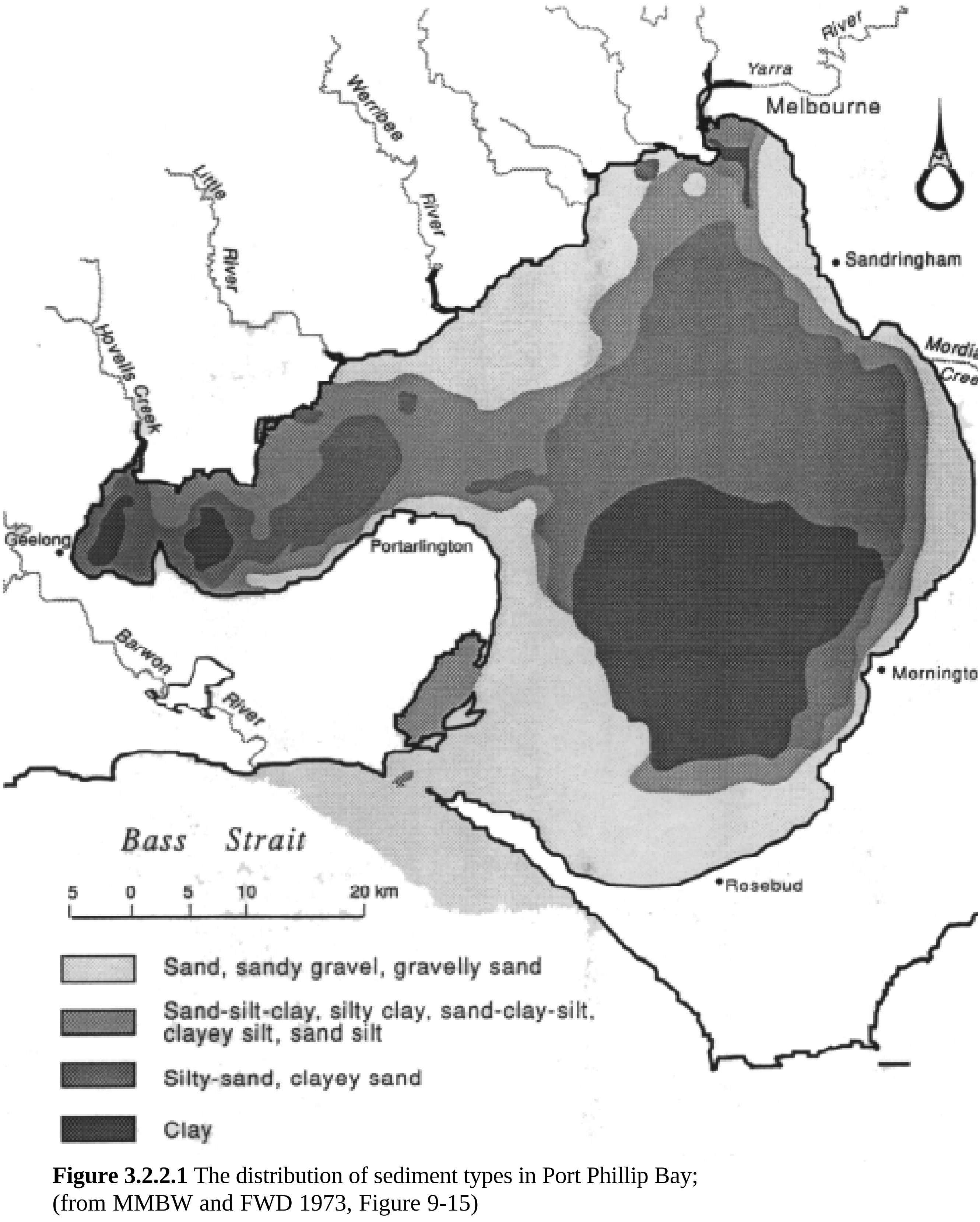

Figure 3.2.2.1 The distribution of sediment types in Port Phillip Bay; (from MMBW and FWD 1973, Figure 9-15)

Bacterial oxidation of organic matter on the Bay floor leads to the recycling of nutrients from organic particulate to dissolved aqueous forms. The chemical species returned to the water column depend on the bacterial pathways involved (see section 3.4). Estimates of nutrient flux from Port Phillip Bay sediments are available from two unpublished preliminary studies.

(i) Gorfine and Abbott: Triplicate sediment cores were collected from 39 sites throughout the Bay on one occasion (August-September 1991) and were incubated. Net nutrient flux across the sediment/water interface was estimated by the change in concentration of inorganic nutrient (PO_4, NH_4, NO_2, NO_3, SiO_4) in the overlying water during incubation over four days. Table 3.2.2.1 lists simple summary statistics for these fluxes (a positive flux indicates movement out of the sediment). During the sampling period there was a net loss of phosphate from the water column and a net release of ammonium, nitrate (plus nitrite) and silicate to the water column.

The dissolved inorganic nitrogen flux was similar from both nearshore and deep central sites. Ammonium release accounted for more than 80% of the nearshore flux, while nitrate accounted for a large proportion (32-72, average 40%) of the inorganic nitrogen released from deep central Bay sites. Sediment from all nearshore sites took up phosphate, while those from the deeper central Bay released phosphate. No other areas of uniform flux were detected; fluxes of nitrate, ammonium and phosphate in Hobsons Bay (two sites), Corio Bay (three sites) and off the WTC (three sites) were indistinguishable from those at other nearshore sites. Silicate fluxes reflected the sediment particle size rather than distance from shore; highest silicate fluxes occurred at sites of small particle size. There is no apparent explanation for this, except that the surficial sediments in these fine deposits are more likely to be anaerobic; also the fine sediments are likely to contain a high percentage of terrestrial silica.

Table 3.2.2.1. Summary statistics for nutrient fluxes from PPB cores, 1991. All units mg m^{-2} d^{-1}; data normally distributed (from Longmore, 1992).

Nutrient	PO_4-P	NH_4-N	NO_{2+3}-N	SiO_4-Si
Minimum	-3.6	-0.04	-0.95	-3.7
Maximum	2.01	22.8	10.0	54.4
Median	-0.47	4.7	1.6	22.4

(ii) Longmore: Duplicate cores were collected from six sites within 1.2 km of shore 17 times in 20 months (over the years 1990-92) and incubated at ambient temperature in the dark for 10 days, in order to detect temporal trends; net fluxes were estimated as in (i) above. Table 3.2.2.2 lists simple summary statistics for these fluxes. The mean was chosen as the median statistic because the flux distributions were non-normal.

Table 3.2.2.2 indicates that, at these nearshore sites, there was a net flux of phosphate into the sediment and net fluxes of inorganic nitrogen out of the sediment (with ammonium fluxes much larger than oxidized nitrogen fluxes). Temporal variation in sediment nutrient flux during 1990-92 was slight, although cyclic regression modelling suggested small annual peaks in summer in all fluxes.

Table 3.2.2.2. Summary statistics for nutrient fluxes from nearshore PPB cores, 1990-92. All units mg m^{-2} d^{-1}.

Nutrient	PO_4-P	NH_4-N	NO_{2+3}-N	SiO_4-Si
Minimum	-3.8	-5.1	-1.7	-7.0
Maximum	45.8	261	12.0	130
Median	-0.3	8.3	0.3	20.3

Phosphate and silicate fluxes were similar in the two studies, but median ammonium fluxes during 1990-92 were twice as high, and oxidized nitrogen fluxes 90% lower than in winter 1991.

Because one data set has good spatial coverage but no temporal information, while the other data set has good temporal coverage but is confined to nearshore areas, an adequate Bay-wide mean annual flux can only be calculated by combining aspects of the two studies. The same approach allows an allocation of mean annual fluxes to the nearshore and deep central areas defined in the Gorfine and Abbott study (Table 3.2.2.3), where the nearshore area is 61% and the central area 39% of the total Bay area.

Table 3.2.2.3. Estimated mean annual nutrient flux from sediments in the nearshore and deep central areas of the Bay (tonnes y^{-1}).

Nutrient	PO_4-P	NO_3-N	NH_4-N	T.I.N. (NH_4 + NO_3)
Near-shore	-214	1800	5600	7400
Central	122	1800	2600	4400
Total Bay-wide	-92	3600	8200	11800

The approximations required to make these estimates lead to an uncertainty in the mean annual fluxes of at least 50%, and the "true" annual inorganic nitrogen flux probably lies between 6000 and 18000 tonnes y^{-1}. With that in mind, net annual inorganic nitrogen fluxes from the sediment may be of the same order as external nitrogen inputs to the Bay (6000-7000 tonnes y^{-1}). Assuming the highest estimated annual efflux of about 12000 tonnes N from the sediment, and assuming the source material had an atomic N:P ratio of 16:1 (7:1 by weight), phosphate sediment flux was estimated at about 2000 tonnes y^{-1}; the observed flux is an uptake of about 100 tonnes. The difference (1900 tonnes) is phosphate sequested in the sediment each year, which exceeds the estimated external input.

3.2.3 **Groundwater**

A detailed review of the groundwater inputs to Port Phillip Bay is given by Otto (1992). Otto describes groundwater discharge rates and nutrient inputs into Port Phillip Bay and his Table containing this information is reproduced in Table 3.2.3.1. A map of the Bay showing the sites of groundwater nutrient discharge is given in Figure 3.2.3.1. This a simplification of a very detailed map produced by Otto (1992).

Discharge Zone	Aquifer System	Discharge Width km	Discharge Rate m^3y^{-1}	Nutrients	Toxicants	Heat $^oC/100$ m	Salinity mgl^{-1}TDS	References
Central	Older Volcanics		?	?	?	very likely	?	
	Werribee F.		$1x10^7$ to Bay and Bass Strait?	likely	likely	very likely	?	Leonard'92
Werribee and	Brighton F.	40	$5.6x10^5$	possible	possible	no	2500-8000	Leonard '79,'92
Altona	Newer Volcanics	60	$4x10^6$?	N, P, C	phenols, PCB PAH, heavy metals	no	1000-14000 $2.9x10^2$ ty^{-1}	Riha et al. '78 Shugg '78
Altona	Newer Volcanics	20	$1.6x10^7$	N, P, C	phenols, PCB	no	1000-14000 $2.9x10^2$ ty^{-1}	Anderson '88 Leonard '79,'92 Riha et al. '78 Shugg '81,'92 Stephan '81 Stewart '88
		15	$1.3x10^7$ Riha et al. '78		PAH, heavy metals			
Werribee	Werribee Delta	19	$2.6x10^6$	34 ty^{-1} N 8.1 ty^{-1}P	heavy metals conc. is low	no	2500-8000	Anderson '89 Coffey '92 Stewart '88
Corio Bay	Bridgewater G.	?	?	very likely	very likely	no	2000-10000	Leonard '79,'92
Bellarine	Bridgewater G	?	?	very likely	possible	no	2500-8000	Leonard '79,'92
Nepean	Werribee F.	?	?			7	?	Shugg 85, '87, '89
	Brighton F.	22	$8x10^6$	N, P 1.2 ty^{-1} NO3	low	no	300-2000	
Southeastern	Older Volcanics	10.5	$8.2x10^5$	?	?	no	?	Leonard '79 Walker '90
	Fyansford F. - Brighton G.	44	$3.6x10^6$	yes, but no data	possible phenols, heavy metals	no	110-7000	
Hobson Bay	Port Melbounre Sd.	?	?		phenols, PCB,PAH, tars, oils, heavy metals (Pb, Hg)	no	2000-40000	Shugg '87,'89

Table 3.2.3.1 Groundwater discharge rates and nutriant toxicant imputs into Port Phillip Bay for selected Discharge Zones and hydrogenic units (from Otto,1992).

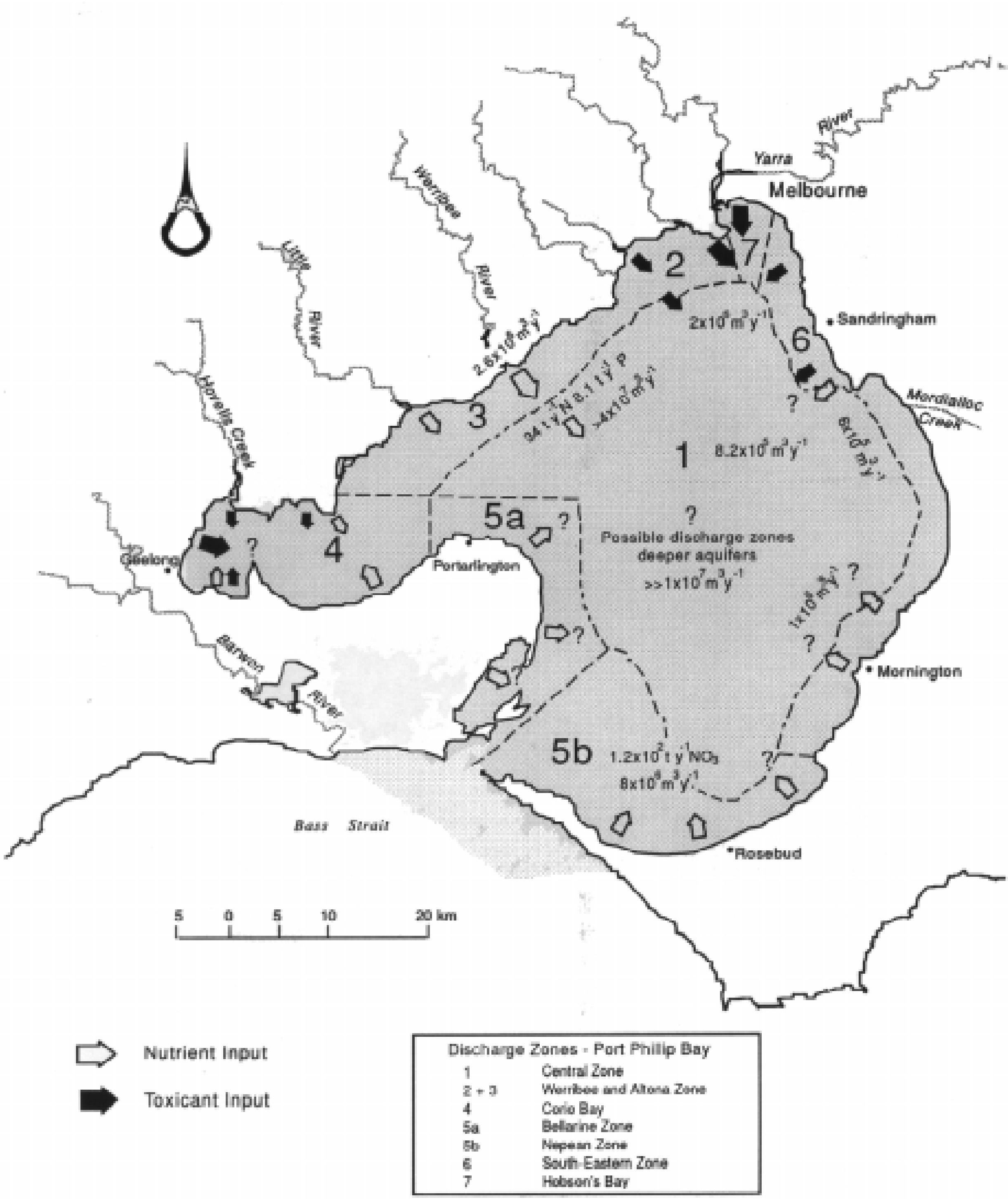

Figure 3.2.3.1 Groundwater discharge rates and nutriant toxicant imputs into Port Phillip Bay for selected Discharge Zones and hydrogenic units (from Otto,1992).

Otto (1992) identified eight discharge zones and eight main aquifer systems. The Werribee Delta aquifer contributed the largest nutrient loads, discharging an estimated 34 tonnes N and 8 tonnes P y^{-1} into the Werribee Zone. There has been an estimated 20% increase in the groundwater nutrient load being discharged into the Werribee Zone, the reason for which is unclear. The Newer Volcanics aquifer discharge nutrients into the Altona Zone but no nutrient data are available. However, it is known that these groundwaters are the most polluted in the region. The Bridgewater aquifer discharges into Corio Bay and the Bellarine Zone. There are no discharge rates for these aquifers but Otto considers that the discharge of nutrients is likely. The Nepean Zone receives groundwater discharge from the Werribee and Brighton aquifers; a small amount of P and 0.3 tonnes N y^{-1} are estimated nutrient loads from the Brighton aquifer. The Fyansford, Brighton and Older Volcanics aquifers discharge groundwater into the South-eastern Zone and Otto considers that nutrient input is likely; there are no data for this large area. Hobson's Bay receives groundwater from the Port Melbourne Sands aquifer. No data are available for nutrients but 50000 m^3 of the latter groundwaters are contaminated with light and heavy oils.

The large Central Zone of the Bay receives discharge water from the Older Volcanics and Werribee aquifers and it is likely that some nutrients are transported into the Bay *via* this route. The total groundwater discharge into the Bay is estimated to be greater than 5.5 x 10^7 m^3 y^{-1} because the inputs to the Central, Corio and Bellarine Zones have not been measured. Compared to the nutrient loads discharged to the Bay from the catchment and the sewage treatment centres, groundwater discharge is very small. However, in most situations the discharge width is 20 to 40 km covering areas which could be locally affected if the discharge loads were to increase.

3.2.4 **Bass Strait**

Because Bass Strait water flows into Port Phillip Bay twice each day, the possibility exists that Bass Strait may be a source of nutrients to Bay waters. Nutrients in Bass Strait water near the Heads have been measured in several studies (MMBW/FWD, 1973; Smith and Longmore, 1980; Cowdell, 1983; Gibbs *et al.,* 1986a; Longmore *et al.,* 1990), producing the relatively narrow range of concentrations listed in Table 3.2.4.1. Although Bass Strait may be considered as a major source of nutrients, the simple water quality model used by Newell to calculate the Bay-wide dynamics of orthophosphate (MMBW/FWD, 1973) indicates that the importing of DON from Bass Strait (which might be quite different from that produced in the Bay) occurs at very slow rates.

Table 3.2.4.1. Nutrient concentrations in Bass Strait waters within 60 km of Port Phillip Bay.

Nutrient	PO_4-P	Total P-P	NO_2-N	NO_3-N	NH_4-N	Total N-N	SiO_4-Si	TOC-C	PO-C
Range (g L^{-1})	6-10	9-13	0.5-1	3-15	1-4	170	11-22	700-900	<100

No measurements have been made of the possible effect that plumes from the Barwon River or Black Rock sewerage plant may have on nutrient fluxes through the Heads.

3.2.5 **The atmosphere**

A detailed review of the aeolian and atmospheric inputs of volatile organic carbon (VOC), nitrogen (as NO_x), and phosphorus (as phosphate) to Port Phillip Bay is given by Carnovale *et al.* (1992a). Winter and summer anthropogenic emissions in the Port Phillip Catchment Region were estimated but because of seasonal variations in rainfall scavenging efficiency, meteorology and photochemical aerosol formation, the total load of VOC to the Bay could not be estimated.

Carnovale and Saunders (1988) concluded from an analysis of 280 rain samples that the wet deposition of atmospheric nitrogen into the Bay is around 600 tonnes y^{-1} (40% nitrate and 60% ammonia). However, from calculations based on Henry's Law, they predict the total atmospheric input to the Bay to be 900 to 1600 tonnes y^{-1} (average 1250 tonnes y^{-1})..

They also estimated that 1200 tonnes N_2O was dry deposited annually to the Bay. Because N_2O is chemically unreactive they considered it unavailable to the Bay ecosystem. However, N_2O can be produced by denitrifying and nitrifying bacteria during the reduction of nitrate and the oxidation of ammonia (Hooper, 1989). Increasing the partial pressure of N_2O may affect the rates of these microbial processes.

Estimates of the atmospheric (dust) inputs of phosphorus to Port Phillip Bay are not available. Because of seasonal variations the input of P to the Bay may vary significantly from year to year. As an example, they cite the fearful dust storm which dumped large quantities of fine soil on Melbourne and Port Phillip Bay in 1983.

3.2.6 **The Werribee Treatment Complex**

Melbourne Water has been using land treatment methods at Werribee (the "Farm", now referred to as the Werribee Treatment Complex - WTC) to process waste water from the city of Melbourne since 1897; with the final effluent being discharged to Port Phillip Bay for the full 92 year period (Bremner and Chiffings, 1991).

3.2.6.1 Quantity of effluent discharged from WTC

Over the 92 year period the population of Melbourne has grown from an estimated 430,000 people to an estimated population of 2,525,000 in 1990. As a result, nutrient loads from the WTC have progressively increased, although accurate estimates of effluent quantity and quality have only been determined on a regular basis since 1975 as part of the monitoring for EPA licences. Prior to the establishment of accurate flow guaging on the outfalls in the 1970s a number of estimates of nutrient loads from the WTC to Port Phillip Bay have been made (Chiffings *et al.*, 1992).

By the mid-1970s nitrogen loads to the Bay from the WTC were estimated to have reached approximately 6000 tonnes per year. Following the commissioning of the South Eastern Purification Plant (SEPP) in 1975 the nitrogen load from WTC was reduced to the 1968 level of 3500 tonnes. Nitrogen loads have since increased to approximately 4000 tonnes per year despite sizeable population increases. The proportionately small increase in nitrogen loads from WTC during that time is at least in part the result of new treatment technologies (Chiffings *et al.*, 1992).

3.2.6.2 Composition of the effluent

The known nutrient composition of the effluent from the five oufalls in 1989 is shown in Table 3.2.6. Most of the nitrogen is inorganic and largely present in the NH_4-N form (63% of the total nitrogen), thereby being readily available for uptake by algae. Organic nitrogen constitutes about 25% of the total, however the relative proportions of DON and PON remain unknown.

Considerable variation occurs in both nitrogen concentrations and loads to the Bay (Table 3.2.6). NH_4-N concentrations, in particular, have a strong seasonal change from high winter-spring values to low summer-autumn values.

Phosphorus contained in the effluent is largely present in the inorganic form, PO_4-P, and therefore also readily available for plant uptake. Seasonal variations in the quantities of phosphorus entering the Bay from WTC are less pronounced than for nitrogen.

The total annual organic carbon load from the WTC for 1989 has been estimated at 4978 tonnes. This is equivalent to four times the estimated pool of phytoplankton carbon in the central segment of the Bay during the Phase 1 sampling period (see below). The detailed chemistry of organic nitrogen and organic phosphate is unknown.

The N:P ratio (calculated using atomic weights) of the effluents averaged 7:1 and ranged between 1.9:1 - 14.4:1. These ratios are lower than the 11:1 N:P ratio obtained by Longmore for seston in the Bay (see Section 3.4.4 below). C:P ratios for the effluents averaged around 10:1 and ranged between 3.1:1 - 29:1.

3.2.7 **Other point sources**

Melbourne Water also operates a small treatment plant (Western No. 2) that services the domestic catchment of Altona. This plant discharges secondary treated effluent to the Bay immediately north of the Laverton Creek main drain. The annual total nitrogen load from this facility is 25 tonnes, and the phosphorus load 29 tonnes. These amounts are insignificant compared with the loads from the WTC, but may be important at a local level.

Four other point sources discharge effluent to the Bay and 1981 estimates of their annual N and P loads total 200 tonnes (Blutstein *et al.*, 1982). Again it can be seen that these loads are very small in comparision with the WTC and major stream inputs.

3.3 **Distribution and pools**

3.3.1 **Waters of Port Phillip Bay**

Large scale studies of Port Phillip Bay water quality commenced in 1969, and six studies have since been carried out (in 1969-71, 1975-76, 1980-84, 1985-86, 1985-present and 1990-present). Results of the first four studies have been published and summarized by Newell (1990). Figures 3.3.1.1.and 3.3.1.2 are reproduced from Figures 8-8 and 8-7 respectively in the Phase 1 Report (MMBW/FWD, 1973). They show the orthophosphate concentrations in surface, mid-depth and bottom waters of the East-West and North-South transects for 1969-70.

	FLOW ML/d	NH_4 mg/L	NO_2 mg/L	NO_3 mg/L	ORG N mg/L	TN mg/L	TN mg-a/L	TNLD T/d	ORT P mg/L	TOT P mg/L	TP mg-a/L	TPLD T/d	TOC mg/L	TOC mg-a/L	TCLD T/d	TSS mg/L	C:N	C:P	N:P
15E OUTLET																			
MEAN	105.5	16.3	0.7	3.8	4.3	25.1	1.8	2.6	6.8	7.8	0.3	0.8	25.0	2.1	2.6	30.2	1.4	8.4	7.2
STD. DEVIATION	26.6	9.1	0.5	2.9	3.1	7.7	0.5	1.0	0.6	0.8	0.0	0.2	9.8	0.8	1.3	8.5	0.9	3.5	2.2
MINIMUM	54.0	1.7	0.1	0.1	0.1	6.7	0.5	0.6	5.3	5.9	0.2	0.4	12.0	1.0	1.0	15.0	0.5	3.1	1.9
MAXIMUM	176.0	30.0	3.1	13.5	16.2	43.4	3.1	6.1	8.0	10.1	0.3	1.4	66.0	5.5	8.2	65.0	5.0	22.7	12.3
TOT. ANN. LOAD	38493.5							952.4				300.8			963.4				
145 W OUTLET																			
MEAN	117	15.3	0.56	2.40	6.2	24	1.74	2.9	6.0	7.6	0.25	0.90	30	2.53	3.54	30.2	1.6	10.5	7.2
STD. DEVIATION	38	8.7	0.47	2.48	4.0	7	0.51	1.4	1.3	1.5	0.05	0.39	14	1.14	2.08	8.5	0.9	4.6	2.4
MINIMUM	50	0.8	0.11	0.1	0.1	6	0.44	0.5	2.9	4.6	0.15	0.31	17	1.42	1.10	15.0	0.7	5.3	3.0
MAXIMUM	200	29.0	2.45	12.60	20.0	39	2.79	7.4	9.3	11.9	0.38	1.99	78	6.50	10.34	65.0	5.6	24.3	14.4
ANNUAL LOAD	42572							1042				329			1291				
WINTER OUTLET – LITTLE RIVER May – Oct. 1989																			
MEAN	45	21.6	0.23	2.1	2.5	26.5	1.89	1.2	6.8	7.9	0.26	0.3	28	2.32	1.2	23	1	9	8
STD. DEVIATION	10	4.5	0.08	1.4	1.9	3.7	0.27	0.3	1.1	1.8	0.06	0.1	8	0.67	0.3	11	0	2	1
MINIMUM	16.	14.	0.1	0.2	0.1	18.7	1.34	0.4	4.6	4.8	0.15	0.2	17.	1.42	0.6	5.	0.79	6.71	4.86
MAXIMUM	57.	30.	0.38	5.5	7.3	33.1	2.36	1.8	9.1	11.4	0.37	0.5	43.	3.58	1.7	50.	1.87	14.10	10.6
ANNUAL LOAD	16362							430.				126.2			437.9				
LAKE BORRIE OUTLET																			
MEAN	54	15.1	0.33	0.9	8.4	24.8	1.77	1.4	6.7	8.7	0.28	0.5	41	3.43	2.3	64	2.1	12.1	6.6
STD. DEVIATION	23	7.5	0.37	2.	4.4	6.1	0.43	0.7	1.4	1.7	0.05	0.2	23	1.88	2.1	50	1.2	5.2	2.2
MINIMUM	11	3.	0.04	0.1	1.	8.8	0.63	0.2	2.1	5.3	0.17	0.1	20	1.67	0.4	15	0.8	5.4	2.4
MAXIMUM	108	29.	2.1	12.2	23.	39.	2.78	4.	10.2	12.3	0.4	1.3	138	11.5	14.2	350	6.1	29.	11.8
ANNUAL LOAD	19696							503.2				172.2			831.4				
MURTCAIM OUTLET																			
MEAN	137	17.2	0.32	1.3	6.	24.9	1.78	3	6.2	7.8	0.25	1	36	2.99	4.	49	1.8	12.2	7.2
STD. DEVIATION	116	5.2	0.2	1.9	4.1	5.	0.36	3	1.3	1.6	0.05	1	13	1.07	3.	34	0.8	4.5	1.4
MINIMUM	14	6.5	0.07	0.1	0.1	14.2	1.01	0	3.5	3.6	0.12	0	20	1.67	0.4	15	0.9	5.6	4.4
MAXIMUM	331	29.	1.15	12.1	21.9	44.4	3.17	10	9.1	11.	0.35	4	75	6.25	9.3	220	5.3	27.3	11.3
ANNUAL LOAD	49970							1253				375			1454.9				
ANNUAL TOTAL LOADS	167094 (ML)							4180 (Tonne)				1303 (Tonne)			4978 (Tonne)				

Table 3.2.6 Average effluent values for WTC outlets in 1989 (MWC unpublished)

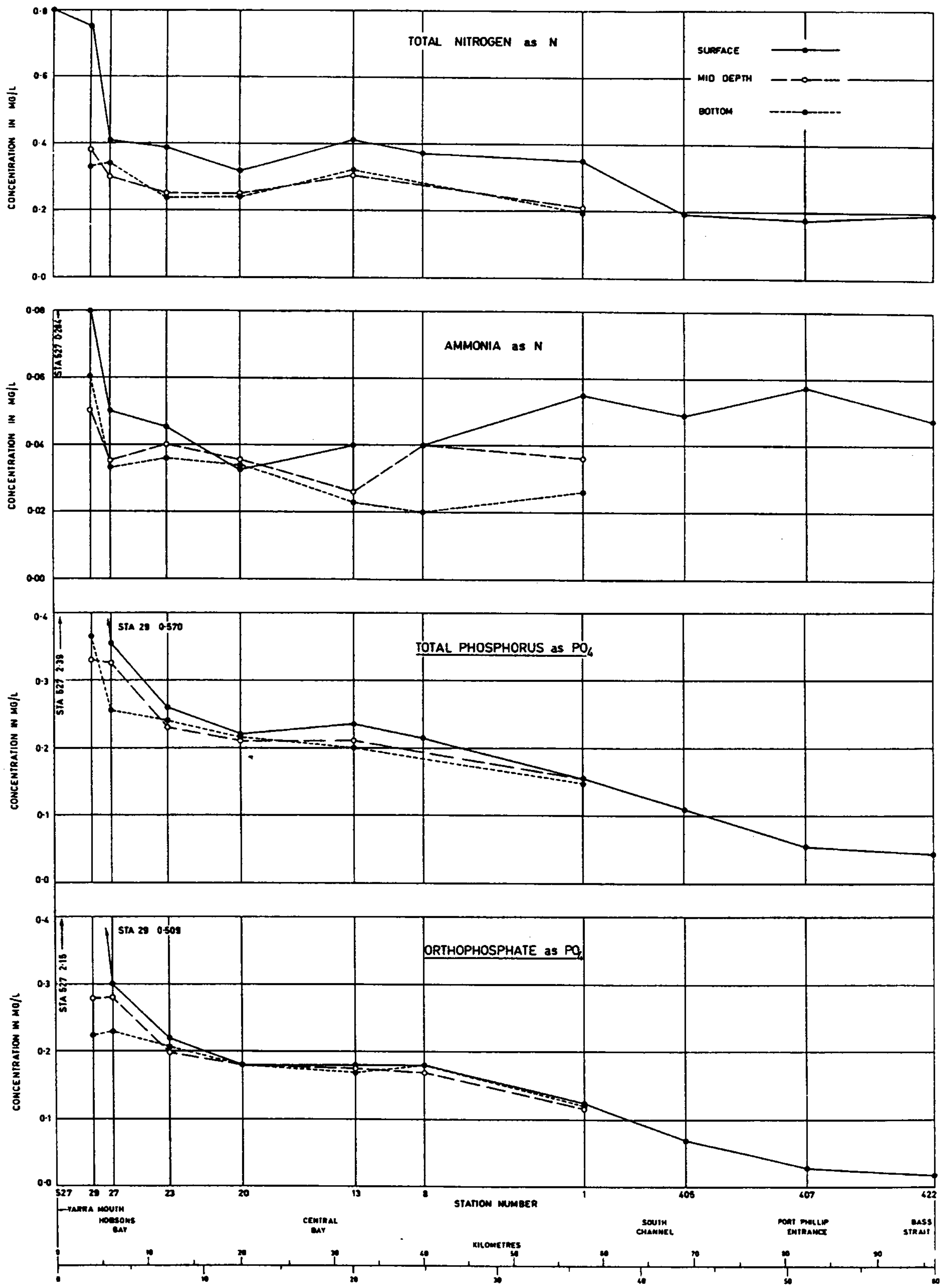

Figure 3.3.1.1 Mean nutrients concentration on stations on North-South Transect during 1969. (from MMBW/FWD,1973, Fig 8.8).

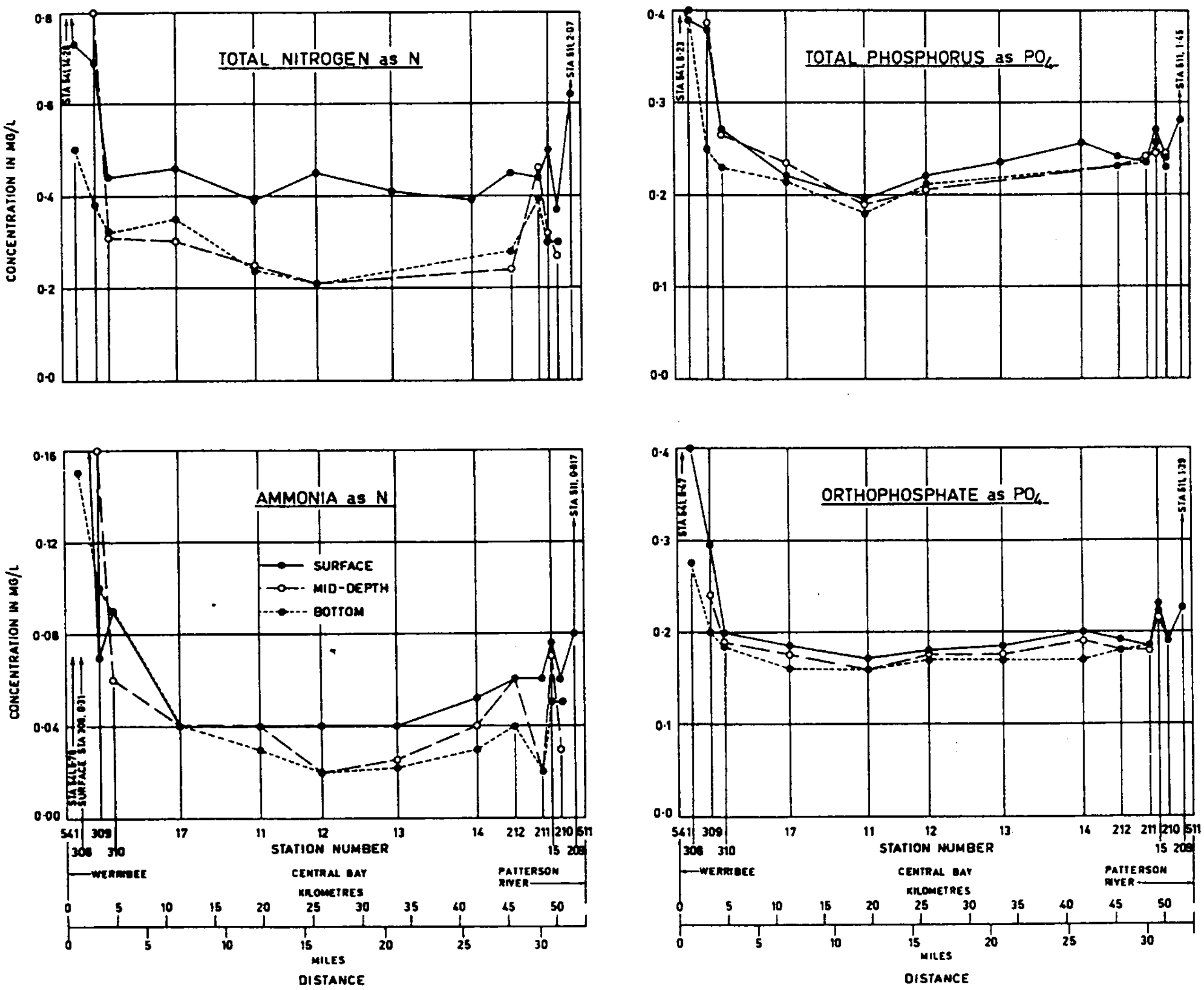

Figure 3.3.1.2 Mean nutrients concentration on stations on East-West Transect during 1969. (from MMBW/FWD,1973, Fig 8.7).

The actual and modelled distributions of orthophosphate show that the highest concentrations of orthophosphate occur in the nearshore waters and that the concentration gradient is relatively steep from the shore to the Central Zone. The distribution patterns are not the same for N (Figures 3.3.1.1 and 3.3.1.2). However there were probably analytical errors due to the low sensitivity of the ammonia method used in Phase 1 (MMBW/FWD, 1973). The biological dynamics of N nutrients is expected to be more influential on the pool sizes and distribution.

3.3.1.1 Spatial distribution

1969-70: Nutrient concentrations were high near-shore, particularly off the WTC and Yarra River, but declined rapidly toward the centre of the Bay. A further decline was observed south of the Sands through to Bass Strait (see Figures 3.3.1.1. and 3.3.1.2.). On this basis the Bay was divided into three hydrodynamic zones: a near-shore mixing zone, a large central zone of relatively uniform nutrient concentration, and an exchange zone. Total nitrogen concentrations were highest near the surface, averaging 420 g L^{-1} in surface waters and 210 g L^{-1} in deeper waters in the centre of the Bay.

1975-77: Random stratified sampling was carried out from surface, mid-depth and bottom waters at 36 sites every six weeks, in nine segments previously identified (MMBW/FWD, 1973) as being of uniform water quality. However, variation of concentrations was often as high within segments as it was between samplings, and no areas of homogeneous water quality were found in this study. The segments with highest nutrient concentrations were Werribee and Hobsons Bay. High concentrations of ammonium, nitrate and total phosphorus in the Werribee segment declined to "background" (central Bay) concentrations 2-3 km from the shore. Total nitrogen and silicate were not measured. Bay-wide mean concentrations of ammonium were about one-tenth of those found in the Phase 1 Study, while nitrate, phosphate and total phosphorus concentrations were similar in the two studies. Newell (1990) suggested that these changes may be due to the reduction of discharge from the WTC in 1975. This is unlikely given that no decrease in phosphate concentration has been observed. It is more likely that analytical errors in the ammonia method of the Phase 1 Study have confounded the situation.

Nutrient concentrations varied little with depth, except near the WTC, where highest ammonium and nitrate concentrations occurred near the surface.

1980-84: Surface waters only were sampled randomly from 16 strata, which were subdivisions of the segments sampled in 1975-77. Highest mean concentrations of inorganic and organic phosphorus, ammonium and total nitrogen occurred in the strata nearest the WTC, while highest mean concentrations of oxidized nitrogen (NO_2 + NO_3) and silicate occurred in Hobsons Bay. Bay-wide mean concentrations of ammonium, nitrate and total phosphorus were slightly higher than those found in 1975-77 because sampling was biased toward the more industrialized north-west of the Bay.

1985-86: Continuous on-board analysis of surface water samples allowed better definition of spatial variation of phosphate, nitrate and nitrite (Longmore *et al.*, 1990; Fig 4) than had been possible from discrete sampling. The same general pattern of high nearshore concentrations was observed off the WTC and the Yarra River, with high concentrations of N and P forms also observed off the Patterson River. High nitrate concentrations were observed off the Werribee River in winter, and nitrate was detectable off the WTC in spring and autumn and in Hobsons Bay in all seasons. There was often no measurable inorganic nitrogen in large areas of the Bay, particularly during summer, while relatively high concentrations of phosphate were always found. The low concentrations of inorganic nitrogen are probably the result of accelerated phytoplankton growth. Sampling from 28 fixed sites also indicated a steep spatial gradient in total phosphorus concentration in winter, from about 250 g P L^{-1} off the WTC, 155 g P L^{-1} off the Werribee River, 125 g P L^{-1} in Corio Bay, 93 g P L^{-1} in Hobsons Bay and off the Patterson River, 60 g P L^{-1} in the centre of the Bay, to 15 g P L^{-1} in Bass Strait. In spring, summer and autumn concentrations were similar in all areas, except off the WTC and Patterson River where they were lower.

1984-92 (EPA fixed sites): Measurements from surface waters at three sites (Hobsons Bay, Corio Bay, central Bay) were made at approximately 2-4 week intervals. Total P and total N were usually highest in Corio Bay, and lowest at the central Bay site. Ammonium, nitrate and silicate were usually highest in Hobsons Bay, and lowest at the central site.

1990-92 (MSL fixed sites): The six sites, sampled fortnightly in this study, are all within 1.2 km of shore. Highest mean concentrations of ammonium, nitrite, phosphate, and organic N occurred off Werribee River (site 11), while highest mean concentrations of organic P, particulate P, nitrate, particulate N, silicate, POC and TOC occurred in Hobsons Bay (site 9). Mean concentrations of all compounds (including nitrate) were lowest near the Rip (site 1). Concentrations of nutrients in surface samples from Hobsons Bay were often (but not always) higher than in bottom samples. Samples from all other sites were depth-integrated due to a freshwater/estuarine wedge.

All of the studies have found high nutrient concentrations near-shore with concentrations declining rapidly toward the centre of the Bay, with a further decline observed south of the Sands through to Bass Strait. Ammonium concentrations reported in the centre of the Bay in 1969-71 were near 40 g L^{-1}, about 10 times higher than those currently observed. The method used for ammonia determinations in Phase 1 had a detection limit of around 14 g N L^{-1}. Since 1980 the Bay-wide concentrations (including those in Bass Strait) of NH_3-N have been ten to twenty times less (Longmore, 1992). We have therefore used the more recent data for ammonia.

Vertical variations in nutrient concentrations through the water column have been observed near freshwater inputs during times of high flow, but are usually small in most areas of the Bay. No diurnal variation was observed in phosphate, total phosphorus, total nitrogen, nitrate, ammonium or silicate concentrations in sampling carried out at hourly intervals for 24 hours during summer and winter at a site in Corio Bay in 1985-86; an area of little tidal influence. Little seasonality has been observed for any nutrient, with increased concentrations related to river flow rather than time of year.

There are no data on the amounts of nutrients mixed into the water column during channel dredging or scallop fishing. However if a dredging operation disturbs the upper 15-20 cm of sediment, up to one tonne of DIN km^{-2} could be released into the water column. Distributed into 1 km^2 of water 10 m deep, the concentration of DIN would be raised by 100 g N L^{-1}.

3.3.1.2 Temporal distribution

1969-70: Bay-wide mean concentrations of total nitrogen in surface waters peaked in summer, but no seasonal pattern was noted for mid-depth or bottom samples. No seasonal pattern was found for phosphate, total phosphorus, or inorganic nitrogen. Mean silicate concentrations increased from 170 g Si L^{-1} in 1969 to 450 g Si L^{-1} in 1970, during which time freshwater discharge to the Bay increased.

1975-77: Highest concentrations of ammonium and nitrate occurred in winter in Hobsons Bay, Exchange, Altona and Geelong segments (the sampling period was for one winter only). Phosphate and total phosphorus concentrations in the Corio segment showed a notional seasonal pattern of summer highs and spring lows. No temporal pattern was detected for any nutrient in other segments.

1980-84: Peaks in total nitrogen concentration occurred in winter in the Werribee and Exchange segments, and in summer in the Central segment, but were probably related to "pulse" inputs rather than a slow seasonal effect. Peaks in ammonium concentrations occurred in winter in the Werribee segment, but no large temporal change was observed in the Central or Exchange segments.

1985-86: Ammonium, nitrate and total phosphorus concentrations were highest in winter; ammonium and nitrate concentrations were very low in summer except near the Yarra mouth.

1985-92 (EPA fixed sites): Peaks in total nitrogen, total phosphorus, silicate, nitrate and ammonium in Hobsons Bay were not seasonal, but coincided with high river flows. Silicate was often lowest at the central site in spring and summer (< 56 g Si L^{-1}).

1990-92 (MSL fixed sites): Ammonium and nitrate concentrations were higher in autumn-winter than in summer, except near the Heads. Ammonium was elevated (>28 g N L^{-1}) off Werribee River for up to 14 weeks during winter-spring 1991. Nitrate concentrations were very low (<1.4 g L^{-1}) for most of the year at Blairgowrie (site 1) and Clifton Springs (site 13). Phosphate concentrations varied little with time at Blairgowrie and Williamstown (site 9), minima were observed in spring at Wooley Reef (site 4) and summer at Clifton Springs (site 13), while frequent peaks at the Werribee River site (site 11) did not relate to seasons. A summer minimum in silicate concentration was observed at Blairgowrie, Wooley Reef and Sandringham (site 6), with minima in winter 1990 and spring 1991 at Werribee River and Clifton Springs. Temporal patterns in TOC concentrations were not evident at any site. Organic nitrogen and organic phosphorus were higher in summer than in winter at all sites, though the seasonal variation was often quite small compared to the background "noise". Seasonal patterns in particulate N and P varied between sites, with spring-summer maxima at Blairgowrie (site 1) and Sandringham (site 6), winter-spring maxima at Wooley Reef (site 4), and no obvious pattern at the other sites.

Dividing the Bay into six zones based on the Phase 1 segments and calculating water volume in each zone and using mean nutrient concentrations found in the 1980-84 study enabled an estimate of nutrient pools in each zone to be made (Table 3.3.1.2). Because it contains over 80% of the total Bay volume, the Central Zone holds by far the largest pool of all nutrients. The Central Zone organic C:N weight ratio of 14:1 is twice that expected by diatomaceous phytoplankton. If the organic material in the Central Zone waters is of phytoplankton origin it has undergone substantial depletion in nitrogen. Alternatively a substantial input of organic C from the atmosphere could also increase the C:N ratio.

Table 3.3.1.2. Nutrient pools in the water column in different zones of the Bay (tonnes) for 1980-84. Zones are defined in the Phase 1 Report; near-coastal includes all areas outside the five other zones.

Zone	Central	Exchange	Hobsons Bay	Corio Bay	WTC	Near-coastal	TOTAL:
PO_4-P	1152	54	11	16	62	153	1447
ORG P	357	18	3	3	18	46	446
NO_2-N	16.7	1.4	0.2	0.1	1.5	2.0	22.0
NO_3-N	47	5	2	1	5	11	71
NH_4-N	84	6	1	1	18	29	138
ORG N	3144	188	26	40	158	402	3960
SiO_4-Si	2059	109	22	25	79	247	2540
TOC-C	43727	2453	325	499	1876	5193	54073

3.3.2 **Sediments**

In addition to their role as a source of nutrients, sediments may also act as nutrient pools. Three data sets exist from which it is possible to calculate Bay-wide pool sizes for total N and total P in the upper 2 cm sediment (Melbourne Water, 1990, 14 sites; Gorfine and Abbott, 1991, 39 sites; Longmore, 1990-92, 6 sites 17 times). Excellent agreement can be observed for mean concentrations from the three studies (Table 3.3.2.1). Highest total nitrogen and total phosphorus concentrations occurred in Corio Bay and in the Central Zone, at sites with sediments consisting of fine mud. High concentrations also occurred off the WTC in sandy sediments. No temporal pattern was observed. The lower mean concentrations observed by Longmore reflect the location of these sites (nearshore, sandy sediments).

Table 3.3.2.1. Mean nutrient concentrations (g-at P (or N) g^{-1} dwt) in surficial (0-2 cm) sediment from Port Phillip Bay.

	Total N	Total P
M.W. (1990)	49.4	9.44
Gorfine & Abbott (1990)	49.8	9.94
Longmore (1990-92)	34.1	7.89

Bay-wide total nutrient loads in the upper 2 cm sediment are estimated at 10900 tonnes P and 25000 tonnes N.

3.3.3 **Phytoplankton epibenthic algae and macroalgae**

As to be expected in a system the size of Port Phillip Bay that displays considerable geographic variability in nutrient point sources etc., the distribution of phytoplankton in the Bay is neither spatially nor temporally homogeneous. Spatial distributions of seston (all suspended matter including phytoplankton) recorded during the Port Phillip Bay Study showed that the highest concentrations generally occurred in the north-eastern region from Hobsons Bay to Carrum (MMBW/FWD, 1973). Other areas showing high concentrations during some surveys included Corio Bay, Geelong Arm, the Werribee coast and the central Bay area . Mean chlorophyll *a* concentrations ranged between 0.6 - 2.0 mg m^{-3}, and the overall range reported was between 0.1 - 4.4 mg m^{-3}. More recent surveys by Longmore (1992) confirm the general distribution patterns observed during Phase 1. Assuming a C:Chl weight ratio of 50:1 and a C:N:P weight ratio of 42:7:1 (Harris, 1986) there is 1900, 200 and 30 tonnes of C, N and P contained in the standing stock of phytoplankton in the Bay.

The toxic algal blooms which have occurred in Port Phillip Bay over the last five years cannot be directly attributed to nutrient excess. However, a continuous input of DIN from the Yarra River, coupled with calm weather, long daylight hours, salinity stratification and slow mixing rates, combine to encourage annual blooms around the northern part of the Bay. It may be possible to reduce the incidence of such opportunistic algal blooms if N input from both the Yarra River and the WTC is controlled during these vulnerable periods.

Axelrad *et al.* (1981) measured 60 to 130 mg m^{-2} chlorophyll *a* concentrations in the upper 1 cm sediment in the Werribee Zone. This chlorophyll is contained in the epibenthic algal mats. Assuming a C:Chl weight ratio of 50:1 and a C:N:P weight ratio of 42:7:1 (Harris, 1986) there is a mean of 4.75, 0.79 and 0.11 g C, N and P m^{-2} in the mat. The area covered by the mat is not known precisely, but assuming 10 km^2 (see Newell, 1990), there is around 47, 8 and 1 tonnes C, N and P contained in the epibenthic algal biomass; Axelrad *et al.* (1981) record that there was little seasonal variation.

The present distribution of phytobenthic algae is not known although the overall distribution is limited. Distribution of phytobenthos is considered to be largely determined by the availability of hard substrate, although in some places extensive beds of *Caulerpa* and other species occur in sand substrates. Other factors influencing the distribution of attached macrophytes include degree of exposure, substrate stability and light availability (Light and Woelkerling, 1992).

Brown *et al.* (1980) found that the basalt reef community at Kirk Point of Werribee had a dry weight biomass ranging from 90 - 312 g m^{-2}, and these values paralleled those for the *Caulerpa* community. Brown *et al.* (1980) estimated the total standing crop biomass of macrophytes in the 40 km^{-2} area from the Werribee River to Point Wilson to be 1.448 x 10^6 kg. This should be considered a maximum estimate because the estimates were based on 100 percent cover. Newell (1990) calculated that this macrophytic biomass contained around 42 tonnes N, about 20% of the N contained in the phytoplankton biomass.

Large quantities of healthy, unattached algae are reported to occur over extensive areas of Port Phillip Bay (Brown, cited in Light and Woelkerling, 1992). The spatial extent, biomass, and productivity of these algae remain unknown. In other coastal systems these algae can make a significant contribution to net productivity (e.g. the temperate coastal areas of Western Australia (Hanson, pers. comm.)).

The N estimated to be contained in the biomass of phytoplankton, epibenthic and macrophytic algae is around 250 tonnes; 80% of which is contained in the phytoplankton. However, the biomass and N content of the unattached algae and seagrass (see next section) remains unknown.

3.3.4 **Seagrasses**

Bulthuis (1981) examined the distribution of seagrasses in Port Phillip Bay over the period 1978-1981. Examinations were made using aerial photography supported by field observations (Figure 3.3.4.1). Extensive seagrass areas occur in Swan Bay, at the Prince George Bank and along the southern margin of Corio Bay and the Geelong Arm. Bulthuis reported that 95% of seagrasses occurred at depths of less than 5m.

While Bulthius found that some changes in distribution had occurred between 1945-50 and 1978-81, he concluded that little overall change in distribution had occurred since 1957-63. Subsequent to this a major reduction in the southern margin of Geelong Arm has been noted (DCNR, unpublished).

Earlier work by Willis (1966) and that of Bulthius (1981) on the distributions of individual species of seagrasses are somewhat at variance (Light and Woelkerling, 1992). This, together with the limited data available from other studies, makes it difficult to draw conclusions with respect to production rates and biomass estimates per unit area for the Bay overall.

3.3.5 **Primary consumers and secondary production**

Figure 3.3.5.1 is a simple illustration showing the relationship between the recycling of nutrients and the primary producers and consumers.

3.3.5.1. Bacteria

Bauld (unpublished) recorded high bacterial numbers in the Werribee Zone, but there are no data on the productivity of heterotrophic, non-photosynthetic bacteria in Port Phillip Bay.

3.3.5.2 Zooplankton

It was an objective of the Phase 1 Study of Port Phillip Bay to correlate data on zooplankton standing crops and seasonal changes with phytoplankton dynamics and physical and chemical factors. Unfortunately this was not accomplished. Newell (1990) notes that, as of 1990, no nutrient transfer models (presumably including zooplankton) have been developed for the Bay. The total crustacean mean standing crop of 3700 individuals m^{-3} in the Bay (from March 1969 to February 1970) is not indicative of eutrophic marine ecosystems where numbers may be 10 or more times larger. A more complete review of water column trophodynamics is given in section 3.5.2.

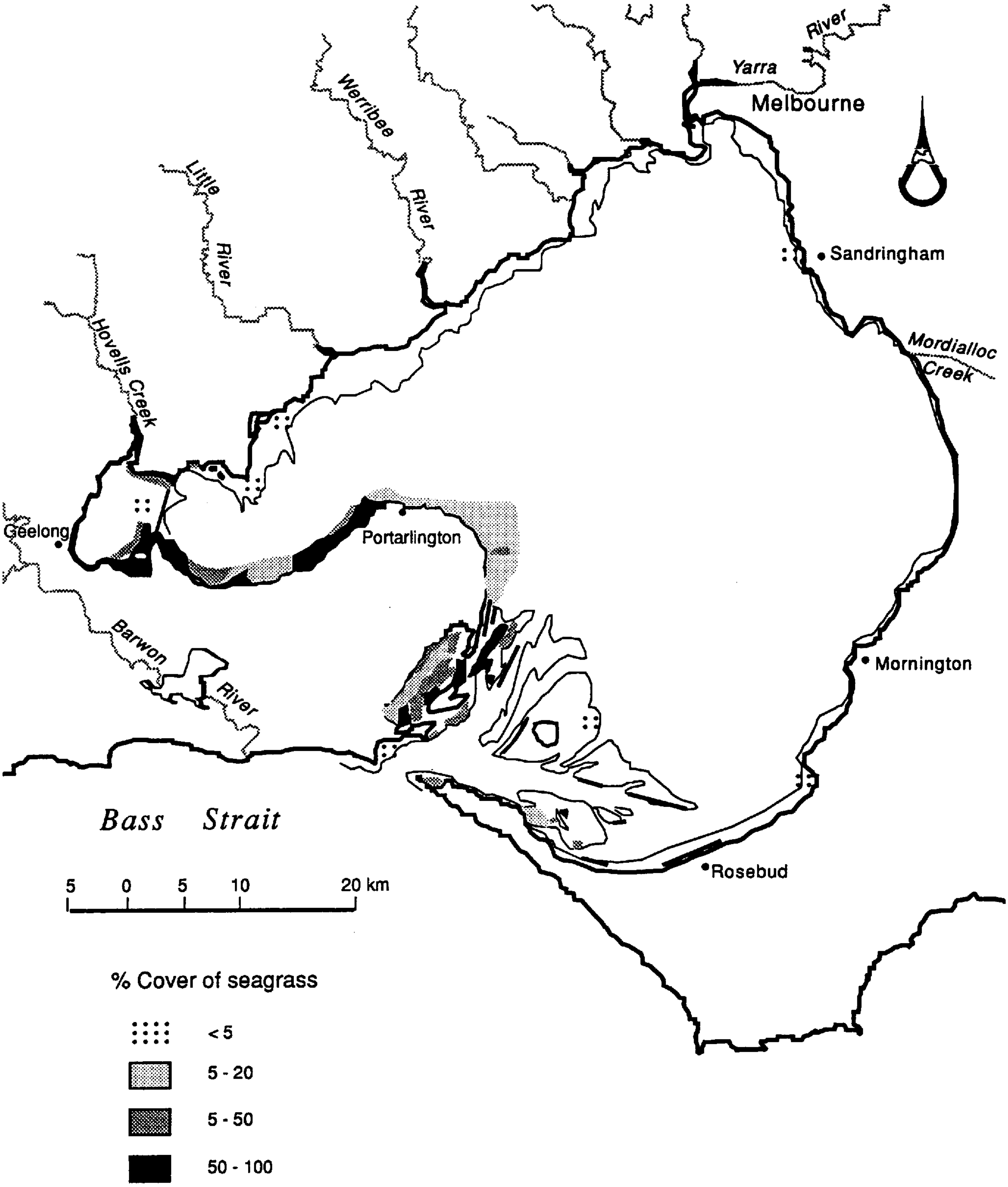

Figure 3.3.4.1 Distribution of the seagrasses in Port Phillip Bay, 1978-81 (after Balthius, 1981).

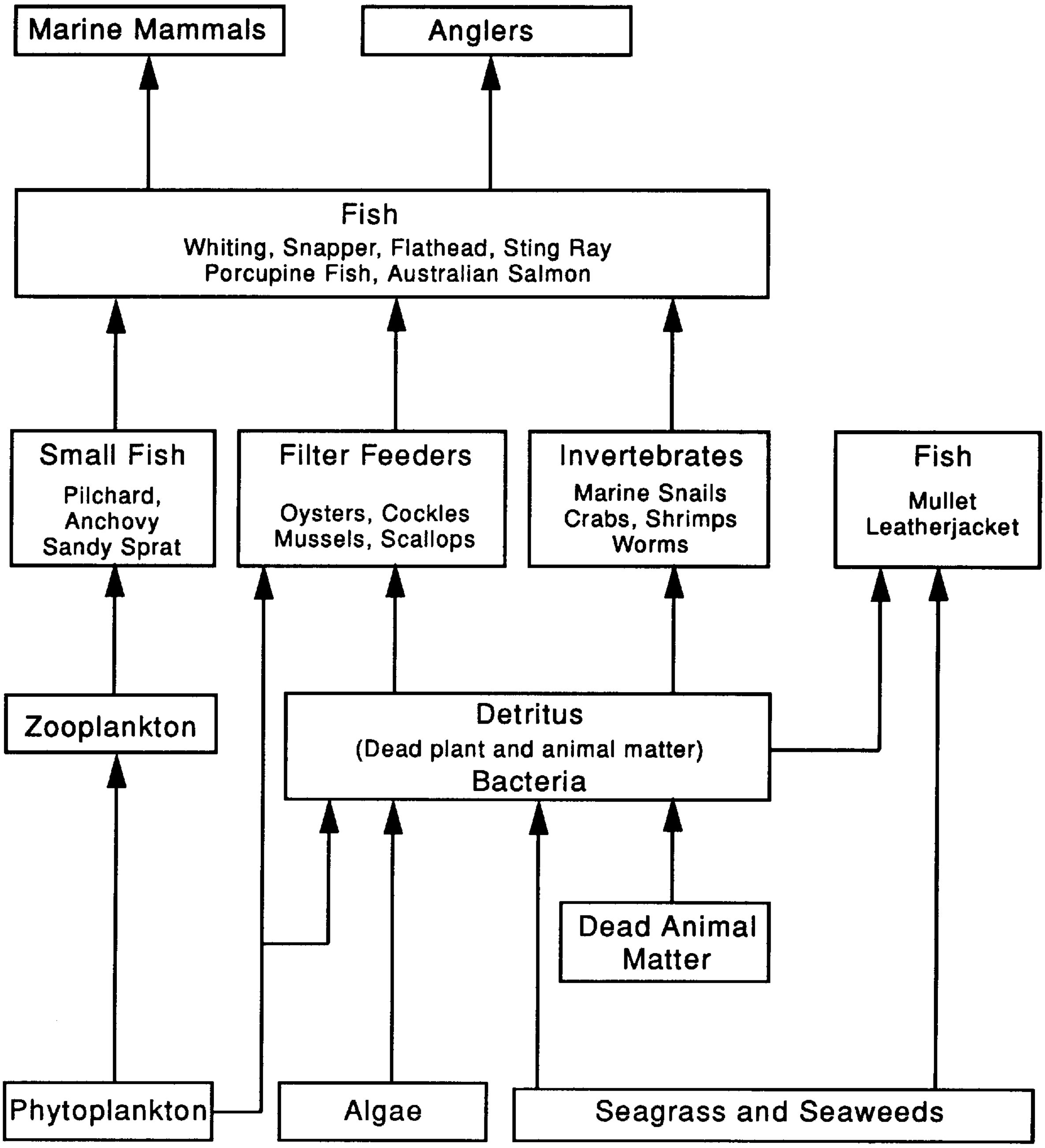

Figure 3.3.4.1 Port Phillip Bay food web; primary producers and consumers.
(from DCE, 1991).

3.3.5.3 Fish

A detailed review of the commercially utilized fish resources of the Bay is given by Nicholson (1992). The general biology (including feeding habits) of commercially important species such as flathead, snapper, King George whiting, pilchard and southern anchovy, southern calamari, blue mussel, scallop and abalone was presented. Production statistics and some preliminary biomass information were also presented. Nicholson (1992) records that the MSL are currently preparing biomass estimates for a variety of fish in the Bay.

Flathead are opportunistic feeders, living on crabs, shrimp, small fish and small invertebrates. Snapper feed mainly on small crabs, isopods, amphipods, polychaete worms, mussels and small fish. Whiting inhabit heavily to sparsely grassed patches and feed on copepods, amphipods and the ghost prawn. Pilchards and southern anchovy feed on copepods, ostracods, crustaceans, mollusc larvae, diatoms and other plankton; adult anchovy feed primarily on zooplankton. Calamari feed on small fish. Abalone inhabit reef environments and feed on red algae, drifting seagrass, detritus and lithophytic algae.

The blue mussel inhabit littoral and sublittoral parts of the Bay on a variety of substrata from compacted mud to rocks and dead shells. They feed on plankton suspended in the water. Scallops inhabit the sandy sediments of the Bay and - being filter feeders - also feed on the plankton in the water. Scallops are the only consumers for which biomass estimates are available; the 1992 population was estimated to be around 2.6 billion individuals (Gorfine *et al.*, 1992). However, the population is quite variable; from 20 million in 1982 to the current large numbers. Figure 3.3.5.2 shows the commercial scallop distribution in the Bay. The average annual production of scallops is about 3000 tonnes meat, wet weight. Assuming a dry weight percent ratio of 10C:3N:1P, the relative nutrient mass removed annually by scallop harvest is 120 tonnes C, 10 tonnes N and 3 tonnes P.. At a total population of four times the catch, the total amount of CNP in scallops Bay-wide is small compared to the total amount in the Bay. We estimate that a large scallop may pump around 20 L d^{-1}; three billion scallops would therefore pump around 0.1% of the total Bay volume each day and this could have significant local effects on nutrient concentrations in the sediments and water and phytoplankton standing crops. It has been estimated by Ward and Jernakoff (Chapter 5) from the scientific literature that filter-feeding molluscs may play a very major role in the trophodynamics of the Bay. Although no measurements have been made, filtering times for the volume of the Bay could be only 24 days.

There is so little available information about the standing stock biomass, and annual productivities of the primary and secondary consumers in Port Phillip Bay that it is not possible to construct comprehensive, Bay-wide nutrient budgets. Poore and Kudenov (1978) determined that mollusc suspension feeders constituted around 75% of the benthic biomass around a WTC drain.

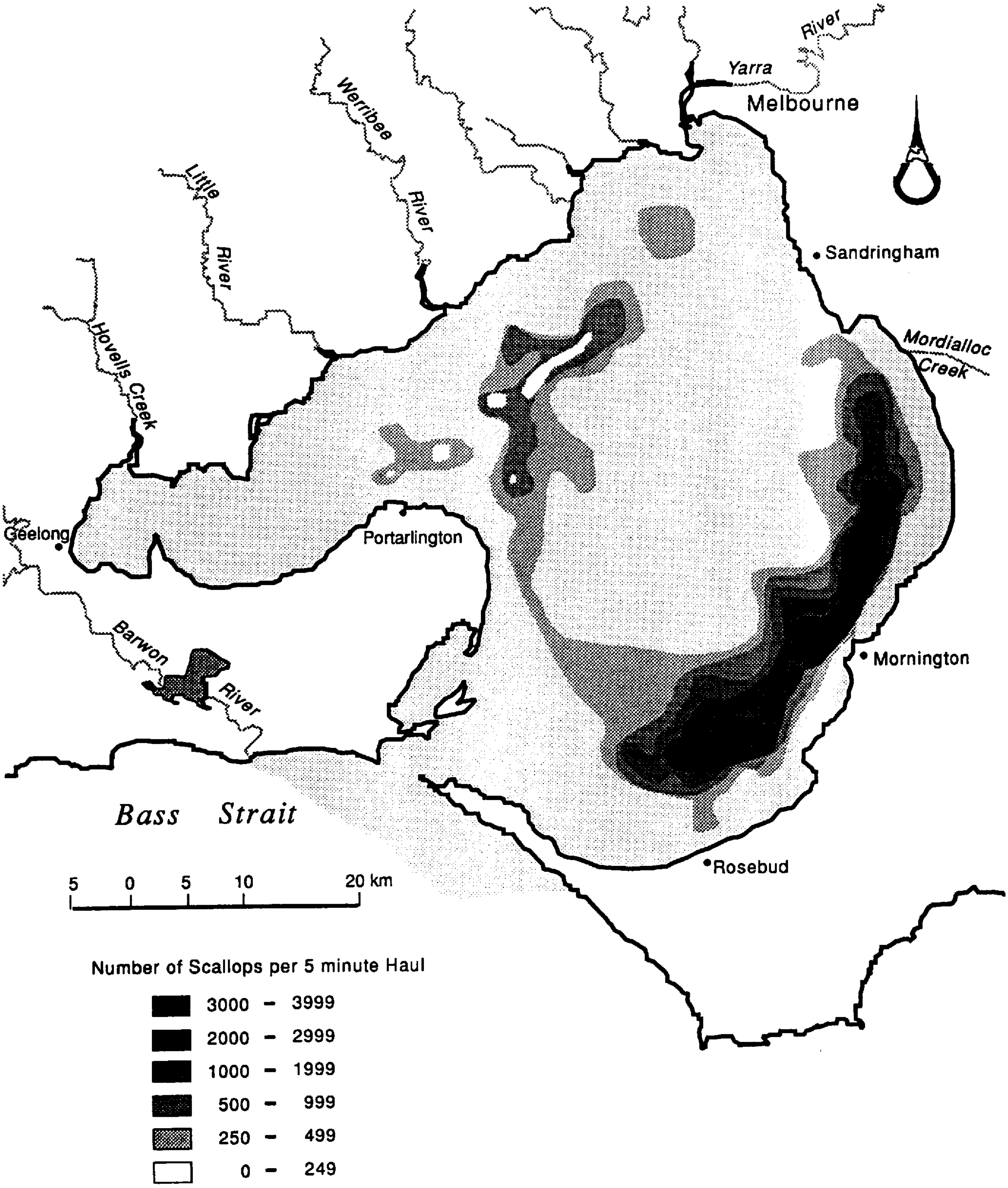

Figure 3.3.5.2 Distribution of commercial scallops in Port Phillip Bay. (after Woo and Woodburn, 1981).

3.4 **Nutrient dynamics**

3.4.1 **Nutrient recycling in coastal marine environments**

3.4.1.1. Nutrient recycling and dynamic processes

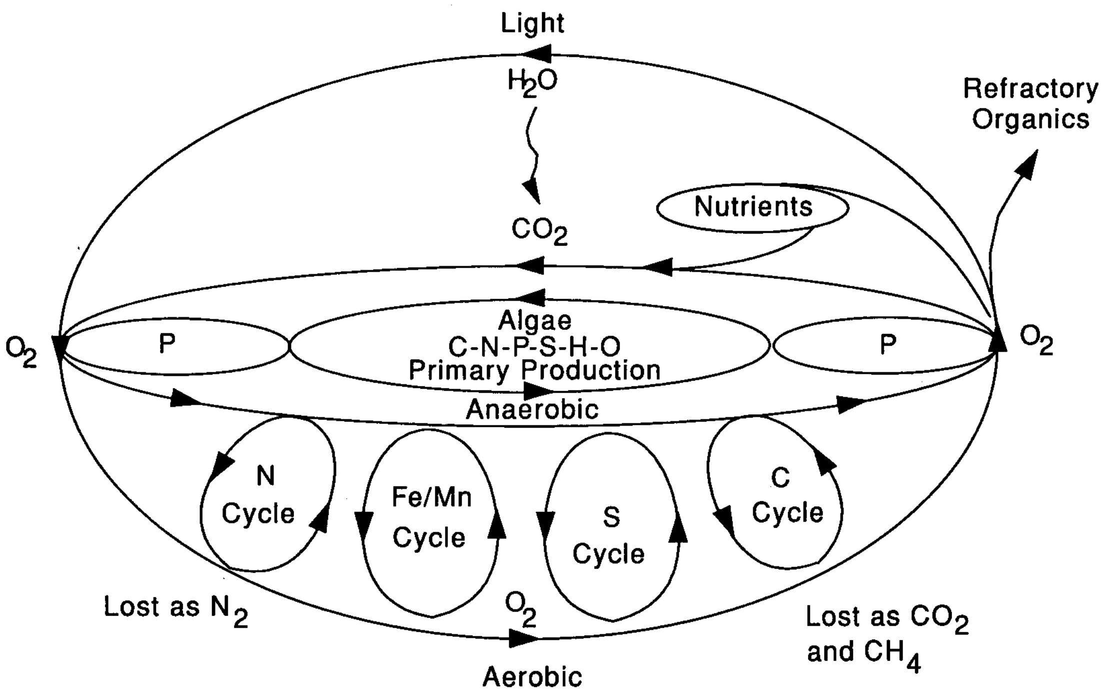

Figure 3.4.1.1 Nutrient cycling in marine and freshwater systems.

.Nutrient dynamics are concerned with the spatial and temporal relationships between the processes which transform and translocate nutrients. Figure 3.4.1.1 illustrates these relationships and process and the following equations illustrate the stoichiometry of the reactions:

(1) Photosynthesis:

$$106CO_2+16(NH_3)+H_3PO_4+122H_2O \rightarrow 106(CH_2O)16(NH_3)(H_3PO_4)+138O_2$$

(2) Oxygen reduction:

$$106(CH_2O)16(NH_3)(H_3PO_4)+138O_2 \rightarrow 106CO_2+16(NH_3)+H_3PO_4+122H_2O$$

(3) Nitrate reduction:

$$106(CH_2O)16(NH_3)(H_3PO_4)+94HNO_3 \rightarrow 106CO_2+55.2N_2+H_3PO_4+177H_2O$$

(4) Ferric iron reduction:

$$106(CH_2O)16(NH_3)(H_3PO_4)+424FeOOH+848H^+ \rightarrow 424Fe^{2+}\ 106CO_2+16(NH_3)+H_3PO_4+742H_2O$$

(5) Sulfate reduction:

$$106(CH_2O)16(NH_3)(H_3PO_4)+53SO_4^{2-} \rightarrow 53S^{2-}+106CO_2+16(NH_3)+H_3PO_4+106H_2O$$

(6) Methanogenesis:

$$106(CH_2O)16(NH_3)(H_3PO_4) \rightarrow 53CO_2+53CH_4+16(NH_3)+H_3PO_4$$

Primary producers (e.g. phytoplankton) synthesise biomass when nutrients and light energy are available (Equation 1). Photosynthesis is a reductive process in which the hydrogen of water is transferred to carbon dioxide with the concomitant synthesis of organic biomass (C-N-P-S) and the release of molecular oxygen. For oceanic phytoplankton the average atomic ratios of C:N:P:S are 106:16:1:0.7 (the Redfield Ratio; by weight, the ratios are 42:7:1:0.7). These ratios may vary considerably in coastal phytoplankton and benthic phototrophs (Harris, 1986; Atkinson and Smith, 1983).

A variety of heterotrophic microorganisms from zooplankton to bacteria consume or degrade the phytoplankton biomass, returning constituent elements as recycled nutrients to the environment. There is a much wider variety of benthic phototroph consumers.

This consumption/degradation process is generally referred to as "mineralisation" (Equations 2 to 6; Figure 3.4.1.1). Degradation of organic biomass is chemically oxidative and it may occur *via* aerobic (Equation 2) or anaerobic (Equations 3 to 6) biogeochemical processes. The major end-products of degradation are CO_2, fatty acids (acetate being predominant), ammonia, phosphate and sulfide. In an aerobic environment, organic carbon is oxidized to CO_2, ammonia is oxidised to nitrate by the nitrifying bacteria and sulfide is oxidized to sulfur/sulfate by the S-oxidizing bacteria. Methanogenesis is the dominant terminal degradative process in anaerobic freshwater environments (Equation 6). In anaerobic marine environments, the sulfate reducers are much better competitors for fatty acids than the methanogens (see Skyring, 1987). There is also a plentiful supply of sulfate in seawater favouring the sulfate-reducing bacteria as the dominant terminal carbon oxidizers in anoxic marine environments. Some methanogenesis occurs in marine sediments in situations where methylamines are available (see Skyring, 1987). In anoxic sediments, oxidised iron and manganese are also reduced; this may result in the solubilisation of some phosphate (Caraco *et al.*, 1989). N_2 fixation is also an important process in the cycle, returning to the biosphere the N which may be lost by denitrification (and ammonia by volatilisation if the pH is high). In a typical estuarine/coastal or oceanic environment, most (*ca* 70%) of the phytoplankton biomass and that of the primary consumers and other secondary producers is rapidly degraded. The remainder is degraded more slowly in the water column or sediments (Jackson, personal communication with Harris, 1986; Aller and Yingst, 1980; Westrich and Berner, 1984). The more refractory organic material may reside in sediments for long periods of geologic time or remain dissolved but undegraded in oceanic and coastal waters (Berner and Raiswell, 1983).

3.4.1.2 Nutrient cycling in the water column

The net production of phototrophic biomass in shallow marine environments dictates the trophic state. Harris (1986) discusses at length the concepts of "limiting nutrients" in both freshwater and the sea. He notes that there has been considerable confusion as to whether a nutrient limits the rate of growth or biomass production. A measurement of the concentration of a nutrient is not a sufficient test of whether or not it is limiting. The nutrient pool size and turnover rate are absolute requirements for investigating limitation phenomena. In general, growth rates in oligotrophic waters are high but an annual net biomass production is low. In a eutrophic environment, the absolute growth rate may be slower but the net biomass productivity higher. The high growth rates depend on very tightly coupled nutrient transfer processes and rapid recycling.

3.4.1.3 Nutrients in the sediments

The amount of biomass which settles to the sediment is exponentially and inversely related to water depth (Suess, 1980). In the deep ocean, very little of the phytoplankton reaches the sediment because most of the degradation occurs in the water column. Except for the dark respiration of primary producers, oxidative processes in the water column are usually aerobic with molecular oxygen being the oxidant (Equation 2). Aerobic oxidation may also occur in the surficial sediments. Ammonia is oxidized to nitrate which effluxes from the sediment into the water column and phosphate remains as precipitated iron phosphate (Caraco *et al.,* 1990); in some instances apatite is formed (Heggie *et al.,* 1990). If biomass production is high, the rate at which algal biomass and detrital organic matter settles to the sediment is high and degradation rates are fast. Ultimately the rate at which O_2 diffuses into the decaying material becomes the limiting process and anaerobic bacteria overtake the degradative process (Figure 3.4.1.2). Diagenetic processes in the sediments may be significantly modified by turbulence due to wind and tidal currents and bioturbation.

It is now known that for most coastal marine sediments, anaerobic processes play a significant and often dominant role in the dynamics of nutrient transformations and translocations (Skyring 1987). Sulfate reduction is quantitatively the most important terminal heterotrophic oxidative process in most coastal sediments. This is because the amount of degradable organic matter deposited during sedimentation is generally high, resulting in high rates of heterotrophic microbial activity and anoxia in the surficial sediments. Sulfate concentrations in seawater are high (about 100 mg SO_4 L^{-1}) and this favours the sulfate reducers. Bacterial sulfate reduction is a respiratory process producing stoichiometric quantities of sulfide from sulfate; ammonia and phosphate are also produced in stoichiometric quantities with respect to the C:N:P ratios of the source material. A positive flux of phosphate to the water column may be maintained by the precipitation of reactable iron as sulfides with the concomitant release of phosphate from iron phosphates into sediment porewater and bottom waters (for this reason, there is always enough orthophosphate available to phototrophs in coastal marine environments (Caraco *et al.,* 1990)). If surficial sediments are anaerobic a positive flux of ammonia into the water may occur. In a badly eutrophied state, sulfide may also efflux from the sediments into the water column.

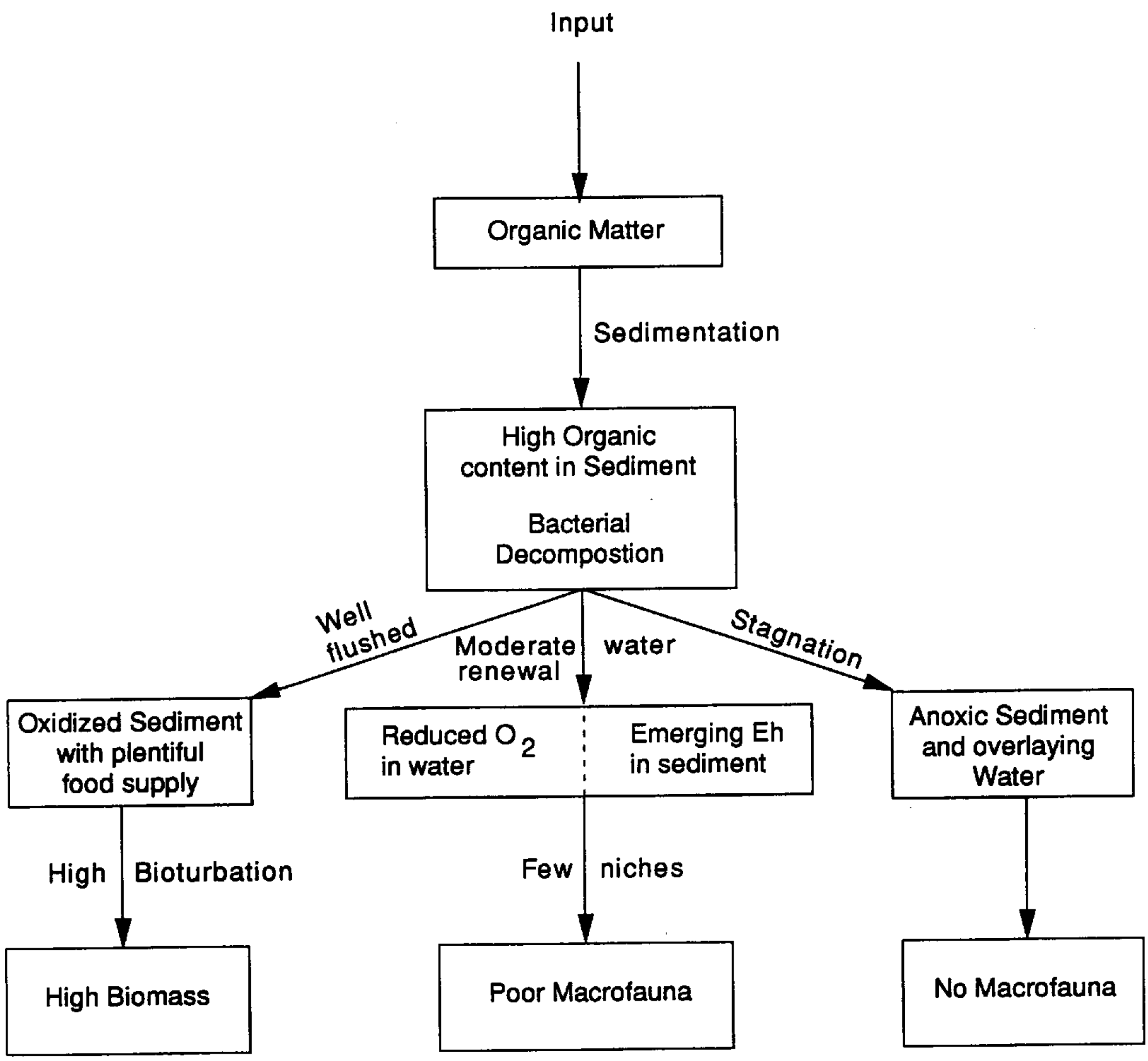

Figure 3.4.1.2 Pathways of organic matter degredation in marine sediments.
(from D McLusky, 1991).

3.4.2 **Nutrient cycles in Port Phillip Bay; a Bay-wide view.**

3.4.2.1 The Phase 1 Study

The Phase 1 Report (MMBW/FWD, 1973) describes the adaption of a hydrodynamic model for Port Phillip Bay for modelling water quality which is pertinent to nutrient dynamics. Evaporation was considered equal to freshwater input. The major hydrodynamic components are defined as the Mixing Zone, the Central Zone and the Exchange Zone (Figure 3.4.2.1). Although chloride and orthophosphate did not show the same concentration distributions, orthophosphate was considered to be conservative and a useful constituent to "test" the hydrodynamic/water quality model. Chloride was not a significant input from terrestrial sources. Sediment loads and sediment nutrient fluxes, atmospheric inputs and groundwater inputs were not included. However, a "loss rate coefficient" was included to account for the combined effects of theoretical "sedimentation, biological uptake, chemical reactions etc". The model was run with values of the loss coefficient set at 0.05, 0.15 and 0.3. Losses due to tidal exchange were excluded. In the model structure, the Bass Strait orthophosphate concentrations were also set at zero. To compensate for the actual concentration of 25 g PO_4 L^{-1} (8 g P L^{-1}), this concentration was subtracted from all the Bay station concentrations to obtain the excess concentrations of orthophosphate.

The dispersion rate, used to describe the mixing and diffusion of phosphate from inputs to the Exchange zone, could not be measured directly; it was calculated as a coefficient of 0.48 when the equilibrium mass for orthophosphate was the same as the observed load to the Bay. Phase 1 did not include sediment/water fluxes and the model was not extended to nitrogen.

The Phase 1 Study identified nine environmental zones (Figure 3.4.2.2):

- Central Zone
- Exchange Zone
- Geelong Channel
- Corio Zone
- Werribee Zone
- Altona Zone
- North-Eastern Zone
- Eastern Zone
- South-Eastern Zone

The South-Eastern Zone is part of the Central Zone and all of the other Zones (except the Exchange Zone) constituted the Mixing Zone; the actual physical Mixing Zone has a lower surface area than the integrated areas of the coastal environmental zones (cf., figures 8.14 and 10.1 in the Phase 1 Report). This is also probably true of the physical Exchange Zone. The EPA have also defined the boundaries of "sectors or segments" which reflect the areas of the Bay important for regulatory and compliance monitoring (Figure 3.4.2.3).

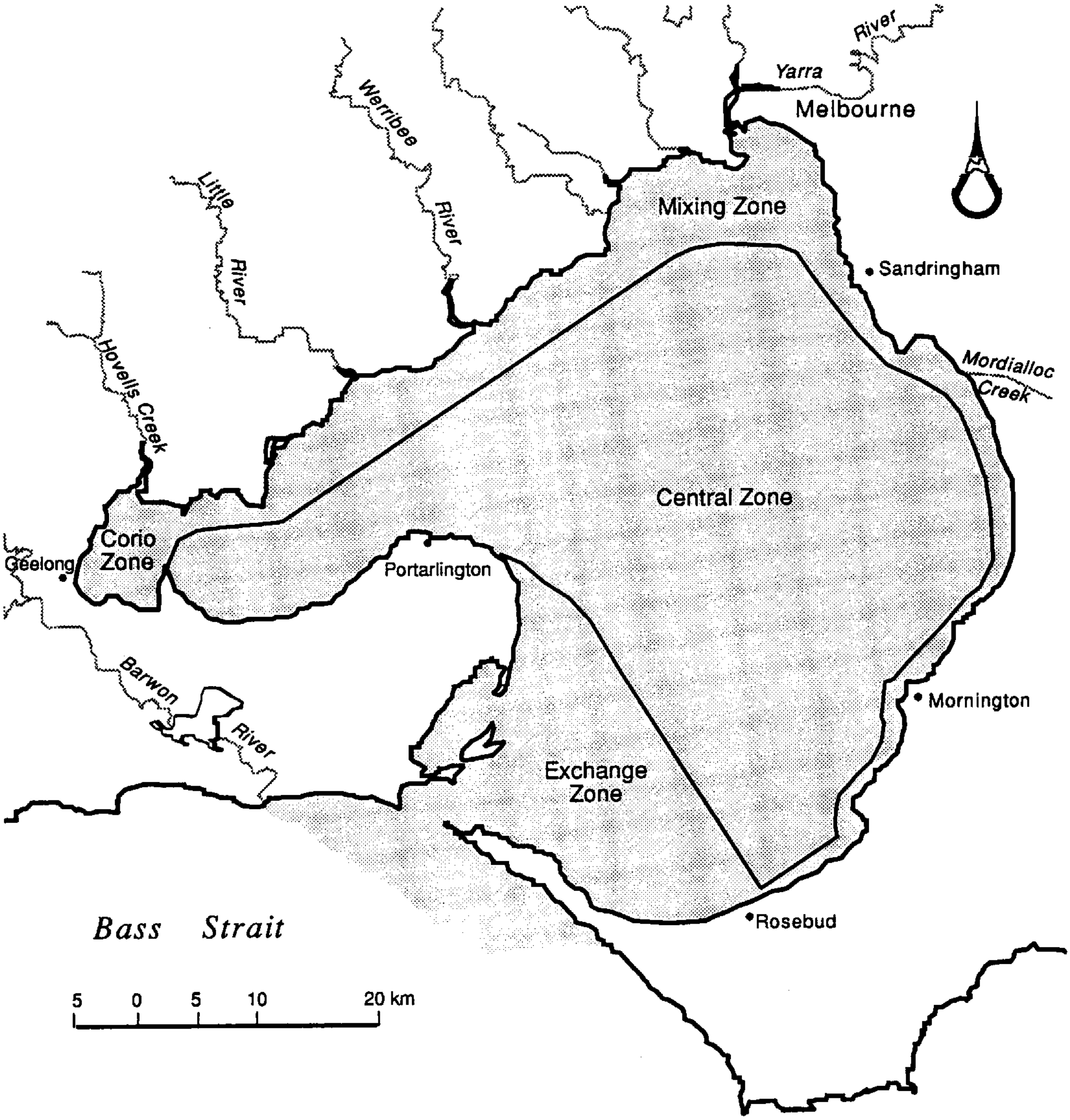

Figure 3.4.2.1 Chemical Quality Zones.
(from MMBW and FWD, 1973, figure 8-14, 1979).

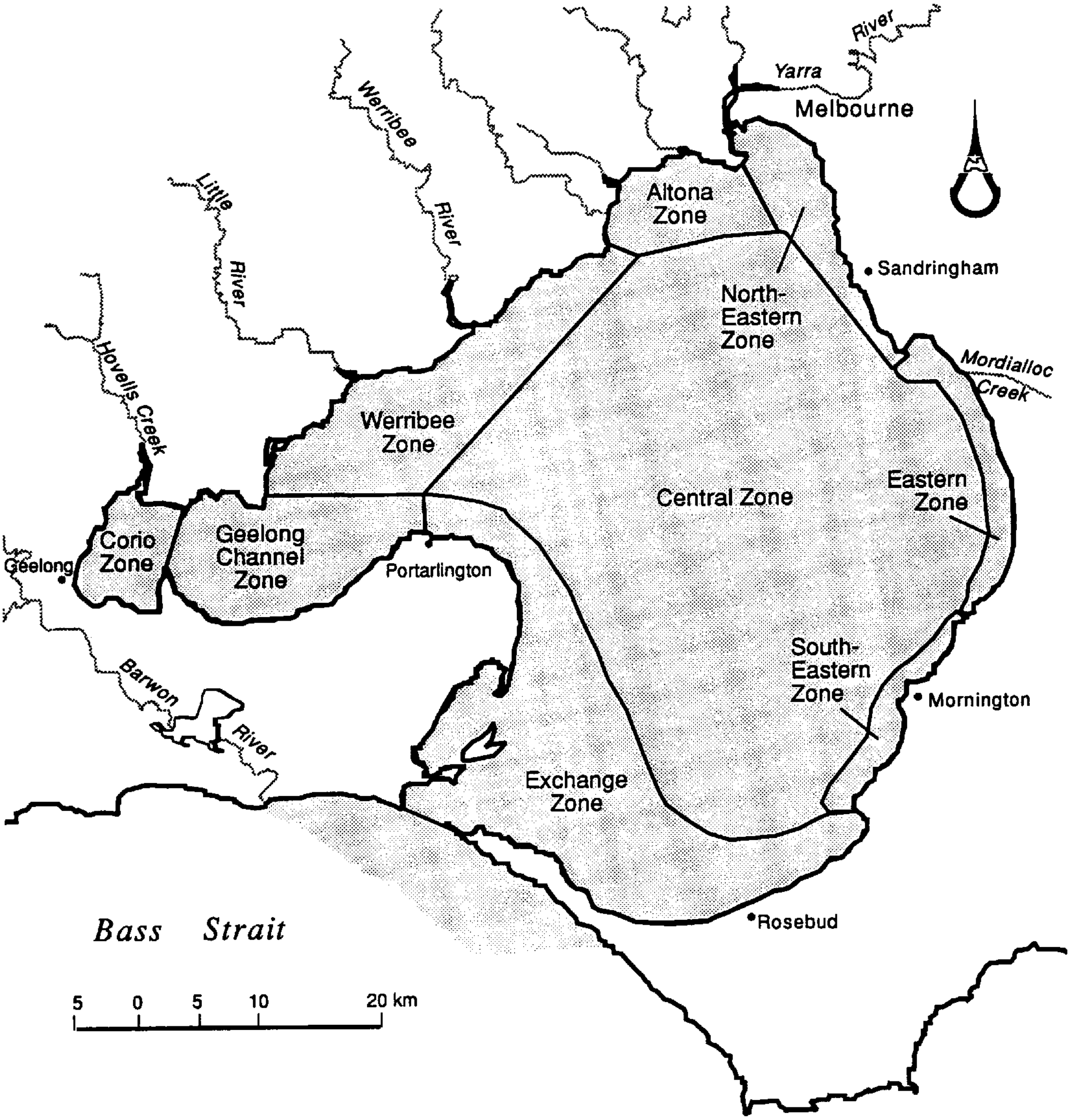

Figure 3.4.2.2 Environmental Zones.
(from MMBW and FWD, 1973, figure 10-1).

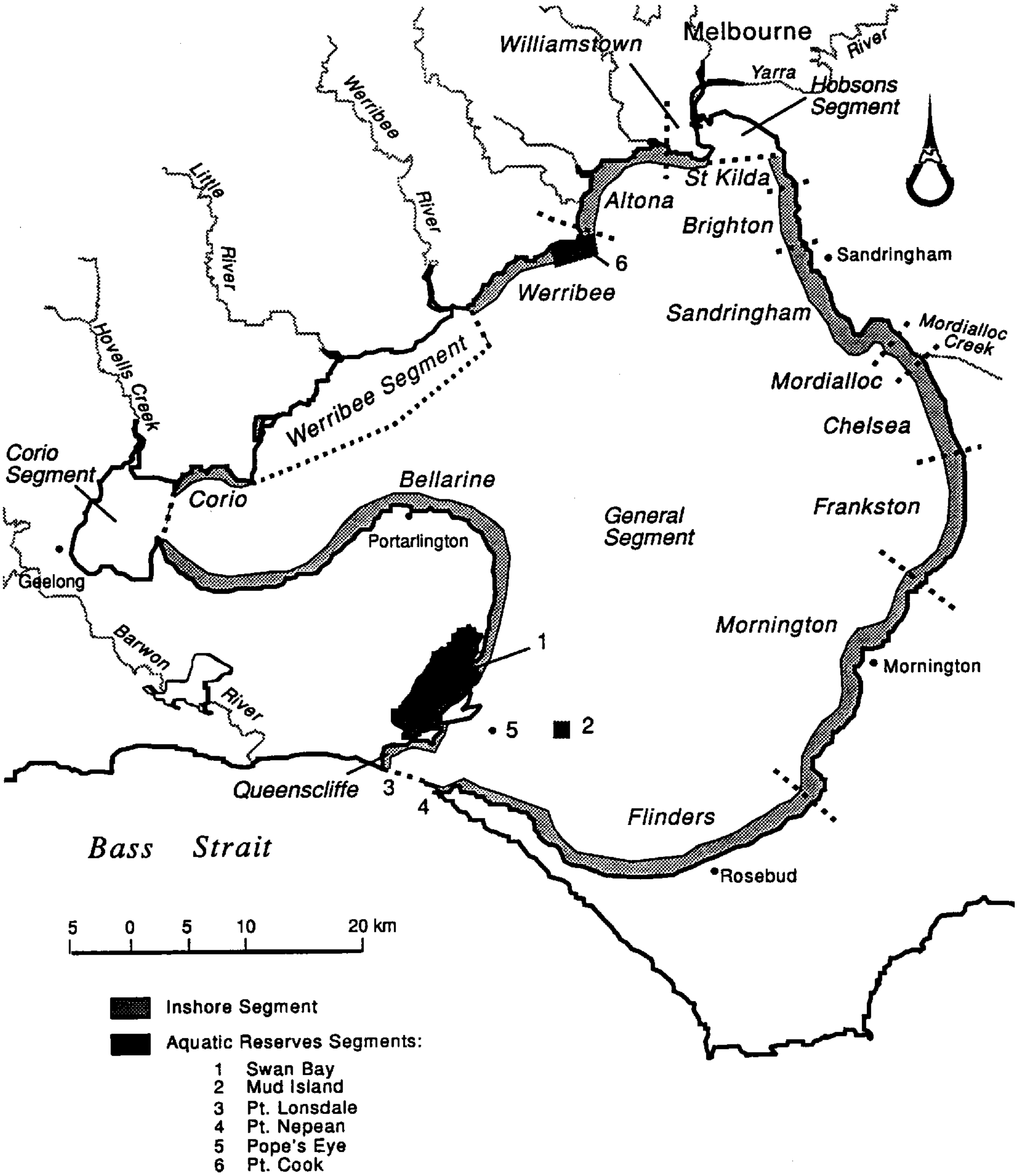

Figure 3.4.2.3 Segment boundaries of Port Phillip Bay.
(from from EPA, 1991a).

The model simulations were for 3 hydrodynamic cases and 30 water quality cases with 7 arrays for varying input conditions (Phase 1 Report, Table 6.2). The annual urban and agricultural runoff was held constant at 513 tonnes P for all arrays, trade wastes at 280 tonnes P for arrays 2 to 7, sullage at 440 tonnes P for arrays 3 to 7, and the treatment plant inputs at 1212, 2424 and 4844 tonnes P in arrays 4 to 7; it is not immediately apparent why arrays 5 and 6 were identical (Phase 1 Report, Table C.3). Bay-wide pool sizes of the major P nutrients were calculated by integrating the zone pool sizes which were determined separately. The main results of the orthophosphate water quality modelling in the Phase 1 Study were:

- the location of the input point or zone was relatively unimportant because mixing with the Central Zone is rapid

- there was little effect over a tidal cycle except in areas close to shore

- the equilibrium mass is different for different inputs but the distribution patterns remain the same

- new equilibrium states are slow to develop; the residence time for orthophosphate in the Central Zone is around 12 months

- the tidal exchange ratios of 0.5 and 0.9 caused only a 10% difference in equilibrium mass which was only slightly affected by wind direction because the main inputs are located in the north-western and northern parts of the shoreline

- the predicted Bay-wide distribution of orthophosphate inputs to the Bay agreed reasonably well with the measured concentrations over two tidal cycles. However, there were significant differences between actual and predicted values in the coastal sectors

3.4.2.2. Phase 2 studies and other investigations

Williams' (1978) principal contribution was to develop the first integrated model of nutrient distribution and cycling in the water column, based on a modern finite-difference circulation model, and including cycling through phytoplankton and zooplankton. He concluded that existing (i.e., for 1975 and including the Phase 1 models) circulation data were insufficient to verify water quality models of the Bay. Williams' work has not been frequently referenced in later model assessments probably because his major conclusions were concerned with the economics of water treatment (Black and Mourtikas, 1992).

Dr R.J. Lukatelich (Kinhill Engineers Pty Ltd, 1990) considered the assimilative capacity of Port Phillip Bay. From studies in the United States in which a loading of 0.75 g P m^{-2} y^{-1} is the maximum permissible to avoid eutrophication in coastal waters, he calculated that an acceptable maximum nitrogen load for Port Phillip Bay is 5.4 g N m^{-2} y^{-1} (10500 tonnes N). He estimated the annual loading to the Bay at 3.75 g N m^{-2} y^{-1} (or 7300 tonnes N). He did not use the water quality model described in the Phase 1 Report (Chapter 8) but recommended that an empirical loading-trophic status model be developed for the Bay and that accurate hydrodynamic models be used to determine the flushing characteristics of the Corio, Werribee and Hobson's segments.

We have estimated the current annual N load to the Bay at 9085 tonnes, not much less than the upper maximum 'safe' load suggested by Lukatelich.

Newell (1990), Longmore (1992) and Chiffings *et al.* (1992) have more recently reviewed most of the available data pertaining to nutrient dynamics in Port Phillip Bay. Brown and Davies (1991) have reviewed the environmental quality of north-western Port Phillip Bay and briefly discussed nutrients. They reviewed the data mainly with respect to compliance with EPA water quality objectives. This publication is most valuable for the amount of information contained in 60 Tables and 114 Figures. They also review the statistical analyses which have been done on a large amount of the Bay data. The reviews by Newell and Longmore deal more specifically with data analysis and data interpretation on a Bay-wide basis; Chiffings and his colleagues were concerned more with the management of nutrient loads.

In his comprehensive review on nutrient budgets and dynamics of Port Phillip Bay Newell concluded the following:

- Port Phillip Bay is in an oligotrophic state and N is probably the limiting nutrient (most of his comments are therefore concerned with N budgets and dynamics)

- the effects of the WTC are restricted to the nearshore Werribee part of the Bay

- nutrient inputs into the periphery of the Bay are rapidly mixed into the Central Zone, passaged through the biota and then exported to Bass Strait *via* the Exchange Zone

- rapid dilution of nutrients by hydrodynamic mixing toward the centre of the Bay and vertical mixing with some light attenuation effects (with some reservations about the latter), may be the major factors limiting phytoplankton growth, particularly in the nearshore Mixing Zone

- the recycling of N-nutrients by phytoplankton is limited by the formation of refractory dissolved organic nitrogen (DON) by phototrophs (and presumably some benthic biota).

- the combined productivity of the benthic epiphytic microalgae, the macrophytic algae and the seagrasses is relatively small, at most 10% of the total primary productivity of the Bay; by inference therefore, phytoplankton are of significance as the major primary producers in the Bay.

3.4.2.2.1. Phytoplankton

Newell's conclusions indicated that nutrient dynamics in the Bay are in steady state. N inputs are balanced by losses to the atmosphere and export to Bass Strait and there are no significant recycled nutrient sources or sinks and primary productivity in the Central Zone poises the nutrient dynamics of the Bay. Newell also calculated from the gross primary productivity data (38 g C m^{-2} y^{-1}) of Axelrad *et al.* (1979) that there were five major cycles of phytoplankton growth each year involving the turnover of 20000 tonnes N. The latter workers used a short incubation period of 2-3 hours in the presence of radioactive CO_2.

Actually, the 20000 tonnes of N calculated by Newell was an over-estimate for two reasons. The figure given by Axelrad *et al*. for Station 3 in the Central Zone was 33 g C m^{-2} y^{-1}. Correcting Newell's estimate for this and using the Redfield ratio (106C:16N:1P), around 10400 tonnes of N would seem more likely and the number of "equal" growth or turnover cycles to be around 50. Newell erroneously calculated the turnover by dividing the N flux through the phytoplankton by the N pool in the Bay instead of the N pool in the phytoplankton. The data from Axelrad *et al*. (1981) indicated the possibility of five temporally distinct 'blooms' in the Central Zone (represented by one station); more recent work indicates that there may be up to ten blooms per annum (Longmore, unpublished). The ratio of 50:1 for C:Chl *a*, which we and the authors of Phase 1 used for estimating the size of C:N:P in the phytoplankton of the Bay, may not in fact reflect the *in situ* phytoplankton composition.

As Harris (1986) notes, daily gross productivity rates in oligotrophic waters may be quite high due to a rapid recycling of nutrients in the vicinity of aggregates of phytoplankton and grazing zooplankton. The pool of organic carbon in the Central Zone is around 44000 tonnes (Table 3.3.1). If the ratio of C:Chl were 250 and all of the organic C in the Central Zone were phytoplankton biomass, then the standing crop of phytoplankton chlorophyll *a* computes to 7 g L^{-1}; this is 6 to 7 times the actual average concentration. On the basis of this calculation, around 85% of the organic carbon in the waters of the Central Zone is not associated with chlorophyll. If the calculation were made on the basis of a C:Chl ratio of 40, the standing crop of phytoplankton chlorophyll *a* computes to 45 g L^{-1}. This situation is most unlikely. Between the possible extremes, it appears that 85 to 97% of the organic carbon and nitrogen in the Bay is not associated with chlorophyll.

3.4.2.2.2. Benthic phototrophs

Benthic primary productivity has been investigated in any detail only in the Werribee Zone (see Axelrad *et al.*, 1981; Brown and Davis, 1991; Newell, 1990). The standing stock of *Caulerpa* is estimated at 1458 tonnes and the net productivity at 2.3 tonnes C y^{-1}. Gross productivities or turnover data are not available. Assuming a C:N weight ratio of 10:1 about 145 tonnes N is contained in the *Caulerpa* biomass in the Werribee Zone. This is a substantial 60% of the N estimated for the standing crop of phytoplankton biomass in the Bay (previous paragraph). Brown and Davis (1991) note that the degradation of *Caulerpa* biomass may be a slow process.

3.4.2.2.3. Sedimentation

The few sediment data which were available to Newell (Stanley, 1977) indicated that the organic carbon concentrations in the Werribee Zone were generally low (<1% organic C; certainly well below eutrophic levels) and that the flux of organic matter to the sediment was low. Newell concluded that the sediments are not "threatening" as a large nutrient pool which could be released into the water column by some catastrophic event. Although Newell did not elaborate, the information on suspended solid inputs to the Bay from the Phase 1 Report indicates that sedimentation in the Central Zone must be extremely slow at less than 1 mm y^{-1} (there is no information on sediment import from the Exchange Zone). This would imply that preservation of sizeable organic-N pools in the sediment would also be a slow process, consistent with Newell's N model for the Bay.

Phase 2 nutrient/sediment studies were concerned mainly with the Mixing Zone offshore from Werribee and the Exchange Zone in the St. Leonards area (Kelly *et al.*, 1987). It is clear from these studies that the N nutrient (mainly ammonia) from the WTC has a dramatic effect on nutrient cycling in the Werribee sector and Newell accounts for the nitrification, denitrification and mineralisation rates in his construction of the N budget for the Bay. There are no data on N_2 fixation processes or rates in the Bay.

3.4.3 **Other factors affecting phytoplankton in Port Phillip Bay**

3.4.3.1 Light

One of the apparent enigmas in Port Phillip Bay is that the productivity (inferred from the standing crop) of the phytoplankton is not commensurate with the supply of readily available ammonia and orthophosphate from WTC and nitrate from rivers and drains; unusually low temperatures do not appear to be a serious limiting factor. The waters of the Bay are generally considered to be turbid and light attenuation has been suggested as a limiting factor to phytoplankton growth. However, Newell (1990) notes that the Secchi disc data indicate that only a few metres of the deepest part of the Bay would be without adequate light for growth. He suggested that much of the algal population spends time in poorly illuminated bottom waters. However, this argument does not hold up well for the inshore zones. Also, both particulate organic nitrogen and phosphorus correlated with the seston concentrations during the Phase 1 studies. Wohlers (1991) statistically analysed a large data set from studies conducted by the EPA from 1980-84 (monthly sampling) and from 1985-86 (quarterly sampling) and found Secchi disc depths to correlate negatively and significantly with nutrient concentrations. For example, in the Central Zone the correlation coefficients for Secchi disc depths versus total P, orthophosphate and chlorophyll *a* were -0.46, -0.54 and -0.62 respectively. He reported that this result was to be expected because visibility would tend to decrease with increasing nutrients, implying an increase in suspended biomass due to primary productivity. His argument seems reasonable unless the light attenuation was due to particulate matter other than phytoplankton.

3.4.3.2 Iron

Table 9-10 in the Phase 1 Report records that the addition of only 3 g Fe and Mn (plus excess nitrate) stimulated algal growth in 7 out of the 17 test flasks. More recently, Mickelson (unpublished) has determined that the waters of Port Phillip Bay are probably not trace metal deficient but that Bass Strait waters are manganese deficient. Also, from a review by Dring (1982), Wood and Beardall (1992) concluded that micronutrients would not be limiting in Port Phillip Bay. Ellaway *et al.* (1980) showed that there were high concentrations of soluble iron in the bottom waters of the Yarra estuary. They concluded this was due to the solublization of iron in the anoxic sediment of the estuary. Large quantities of orthophosphate are discharged in the northern coastal zones where it is expected that the major inputs of iron would occur from the Yarra River. Any reactive iron would be rapidly precipitated as insoluble hydrated iron phosphates in the presence of excess orthophosphate of 80 to 160 g P L^{-1}. Is it possible that the waters in the mixing zones are depleted in available iron to such an extent that the standing crop of phytoplankton is limited by iron availability?

3.4.3.3 Silica

The studies in Phase 1 established that diatoms were the predominant phytoplankton species in Port Phillip Bay and current MSL studies indicate that this is generally the case. Silica must be considered therefore as an essential nutrient. The annual average concentrations of SiO_2 were 0.17 (1969) and 45 mg L^{-1} (1970) respectively. The higher concentrations in 1970 were in response to high freshwater inflows. Silica is taken up by diatoms in about the same ratio to P as is N, so Si could not be considered as limiting phytoplankton productivity in Port Phillip Bay. The silicate frustules of diatoms which are incorporated into the sediment undergo solubilisation and 50 to 90% of the Si is returned to the water column (Ohler 1979).
Gorfine and Abbott (1992, unpublished) showed that silicate was effluxed from most sediments in the Bay at rates from zero to 72 +/- 7 mg Si m^{-2} d^{-1}.

3.4.4 **A closer examination of models for Bay-wide nutrient dynamics**.

As a first approximation of the N budget in the Bay, Newell's innovative interpretation seems plausible and while most of the interpretational points which he discusses are mentioned in the Phase 1 Report and Phase 2 studies, his review synthesised a perceptive nutrient model for the Bay which was not previously available.

Longmore (1992) examined the data more in terms of compliance with the State Environment Protection Policy (SEPP) and "better indicators" of water quality. However, in taking this approach, he also had to develop a conceptual nutrient model for the Bay. He concluded that, from the current data and statistical analyses, it was not possible to detect significant changes in the nutrient status of the Bay which may have occurred over the last 15 years (this is post the reduction of the nutrient load from the WTC in 1975-76). His assessment also indicated that nutrient dynamics of the Bay are in probably in a net steady state. This is for the most part similar to Newell's version of nutrient dynamics in Port Phillip Bay. However, Longmore did not rule out the notion that the Bay-wide sediments could be nutrient sinks. This is the major difference between Newell's and Longmore's interpretation of essentially the same data. Newell's conceptual model exports all nitrogen (as the important limiting nutrient) and Longmore's model is that significant nutrient N pools could be sequestered in sediments. The comparative data for the 1970s and 1990s are presented in Figures 3.4.4.1 and 3.4.4.2 respectively. The difference between the annual input of 9085 tonnes N and the average pool size of 4300 tonnes TN (Figure 3.4.4.2) must be balanced by the N sequestered in biomass, the N lost to the atmosphere by denitrigication and ammoniavolatilization, sedimentation and export to Bass Strait. Denitrification is the only measured N loss and this accounts for around 20% of the difference between input N and the average TN pool in Port Phillip Bay.

Longmore's (and Gorfine and Abbott's) calculations show that the Bay-wide sediments constitute a sizeable N pool and also that the efflux of N into the water column is significant with respect to input loads, Bay-wide pool sizes and dynamics. The new information from these studies is as follows:

- there are 10900 tonnes of total P and 25000 tonnes total N sequestered in the upper 2 cm of Bay-wide sediment

- sediments from Corio Bay, Hobsons Bay and the Central Zone had organic carbon contents of between 3% and 5% in the upper 2 cm

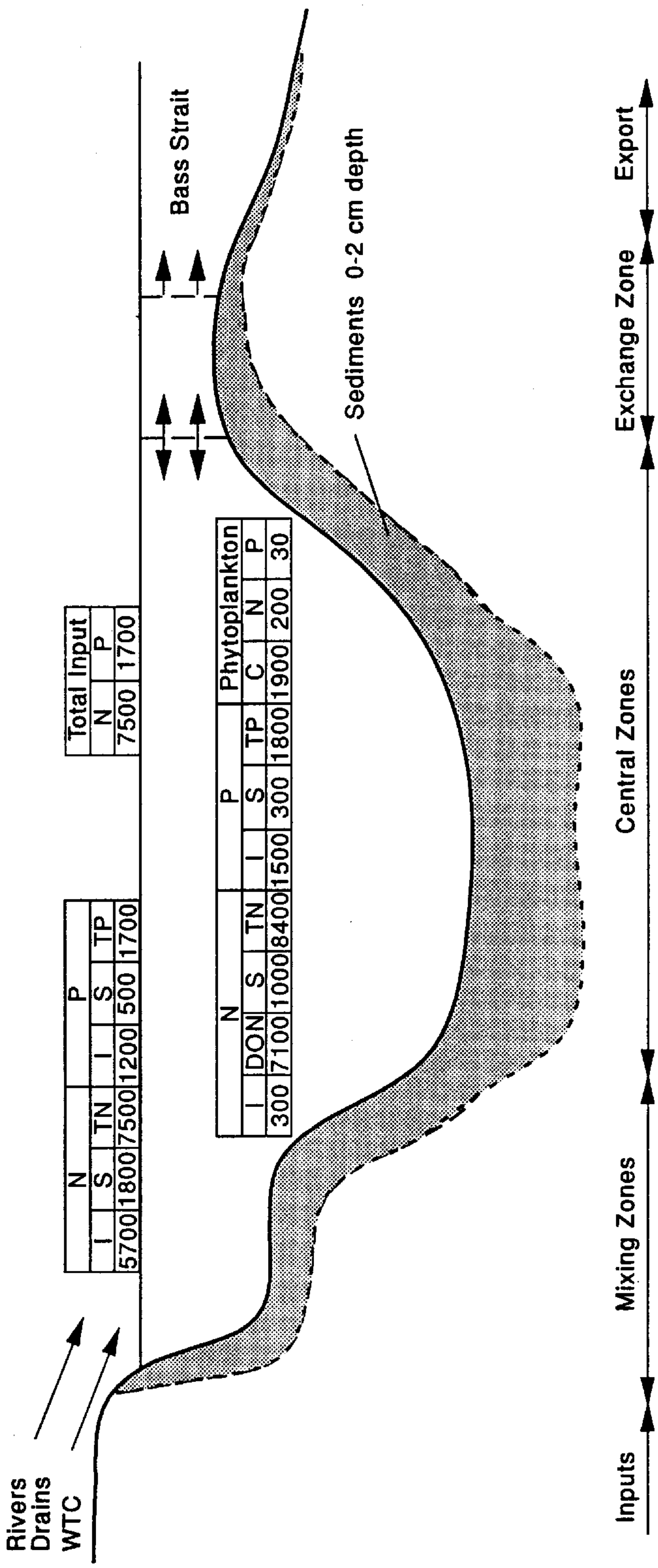

Figure 3.4.4.1 Measurements (tonnes) of annual nutrient imputs and nutrient pool sizes in Port Phillip Bay, 1969/70

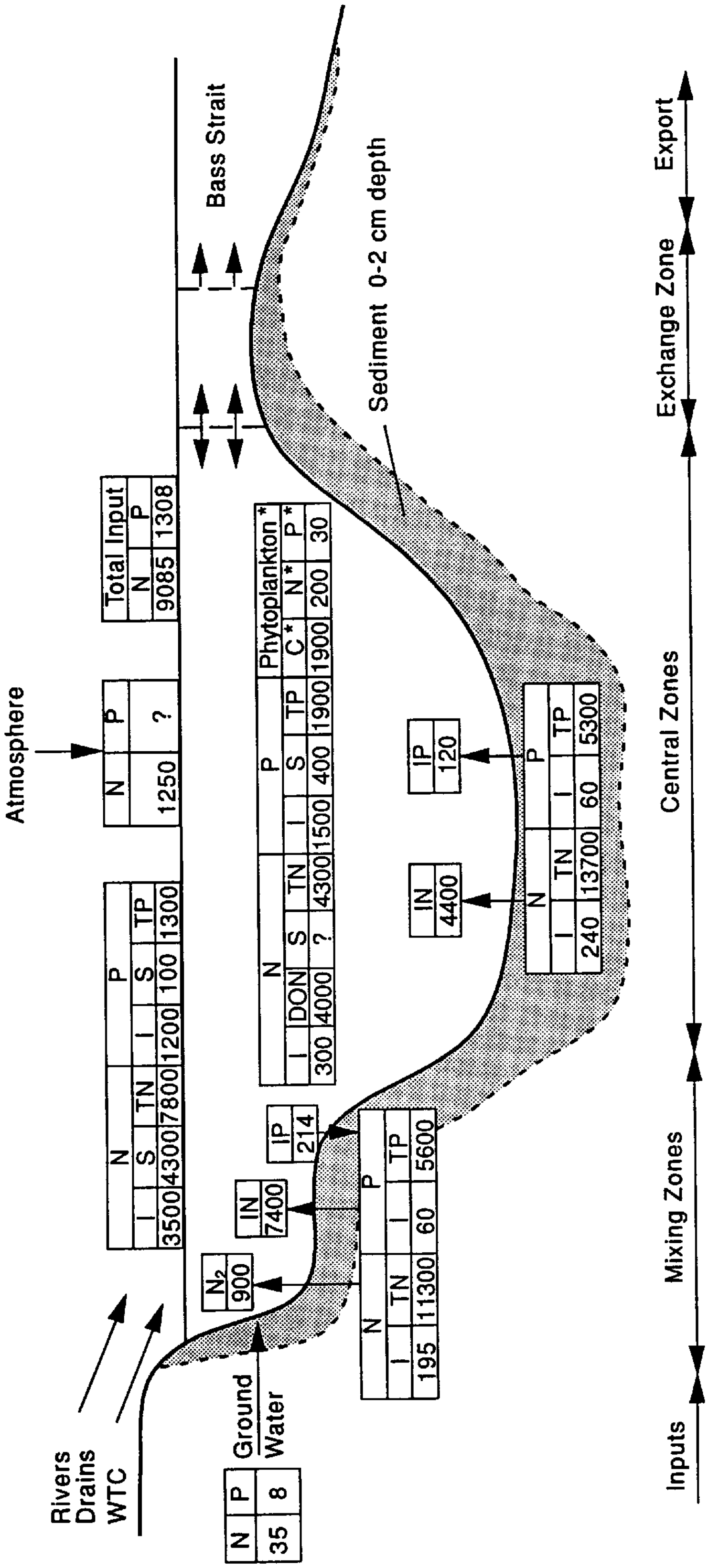

Figure 3.4.4.2 Measurements (tonnes) of annual nutrient imputs and losses and nutrient pool sizes in Port Phillip Bay, 1990/92. (refers to Central Zone only)

- there is a Bay-wide net annual influx of around 200 tonnes P for orthophosphate and an efflux of 11800 tonnes DIN for N
- there is an efflux of 120 tonnes of orthophosphate-P from the sediments of the Central Zone and an influx of 214 tonnes P to the sediments in the Mixing Zone

- the efflux of 4400 tonnes N from the Central Zone sediments is made up of 1800 tonnes NO_3-N and 2600 tonnes NH_3-N

- the efflux of 7400 tonnes N from the Mixing Zone sediments is mainly (82%) NH_3

- the fluxes of N and P from the sediments of the Central Zone are quite low, comparable to oligotrophic coastal environments.

The organic content of 3-5% for the sediments of the Central Zone (Gorfine and Abbott, unpublished) is higher than that expected for oligotrophic, shallow marine environments. Low oxygen concentrations have been reported occasionally for the bottom waters of the Central Zone and this is also unusual for an oligotrophic environment. However, these sediments did not have the high oxygen demand which was characteristic of several of the nearshore sediments. The lower degradation rates may indicate an older, more refractory organic material than that in the nearshore sediments. Gorfine and Abbott (1992) suggest that this conundrum may be due to inherent depositional conditions rather than different respiration processes.

The potential loss of significant amounts of ammonia by volatilization has not been considered in any of the reviews on nutrient dynamics in Port Phillip Bay. However, the partial pressure of dissolved ammonia gas is related to the ammonia concentration by Henry's Law. At pH 8, about 5% of the ammonia in solution exists as NH_3 gas. The recorded pH values for the waters of Port Phillip Bay are close to or greater than 8 and from the turbulence created by winds, significant quantities of NH_3 could be lost to the atmosphere by a simple equilibrium displacement.

The atomic ratio between total N and total P in the upper 2 cm sediment is 5.5. This is similar to the water column ratio of 5.2. The average weight N:P ratios of the total particulate N and P in the Bay from April 1970 to March 1971 was 15 (atomic ratio of 37), certainly well above the atomic ratio of 16 expected for phytoplankton. More recent data (Longmore, unpublished) indicate the N:P ratio of the total seston to be around 11:1. In the Phase 1 work, the phytoplankton standing crop for the same period was estimated to be from 10 to 20% of the total particulate C load the remainder of which was assumed to be calcium carbonate. The organic part of the seston appears to have been enriched in P with respect to N from 1969-70 to 1990-92. If the presumption is made that the seston precipitates to the sediment and is there degraded, around 80% of the nitrogen load to the sediment is mineralised or denitrified. A flux of soluble N from the sediments would therefore be expected and this has been confirmed by Longmore and his colleagues. Unfortunately, there is not enough information to calculate a sedimentation rate to the Central Zone, but the following items indicate very slow sedimentation:

- the total annual suspended solid (mostly quartz) loads from the rivers, drains and treatment works are small inputs to a basin as large as Port Phillip Bay; if an average annual load was spread over the floor of the Bay the sediment rate would be much less than 1 mm; most of the suspended solids discharged into the Bay precipitate near the input location, except during significant storm and flood events.

- most of the sediments in the Bay are re-worked deposits which were laid down before the Bay was flooded by the sea 7000 years ago

- active resuspension of sediments is indicated by the predominance of calcium carbonate (not derived from the modern catchments) in the carbonate producing zones of the sediment; unless of course the calcium carbonate is produced in the water column.

- significant contribution of input suspended solids to the Central Zone sediments would occur only during run-off from large storms and floods

- even though the organic content of the Central Zone sediments is high the nutrient efflux and oxygen demand are low, suggesting that the organic matter is either old or refractory

Returning to Longmore's pool and flux data, if the net annual production of phytoplankton - representing around 9000 tonnes N (the lowest of the estimated values) - were deposited and degraded in the sediment without phosphate loss, the pool of TP in the upper 2 cm sediment would have taken around 10 years to accumulate. Concomitantly, 90000 tonnes N would have been sequestered into the sediment. Longmore calculates that there is about 25000 tonnes N in the sediment so around 65000 tonnes N have been lost from the sediments, presumably through mineralisation and denitrification with N_2 effluxing from the sediment or by mixing to greater depths. This is equal to an average loss of 6500 tonnes N y^{-1}; a very slow oligotrophic flux of around 3.3 g N m^{-2} y^{-1}. Gorfine and Abbott (unpublished) measured the average N flux from the sediments of the Central Zone at 4400 tonnes y^{-1} which is about equal to that which we calculated from the remineralisation of phytoplankton estimated to have settled to the sediment. Longmore also measured the aggregate N flux from sediments in the Mixing Zone at 7400 tonnes y^{-1}. Clearly there is not enough nitrogen input to balance the total annual N flux budgets for the whole Bay unless there are significant recycling coefficients in the efflux equations.

There are obvious dangers in extrapolating averages and aggregates to very large ecosystems such as Port Phillip Bay. This is clearly part of the problem with the simple models which have been used to construct budgets and dynamic models. Even when the facts satisfy model equations, the apparent correlations may not reflect the actual function of the system.

An absolute requirement of Newell's N budget and dynamic model for Port Phillip Bay is that there is a net annual conversion of the input of DIN (mainly ammonia and nitrate) to DON by the biota of the Bay. Since it was shown in the Phase 1 work (Chapter 9) and by Mickelson (unpublished) that the DON of the Bay is not available to phytoplankton, it would seem most unusual that the biogeochemical cycling of N produced so much highly refractory DON so rapidly. However, Harris (1986) records a personal communication with G A Jackson who considers that DON plays an important role in the regeneration of N in the water column of the ocean. In the ocean, the DIN pools are very small compared to the DON pools but Jackson calculated that the DON pool needs to turn over only by 3% to have a significant impact on the availability of N in the surface waters of the ocean. Also, Fisher and Cowdell (1982) showed that four diatomaceous species from Port Phillip Bay were able to utilize DON as efficiently as DIN.

A comprehensive understanding of the N budget and dynamics is essential for the effective management of nutrients in Port Phillip Bay. Figure 3.4.4.3 gives some idea of the present information based on measurements; good (G), poor (P) and nil (N) are the scores. Except for the WTC inputs and the pool sizes of soluble and refractory N in the water, measurements have been minimal to nil. Even the WTC inputs are known only within +/- 30% and Longmore (unpublished) places a +/- 50% error on sediment nutrient flux and pool sizes. The large uncertainties in these and other measurements of nutrient pool sizes and fluxes permit Bay-wide model situations ranging from those in which most of the nutrient inputs are retained in the water and sediment, or exported to Bass Strait and the atmosphere. From the prevalence of 'poor' and 'nil' entries in Figure 3.4.4.3 it is evident that the current measurements and the gross estimates derived from them are inadequate to provide even a simple budget for N in Port Phillip Bay. A concept model is therefore urgently needed to provide some estimates for the accuracy and sensitivity of measurements required so that the impacts of natural and/or managed changes (real or imaginary) to nutrient input can be quantitatively assessed. Such a concept model will also guide the evolving work plans for Phase 3 investigations. From the prevalence of 'poor' and 'nil' entries on Figure 3.4.4.3 it is evident that a credible N budget cannot yet be constructed. It remains for Phase 3 to provide a reliable model for N and P transformations and a detailed three dimensional hydrodynamic models using measured and detailed mixing coefficients and tidal exchange ratio.

3.4.5 **Nutrient cycling and dynamics in the Werribee and Hobsons Bay areas.**

The nutrient pools of the shoreline sectors of Port Phillip Bay are small compared with the Central Zone (Table 3.3.1.2). However, these sectors are the most visible and most used by the general public. They are also the areas most likely to suffer eutrophication if the input of nutrients exceeds the dispersal capacity of the tides and currents. We have chosen to examine the Werribee and Hobsons Bay areas in more detail because they are subject to the largest nutrient input from sewage and nutrients from rivers and drains.

The Werribee and Hobsons Bay areas are contained by the Werribee, Altona and North-eastern environmental zones defined in the Phase 1 Report (MMBW/FWD, 1973, Figure 10.1). According to current hydrodynamic modelling, the waters of the nearshore areas are well mixed with the Central Zone but Corio Bay does not exchange rapidly and it is viewed as a sediment trap (Newell, 1990).

3.4.5.1 The Werribee Zone

The dominant feature of the Werribee Zone is the discharge of effluent from the WTC. The Phase 1 Report (MMBW/FWD, 1973) briefly describes the sediments and biology of the Zone as it was 20 years ago. It was concluded that the nutrients from the WTC had little effect on the growth of seagrass (except close to the shore). It was also noted that beaches in the zone were subject to regular heavy weed deposition (still common to many parts of the shoreline in the Bay). The growth of phytoplankton was significantly less than that expected from the concentration of readily available ammonia and orthophosphate.

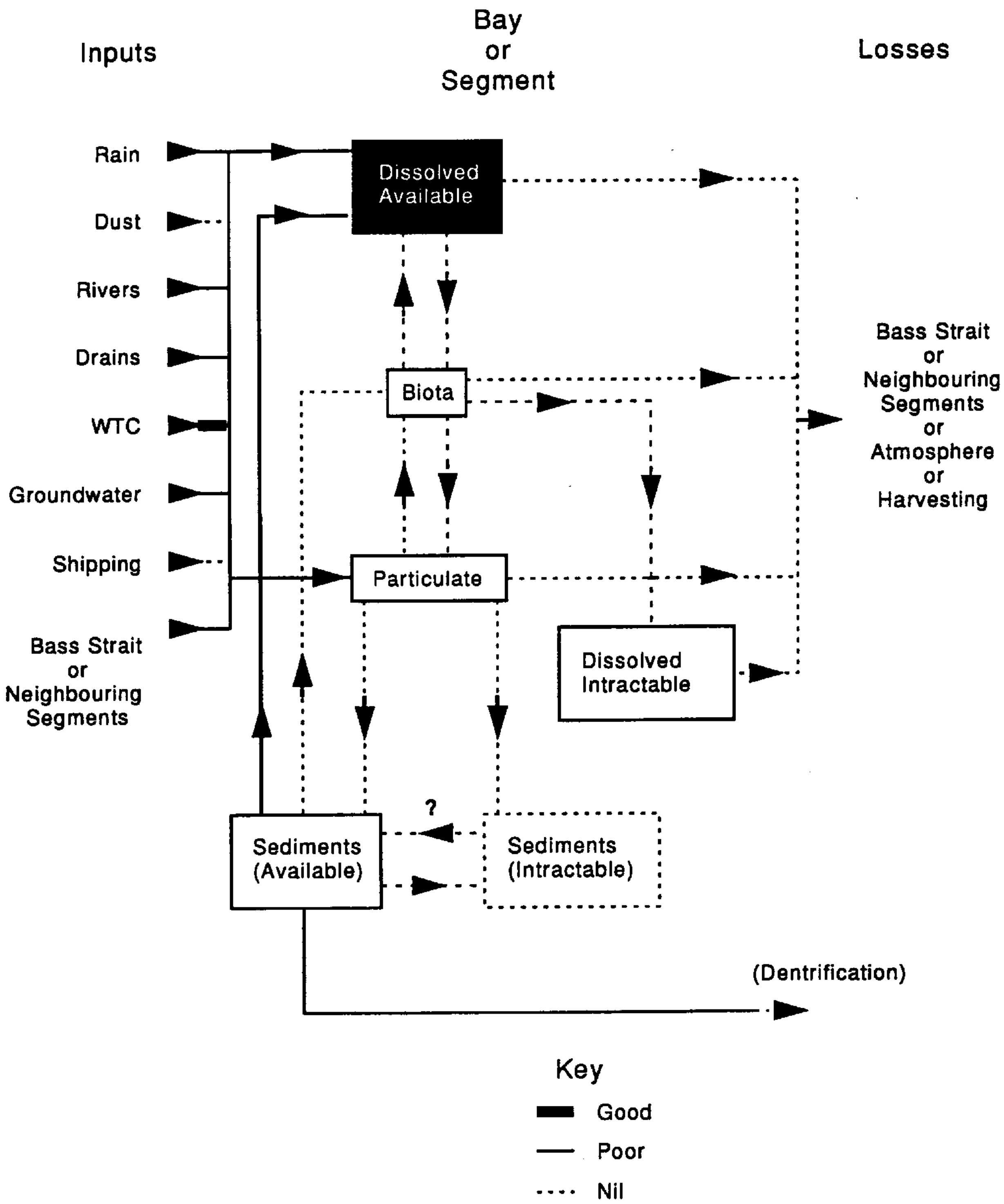

Figure 3.4.4.1 Measurements (tonnes) of annual nutrient imputs and nutrient pool sizes in Port Phillip Bay, 1969/70

Much of the work of Phase 2 (see Axelrad *et al.*, 1981; Brown and Davies, 1991) was concentrated on the Werribee Zone and this is reviewed in detail by Newell (1990). Phase 2 brought a much more detailed understanding of the nutrient status and dynamics of the Zone and this work has been complemented by studies on nutrient fluxes at the sediment/water interface by Longmore and Abbott and Gorfine (unpublished) from the early eighties to the present.

An estimated 4380 (+/- 1800) tonnes organic C is discharged from the WTC annually. Its' chemical composition is not known but some fraction would probably be biologically available. The quantity is around 8% of the estimated Bay-wide pool of organic C. The available fraction may contribute to heterotrophic metabolism, particularly microbial processes such as aerobic diagenesis, fermentation, denitrification, sulphate reduction and nitrogen fixation. There is no information on the latter in Port Phillip Bay. It has been shown that N_2-fixation may account for 1 to 10% of the N required for primary production in shallow, temperate coastal environments like Port Phillip Bay.

3.4.5.1.1 Phytoplankton

Axelrad *et al*. (1981) determined phytoplankton production at one station in the Werribee Zone and compared that to productivity in the Central Zone. Productivity in the water column reached its highest values over the summer months. The annual productivity in the Werribee Zone was estimated at 73 g C m^{-2} y^{-1}; in the Central Zone the estimate was 33 g C m^{-2} y^{-1}. For most of the year (1976) the chlorophyll *a* concentrations were similar in the Werribee and Central Zones. This was not consistent with the relatively high phytoplankton productivity. Axelrad *et al.* (1981) suggest that this was due to the rapid mixing of the phytoplankton biomass from the Werribee Zone to the Central Zone. The export of the phytoplankton biomass from the Werribee Zone (if it is as significant as the estimates indicate) clearly mitigates any serious eutrophying effect the discharge of nutrients from the WTC may have. However, the zooplankton numbers in the Werribee Zone included the highest densities recorded and, although the grazing populations were low at other times, it is clear that export estimates must be modified by grazing and also the growth of filter feeders in the sediments.

3.4.5.1.2. Benthic communities

An important obseration by Axelrad *et al.* (1981) was the very high primary productivity of the epibenthic community of diatoms colonizing the shallow water sandy sediments. The productivity rate of 236 +/- 19 g C m^{-2} y^{-1} was the highest to be reported for shallow-water marine environments. The incubation period was twelve hours so it was probably a net productivity which was being measured. Within the zone, the epibenthic and microalgal communities would have a significant role in nutrient cycling and dynamics. The standing crop of epibenthic algae was also very high, from 60 to 130 mg chlorophyll *a* m^{-2}.

The intertidal zone was also characterized by blooms of macrophytic algae which was indicative of nutrient enrichment (compared to other temperate, coastal environments). The growth of benthic macrofauna was stimulated by the increased nutrient supply from the WTC, both as dissolved and particulate forms. There appeared to be no deleterious effects of the effluent on these benthic communities. It can be inferred that eutrophication of the sediments was not a characteristic of the zone, at least up to 1975/76. On the other hand, Bauld and Millis (1977) showed that the sediments close to the shore had high ammonia concentrations to depths of 25 cm; this is indicative of high rates of anaerobic microbial activity.

3.4.5.1.3. Sediments

From measurements of nutrient fluxes at the sediment/water interface, Gorfine and Abbott (1992) showed that ammonium was the major nutrient efflux from nearshore sediments and the efflux from the sediments of the Werribee Zone were similar to those at other nearshore sites. Longmore (1992) measured nutrient fluxes at six nearshore sites (1.2 km seaward of the shore) on 17 occasions over a 20 months period and the results of these studies are given in Table 3.2.2.2. On an areal basis, the nearshore efflux rates for DIN were high and the estimated annual fluxes from sediments in the nearshore and central zones are given in Table 3.2.2.3. Longmore calculated an annual efflux of 7400 tonnes DIN from near-shore sediments. If this were ultimately supplied by primary producers in the zone, a total productivity of 44400 tonnes or more of organic C by phytoplankton is implied, depending on the C:N ratio of the prime sources. Benthic phototrophs may have atomic C:N ratios of 10 or more (Atkinson and Smith, 1983). Also, about 5000 tonnes organic C-N is discharged by the WTC and this could be a significant contributor to the activity of the degradative heterotrophic bacteria in the upper few centimetres of sediment.

Axelrad *et al*., (1981)(see also Poore and Kudenov, 1978) showed that most of the benthic biomass in an area around a WTC drain was contributed by suspension feeding molluscs. The food intake by the fish populations in the nearshore areas were six times that of the offshore fish: this was due to a population age structure.

3.4.5.1.4. Eutrophication

There is a notion that the nearshore zones in Port Phillip Bay are protected from stagnation and eutrophication by rapid mixing of the inshore waters toward the Central Zone. While the current climatic and hydrodynamic conditions prevail, this is probably the case. However, maintenance of the environmental quality of these nearshore zones clearly requires detailed information on nutrient pool sizes in the sediments and benthic communities and and also on the trophic state of the sediments and waters. The efflux of ammonia from the Werribee sediments, as the predominant form of DIN, indicates that the surficial sediments are predominantly (or maybe entirely) anaerobic (Gorfine and Abbott, 1992; Longmore, 1992).

Future management plans clearly must continue to measure the efflux of ammonia from Werribee sediments because this will be a useful indicator of the trophic state of the zone. An efflux of sulfide into bottom waters would be alarming. There are several examples in the coastal lakes of eastern Australia, where inshore zones have become badly eutrophied. They are now characterized by restricted flushing due to dense standing crops of macrophytic algae and the ebullition of hydrogen sulfide into the atmosphere (e.g. Tuggerah Lakes).

3.4.5.2. Hobsons Bay

The following is a discussion of the work of Grant and Gaylor (1982) and Longmore (unpublished, 1990-present).

Salinity in the bottom waters of Hobsons Bay approximates that of the central Bay, but decreases toward the surface. Proximity to the Yarra mouth and the rate of river flow influence the salinity characteristics of the upper waters of Hobsons Bay. Fortnightly measurements over the period August 1990-May 1992 from a site within Hobsons Bay demonstrated persistent salinity differences between surface and bottom water of 0.03-16 PSU (mean 2.6). At times of high flow, surface salinity fell to as low as 7-11 PSU, though the low-salinity surface layer was often less than 2 m thick. The surface freshwater plume may move to the east or to the west, depending on prevailing winds, though it is more often observed close to the eastern coast. Continuous salinity records from three metres depth at a site four nautical miles due south of the river mouth (i.e. outside Hobsons Bay) showed salinity to vary in the range 33-35.5 over the period August 1990-May 1992.

Photosynthetically Active Radiation (PAR) profiles show that sufficient light (>1% incident) reaches the bottom in waters less than 5 m deep on most occasions. Attenuation coefficients near Williamstown vary in the range 0.2-0.8 m^{-1}. During 1990-92, nutrients were highest in surface waters after heavy rains in January 1991 (ammonium to 220 g N L^{-1}, nitrate to 200 g N L^{-1}, phosphate to 240 g P L^{-1} and silicate to 1120 g Si L^{-1}), but there appeared to be no seasonality in concentrations for the rest of the year (ammonium 14-30 g N L^{-1}, phosphate 60-120 g P L^{-1}, nitrate 2-140 g N L^{-1}, silicate 150-450 g Si L^{-1}).

In the 1974-75 study chlorophyll *a* concentrations were greatest in February when the entire Bay displayed concentrations of >10 g L^{-1}, while concentrations in mid-winter were less than 3 g L^{-1}. High biomass was confined to the top 3 m of the water column. Longmore's unpublished data from a single site near Williamstown show a similar pattern but a wider range, with peaks to 6 g L^{-1} in January 1991 and 20 g L^{-1} in January 1992 superimposed on a background of 0.8-4 g L^{-1}. Phytoplankton were dominated by diatoms for most of the year, but the intense peak in January 1992 was due to a dinoflagellate bloom.

Primary productivity during 1974-75 was divided into two periods: winter, with low chlorophyll (<1 g L^{-1}) and low productivity (<2 mg C m^{-3} hr^{-1}), and summer, with higher chlorophyll (5-6 g L^{-1}) and higher productivity (10 mg C m^{-3} hr^{-1}). In spring, biomass increased, but productivity remained low. In late spring, biomass decreased, while productivity per unit biomass was increasing. Grant and Gaylor (1982) concluded that productivity was limited by light or temperature (or both) in winter, while herbivore grazing reduced the standing crop in late spring and summer.

3.5 Knowledge gaps and research needs

The important results of the detailed review of nutrients in Port Phillip Bay were:

1. An annual nutrient budget for nitrogen cannot yet be constucted because of high percentage errors associated with currently measured inputs.

2. Atmospheric inputs to the Bay waters may be as high as 30% of the input from the WTC.

3. Uncertainties about the exchange between Bay and Bass Straits waters complicate the assessment of residence times of chemically conservative compounds which are discharged into the Bay.

4. The sediments of the Bay are large and significant sinks and sources of nutrients.

5. The sediments of the Central Zone are of prime importance in quantifying nutrient dynamics in the Bay.

6. It may be necessary to measure the effects of phytoplankton grazers and consumers to obtain a credible N budget for the Bay.

7. A nutrient model for the Bay, which compliments and extends those discussed by the Phase 1 study (MMBW/FWD, 1973), Williams (1978), Newell (1990) and Longmore (1992), is required to substantiate the N tasks for Phase 3.

8. Modelling, understanding and monitoring for nutrient dynamics in Port Phillip Bay is completely dependent on a hydrodynamic model which describes the details of mixing, sedimentation, exchange, import and export.

The following refers to the N tasks given in the Project Design.

Task N1.1 Nutrient Inputs/Loads - WTC and other point sources

Review of WTC data show that loads are estimated +/- 30%. This is too high for the determination of a budget which would be useful to managers of water quality in the Bay. Methods should be accurate enough to account for daily and seasonal variations in the discharge of all WTC effluent drains to a +/-10% confidence level. Is this feasible and are the analytical methods sensitive enough to achieve this level of accuracy?

Does the temperature need to be measured to $0.01^{o}C$?

Recommendation: Accurate measurements of total annual loads of N and P are essential for nutrient management in Port Phillip Bay. This task should proceed with provision for the measurement of daily fluctuations in summer and winter to permit a more accurate assessment of annual loads.

Task N1.2 Nutrient Inputs/Loads - Urban Drains

Recommendation : This task should proceed as planned.

Does the temperature need to be measured to $0.01^{o}C$?

Task N1.3 Nutrient Inputs/Loads - Rivers and Streams

Recommendation: Accurate measurements of total annual loads of N and P from rivers and streams are essential for nutrient management in Port Phillip Bay. This task should proceed as planned with the addition of chlorophyll *a* in flows from the major rivers.

Task N1.4 Nutrient Inputs/Loads - Ground water

Recommendation: This task should proceed as planned.

Task N1.5 Nutrient Inputs/Loads - Aeolian and Atmosphere

Current estimates for the atmospheric input of organic carbon and inorganic nitrogen to Port Phillip Bay indicates it is equivalent to about 30% of the WTC input. This load is predicted to increase with increasing urbanization of the Melbourne area. It is therefore essential to have accurate measurements of atmospheric inputs.

Recommendation: This task should proceed as planned.

Tasks N2.1 and N2.2. Nutrient Pathways/Budgets - Ambient/extreme Bay concentrations and phytoplankton dynamics/fluxes

Recommendation: These tasks should be combined and proceed as planned. It would be valuable to obtain measurements of zooplankton populations during the course of this project.

Tasks N2.3 and N2.4. Nutrient Pathways/Budgets algae/seagrasses dynamics/fluxes and algal mats/turf dynamics/fluxes

Recommendation: These tasks should be combined and proceed as planned.

Tasks N3.1, N3.2, N3.3 and N3.5. The nutrient status of PPB sediments, the efflux of nutrients from sediments into the water column, oxygen/nitrate/sulfate reduction rates and N losses.

Recommendation: These tasks should be combined and proceed as planned. The estimation of sedimentation rates for the major sediment types in the Central Zone should be additional to the work plan for these tasks.

Task N3.4 The distribution and concentrations of biomarkers for sewage, primary producers and bacteria in the major sediment types of PPB.

Recommendation: This task should proceed generally as planned but the work should be done in close collaboration with the other N3 tasks.

3.6 **Acknowledgements**

We thank our colleagues in Melbourne Water, the Environmental Protection Authority, the Department of Conservation and Natural Resources and the Marine Science Laboratories, who have made the time and effort to make us aware of the extensive work on nutrients which has been done in Port Phillip Bay.

Chapter 4. Toxicants

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4.1 Introduction

Port Phillip Bay has been the recipient of atmospheric and water-borne industrial and urban contaminants for over 100 years. Historically there has been evidence of site-specific impacts from toxicants including cadmium, mercury and hydrocarbons which have threatened commercial and recreational uses of the Bay. Despite improvements in controls, and the introduction of a State Environmental Protection Policy (SEPP) for waters of the Bay (EPA, 1991a), the impact of accumulated sediment-bound contaminants, and the capacity of the Bay to accept continuing contaminant loads in the future, are vital management questions.

The loads and species of toxicants entering Port Phillip Bay from both diffuse and point sources are likely to have changed considerably since the studies of the 1970s. A major focus of the toxicant program will be to assess the extent of these changes to permit an evaluation of their impact on the Bay. It should address all potential inputs, both aqueous and atmospheric, and the distribution and fate of toxicants entering the Bay. It should also consider the status of biota, including both bioaccumulation of toxicants and the impact of such toxicants on the health of biota, especially fish. The study will permit an evaluation of how well the Bay ecosystem is coping with current toxicant inputs, both in terms of the compartments into which they partition and their impacts in these compartments, and how significant an alteration in inputs from particular point sources might modify these impacts. To do so requires a more comprehensive knowledge of the physical processes in the Bay, of the general ecology of the Bay, and a consideration of the overlying effects of existing and possibly variable nutrient loads on the system. These data will be brought together in a process model which will be able to be used by management to predict the effects of potential management options.

It should be remembered that such an approach is not new, and that aspects of this type of study have been and are being applied to estuarine systems elsewhere in Australia and abroad. The toxicant program will not attempt to duplicate fundamental aspects of this research, but will utilise such findings wherever possible in the planning of its program.

This report reviews past research on toxicants in Port Phillip Bay in order to provide a sound basis for the proposed toxicant research tasks before studies are commenced. There have been two recent major reviews on toxicants, by Coller et al., (1991) and Phillips et al., (1992). Information from these reports will be critically reviewed together with those from the many specific studies undertaken in the last twenty years, but with a particular focus on the proposed research tasks.

4.2 **Toxicant inputs to Port Phillip Bay**

4.2.1 **Industrial point sources**

4.2.1.1 Werribee Treatment Complex

Melbourne Water Corporations Werribee Treatment Complex (WTC) is a major potential source of toxicants to Port Phillip Bay, if only through the sheer volume of sewage. On average 470 ML of sewage - which has been previously purified by land treatment and lagooning - is discharged each day into the Bay from the WTC.
Early investigations of the quality of discharged water concentrated on nutrients and major ions. The WTC currently receives licensed industrial discharges of metals and organic toxicants, however most of these are expected to be removed during the primary sedimentation with further precipitation in the anaerobic lagoons. It has been reported that about 90% of the metals and 66% of the nitrogen entering WTC are removed during treatment (Wood *et al.,* 1991). Melbourne Water Corporation and EPA data for 1980 and 1981 show annual loads of chromium, nickel, copper and lead from WTC to be of the order of 5000, 6000, 3000 and 1500 kg respectively. A survey of heavy metals entering the Bay from input sources, creeks and drains (EPA, 1979) concluded that WTC was one of the major sources, especially of zinc. More recent data (Coller *et al.,* 1991) are shown in Table 4.1.

Table 4.1. Heavy metals in sewage effluent from Werribee Treatment Complex, 1989/90

Metal	Mean concentration (g L^{-1})	Annual load (tonne y^{-1})
Iron	1600	304
Zinc	73	13.5
Copper	29	5.4
Nickel	39	7.3
Chromium	22	4.1
Lead	15	2.8
Cadmium	1.1	0.2
Mercury	0.2	0.04

It has been estimated that these data in general represent an increase of between 20% and 50% compared to 1980-81 data. The concentrations, when assessed with respect to the Australian and New Zealand Environment Conservation Council (ANZECC) Water Quality Guidelines (ANZECC, 1992)(Table 4.2), show high concentrations of copper, chromium and lead. The impact of these concentrations on aquatic biota will also be dependent on the chemical form of the metal. Guidelines are based on ionic metal in its most toxic form. Chromium is most likely in the reduced and less toxic chromium (III) form, and copper, and to a lesser extent lead, are likely to be complexed by the high dissolved organic content of the water or adsorbed onto particulates. The values are therefore not *a priori* a cause for alarm.

Table 4.2. ANZECC water quality guidelines for inorganic and organic toxicants in fresh and marine waters (from ANZECC, 1992)

Toxicant	Guideline concentration (g L^{-1}) for the protection of aquatic life Freshwater	Marine Waters
INORGANICS		
Ammonia	30	500
Arsenic	50	50
Cadmium	2	2
Chromium	2	2
Copper	2-5 [a]	5
Cyanide	5	5
Iron	1000	-
Lead	1-5 [a]	5
Mercury	0.1	0.1
Nickel	15-150 [a]	15
Selenium	5	70
Sulfide	2	2
Zinc	5-50 [a]	50
ORGANICS [b]		
Benzene	300	300
PAHs	0.3	0.3
PCBs	0.001	0.004
Aldrin	0.010	0.013
Chlordane	0.004	0.004
DDE	0.014	0.014
DDT	0.0005	0.001
Dieldrin	0.002	0.002
Endosulfan	0.001	0.009
Endrin	0.003	0.002
Heptachlor	0.0003	0.04
Lindane	0.003	0.002
Methoxychlor	0.04	0.03
Mirex	0.001	0.001
Toxaphene	0.002	0.0002
Chlorpyrifos	0.001	0.006
Malathion	0.07	0.10
Parathion	4	4
Pentachlorophenol	0.5	11
Monochlorophenol	7	-
Phenol	50	50
Dibutylphthalate	4	NR

[a] Dependent on hardness [b] Some refer to unfiltered samples

Only recently have organic toxicants in WTC received serious consideration, following allegations of discharges to WTC from a plant producing 2,4 D. Only minor concentrations of organochlorines and dioxins were detected (MMBW, 1990b). Given the expected partitioning of these compounds, it is likely that they would be preferentially attaching to organic matter and clays in the sediments rather than being present in solution.

4.2.1.2 Other treatment plants

In addition to WTC the Western No 2 Purification Plant, at Altona, treats 1% of Melbourne's sewage by biological filtration. Some 13 other small treatment plants operated by Melbourne Water Corporation collectively treat about 3% of the total sewage flow which is subsequently discharged into local watercourses.
The Dandenong-Springvale Water Board (DSWB) and the Mornington Peninsula District Water Board (MPWB) both treat sewage which is discharged to Mordialloc and Kananook Creeks respectively which flow into the Bay. Both plants use secondary treatment and are likely to constitute minor sources of nutrients and toxicants to the Bay.

4.2.1.3 Corio Bay industry

Corio Bay is a 60 km^2 body of water forming the western extension of Port Phillip Bay. Its boundaries are home to a number of major industries. These include the Shell oil refinery, Pivot's Phosphate Fertilizer Cooperative, the Ford Motor Company's car plant (including a casting plant), Alcoa's aluminium smelter and the extensive light industry of the City of Geelong (Fabris, 1983). All of these are likely sources of heavy metals and organic contaminants.
In 1974 it was found that Lindex Chemicals, a paint pigments manufacturer, had been discharging large quantities of cadmium and lead into Corio Bay *via* a stormwater drain. This resulted in extensive contamination of sediments and mussels in most parts of the Bay (Fabris, 1991; Nicholson *et al.,* 1991a). Recent sediment data suggest that concentrations have decreased as a result of burial, remobilisation or transport of sediment.

Corio Bay industry has also been implicated in the high concentrations of mercury found in sediments. Kelly (1978) reported mercury in effluent entering the Bay. High sediment mercury concentrations were detected in the vicinity of the Shell Refinery Pier where recycled seawater used as cooling water in the refinery is discharged (Nicholson *et al.*, 1990). Other potential mercury sources include the Shire of Corio tip and the Pivot Phosphate Cooperative Company. Drainage from the latter analysed on two occasions in 1989 contained 1800 and 40 ng L^{-1} respectively (Coller *et al.,* 1991; Jackson *et al.,* 1986; Nicholson *et al.,* 1990).

4.2.1.4 Altona industry

The Altona region of Port Phillip Bay - north of Point Cook - hosts a number of substantial industries. These include the Altona Petrochemical Complex of Hoechst, BF Goodrich's tyre plant, Dow Chemicals, ICI plastics and pigments, BASF, Toyota, BP, Petroleum Refineries Australia, the Western No 2 Treatment Works and the Newport Power Station. These have the potential to contribute both atmospheric and aqueous emissions.

Previous measurements have shown evidence of aqueous waste discharges to waterways in the region such as Kororoit Creek, and the sediments of the waterways reflect the history of such activities (Kunert, 1990; Reed, 1990). These effluents are now diverted to the sewer that flows into the WTC.

4.2.1.5 Hobsons Bay industry

Hobsons Bay includes the mouth of the Yarra estuary, and contains the major dockyard facilities for the Port of Melbourne. Cooling water from the Newport Power Station is also discharged in this region. These activities, together with oil terminals and Australian Glass Manufacturers plant, are potential sources of heavy metals and organics - especially hydrocarbons. Limited data are available (Coller *et al.,* 1991; Phillips *et al.,* 1992).

4.2.1.6 Eastern Bay industry

This area comprises the inputs of Mordialloc Creek and the Patterson River. Both are the sites of substantial industry, including the Dandenong-Springvale sewage treatment facility, and a range of light industries.

4.2.1.7 Licensed discharges

The Victorian EPA licenses discharges by a number of industries to drains and creeks around Port Phillip Bay. The conditions for such licenses have altered considerably over the last twenty years, particularly during the 1970s when many of the discharges were diverted directly to the sewers. Fabris (1983) listed the licensed discharges to Corio Bay at that time. Conditions specify a maximum concentration and a calculated maximum load. Industries holding current licenses are self-regulating, with reports of discharged concentrations being supplied to EPA who do not have the resources to undertake checks of these data.

4.2.2 **Rivers and creeks**

Previous studies have identified rivers, creeks and drains as the second most important source of both nutrients and toxicants to the Bay after the WTC (EPA, 1979). Three major rivers, the Yarra, Maribyrnong and Werribee, drain into the Bay as well as more than 300 creeks, drains and effluent pipes.

A summary of some important input streams is given in Table 4.3. There appear to be no data more recent than 1979 on the toxicant loads of these streams, and even then the data were extremely limited in the species analysed and their relationship to present loads is questionable. For comparison the summed mean daily metal load is included. This is of dubious value except as some indication of relative inputs. There is clearly a requirement to quantify current loads in terms of specific contaminants, both organic and inorganic. Suspended sediment association of these contaminants in the input waters is also important.

Given that the flow of the Yarra River (approximately 2500 ML d^{-1}) is about six times that of the WTC and far greater than any of the other input streams, particular attention should be given to its' contaminant loads.

Table 4.3. Important input streams to Port Phillip Bay

Input stream	Land use	Pollutant loading [a]
Werribee Treatment Works	Sewage farm	V. High (175)
Yarra/Maribyrnong River	Major urban/industrial	High (?)
Patterson River	Urban/light ind.	High (60)
Mordialloc Creek	Urban/light ind.	Low (3)
Kororoit Creek, Altona	Grazing/ major ind.	Low (3)
Kananook Creek, Frankston	Urban/grazing	Low (?)
Cowies Creek, Geelong	Urban/industrial	Low (1)
Werribee River	Grazing	Very Low (Background)

[a] Figures in parentheses represent summed mean daily load (in kg) of lead, zinc, cadmium, copper, arsenic, chromium and nickel, measured on several occasions in 1977 (EPA, 1979).
The latest published EPA data for rivers and creeks discharging into Port Phillip Bay (EPA, 1992) are shown in Table 4.4. The data are means of between 13 and 24 measurements dating from 1989. In many instances high concentrations have been recorded and - relative to other inputs - their contributions can be considered as significant. Many of these dissolved pollutants (including iron) will precipitate during mixing with the saline waters of the Bay, resulting in high heavy metal accumulation in the estuarine zones close to the river mouths.

Table 4.4. Typical metal concentrations in Port Phillip Bay rivers and creeks 1989 [(a)]

Input	Location	Zn	Cu	Pb	Cd	Cr	Ni	Hg
		(g L^{-1})						(ng L^{-1})
Barwon River	Queens Park, Geelong	22.6 (99)	-	3.4 (16)	-	-	-	-
Werribee River	Werribee Gorge	4.0 (12)	-	2.6 (29)	-	-	-	-
Kororoit Creek	Station Road, Deer Park	45 (150)	9.8 (24)	12.2 (29)	0.5 (1.3)	35 (78)	7.7 (33)	-
Maribyrnong River	Canning Street	11.0 (35)	-	3.3 (9)	-	-	-	-
Yarra River	Chandler Hwy	35 (81)	4.4 (9)	9.0 (32)	-	-	-	-
Dandenong Creek	Olive Grove	110 (350)	10.4 (28)	14.2 (45)	0.5 (1.1)	3.9 (18)	4.8 (34)	25 (<50)
Patterson River	Upstream of freeway	64 (340)	6.8 (16)	11.3 (17)	0.2 (0.9)	3.3 (26)	3.2 (7)	25 (<50)
Mordialloc Creek	Wells Road	109 (180)	17.1 (34)	17.0 (41)	0.5 (2)	8.5 (31)	7.8 (21)	25 (<50)
La Trobe River	Rosedale	21.2 (100)	4.7 (7)	0.9 (2)	0.3 (<0.5)	1.6 (4)	2.1 (6)	100 (<200)

[a] Figures in parenthesis are maximum measured concentrations

4.2.3 **Drains and stormwater**

Stormwater has the potential to transport terrestrial pollution to the Bay. Its contribution is measurable initially in the many drains designed to collect such inputs. In assessing stormwater inputs it is the pollutant load that is important. This cannot be determined directly by flows since the flux of contaminants will vary both with the pollutant and with the frequency and intensity of storms.

It is usual for the first flush concentrations to be extremely high, especially following an extended dry spell, and this may result in a measurable impact on the biota of the Bay. Dilution will be important in moderating this, however, excessive dilution brought about by a long and intense rainfall event will have dramatic effects on the salinity of the Bay near the point of input, and the extent of transportation and deposition of pollutants in bottom sediments will depend on the size of the reduced salinity zone. Accompanying soluble contaminants will be a variable suspended sediment load, and the relationship between this profile and that for the contaminants will be important in defining their behaviour and ultimate impact on the Bay.

The 1979 EPA survey of input streams contains data for a range of drains sampled over 12 months both in normal low-flow conditions and during and after rainfall. The study enabled some idea of flows and pollutant loads to be determined, but, given the changes in inputs to these drains and in the nature of the surrounding terrestrial environment, it probably is not relevant to loads and flows in 1992. The drains considered are shown in Table 4.5. The toxicants transported in low flow conditions would be those associated with discharges of secondary-treated sewage, cooling water and other industrial licensed discharges. In storm conditions sewage overflows, heavy metals and hydrocarbons from roadways and pesticides from agricultural soils would be likely contaminants. The major metal inputs are likely to be lead and zinc, together with lesser amounts of copper.

Table 4.5. Drains surveyed in previous EPA studies

Drain	Drainage land use	Pollutant load
Cowderoy Street Drain, St Kilda	Residential, commercial	High
Elwood Canal, Elwood	Residential, commercial	High
Park Street Drain, Brighton	Resid., comm., ind., grazing	High
Heatherton Drain, Mordialloc	Resid., comm., ind., grazing	High
Mordialloc Settlement Drain	Resid., comm., ind., grazing	High
Mordialloc Main Drain	Residential, industrial	High
Paisley Street Drain, Altona	Residential, commercial	Moderate
Mulga Street Drain, Altona	Residential, comm., industrial	Low
Rippleside Drain, Geelong	Residential, comm., industrial	High
Cunningham Pier Drain,Geelong	Residential, comm., industrial	High

The 1979 EPA survey of input streams contains data for a range of drains sampled over 12 months both in normal low-flow conditions and during and after rainfall. The study enabled some idea of flows and pollutant loads to be determined, but, given the changes in inputs to these drains and in the nature of the surrounding terrestrial environment, it probably is not relevant to loads and flows in 1992. The drains considered are shown in Table 4.5. The toxicants transported in low flow conditions would be those associated with discharges of secondary-treated sewage, cooling water and other industrial licensed discharges. In storm conditions sewage overflows, heavy metals and hydrocarbons from roadways and pesticides from agricultural soils would be likely contaminants. The major metal inputs are likely to be lead and zinc, together with lesser amounts of copper.

Of more value is the comprehensive study of a number of stormwater drains undertaken for the Australian Water Resources Council by Gutteridge, Haskins and Davey Pty Ltd and the Victorian EPA (AWRC, 1981). This study examined in some detail the pollutant hydrographs for sites ranging from high density industrial, urban and light urban to agricultural over at least six storm events. A summary of typical results is shown in Table 4.6 with a representative toxicant hydrograph provided in Figure 4.1.

It is important to note that the major river systems - the Yarra, Maribyrnong, Patterson and Werribee Rivers - as well as many of the creeks will also be the recipients of stormwater. As a result any consideration of the fluxes of stormwater toxicants should not ignore their loads which, given the size of the drainage basins, may well exceed those from the smaller drains described above.

It will also be important to consider the fate of sediment-associated toxicants in such river and creek systems which in some cases may never reach the Bay. By comparison most of the toxicants loads to the smaller drains will flow to the Bay.

Table 4.6. Typical results from survey of stormwater inputs from drains

Site	Catchment size (ha)	Mean pollutant load per storm (kg)			
		Pb	Zn	Cd	Cu
Bulli Rd Drain, E. Moorabbin	78	0.8	2.3	0.02	0.8
Hawthorn Main Drain	800	3.8	7.1	0.09	0.4
George St Drain, E. Doncaster	227	0.4	0.9	0.02	0.05
MMBW Drain, Reynolds Rd, E. Doncaster	189	0.06	0.05	0.004	0.01

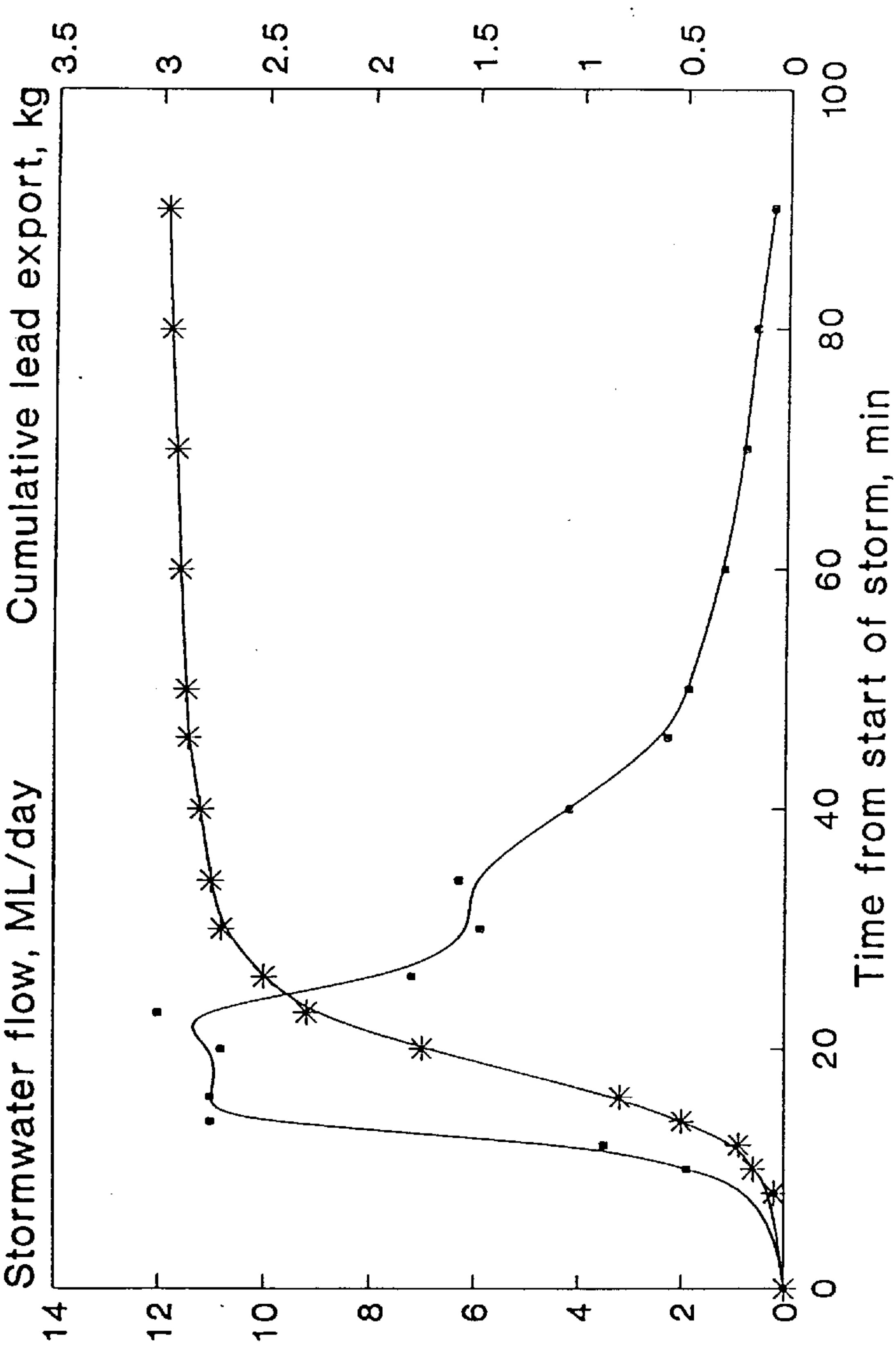

Figure 4.1 Storm hydrograph and cumulative lead export for a drain in East Moorabin (AWRC, 1981).

4.2.4 **Shipping**

The Port of Melbourne is a major shipping port and important shipping channels are maintained in Port Phillip Bay. It is estimated that about 50 vessels traverse the Bay each week to the Port of Melbourne, in addition to about 7 servicing the Port of Geelong. Total shipping in the Bay amounts to 35 million gross registered tonnes per year, with an average size of 10,000 tonnes. During their passage across the Bay ships invariably discharge bilge water, release antifouling paint toxins from their hulls, and also contribute oil and other hydrocarbons to the water column. The impact of these releases are presumed to be even greater whilst docked in port when the vessels are stationary.

One of the major concerns is for the releases of tributyltin (TBT) from antifouling paints, because of its potential impact on the Bays' scallop industry. The use of TBT is now banned on boats under 25 m long in most states of Australia primarily because of the adverse effects on the various shellfish industries. The ban, however, does not apply to large ships. Estimates indicate that the average ship releases some 2.5 g of TBT each day to the water column. Freshly painted ships leach at a rate of an order of magnitude greater. The Port of Melbourne Authority advises that very little antifouling is carried out in port, although the planned introduction into the Bay of a large dry dock in the near future has been of concern to the EPA given that wastes from hydroblasting and other maintenance operations will need to be strictly controlled.

Considering that TBT concentrations in water of 40 ng L^{-1} would be sufficient to have an effect on scallop spat, there may be cause for concern for this source. The recent decline in spat fall may well have been associated with TBT, but may also be associated with the combined effects of both commercial and recreational boating. Examination of records of prevailing winds and the areas where scallop numbers were reduced revealed a coincidence, suggesting that TBT contained in the surface microlayer, and moved by the wind, may have been toxic to free swimming scallop spat which, in their first weeks of life, will feed on plankton in the upper layers. Since the introduction of a ban on the use of TBT on recreational boats record scallop spat falls have been recorded.

The impacts of recreational boating, as a further source of contaminants, is likely to be minimal in the open waters of the Bay - however impacts are to be expected at marinas and slipways where the concentration of boats is high. Wastes from the hydroblasting of hulls is generally poorly contained and can reasonably be expected to find its way into the waters accumulating, eventually, in bottom sediments.

The impacts of shipping have not yet been satisfactorily addressed in Port Phillip Bay although some monitoring of TBT has been carried out at a number of key sites by the EPA (1990a). In general water column concentrations of TBT were found to be low, although sediment concentrations near marinas were high as was to be expected.

No measurements are available for oil and other hydrocarbons in dock areas or shipping lanes. Routine monitoring is carried out by the EPA at a single Hobsons Bay site, however it is likely that results obtained from this site may reflect - at least in part - inputs from the Yarra River.

4.2.5 **Groundwater**

Groundwater inputs have been the subject of an independent review by Otto (1992). Aquifer contamination in the Port Phillip Bay region is reportedly serious and widespread.

Near-shore discharge areas have been subdivided into zones comprising the Central Zone, the Werribee and Altona Zone, Corio Bay, the Bellarine Zone, the Nepean Zone, the South-eastern Zone and Hobsons Bay. Estimated discharges from these zones are 1 and 3.6 x 10^4 ML y^{-1} for the Central and the Werribee and Altona Zones respectively, and 0.8 and 0.4 x 10^4 ML y^{-1} from the Nepean and South-eastern Zones respectively. Excluding input to the Bays Central Zone by point or diffuse discharge, discharge into Corio Bay, and discharge from the aquifers of the Bellarine Peninsula, the groundwater discharge to the Bay has been estimated at 5.5 x 10^4 ML y^{-1}, which is about one third of the discharge from WTC.

Data on toxicant content is sparse and dated. The Werribee-Altona Zone has been well-studied. Many older reports of heavy metals and organics were associated with discharge practices which have since been discontinued, while sampling points were often located close to point-source inputs thereby biasing results. The Altona Zone has reputedly the most polluted groundwater in the Port Phillip Bay region and is a source of contaminated groundwater input to the Bay. A reassessment of accessible boreholes in the Zone to assess the current levels of pollution would be desirable.

Recent studies of the Werribee Delta Aquifer did not indicate significant contamination (Otto, 1992). No recent data are available on groundwaters in the Corio Bay and Hobsons Bay areas and these should be examined to determine the extent of their toxicant loads.

4.2.6 **Atmospheric inputs**

A recent inventory of emissions to air for the Port Phillip Bay Control Region has estimated that on an annual basis 167 kilotonnes of volatile organic compounds, 0.5 kilotonnes of lead and 27 kilotonnes of particulates were emitted in 1990 (Carnovale *et al.*, 1992b). Over 90% of the lead is estimated to originate from motor vehicle emission, principally in the form PbBrCl.

The major contribution to volatile organics are petroleum hydrocarbons. Polycyclic aromatic hydrocarbons (PAHs) originate from incomplete combustion and will come from motor vehicle and power station emissions. They were not inventoried specifically but are included in the estimates of volatile organic carbon by Carnovale *et al.* (1992b).

There are no measured data available on atmospheric inputs for Port Phillip Bay (Carnovale *et al.,* 1992b). Dry depositions to the Bay are able to be estimated from available data, but to date this has not been done. Such estimates will be important in identifying the significance of atmospheric sources of toxicants in relation to the other input sources. It should be noted that dry terrestrial deposition of toxicants within the catchment will be measured in stormwater inputs which will carry toxicants deposited in the catchment to the Bay *via* the drainage system.

4.3 **Toxicant distribution in Port Phillip Bay**

4.3.1 **Surface waters**

The first large scale monitoring for toxicants in Port Phillip Bay was commenced in 1974. This was initially restricted to heavy metals, but in the early 1980s was expanded to include organics. As a consequence of elevated concentrations of metals being found in mussels and sediments in areas of Port Phillip Bay, the EPA undertook a review of licensed discharges, many being stopped and others diverted to the sewer. The SEPP for the Bay provided for surveys of water quality to provide information on trends over time. These were commenced in 1980.

4.3.1.1 Trace elements

Monthly monitoring of Port Phillip Bay is carried out by the EPA at the stations shown in Figure 4.2. Until 1990, three sites only were monitored routinely. Data for these sites - Hobsons Bay, Corio Bay and a central Bay site - are summarised by Phillips *et al.* (1992) and are reproduced in Table 4.7. Three additional sites - Dromana Bay, Patterson River and Long Reef - are now included in all surveys. Samples have been analysed for zinc, lead, arsenic, mercury and copper, although copper analysis is no longer undertaken.

It is difficult to compare analyses over time in this concentration range without a thorough knowledge both of the competency of the laboratories involved and the analytical methodologies employed. To verify this, it is desirable that the variability of data for replicate samples and for replicate samplings at the same site be reported. Such procedures are now mandatory as part of quality assurance and quality control protocols, but were certainly not widely practised at the time the data below were collected. Even more critical is the sampling procedure, and there is now ample evidence that the variability seen over time - in, for example, the data for zinc in Table 4.7 - could be reduced by the imposition of a rigid sampling protocol. Water sampling was undertaken using Niskin bottles, and unless these have Teflon-coated components, the prospects for contamination are high especially where concentrations might be expected to be below 1 g L^{-1}. The most recent data are not abnormal for an urban estuary but, given the size of the Bay, lower concentrations of heavy metals might have been expected at least at the central Bay site.

In the studies reported in Table 4.7, measurements were undertaken on filtered samples. Filtration was carried out in the laboratory, not in the field.

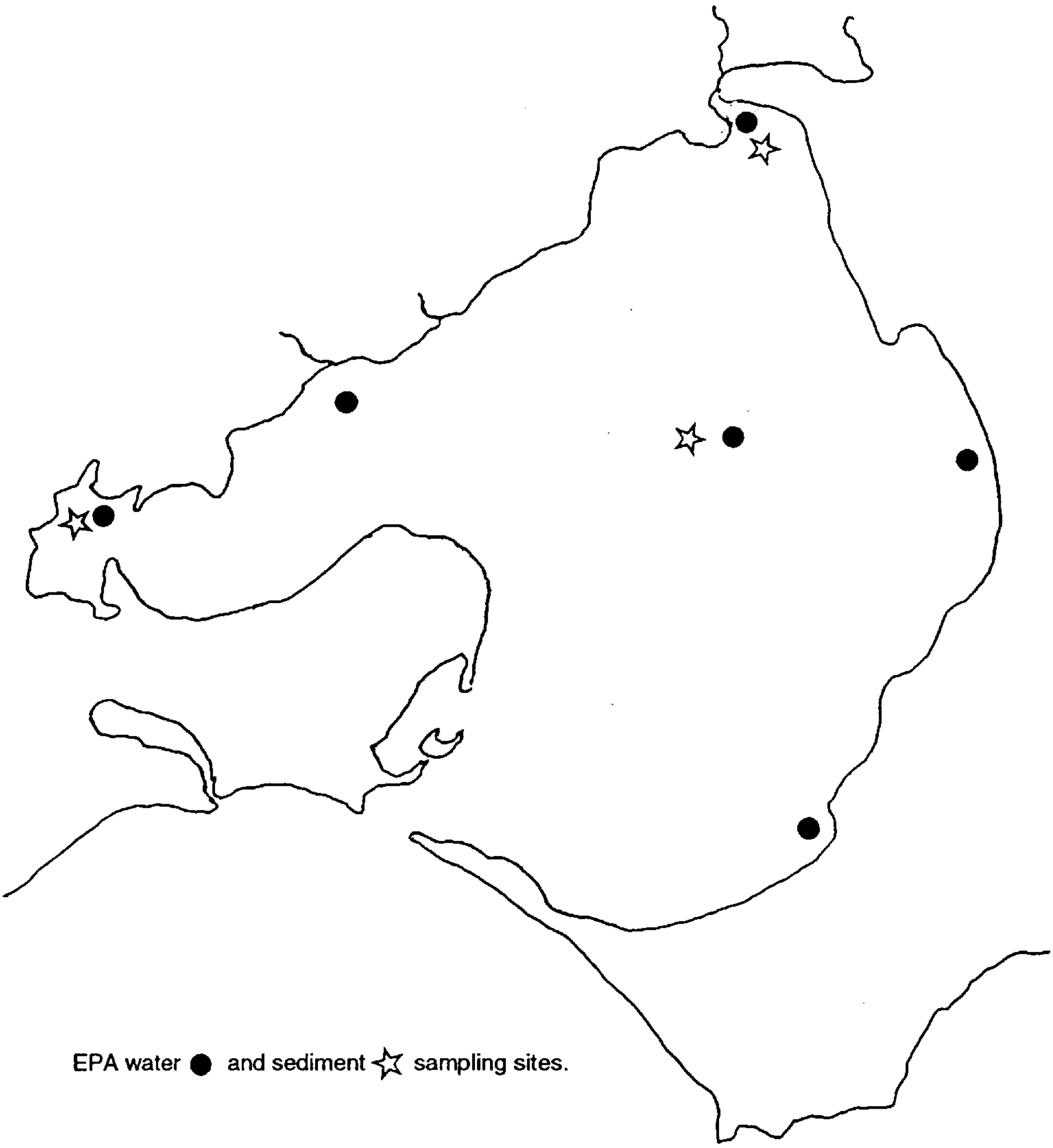

Figure 4.2 Map of Port Phillip Bay showing EPA water and sediment sampling sites.

Table 4.7. Total trace metal concentrations (g L^{-1}) in Port Phillip Bay waters, 1975-1989

Site	Year	Zn	Ni	Pb	Cu	As	Cd	Hg	Reference
Hobsons Bay	1980-82	9	1.1	3.6	-	2.3	< 0.1	0.06	EPA, 1983
	1983	2	1.2	0.7	0.8	1.1	0.1	0.006	MSL, unpub.
	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	Longmore *et al.*, 1990
	1986	< 2	1.4	< 0.8	0.8	-	< 0.05	-	"
	1987	15	3.4	3.2	2.0	2.0	0.3	0.007	EPA, 1989
	1988	4	1.0	1.4	1.0	3.1	0.1	0.003	EPA, 1990b
	1989	3	1.1	< 0.8	1.1	3.0	0.1	< 0.002	EPA, 1991b
Corio Bay	1975	-	-	1.5	3.2	-	0.3	-	Smith, 1978
	1977	30	-	0.8	5.6	-	0.4	-	"
	1980-82	< 2	1.5	1.6	-	2.6	0.3	0.05	EPA, 1983
	1983	4	0.8	1.4	0.7	0.9	0.2	0.003	MSL, unpub.
	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	Longmore *et al.*, 1990
	1986	3	1.8	< 0.8	1.6	-	0.4	-	"
	1988	< 2	1.8	< 0.8	1.1	3.3	0.3	0.004	EPA, 1990b
	1989	< 2	1.5	< 0.8	1.1	3.2	0.2	< 0.002	EPA, 1991b
Central	1980-82	< 2	0.9	1.1	-	2.6	< 0.1	0.05	EPA, 1983
	1983	< 2	1.0	< 0.8	0.9	1.2	0.1	0.003	MSL, unpub.
	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	Longmore *et al.*,1990
	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	EPA, 1990b
	1988	< 2	0.8	< 0.8	5.5	3.0	0.1	< 0.002	EPA, 1990b
	1989	< 2	0.9	< 0.8	0.6	2.8	< 0.05	-	EPA, 1991b

4.3.1.2 Organics

Very few data are available for organics in waters from the Bay. This is not surprising because it is likely that the concentrations are extremely low and are possibly better reflected in integrated concentrations accumulated by fish or other biota. Organics are not monitored in the EPA monthly water samples. Smith and Maher (1984) and Murray *et al*. (1991) have reported results for dissolved and particulate hydrocarbons, PAHs and linear alkylbenzenes in the Bay. Total hydrocarbon concentrations measured in 1977 (Burns and Smith, 1981) were highest in Hobsons Bay at 22.6 g L^{-1} compared to values below 1 g L^{-1} obtained in 1986 (Murray *et al.*, 1991). In the latter study concentrations of dissolved PAHs of the order of 0.05 ng L^{-1} and around 0.2 ng L^{-1} particulate concentrations were determined using a Seastar integrating sampler giving a concentration factor of around 10^4.

Especially where ambient concentrations are extremely low, the use of *in situ* integrating samplers offers potential advantages for both organics - which are preconcentrated on a polymeric adsorbant - and metals, collected on an 8-hydroxyquinoline-containing resin column. These were applied by the NSW EPA in a recent study of contamination associated with Sydney's ocean sewage outfalls (SPCC, 1992). A disadvantage is their cost of near $10,000 (Richardson, 1990). Of greater promise are solvent-filled dialysis cells, which can integrate organic contaminants over several days (Apte and Batley, unpublished results). These are, however, not yet commercially available. Such samplers overcome the problem of fluctuations in concentration over short intervals that has been reported in Port Phillip Bay (Fabris *et al.*, 1986).

Where the concentrations of dissolved toxicants are below 0.1 g L^{-1}, as with PAHs, one would need to seriously question the value of such measurements. More important is the extent to which the contaminant can accumulate in sediments, and ultimately the effect that both sediment and water-borne toxicants have on biota.

4.3.2 **Sediments**

Sediments are the ultimate sink for most of the toxicants entering Port Phillip Bay. An examination of sediment cores for intractable pollutants such as metals will reveal the pollution history. It is important to recognise the rates at which sedimentation is occurring and the historical window that a surface sediment sample is likely to represent. In the absence of such data for Port Phillip Bay, but with reference to other estuaries (Batley *et al.*, 1990), it would be reasonable to assume that near input sources such as creeks or rivers that rates of sedimentation may be above 7 mm y^{-1} and as high as 10 mm y^{-1}. In the centre of the Bay values might well be lower, of the order of 3 mm y^{-1}. A 10 cm deep nearshore sediment core could therefore represent 10-14 years accumulation, whereas further from the source the same depth core could represent 20 years. To look at recent deposition it is probably desirable to examine only the top 5 or even 2 cm.

Benthic biota may be capable of penetrating sediments to depths in excess of 20 cm. If sediments were being examined to assess the bioavailability of their toxicant load, then it would be appropriate to analyse to these depths. It is also important to have some feeling for the biological population within a sediment deposit and the extent to which burrowing organisms are turning over the sediments.

In the case of Port Phillip Bay the question to be answered is more likely to be what are the toxicants present in surface sediments, and are they in sufficient concentrations that they will have a measurable impact on benthic biota?

4.3.2.1 Trace elements

Sediment samples are collected quarterly by EPA from 3 sites in the Bay as shown in Figure 4.2 and discussed in section 4.3.1.1. Grab samples are obtained and are analysed for heavy metals. Recent sampling has included analyses for TBT. Total petroleum hydrocarbons were scheduled for analysis but have not been done. Samples are being retained for this analysis should it be required later.

Because sediments are easier than waters to analyse for toxicants, it is not surprising that there have been a number of studies of potentially impacted areas of Port Phillip Bay and of the input streams. Sediments have been analysed by the Port of Melbourne Authority in association with their dredging program but the report on these analyses was not available at the time of writing. The Port of Geelong Authority is planning an environmental impact assessment for a proposed dredging program which will generate considerable data on sediment composition in Corio Bay.

Sediments from Corio Bay have been extensively studied by the Marine Science Laboratories at Queenscliff (Fabris, 1983; Nicholson *et al.*, 1990). Core samples were taken over a range of sites. Depth profiling was examined at selected sites, while for the remainder the top 5 cm was analysed. The study focussed on the anthropogenic trace metal fraction and both an exchangeable and a 0.5 M nitric acid leachable fraction were measured. In general, metal concentrations in surface sediments were not exceptionally high by comparison with contaminated estuaries elsewhere in Australia (Table 4.8). It is important to note that metal accumulation by sediments is a function of grain size, with the largest concentration in the 63 m fraction. It has not been the practice to record grain size distributions with past results, but at least the fraction <63 m should be recorded in future studies to identify the sediment type. The low metal contents of some of the Port Phillip Bay sediments in Table 4.8 suggest that they must be sandy.

There was evidence from cores of historical contamination associated with discharges from a period when environmental controls were less stringent. For example, the effluents from Lindex Chemicals, containing cadmium paint pigments which were discharged between 1966 and 1974, showed as a clear spike at about 30 cm deep in some cores. Sedimentation rates estimated from such measurements were high, at 15 mm y^{-1}, but perhaps not unexpected for an enclosed bay receiving considerable drainage input.

It is important to keep in perspective the significance of contaminant concentrations in sediments. For reference some existing US sediment quality guidelines are indicated in Table 4.9. According to these criteria there are examples in Port Phillip Bay where the sediments exceed the pollution classification, such as sites in Corio Bay and Hobsons Bay where pollutant inputs have been significant as might be expected from the extent of unchecked industrial activity over past years. A recent survey of sediment contamination at 18 Bay sites was undertaken by Melbourne Water (MMBW, 1990c). Nine of these sites were adjacent to the WTC. While no statistical analysis of data was possible, there were examples of high heavy metal concentrations at a number of sites, in particular those near the WTC. One unidentified site contained 790 g Zn g^{-1}, 300 g Pb g^{-1} and 140 g Cu g^{-1}. No details of grain size were included.

Table 4.8. Trace metals (g g$^{-1)}$ in Port Phillip Bay sediments compared with data for NSW estuarine sediments

Site	Extract	Depth cm	Zn	Pb	Cu	Cd	Reference
Corio Bay (offshore)	Total	Surface	4-400	2-210	2-50	0.1-13	Smith, 1978
Corio Bay (mid)		0.5M					
	HN03	Surface	14-166	14-100	4-35	0.2-9	Fabris, 1983
Port Phillip Bay (nearshore)	Total	Surface	21	8	1.5	0.8	Talbot *et al*., 1976
Port Phillip Bay, (offshore)	Total	Surface	40	22	8	2	EPA, 1976
Port Phillip Bay, WTC	Total	Surface	9-300	<20-140	<5-75	<5	MMBW, 1991
Lake Munmorah	Total	5	150	40	70		Batley *et al*, 1990
Tuggerah Lake	Total	5	110	40	20		Batley *et al*, 1990
	Total	70	80	30	30		Batley *et al*, 1990
Lake Macquarie:							
- North	EDTA	5	1100	400	6	40	Batley, 1987
	Total	5	2400	1200	170	160	Batley, 1987
- South	EDTA	5	24	15	3	1	Batley, 1987
	Total	5	150	68	20	4	Batley, 1987
Blackwattle Bay, Port Jackson	EDTA	10	180	180	2	0.2	Batley, 1986
	Total	10	1150	520	180	3	Batley, 1986
Port Kembla Harbour	EDTA	10	230	8	8	1	Batley and Low,
1985							
	Total	10	380	113	113	2	
Quibray Bay, Botany Bay		EDTA	10	20	6	0.6	0.4
	Batley,						
							unpubl. results
	Total	10	25	10	3	0.5	

To assess the impact of recent pollution is difficult unless only the very uppermost sediment can be sampled, bearing in mind that the recent sedimentation rate is of the order of 10 mm y^{-1}. This might be adequate if surface sediment movement were insignificant or if the effects of bioturbation were absent. The former cannot be ignored, since according to Seedsman and Marsden (1980) strong winds are capable of creating waves that disturb the sediment floor in areas up to 6 metres deep. The extent of bioturbation cannot be estimated without some knowledge of the presence of worms or other sediment-dwelling biota, however, based on the typical depth of burrows it is likely that bioturbation adds no more than a 50-100 mm window to the depth at which a pollution event is indicated.

Table 4.9 Sediment Quality Guidelines

Element	EPA Region V Guidelines for Pollution Classification (mg kg^{-1})	USGS Sediment Alert Levels (mg kg^{-1})	Ontario Ministry of Environment Dredge Spoil Guidelines (mg kg^{-1})	Interstitial Water or Elutriate Water as sediment quality indicators - EPA Region VI (g L^{-1})
Arsenic	3	200	8	440
Cadmium	6	20	1	
Chromium	25	200	25	0.3 (CrVI)
Copper	25	2000	25	5.6
Iron	17,000		10,000	
Lead	40	500	50	
Mercury	1	20	0.3	0.0006
Nickel	20	2000	25	
Selenium		20		35
Silver		1000		
Zinc	90	5000	100	47

Recent concern for TBT as a contaminant has led to extensive studies of sediments and biota by EPA (1990a). As anticipated, elevated concentrations were found in the vicinity of marinas (up to 2400 ng Sn g^{-1} in Hobsons Bay). This is not unexpected given the historical usage, and would be expected to decrease - at least near marinas - with the recent banning of TBT for use on boats under 25 m long.

4.3.2.2 Organics

As with waters, organics are not being addressed in routine monitoring of sediments in Port Phillip Bay. Specific studies have considered particular contaminants.

Several studies have addressed the concentrations of hydrocarbons in Bay sediments (Burns and Smith, 1978, 1982). As expected, high concentrations were found at nearshore sites in Corio and Hobsons Bays. Polycyclic aromatic hydrocarbons have been determined by Bagg *et al.* (1981), Smith and Maher (1984), and Murray (1987). In Hobsons Bay, for example, a sediment core contained 570 ng g^{-1} of benzo(a)pyrene at the surface, increasing to 1140 ng g^{-1} at 15 cm depth. This was in the presence of a total hydrocarbon concentration at the surface of 177 g g^{-1} and 313 g g^{-1} at 15 cm.

Following the discovery of dieldrin and PCBs in cattle at the WTC in 1988, an extensive study was initiated to detect the presence of such organics both at Werribee and offshore. This was accelerated by public concern aroused by Greenpeace in 1990 over the discharge to the sewer from Nufarm of wastes potentially containing dioxins and furans. Measurements of water supply samples revealed no detectable levels of the latter compounds (MMBW, 1990b). Analyses of sediments from Port Phillip Bay showed concentrations of polychlorinated dibenzodioxins (PCDDs) ranging from 9 to 62000 pg g^{-1} (parts per trillion), with polychlorinated dibenzofurans (PCDFs) lying between 0.2 and 1700 pg g^{-1}. The highest concentrations were in the Yarra and Maribyrnong estuaries, with the origin thought to be combustion processes. The main congeners were the octachlorinated species which have low toxicity. Values for the highly toxic 2,3,7,8 tetrachlorodibenzodioxin were below 0.3 pg g^{-1} in Bay sediments and 1.2-4.3 pg g^{-1} in the estuaries. Results generally indicated that in addition to the estuarine (urban) input, Corio Bays petroleum complex and WTC were also sources of the less toxic PCDDs.

Richardson and Waid (1980) undertook a detailed survey of sediments for PCBs. Highest concentrations were found in Hobsons Bay (55-60 ng g^{-1}), inputs to Corio Bay, where values ranged from 230 ng g^{-1} near the Corio tip drain to 810 ng g^{-1} inside the Rippleside drain, and in the Corio Channel (86 ng g^{-1}). All other Bay sites were below 10 ng g^{-1}. The inshore sites are not excessive by world standards. More important is the extent of accumulation by aquatic biota, as will be discussed later.

4.3.3 **Biota**

Biota are the final integrator of toxicant load in the Bay. Depending upon the species, they may accumulate from both the water column or the sediments. The bioaccumulated dose will be significant if it represents a threat to aquatic life or is likely to be biomagnified in higher organisms, or if it represents a health risk where the species are harvested for human consumption.

It is not usual to define acceptable concentrations in plankton, algae and lower organisms, rather the impact on these species is usually defined on the basis of water quality criteria for the protection of aquatic ecosystems (Table 4.2), which in turn are derived from LC50 values for the most sensitive species, or some ″no-observed-effect″ concentrations. Sediment quality criteria are applicable to sediment-dwelling biota.

Of the commercially-harvested species, fish, mussels, oysters, scallops and abalone will be important and the extent of contamination will be defined by the National Health and Medical Research Council guidelines (NH&RMC, 1987, 1988)(Table 4.10)

Table 4.10 NH&MRC guidelines for toxicants in aquatic biota for human consumption

Element	Maximum permitted concentration ($g\ g^{-1}$) Fish	Shellfish
Antimony	1.5	-
Arsenic	1.0	1.0
Cadmium	0.2	2.0
Copper	10	70
Lead	1.5	2.5
Mercury	0.5	0.5
Selenium	1.0	-
Zinc	150	150 (mussels) 1000 (oysters, scallops)

A more detailed review of toxicants in Port Phillip Bay fisheries can be found in the report by Nicholson (1992).

4.3.3.1 Fish

The bioaccumulation by fish of both inorganic and organic toxicants from Port Phillip Bay has received only limited attention. This is despite the rather considerable commercial and recreational fishing taking place in the Bay. The mean annual commercial catch of finfish from the Bay was 1675 tonnes for the years 1990 and 1991. By far the greatest catch weight of any given species was pilchards (about 750 tonnes per annum) compared to whiting and snapper (*ca* 120 tonnes each) and flathead and anchovy (about 25 and 16 tonnes respectively) (Nicholson, 1992). Surprisingly little data are available on contaminants in pilchards or most of the other commercially and recreationally important species. Flathead represented at least half of the total recreational catch weight in 1982-83, principally as the sand flathead (*Platycephalus bassensis*)(MacDonald and Hall, 1987).

The concern for mercury contamination in the 1970s led to monitoring studies being reported on sand flathead in the Bay. Walker (1982) found concentrations exceeding the NH&MRC limit of 0.5 g Hg g^{-1} compared to <0.2 g Hg g^{-1} in Bass Strait. By 1990 these concentrations had declined to about half of the maximum recommended limit (Fabris *et al.*, 1991), although some samples from Corio Bay were still close to and sometimes over the limit. Data were also collected for mercury in snapper, King George whiting, southern anchovy and pilchards (Walker, 1988).

In response to fish kills and evidence of visible lesions on sand flathead and globefish in the Bay in 1984, analyses for metals were carried out and elevated concentrations of cadmium and lead were recorded (Fabris and Gibbs, 1985). Measurements were made on fish collected from three sites - one in the vicinity of WTC and the other two east of Portarlington. Cadmium enrichment was observed in the liver of flathead compared to those inhabiting Bass Strait control, while lead was higher in the bones of both species relative to the control.

Flathead and mullet were the subject of a major study of organic toxicants by the Melbourne Metroploitan Board of Works (MMBW, 1991). Phthalate esters, chlorinated organic hydrocarbons, PCBs, phenols, organochlorine pesticides and PAHs were detected in many instances. Both species contained DDE, dieldrin and DDD, PCBs (principally with 4, 5 and 6 chlorines), bis(2-ethylhexyl)phthalate, 1,2- as well as 1,3/1,4 dichlorobenzene, and very low concentrations of chlorophenols and PAHs.

A more detailed study of petroleum and chlorinated hydrocarbons in sand flathead from ten Bay sites was reported by Nicholson *et al.* (1991b). Petroleum hydrocarbons exceeded the threshold of 10 g g^{-1} for taste tainting in samples from the Geelong Arm and two sites offshore of St Leonards, but PAHs were barely detectable. Dieldrin, DDT, DDD and DDE were widespread contaminants in fillet and liver from fish throughout the Bay, the highest concentrations being in fish from the Geelong Arm. There, dieldrin in fish liver was twice the health limit of 0.1 g g^{-1}. Lindane and heptachlor occurred in patches throughout the Bay, while arochlors 1254 and 1260 were widespread PCB contaminants, especially in Corio Bay and the Geelong Arm. Results suggest that the Bay is chronically contaminated with petroleum and chlorinated hydrocarbons.

4.3.3.2 Shellfish

The use of mussels worldwide as indicator organisms has resulted in a number of studies of the impact of toxicants in Port Phillip Bay on common blue mussel, *Mytilus edulis planulatus*. As early as 1974, mussels from Rippleside Pier in Corio Bay provided evidence of the cadmium discharges discussed earlier (Adler, 1975), with cadmium concentrations some 20 times the NH&MRC recommended limit of 2 g g^{-1} wet weight. Concentrations are now close to the limit in the Rippleside area but elsewhere in Corio Bay are below this value (Fabris, 1991; Nicholson *et al.*, 1991a).

Copper and zinc in mussels were never found to exceed the NH&MRC limit between 1975 and 1988, remaining relatively constant with time at 24-42 g Zn g^{-1} and 0.7-0.8 g Cd g^{-1} wet weight (Fabris, 1991). Lead concentrations in offshore samples were also well below the health limits, but data for intertidal mussels from Corio Bay from 1977-78 showed 0.4-14 g Pb g^{-1}. Mussels from sites such as the Rippleside area still exceed the limit, while elsewhere concentrations appear to have decreased. Hickman *et al.* (1984) have suggested alert levels for heavy metals to detect future degradation of water quality. On a dry weight basis these were respectively 128 g Zn g^{-1}, 2.0 g Pb g^{-1}, 6.4 g Cu g^{-1} and 2.7 g Cd g^{-1}. These were based on the sum of the mean and standard deviation from a years' monthly sampling.

Arsenic in mussels from Corio Bay was close to the limit of 1 g g^{-1}, but did not pose a threat because it was mostly present in non-toxic forms (Walker, 1989). Mercury concentrations in mussels were well below the limit, and accumulation by mussels appeared to be less significant than for fish which are more prone to biomagnification through the food chain.

Sampling of oysters has been limited to selected sites in the Bay. Except for cadmium, metals have always been within the statutory limits. Cadmium in Corio Bay oysters has been close to the limit.

A similar dearth of data applies to scallops (*Pecten alba* Tate). Corio Bay concentrations exceeded the health limits for cadmium and arsenic in 1972, whilst samples from other parts of the Bay also contained elevated cadmium. Concentrations of cadmium are now within health limits and concentrations of other elements are generally unchanged (Fabris, 1991).

The presence of TBT in shellfish has received some attention. Measurements taken in 1989 showed that in most instances concentrations were below 10 ng Sn g^{-1} fresh weight. By comparison, in oysters impacted by TBT, concentrations exceeding 40 ng Sn g^{-1} induced shell deformities and reduced growth, and values as high as 100 ng Sn g^{-1} were not uncommon. Any impact on scallops is likely to be more significant for spat, since being sub-tidal, scallops are not exposed to the high surface film concentrations that affect intertidal shellfish (see Batley and Scammell, 1991). The subtidal mud oyster, *Ostrea angasi*, common in Port Phillip Bay, is also likely to be less seriously affected than intertidal oysters.

Only one set of data is available for abalone (Walker, 1979), and all trace elements measured - except arsenic - were within guidelines: a reflection on inadequate guidelines rather than an environmental concern.

The accumulation of organics by mussels has been reasonably well-studied. Hydrocarbons in mussels were investigated by Burns and Smith (1982), and, as with fish, the highest concentrations were found in Corio and Hobsons Bays. Concentrations appear to have remained unchanged since then and remain a concern in the Bay generally. The value of mussels as bioindicators of hydrocarbons has been assessed by Murray *et al.* (1991). Bioconcentration factors of around 10^6 were estimated on the basis of a three month deployment of mussels and 4-day samplings of water using Seastar *in situ* water samplers at times during the three months. Their water data are the only so far obtained using an integrating sampler. Richardson (1990) makes a strong case for the use of mussels as biomonitors.

Data on organochlorine and organophosphate pesticides and PCBs in mussels from the Bay have been summarised by Fabris (1991). These are limited and more data plus an extension of measurements to chlorinated phenolics or other compounds not previously investigated could be justified.

4.3.3.3 Macroalgae

The interactions of toxicants with seagrasses and other macroalgae has been reviewed by Light and Woelkerling (1992). The accumulation of metals by seagrasses has been widely discussed elsewhere (Batley, 1987; Ward, 1989; Batley *et al.*, 1990). Uptake can occur both from sediments and from the water column.

Fabris *et al.* (1982) carried out experiments with *Heterozostera tasmanica* and demonstrated that cadmium uptake occurred principally from the water column *via* the leaves. Seagrasses are good bioindicators of metal contamination.

4.4 **Dynamics of toxicants in Port Phillip Bay**

4.4.1 **Dynamics of toxicants in surface waters**

While physical transport is important in describing the movement of toxicants from input sources to the Bay, the concentrations of these contaminants are far from conservative. A number of important processes have to be considered. These will be governed by the salinity of the water, the presence of suspended particulate matter, the chemical stability of the toxicant and its chemical speciation.

The behaviour of heavy metal toxicants has been thoroughly reviewed (Florence and Batley, 1980). In solution, only a small percentage of metals are present as simple ionic species, with many bound in organic or inorganic complexes or associated with colloidal or particulate matter. Their availability to organisms for bioaccumulation will be defined by this speciation. An important finding has been the large proportion of metals in colloidal forms associated with hydrous iron oxides and stabilised by humic and fulvic acids. As freshwaters mix with saline waters these colloids aggregate into particulates which gradually settle to the bottom sediments carrying with them entrained metals. There is therefore likely to be high concentrations of metals in sediments in the mixing zones of estuaries where the salinity varies between 0 and 35 parts per thousand (ppt). By comparison, the binding of metals with algal exudates, humic acids or other soluble organic ligands is likely to retain metals in a soluble form, although this fraction is relatively minor as entrainment in particulates is likely to be an important removal process. Changes in redox potential, for example in bottom waters in a river or in the deeper parts of the Bay, will also modify solution chemistry. Importantly, many metals will form insoluble sulfides, while iron and manganese are likely to be solubilised in their divalent states. Such stratification is not uncommon, particularly where different salinity waters are capped by large inflows of freshwater.

Biodegradation processes are unlikely to be significant for dissolved metal species.

The equilibrium solubility of organic contaminants can be largely predicted on the basis of their octanol-water partition coefficients. These can be measured directly or predicted on the basis of QSARs, quantitative structure activity relationships. Compounds with log k_{ow} values exceeding 2 will preferentially partition to particulates or to the surface microlayer.

While inorganic toxicants are relatively stable to degradation processes, organometals and organics can be transformed by both photolysis and hydrolysis, the latter often being more important for many pesticides. Microbiological processes can also be significant.

4.4.2 **Dynamics of toxicants in sediments**

4.4.2.1 Decomposition and mobilisation

Toxicants accumulated in sediments will be present in a number of chemical forms. Inorganics will be adsorbed in ion exchangeable forms, coprecipitated with hydrous oxides, bound to carbonates, precipitated as sulfides, bound to organic matter or in mineralised forms. Changes in redox potential will result in the solubilisation of iron and manganese, but precipitation of many heavy metals as insoluble sulfides will occur.

The bioavailability of metal toxicants will principally be controlled by the free or labile metal ion concentration in the sediment interstitial waters. This will be moderated by the sulfide-binding capacity of the sediments which will remove metals as insoluble sulfides.

Organics are most likely to be adsorbed to clays or organic matter. These will be in equilibrium with sediment interstitial water. In sediments, microbial degradation processes become important. These can degrade organics but also result in methylation of some metal and organometal species.

There is evidence to suggest that the availability to biota of metal toxicants in sediments is controlled by the free metal ion concentration in the interstitial water, and that of organics by their equilibrium partitioning (Di Toro *et al.*, 1991). This makes measurements of interstitial water concentrations important in any sediment study. The availability of metals in the interstitial water will be modified by the acid volatile sulfide concentration, since by binding metals sulfides can reduce toxicity (Di Toro *et al.*, (1990). Interstitial water concentrations can be effectively sampled using sediment peepers - chambered Perspex cells containing seawater covered with a membrane which can be placed in the sediment and allowed to equilibrate with the pore waters. The chambers can later be sampled and analysed for the equilibrium concentrations of metals or organics.

4.4.2.2 Impact of dredging

Regular maintenance dredging is carried out in dock areas and shipping channels in Port Phillip Bay to remove accumulated sediments. These sediments may contain high toxicant concentrations, especially in the inshore port areas in Hobsons Bay where sedimentation associated with the Yarra River is combined with oil, PAHs and antifouling paint constituents associated with high shipping densities. During dredging there is the possibility for remobilisation of toxicants as anoxic sediments are exposed to oxygenated seawater. Overlying seawater is recycled to the Bay during the dredging operation, and the dredge spoil is dumped elsewhere in the Bay. Toxicant release may also occur during the dumping operation. A secondary effect of dredging is the physical smothering of existing biota.

The Port of Melbourne Authority controls and administers shipping channels and is entrusted with the maintenance of guaranteed depths in the "Rip" and South and West Channels and in the port areas on the Yarra River including the Port Melbourne Channels (EPA, 1991c). Dredging in the South Channel involves removal every two years of about 60,000 m^3 of mainly redistributed sand. The West Channel is reasonably stabilised, but increasing the depth by an additional 0.5-1.0 m may be considered. Within the Port, approximately 500,000 m^3 of material, mainly silt, is dredged annually. The Port of Geelong Authority undertakes dredging in Corio Bay. Both Authorities have plans for major dredging works in the next decade.

Melbourne Water is responsible for drainage and waterways including the Yarra River upstream of Spencer Street Bridge and the Maribyrnong River upstream of Footscray Road Bridge to its junction with Jacksons Creek at Keilor. Annual dredged volumes are typically 40,000 m^3 for the Yarra and 30,000 m^3 for the Maribyrnong. Major maintenance dredging has been suspended since 1989 with the introduction of a comprehensive silt monitoring system, but minor dredging is still carried out.

The dumping ground for Yarra spoil is an area 3 miles x 1 mile, and 14-20 m deep. It has been used for 50 years. It is surveyed annually to ensure that the depth is more than 14 m. Spoil dispersion is slow (PMA, 1989).

Scallop dredging also has the potential to disturb contaminated sediments, although with less impact than channel dredging. Because of the public concern for this activity, it is desirable that the extent of any impact with respect to toxicant release be assessed. Fabris (1981) discussed the likelihood of heavy metal mobilisation during dredging in relation to the sedimentology of the areas concerned. Heavy metals will be predominantly associated with the finer clay and silt sediments rather than sand substrates. The distribution of scallop beds covers both sand and clay sediments. Resuspension of the latter will not result in large pH changes but - in contact with aerobic seawater - slow oxidation of metal sulfides is possible together with complexation with chloride ions and soluble organics and desorption through competitive cation exchange reactions.

Data available on metal concentrations in the areas of scallop beds show that metal concentrations are generally not high, with the possible exception of cadmium. This would have a possible short-term impact on the water concentration. An analogy is the effect of metal release during the ocean dumping of anoxic sediments. In experiments where metals were measured in the plume of the dumped material, only minor increases in metal concentrations were observed where sediment concentrations were considered to be high (Batley and Low, 1985).

The impact of dredging is listed as a specific objective of the Port Phillip Bay study, however, any process that remobilises toxicants is effectively creating an additional input, and as such should be considered in any model of the Bay. Clearly the heavy deposition of sediments that occurs in channels and ports will always need maintenance dredging. Should a toxicant problem be revealed in this study, alternative dredging methods, with the possibility of ocean dumping, might need to be considered as management options.

The EPA have prepared a protocol for dredging and disposal of dredged material (EPA, 1991d). In essence this is based on that of the London Dumping Convention used to control ocean dumping of dredged material, but taking into account current knowledge of contaminant behaviour. Concentrated acid extractions are recommended for total extractable species, with elutriate tests for any suspected problem contaminants. Material to be dumped should, in the case of heavy metals, be no more than three times the ambient metal concentration at the dump site. For organics the factor varies with compound between 10 and 100. This seems an overly stringent requirement, unless some consideration is made of sediment types. A more important assessment is the possibility for short-term and longer-term mobilisation. The former assessed by elutriate tests, the latter - taking into account microbial processes - usually results in a slower release which must be considered alongside water quality criteria.

4.4.2.3 Sediment toxicity

Since the ultimate sink for most toxicants is the sediments, it would be desirable to be able to rank sediments on some scale of potential toxicity. A generic test method such as the Microtox photoluminescent bacterial bioassay is the logical first step.

Microtox testing equipment is currently available in at least three Victorian laboratories, and there are established methods for its application to sediments. Prior to initiating this task, other local and overseas experience on sediment toxicity testing by this method would need to be evaluated in the light of measured toxicant concentrations to assess its feasibility. If it looks promising, this small pilot study would be carried out and the results related to sediment data to determine its value as an early warning indicator of toxicants in sediments.

Other assays designed to assess sediment toxicity have been based on the use of burrowing amphipods, or on algal or other bioassays undertaken on the sediment pore waters.

4.4.3 **Fish health**

Sub-lethal effects of toxicants on biological communities are not necessarily indicated by bioaccumulation data, or from ecological studies of species diversity and abundance. Some measure of non-acute impact (on abundance) is important to the fisheries resources and the food chains on which they depend.

The presence of tainting or any visible anatomical abnormalities in fish or other biota will certainly have an immediate impact on the consumer and a devastating effect on value of the commercial and recreational fisheries in Port Phillip Bay. The existence of any such symptoms should be the first target of this task, with an attempt being made to relate its cause to any toxicant identified by chemical analysis.

A major fish kill in 1984 was accompanied by lesions, focal haemorrhages and cutaneous erythema in spikey globefish (Gibbs *et al.*, 1986b). Sand flathead had no external lesions but had severe visceral haemosiderosis. The condition was accompanied by high tissue cadmium and lead, as discussed above, but this was not considered to be responsible for their condition. There is circumstantial evidence that organic toxicants such as PAHs may be involved in chronic haemosiderosis, but their presence in this case was not confirmed. A further possibility is that toxic dinoflagellates were involved.

The toxic effects of some marine algal blooms is an area that has yet to be fully investigated. Parry *et al*. (1989) showed that a bloom of the diatom *Rhizosolenia chunii* resulted in a bitter taste in flat oysters (*Ostrea angasi*), mussels and scallops in late 1987. Mussels also showed digestive gland lesions. High mortality ensued for some 3 to 8 months after the bloom. An early detection method for impending toxic algal blooms has yet to be devised, however, a comprehensive biotoxin surveillance project has been devised for implementation in 1991 (Arnott, 1990a). This involves frequent phytoplankton monitoring, routine mouse bioassays to determine toxin concentrations in mussels and an early warning network to report abnormal environmental phenomena such as water discoloration and fish or shellfish kills.

A more sophisticated approach to sub-acute toxicity in fish and other biota is to look for specific biochemical or physiological indicators. This may involve the measurement of enzyme activity, liver function and a range of other parameters. Many such tests are being applied to fish in the more contaminated rivers of Europe and in the North Sea. In Port Phillip Bay, liver enzyme biomarkers are currently being studied in a project at the Royal Melbourne Institute of Technology (RMIT). Such studies are also being undertaken as part of the National Pulp Mill Research Project, and in other laboratories in Australia and abroad.

While research is being actively pursued to develop new biomarker techniques there is a need to apply appropriate existing tests as additional monitoring tools to assess toxicant impacts. From a management point of view it is important that these be linked to known causes, toxicants or otherwise, to provide data on the overall health of the Bay.

4.5 **Modelling toxicant behaviour**

The main routes for transport, transformation and biological uptake of toxicants in estuarine systems are illustrated in Figure 4.3. In the case of organic contaminants, the partitioning between dissolved and particulate forms can be readily modelled. Indeed there are a number of fate and transport models which take into account degradation processes and predict with some reliability the concentration of toxicants as a function of time.

Numerous simulation models have been developed for toxicants in surface water systems and these have been reviewed by Ambrose *et al.* (1988) and Waite (1989). For general application, a model must include benthic and water column compartments, transport and partitioning processes and transformation reactions. Four such models are MICHREV, a steady-state model for organics and metals; EXAMS-II, a quasidynamic model for organics, and WASP4 and HSPF, dynamic compartment models for organics and metals. There are several both 1- and 2-dimensional dynamic models for organics and metals, a dynamic multimedia model, and a steady state 1-dimensional model. MINTEQ is an equilibrium geochemical model for metal speciation which has special application to solid phase metal inputs.

4.6 **Monitoring**

There is already a large body of knowledge concerning toxicant inputs and distributions in the Bay obtained by various government and academic research groups. With the exception of the long-term fixed sites monitoring program, many of the data are old and both site and contaminant specific. The advances in analytical methodology and awareness of the existence and effects of new toxicants limit the value of some of the past data. Coolection of past data has usually been driven by specific impacts such as fish kills or disease, accumulation in biota, expedience of health standards, or events such as known spills. There is a need to integrate and evaluate the existing data and supplement this by research and monitoring programs which will permit the evaluation of implications of toxicant concentrations in waters, sediments and biota on the long term environmental health of the Bay.

The important input data for any model of toxicant behaviour in an estuary will be the total concentrations in both dissolved and particulate forms at the point of entry into the aquatic system. The observed concentrations in both waters and sediments in the estuary exist under pH, salinity and redox regimes that may differ from that in the input water, and these will be important in defining the spatial and temporal variabilty of the contaminants and important for verifying the model. The chemical form of metal species will also be important in determining bioavailability, but any detailed speciation measurements would be beyond the scope of this study.

Monitoring biota is the ultimate test of the bioavailability of contaminants. The integrated dose received by fish, molluscs and crustacea defines their suitability for human consumption, and, should they be unsuitable, will impact heavily on the public perception of the health of the Bay.

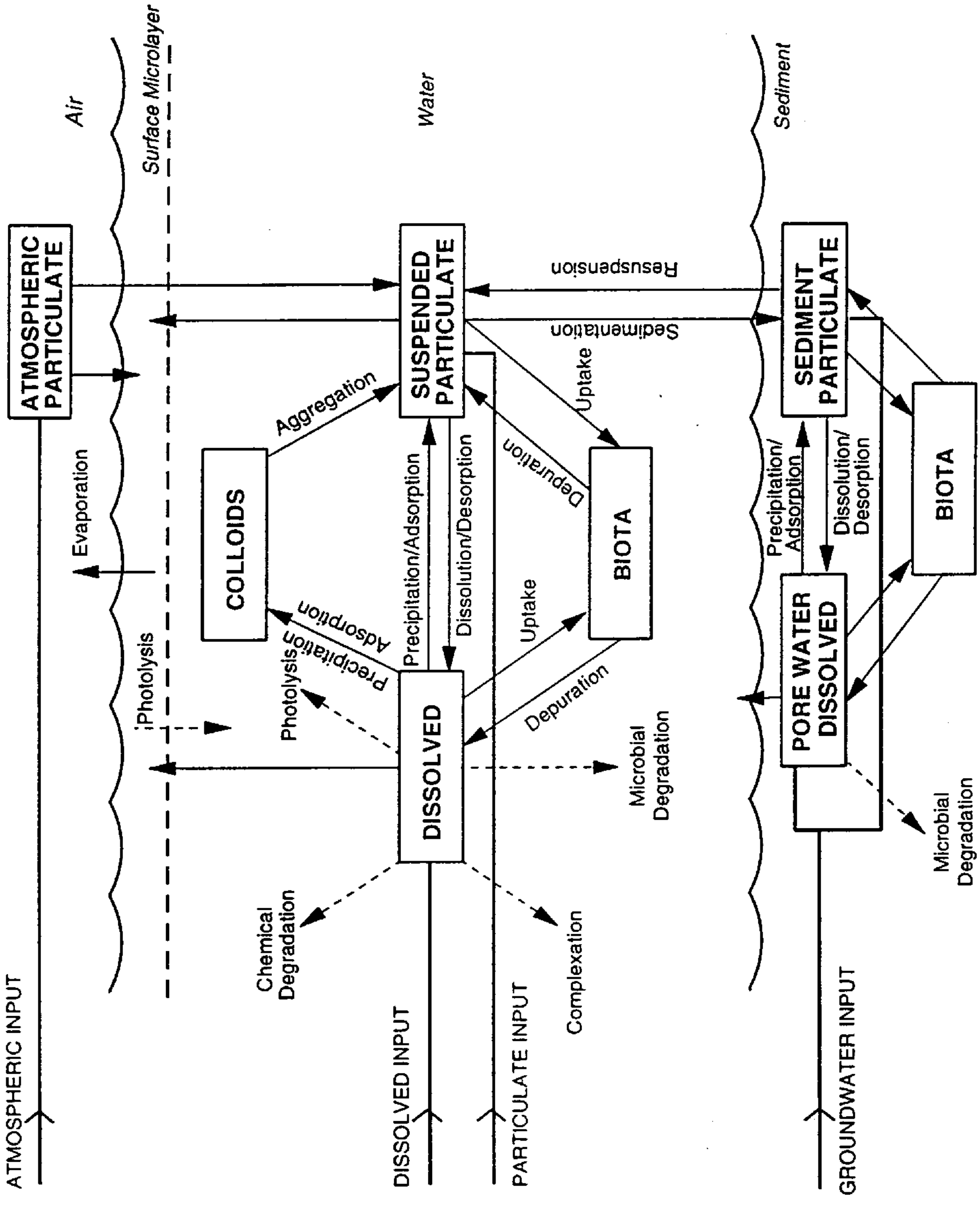

Figure 4.3 Transport and degredation routes for toxicants in Port Phillip Bay.

The need for demonstrated expertise in undertaking any monitoring exercise for trace contaminants cannot be too highly stressed, and this begins with the sampling protocol and includes all of the precautions and complexities of the procedures for preconcentration and measurement. It is important that the needs for achieving the best available detection limits are tempered by an awareness of the required detection limits in order to assess the health of the water, sediment or biota with respect to an unimpacted or control site.

In terms of monitoring the impacts of toxicants, bioassays can be of value. Systems such as the Microtox bacterial bioassay, algal bioassays and chronic fish bioassays can all contribute in evaluating the ultimate impact. In the assessment of the impact of dredge spoil, US criteria stipulate the need for bioassays on three species of all sediment elutriates. Sediment quality criteria are being developed on the basis of bioassays with amphipods and other key species. Critical factors in the acceptance of these approaches are cost, time and relevance. A 28-day test costing $20,000 will be of little value because it takes too long and costs too much. Shorter more cost-effective tests will gain acceptance if applications on lower organisms can be related to impacts further up the food chain.

4.7 **Knowledge gaps and research needs**

While it is evident from this review of toxicants that a large amount of data has been accumulated for Port Phillip Bay, it has largely been the result of isolated studies. Although existing information enables an overview of the status of various parts of the Bay with respect to toxicants, the development of an end to end understanding of toxicant fate and transport in the Bay is not possible. The comprehensive, integrated study of nutrients and toxicants in inputs, waters, and sediments proposed in the Port Phillip Bay Environmental Study should provide a better basis on which to assess the health of the Bay.

It is important that all of the deficiencies noted in the existing information be included in determining the needed research and monitoring tasks in order to maximise their value.

The originally proposed water monitoring task may need to be streamlined, and the proposed detection limits for many of the species re-evaluated, especially if these are well below the levels known to have chronic biological impacts. Since previous studies have shown that PAHs and many other organics are not present in significant concentrations in solution, while as predicted by equilibrium partitioning they are almost entirely bound to sediments, then their measurement (at considerable expense) in water would seem unwarranted. The frequency of monitoring should also be considered. Should we be looking for statistical variability of the Bay concentrations or an estimate of typical concentrations at several times throughout a year? The difficulty in obtaining meaningful results considering the potential variability of toxicant concentrations should be weighed against the option of using integrators of pollutant load such as biota and sediments.

Certainly more useful data can be obtained from input river and stream analysis. There are good EPA data but not for all parameters. Knowing their findings, and in consideration of those compounds found in mussels and fish, the input monitoring could be extended to include selected organics and other elements - possibly selenium and arsenic - should these be found in biota.

Other inputs that have not been fully characterised include stormwater, atmospheric deposition and groundwater. Stormwater has not been satisfactorily studied for some 11 years. In addition measurements of a major stormwater receiving system such as the Yarra over a storm period might be enlightening. Calculations of estimated dry deposition might suffice in the first instance to estimate the significance of this source of toxicants, but there is a need to measure the toxicants in groundwater from the sites that have historically shown contamination.

There is a gross deficiency in sediment data for toxicants. These must be related to the grain size of the sediments in order to make meaningful assessment of pollutant distribution and bioavailability. The depth of sediment sampled must be carefully considered to determine whether the data refer to recent or past deposition. There would be some value in screening sediments for toxicity using amphipods or the Microtox bacterial bioassay.

Toxicant concentrations in biota remain a high priority and should precede some of the water monitoring to highlight toxicants of concern. It is surprising that there are no data on toxicants in pilchards - which represent the largest commercially-harvested fish species in the Bay - and these should be included in any study. There is also a need for the development and application of procedures to assess chronic impacts.

The impact of shipping needs to be more fully evaluated. Measurements of toxicants in waters, biota and sediments in and around the port areas in both Melbourne and Geelong which are subjected to dredging would be of value, together with information on the fate of sediment-bound toxicants, their mobilisation by dredging and their potential toxicity.

In all proposed studies, integration of the sampling and results of the toxicant program with those of the physical, nutrient and ecology programs at all stages is vital. The research tasks should be seen as not set in stone, but able to be modified on the basis of a continuing review of the integrated results. While this may cause problems because it makes the contractual agreements more difficult to define, it is essential if we are to achieve a useful understanding of the health of the Bay, how it operates and its assimilative capacity for altered inputs.

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Chapter 5. Ecology

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5.1 Introduction

This is a Study Update: a reassessment of the Study Design based on information contained in commissioned specialist reviews of Port Phillip Bay ecology. These reviews investigated and reported on the state of existing knowledge of the Bay ecology, including physical, chemical and biological processes controlling the interaction of ecological components with nutrients and toxicants and the knowledge needed to predict the ecological impact of a 50% increase in nutrient input. Areas covered by the specialist reviews included phytoplankton (Wood and Beardall, 1992), benthic flora (Light and Woelkerling, 1992), benthic fauna (Poore, 1992), fisheries (Nicholson, 1992) and water column trophodynamics (Crawford *et al.*, 1992). A workshop on marine birds and mammals was convened (9th September 1992, at Melbourne Water) to assess the state of knowledge on these biota with respect to the Study Objectives. Information reported during the workshop has also been used to evaluate the Study Design. The Port Phillip Bay study update also considered recent publications such as the appraisal of the effects of Werribee Treatment Complex (WTC) on Port Phillip Bay (Newell, 1990), the identification and use of indicators for Victorian coastal and marine environments (Coleman *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, 1991).

The objectives outlined for the Port Phillip Bay Study fall into three generic categories:

- determine the existing status ("environmental health") of the Bay;
- determine whether the "health" of the Bay is changing over time; and
- predict the impacts of nutrients and toxicants.

Ecological systems such as Port Phillip Bay are naturally variable and largely unpredictable. In the absence of a detailed knowledge of the systems and their natural behaviour, it is generally not possible to precisely and accurately predict the ecological impacts of discharges of nutrients and toxicants, except at very local scales and in respect of gross impacts and on short time scales. In this study we propose to predict the ecological impacts based on simple linkages between water and/or sediment concentrations and biological effects derived from existing ecotoxicological literature and from patterns emerging from observational studies in Port Phillip Bay.

Therefore there is a critical need to develop a set of studies which will provide early indicators of adverse change and expose the linkages with changing levels of nutrients and toxicants. When the fate of nutrient and toxicant distributions can be modelled and combined by empirical linkages with ecological impacts this will provide a general predictive capability. As the Study progresses and the models are improved, critical gaps in process level understanding will emerge and these will need to be addressed by targeted experimental studies. The behaviour of the nutrients and toxicants and apparent cause and effect relationships should be tested in appropriate field and laboratory experiments.

These experiments should be designed to test explicit hypotheses derived from the combination of real management options for control/management of the nutrients and toxicants and their predicted behaviour in Port Phillip Bay.

The Port Phillip Bay Environmental Study aims to:
(1) conduct a set of observational studies;
(2) correlate the ecological observations with information on nutrients and toxicants; and
(3) formulate empirical models for interactions of nutrients and toxicants with the valued attributes of the Bay.

Most of the ecological tasks described in the Study represent points 1 and 2 above: providing biological data to integrate with the other nutrient and toxicant investigations. These ecological studies would build on data from earlier studies of the Bay and would be aimed at demonstrating ecological trends in time and space and identifying empirical relationships between nutrients and toxicants.

These studies will describe and measure in detail the ecosystem and population level indicators that may be used to represent the effect of nutrients and toxicants on the attributes of the Bay system, with particular emphasis on indicators that produce information of value to management agencies. The attributes are defined as valued properties of Port Phillip Bay, ranging from the commercial resources such as fisheries products to the purely aesthetic attributes such as perceptions of environmental quality. Measurement of the appropriate indicators (i.e. those that are linked to nutrients and toxicants and are measurable in a quantitative sense) over sufficiently long periods of time and appropriate spatial sampling intensities is probably the most efficient and reliable way to assess the present ecological status. Such temporal measures will also indicate the direction of change for key ecological components. The indicators are chosen to provide an optimal amount of information about the system attributes. Recent reviews on indicators by Coleman *et al.*, (1991) and Quinn and Keough (1991) provide valuable input about this subject.

The emphasis on the Port Phillip Bay Study is on indicators of primary production because these are expected to respond earlier and change more rapidly than secondary producers. However, secondary producers and other valued species are also important and appropriate indicators have been included.

Where existing information is scant or non existent, studies will also be initiated to obtain suitable baseline data from which a long-term monitoring program can be developed. These initial studies will include detailed measures of the natural scales of variability in space and time, if such information does not already exist in the Port Phillip Bay literature. For a long-term monitoring program, the design (particularly time and space scales) should be determined only after this initial set of field studies.

5.2 **Distribution, abundance and time series of biota**

This section provides a brief description of spatial and temporal patterns of abundance of biota within Port Phillip Bay. Where possible it assesses the suitability of data on patterns and processes to meet the objectives of the present Study Design. More detailed information is available from the specialist reviews.

5.2.1 **Phytoplankton**

Most groups of coastal and oceanic phytoplankton are represented in Port Phillip Bay, though the dominant group is usually diatoms, followed by cryptophytes and dinoflagellates.

The general trend of succession in Australian coastal phytoplankton communities is for a spring bloom of diatoms to be followed in summer by coccolithophorids, microflagellates and dinoflagellates. Late summer/autumn often sees a second diatom bloom and occasionally short increases in cryptomonads and silicoflagellates before the population settles to a winter minimum (Jeffrey and Hallegraeff, 1990). However, published reports concerning the overall spatial and temporal distribution of phytoplankton in Port Phillip Bay are almost nonexistent, though limited data is available from 1968-1971.

From what is known, spatial and temporal phytoplankton distributions in the Bay appear to follow complex patterns. These patterns are related to factors such as water temperature, light availability, nutrient availability and possibly predation pressure. Some of these factors follow seasonal patterns, while others fluctuate more randomly (though patterns of input of waste into Port Phillip Bay from WTC are highly controlled in occurrence). General patterns of phytoplankton distribution can be surmized from chlorophyll *a* data, though problems in interpretation exist because species cannot be distinguished and the linkage to phytoplankton abundance is uncertain.

Research into remote sensing for chlorophyll *a* is continuing and, if the problems associated with this can be reduced, more complete chlorophyll *a* distribution descriptions (spatial and temporal) in Port Phillip Bay will be possible. A recent evaluation of mapping of chlorophyll *a* by remote sensing technology (Howard and MacDonald, 1990) suggests that while there are still problems to be overcome, the method has great potential to detect phytoplankton blooms at an early stage and to assess changes in bloom distribution and abundance over time and space.

The occurrence of phytoplankton blooms, in particular those of toxic species, are of particular interest in assessing changes in the Bay with respect to nutrients and toxicants. Blooms are a natural occurrence, though the increase in nutrient levels as a result of human activity may have increased the frequency, intensity and length of these events (Hallegraeff and Bolch, 1991). In Port Phillip Bay the surveillance of phytoplankton populations is important as an 'early warning' system for algal blooms, particularly with respect to toxic species (Arnott, 1990a). Blooms of non-toxic phytoplankton are apparently not a problem in Port Phillip Bay, and records of their occurrences (if any) are not readily available. Massive fish kills have, however, been associated with non-toxic microalgal blooms in various marine systems (Bulthuis and Cowdell, 1982), mainly as a result of depleted oxygen in marine waters (anoxia). No reports of fish kills in Port Phillip Bay have been associated with anoxia caused by phytoplankton blooms.

Overall there is a general paucity of knowledge regarding the ecology of phytoplankton in Port Phillip Bay. The distribution in space and time is patchy and biomass levels are generally low compared with Australian coastal waters. Only recently have phytoplankton community monitoring programs been used as a tool to assess phytoplankton in Port Phillip Bay, and even these programs are not complete in their coverage of general Bay areas, nor do they include the smaller but often extremely important microalgae.

Present monitoring of phytoplankton blooms appears to be restricted to toxic-species. Data regarding the occurrence of all blooms in Port Phillip Bay is important, because knowledge of the processes underlying bloom occurrence can be gained, allowing the more accurate future prediction of both toxic and non-toxic blooms.

5.2.2 Seagrass and benthic algae

Port Phillip Bay has eight recorded species of seagrasses and 95% are found in water < 5m deep. The distribution of seagrasses in the Bay was mapped from field studies in 1957-1963, and changes in cover over time were estimated from historic aerial photos by Bulthuis in 1981. At a number of locations (e.g. Swan Bay, Bellarine Peninsula, Corio Bay, Werribee, Altona Bay) more detailed field and aerial photography studies have been conducted of varying quality and vigour, usually covering small areas or in response to specific issues. A significant problem in using aerial photography to map seagrass meadows is that it is often difficult to locate the position and edge of a meadow with accuracy and precision. This can make accurate detection of changes over time of seagrass distribution very difficult.

Data on biomass and species composition of seagrasses in Port Phillip Bay are limited mainly to Swan Bay with several limited data sets from the Bellarine Peninsula and the Werribee regions. These data indicate that seagrass biomass changes with season and species. There are only limited data on seagrass interactions with nutrients, and these involve few sites and have been collected across different temporal and spatial scales.

There is no comprehensive floristic analysis of benthic algae - macro or microalgae - in Port Phillip Bay. With few exceptions studies have been limited in scope, were localised and had markedly different objectives. Past studies have been mainly based on single static sampling events or on a short series of sampling events covering varying time-spans. Most of the data on the benthic vegetation of the Bay are qualitative. This is particularly true with respect to macroalgae and non-planktonic microalgae. In addition many of the study designs appear to lack the rigour required to generate reliable and accurate data so that interpretations arising from them must be treated with caution. According to the specialist reviewers (Light and Woelkerling, 1992) it is virtually impossible to combine the extant data sets into a single meaningful and useful database which will adequately address the required information on temporal and spatial variation of benthic algae within Port Phillip Bay.

Seasonal variation in species composition and abundance of benthic macroalgae appears to occur at all Port Phillip Bay localities although there are no clear patterns evident in the available data. This is most likely because most of the studies were carried out for one year or less. Although there is evidence both nationally and internationally of a negative correlation of epiphyte biomass and seagrass abundance (e.g. Larkum, 1976; Cambridge, 1979; Sand-Jensen, 1977; Orth and van Montfrans, 1984) there are virtually no data on this phenomena in Port Phillip Bay with the exception of Bulthuis and Woelkerling (1983). There also appears to be a lack of quantitative data regarding the noticeable absence of large brown algae near effluent outfalls, and the reason for the absence of these algae is uncertain.

The effect of environmental factors on macroalgae has been most comprehensively studied in the Werribee region of the Bay. The main factors there appear to be wind, wave, current, sediment type and, possibly, proximity to urban drains. Information on seasonal changes and productivity in the macroalgae is restricted to a range of localised studies of varying quality.

There is little reliable information about the distribution of microalgae in the Bay, although, as for macroalgae, the areas near Werribee and Hobsons Bay have been subjected to some more detailed studies. However, a number of assumptions and approximations were made in arriving at stated productivity values and the interpretation of the data requires care.

5.2.3 **Benthic invertebrates**

Bay-wide and local surveys of the benthic invertebrates in Port Phillip Bay have revealed that Port Phillip Bay appears to support a benthos exceptionally rich in species compared to similar ecosystems in other parts of the world. This may be due, in part, to the intensity of study of the benthos within the Bay compared to other areas. These studies have indicated that small-scale spatial and temporal variability in community indices and species densities is also high.

The diversity of benthic communities is primarily correlated with the nature of the sediment of the Bay (Poore *et al.*, 1975; Poore and Rainer 1979). They range 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 and suspension feeding molluscs may process a significant fraction of primary planktonic production.

On a local scale, the fauna of the Yarra River is more depressed in terms of numbers of species than might be expected from sedimentary properties alone. The fauna of Corio Bay and the Geelong Arm seems to have changed over the period of 20 years for which data are available, with suspected lower densities of macrobenthos and the spread of at least two introduced species of molluscs and one polychaete species. There is no clear indication from the only quantitative Bay-wide survey of an effect on benthic community structure of the effluent from the WTC except for within about 300 m from the drains (Poore and Kudenov 1978). In the immediate vicinity of the WTC drains epifloral microalgae process nutrients and probably contribute directly to deposit feeders which dominate there.

5.2.4 **Finfish and shellfish**

Port Phillip Bay is the most productive bay and inlet commercial fishery in Victoria with over 60 species of finfish being recorded from commercial catches (Hall and MacDonald, 1986). In 1989-90, 1900 tonnes of finfish with a wholesale market value of about $5 million was obtained from the Bay (NREC, 1991). However, production is low in comparison with other Australian locations such as Sydney Estuary, Jervis Bay, Botany Bay, Tuggerah Lakes and Cockburn Sound (Gwyther, 1990). When the shellfish contribution to total production of the Bay is considered, however, Port Phillip Bay compares far better with other locations in terms of fishery production. This could be considered supportive of the view that scallop, mussel and oyster growth rates within Port Phillip Bay are related to the levels of anthropogenic nutrient inputs into the Bay (Gwyther, 1990). Studies over a 20 year period show evidence of an increase in growth rate, possibly related to density-dependent effects resulting from 20 years of commercial fishing, as well as increased nutrient input from the catchment area of the Bay (Gwyther and McShane, 1988).

Major commercial species include flathead, snapper, King George whiting, southern anchovy, pilchard, calamari, mussels, scallops and abalone. Pilchards in some years contribute more than 50% of the commercial catch. Some of these species are also taken by recreational anglers.

Presently there are no biomass estimates of any commercial species in the Bay except scallops, although commercial catch data for some species exists from 1914. A computerised database of catch and effort from 1978 to the present is held by Marine Science Laboratories (MSL). Collection of data from recreational fisheries began in 1990 although a 12 month survey of recreational fishing was undertaken in 1982-83.

The following brief description of the distribution and abundance of finfish and shellfish species is based largely on the commercial and recreational catch statistics.

Flathead: A number of flathead species are found in Port Phillip Bay. Of major commercial and recreational importance are the sand flathead (*Platycephalus bassensis*), long-nose or yank flathead (*Platycephalus speculator*) and rock flathead (*Platycephalus laevigatus*) (MacDonald and Hall, 1987). Sand flathead are a demersal, non-migratory finfish (Dix *et al.*, 1975). Yank flathead live mainly on shallow sandy patches between seagrass beds while rock flathead live in shallow water habitats on reefs or among seagrasses and algae (Last *et al.*, 1983).

The greatest commercial catches of flathead (all species) are taken during the late autumn and winter. This appears to be a reflection of regulations prohibiting the use of mesh nets extending from late spring to early autumn. Sand flathead are taken in all areas of Port Phillip Bay, with most being taken in nearshore, shallow waters over little or no vegetation (Hall and MacDonald, 1986). Over 60% of the rock flathead and yank flathead are taken from the Geelong Arm (Hall and MacDonald, 1986).

Data from the 12 month recreational fishing survey beginning from October 1982 indicates that the greatest catch rate for all species of flathead by recreational anglers occurred during December, January and June. The better catch rates for flathead were obtained in the eastern and southern regions of the Bay (MacDonald and Hall, 1987).

Snapper: Snapper are an important commercial species in Port Phillip Bay and also actively sought fish by recreational anglers. Snapper congregate in the vicinity of broken reef or seagrass (Winstanley, 1983). Data from the 12 month recreational fishing survey in 1982-1983 indicates that the months in which the greatest angling success rates for recreational anglers fishing for snapper occurred were February, March and September. The bulk of the snapper caught were taken from the southern part of the Bay (MacDonald and Hall, 1987).

King George whiting: King George whiting is the largest species of whiting found in Australian waters. In a study based in Westernport Bay juvenile whiting were found to inhabit heavily grassed patches of seagrass. As the fish grow older they inhabit less dense grassy areas (Robertson, 1977).

The greatest commercial catches of King George whiting in Port Phillip Bay are taken during late autumn, although they can be caught all year. The lowest catches were obtained during late winter and early spring. Almost 70% of the annual whiting catch is taken from channels, 25% from other deep water habitats and 6% from shallow water habitats. It is likely that a larger portion of the catch is taken from more shallow areas, and this anomaly is blamed on misinterpretation by fishermen when filling out catch returns. In addition, seine nets set in deep water are hauled into more shallow waters for final sorting of the catch (Hall and MacDonald, 1986).

Data from the 12 month recreational fishing survey in 1982-1983 indicated that the best catch rate by recreational anglers fishing for King George whiting in the Bay occurred in October. Other months during which relatively good catch rates were obtained were from November to June (MacDonald and Hall, 1987).

Pilchard and southern anchovy: Pilchards and anchovy have historically been fished in the Bay as both baitfish and for human consumption. Pilchards are most abundant in Port Phillip Bay from August to May.

They are taken throughout the year, with the largest proportion being taken during autumn. About 70% of the pilchard catch comes from the northernmost areas of the Bay, with 17% from the Geelong Arm and 10% from the east of the Bay. Anchovy are taken all year and throughout the Bay. The largest proportion is taken in autumn along the western shoreline and where freshwater flows into the northern parts of the Bay (Hall & MacDonald, 1986). Younger anchovies prefer inlets and estuarine environments, with three year old fish moving out to sea.

Scallops: The scallop (*Pecten alba*) is very abundant and supports the major fishery in Port Phillip Bay. *P. alba* is a filter feeder and may well undergo altered growth rates according to the species composition of the algal food available (Bayne, 1976). They are found on sandy substrates into which they work themselves, 'blowing' out sand to create a suitable depression (Yonge and Thompson, 1976). The January 1992 dive survey estimated the scallop population in Port Phillip Bay to be about 2.6 billion (Gorfine *et al.*, 1992). There are annual population estimates of scallop numbers in the Bay extending back to 1982.

The scallop fishing season in Port Phillip Bay lies within the period April to December, although the exact time of the opening and closure of the season is dependant on scallop size, condition and catch rate. The optimum sites are locations on the east and west sides of the Bay in water deeper than 10 m (Hall and MacDonald, 1986; Coleman and Saddlier, 1989).

Blue mussel: The blue mussel is a filter feeder and is extremely common in Port Phillip Bay (Macpherson and Gabriel, 1962). Mussels are generally found in littoral and shallow sublittoral waters, although they have been recorded from deeper water. The species lives on a variety of substrata, such as rock, stones, shingle, dead shells, compacted mud and sand, pier piling's and harbour walls (Seed, 1976).

Mussel harvesting from mariculture facilities commences in July-August and continues through to February (Arnott *et al.*, 1990). The mariculture areas for mussels in the Bay are Portarlington, Clifton Springs, Beaumaris and Dromana Bay (Arnott, 1990b).

Abalone: The abalone is a gastropod mollusc which can grow up to 14 cm long and may live for more than 10 years (McShane and Smith, 1986; McShane, 1989).

Blacklip abalone occur in reef environments, preferring caves, crevices or vertical faces at depths from 1-2 m to 10 m and, rarely, to 25 m (Shepherd, 1973). Sticks Reef in Port Phillip Bay is a shallow reef (4 m) and has flourishing macro-algal communities. It is an example of the typical reef environment inhabited by abalone in the Bay (McShane and Smith, 1986). Approximately 8 km^2 of low relief reef supports the abalone population in Port Phillip Bay (McShane *et al.*, 1986).

Abalone are harvested throughout the year. The three major reefs in Port Phillip Bay that provide the bulk of the abalone catch are located on the western side of the Bay between Point Cook and Kirks Point, bounding the WTC (McShane *et al.*, 1986).

Southern calamari: In Port Phillip Bay southern calamari inhabit vegetated and reef areas (Hall and MacDonald, 1986). Although calamari are taken commercially all year, highest catches are obtained during September and October. More than 50% of the annual calamari catch is taken from the southern shores of the Geelong Arm and 32% is taken near Port Phillip Heads (Hall and MacDonald, 1986).

Data from the 12 month recreational fishing survey in 1982-1983 indicated that the greatest catch rate by anglers of calamari was from June to August from both the Geelong Arm and near Port Phillip Bay heads (MacDonald and Hall, 1987).

5.2.5 Water column trophodynamics

Trophic linkages in the Bay appear to be very complex, but there is very limited available data on the pathways and fluxes of biomass in the Port Phillip Bay water column biota. Although there have been some studies of zooplankton and phytoplankton species abundance, distribution and biomass (e.g. Jenkins 1986,1987,1988; Fancett 1988a,b), assessment of changes of these parameters in time and space is difficult because of the highly variable nature of the populations and different sampling strategies (including sampling mesh sizes) used in different studies.

Trophic linkages have been studied for scyphomedusae, larval fish and cladocerans, but only over limited time periods (Fancett 1988a,b; Fancett and Jenkins 1988). The scyphomedusae had little impact on zooplankton, but may have had an impact on fish eggs and larvae in localised areas. Pilchards and anchovies may have an impact on copepod populations in summer when juvenile anchovies and pilchards are most abundant.

The most important trophic pathway for phytoplankton production in the pelagic ecosystem is probably *via* zooplankton to pelagic clupeoids (Fancett, 1988a). The pelagic uptake of carbon and nutrients may be highly important in Port Phillip Bay if commercial fish catches are taken as an indication. Recent catches of clupeoids (pilchards and anchovies) have ranged from 700 to 1400 tonnes compared to 500 to 600 tonnes for demersal fish (Crawford *et al.*, 1992). The actual population may be much greater because the clupeoid fishery is believed to be largely under exploited (although the anchovy catch has shown a dramatic recent decline [Nicholson, 1992]) whilst the demersal fishery appears to be fully or over exploited. Based on general calculations, Crawford *et al.*, (1992) estimated that if a total population of 5000 tonnes of clupeoids is assumed with an average weight of 10 g, and the clearance rate of zooplankton for a 10 g fish is taken to be 1 litre min^{-1} (James and Findlay, 1989), then constant feeding would see the volume of Port Phillip Bay (2.51×10^{10} m^3) cleared of zooplankton each 10 days. It appears possible therefore that large populations of clupeoids in Port Phillip Bay may graze most zooplankton production and only when zooplankton production increases sufficiently to outstrip this heavy grazing pressure will greater zooplankton biomass occur.

Kimmerer and McKinnon (1989) found that the dominant calanoid in Port Phillip Bay, *Paracalanus indicus,* had a higher clearance rate of the small flagellate *Tetraselmis chui* (*ca*. 10 ml copepod d^{-1}) than did *Acartia tranteri* (*ca*. 2 ml copepod d^{-1}). If the grazing rate of *P. indicus* is related to total holoplankton (> 150 m) densities of 4000 individuals m^{-3} (Fancett 1988a), then the animals present in $1m^3$ would clear ≈ 40 litres per day, giving an approximate clearance time for the volume of the Bay of 25 d. Clearance rates by *P. indicus* for larger diatoms and dinoflagellates which may bloom in Port Phillip Bay would probably be much higher due to more efficient handling. To date there are no estimates of the grazing impact of zooplankton on larger phytoplankton or the impact of planktivorous clupeoids on phytoplankton and zooplankton in the laboratory, let alone any field based estimates for Port Phillip Bay.

5.2.6 Higher trophic levels

Higher trophic levels within Port Phillip Bay consist of marine mammals and marine/aquatic birds.

Little quantitative information on marine mammals (seals, whales and dolphins) is available for Port Phillip Bay, and that which does exist is extremely patchy (Warneke, Victorian Department of Conservation and Natural Resources, personal communication). Of all of the mammals that inhabit the Bay the bottle-nosed dolphin is probably the most numerous and well known. It is thought that there may be a resident population within the Bay. Tissue samples are taken from carcasses of this species on an opportunistic basis. Sightings of many of the other marine mammals within the Bay suggest that their occurrence may be rare and they would be difficult to study in a quantitative manner.

There appears to be some observational information on the diet of water rats within Port Phillip Bay. Although there is a potential for these mammals to act as ecological indicators of change in response to toxicants, their complex behaviour and our lack of knowledge of their ecology suggests that other organisms already identified in the Study Design may be better candidates.

Port Phillip Bay supports large populations of both resident and migratory bird species and it is also an important site for wading species. Temporal surveys on selected species are carried out as part of the activities of ornithological societies. Although studies of the trophic pathways and detailed dietary analyses are virtually non existent, observational studies suggest that some species eat small intertidal and salt marsh invertebrate fauna whilst others utilise small fish in the Bay such as the anchovies and pilchards.

5.3 Interactions of nutrients and toxicants with biota: Knowledge gaps and research needs

This section assesses information from the specialist reviews in the light of the Study Objectives, Design and Tasks. In cases where significant omissions in task activities have been identified, recommendations are made to update the study design to accommodate those gaps.

5.3.1 Phytoplankton

The complex spatial and temporal patterns in the phytoplankton populations in Port Phillip Bay and the lack of historical coordinated data collection indicates that additional intensive and coordinated monitoring is needed to establish correlations with environmental factors. This is partly addressed by existing MSL work, and this program could be expanded to meet the Study Objectives (Tasks N2.1 and N2.2). Given the cost of intensive monitoring of phytoplankton populations, it is important that new cost-effective remote sensing technologies be developed (Task G2.1). This would involve coupling *in situ* monitoring, including both routine and event-driven sampling, in Tasks N2.1 and N2.2, to remotely sensed data. A report produced in 1990 evaluating remote sensing to assist in monitoring chlorophyll *a* in Port Phillip Bay will provide valuable information on this subject (Howard and MacDonald 1990).

In pilot sampling of plankton in the Bay the relative contribution of nanoplankton and picoplankton should be addressed in Task N2.2 to obtain their current status. Such findings will need to be taken into account when designing sampling regimes. Studies of the frequency of toxic and non toxic bloom species will be addressed through Task N2.2.

5.3.2 Seagrasses and benthic algae

The updated mapping of seagrasses and other benthic flora using modern navigational aids, being reviewed in Task G2, will provide information on time series changes in the Bay when validated against ground-truthed maps from previous surveys. Information on seagrass distribution is likely to be the most valuable indicator for monitoring of changes, given the historic database of aerial photography. The apparent negative correlation of the presence of large brown algae with proximity to outfalls may also provide an indicator for monitoring the effect of increased nutrients and toxicants in the future into the Bay. Their distribution will be mapped in Task G2 and estimates of biomass, abundance and productivity with distance from the outfalls may be possible through tasks N2.3/4.

The use of seagrass epiphytes as an indicator of nutrient load will be addressed through task N2.3/4 although the corresponding role of herbivores in modifying any direct effects of an increase in nutrient load on benthic flora has not been directly addressed in Study Design. The investigation of grazing rates by herbivores on seagrass epiphytes may be necessary if the empirical linkage between epiphytes and nutrients fails to be an adequate predictor of epiphyte abundance.

5.3.3 Benthic invertebrates

The comprehensive survey of benthic invertebrates between 1968-1973 established a baseline for distribution and abundance of many species in the Bay. Changes in this fauna over the intervening period will be established in Task E5, and when coupled with interannual variability estimated from both the previous and the new data, will permit a first approximation of the nature of ecological change in the Bay. This will also establish the basis for continued quantitative monitoring of the Bay in the future.

Knowledge of nutrient pathways involving the benthic fauna is based on overseas information. Some pathways have been suggested by the specialist reviews but few have been observed and no rates have been measured. Benthic molluscs contribute a large fraction to the biomass of the Bay-wide fauna and suspension feeding molluscs may process a significant fraction of primary planktonic production. Although actual rates have not been measured, estimates using literature values for long-term filtering rates indicate a filtering time for the volume of Port Phillip Bay of 24 days. The possible very major role for filter-feeding molluscs in the Bay suggests that the benthic invertebrate studies should initially include a review of the national and international literature on filter feeding rates of molluscs, extended to a small experimental study if necessary.

5.3.4 **Finfish and shellfish**

Fishing-independent estimates of demersal fish populations and their change over time will be made in Task E8. An increased understanding of trophic link will also be attempted in this task.

The major role of small pelagic species in the Bay (pilchards and anchovies constitute about 50% of the total catch since 1981) indicates that additional effort needs to be devoted to develop fishing-independent estimates of population sizes, and their spatial and temporal distribution. Although this research falls under the responsibility of the MSL and outside the direct brief of the Port Phillip Bay Environmental Study, close collaboration and liaison with MSL will assist in clarifying trends in space and time of pelagic species.

Annual scallop surveys conducted by MSL provide the only biomass estimates of commercial scallops in the Bay. These estimates indicate wide fluctuations in population size as a result of highly variable recruitment, a well known feature of scallop populations. This extreme variability probably precludes correlations with nutrient or toxicant effects.

5.3.5 **Water column trophodynamics**

The apparently major role played by zooplankton and planktivorous fish in grazing phytoplankton populations indicates that better information is needed on their respective grazing rates, and their spatial and temporal basis. A review of the national and international literature would permit an analysis of whether additional research on this topic is warranted.

The apparently major role of clupeoid fish also indicates that, in Task T8, representative species of clupeoids should be included in the surveys of body burdens of toxicants. Monitoring of toxicant burdens in this group of fish would estimate changes in response for higher order predators (such as some birds and mammals) which rely heavily on anchovies and pilchards for food.

5.3.6 **Higher trophic levels**

There are significant problems (in terms of logistics and costs) in mounting comprehensive surveys of the effects of nutrient and toxicant loads on marine mammal and seabird populations of Port Phillip Bay. As an indicator of change or health of the Bay, it would be more effective to concentrate on populations lower down the food web that turn over rapidly and show faster response times. There are also difficulties in attributing changes or fluctuations in populations of birds in response to nutrients and toxicants unless the birds are permanent residents of the Bay (i.e. changes in abundance, toxicant content etc. may be in response to conditions along migratory paths or at end destinations).

A small program monitoring the levels of toxicants in resident species may provide a baseline of information regarding any time series in toxicant accumulation. This may be an option if samples can be taken from live animals, or at worst from fresh carcasses before internal chemistry changes.

The birds/mammals workshop on 9th September 1992 also identified the issue of the nutrients from WTC providing enhanced productivity in adjacent flora and invertebrate fauna leading to enhanced food supply to bird populations. The Port Phillip Bay Environmental Study will be investigating the link between nutrient levels and productivity of primary producers in these areas (Tasks N2.3/4). Although large populations of waders are known to feed in the vicinity of the WTC, there appears to be no evidence that bird populations at WTC are food-limited or that any changes in bird abundance are linked to nutrient output from the Werribee site. If this issue is viewed by managers to be significant it may be possible to co-locate some benthic sampling stations for algal turfs (Task N2.4) and invertebrates (Task E5) to monitor for changes in the abundance of biota in response to changes in nutrient levels and relate these changes to results from sampling of resident bird populations carried out by ornithological societies. However it should be remembered that in terms of the ecological work within the Study Design, research on trophic flows falls outside the brief of the Study which, due to logistical and resourcing constraints, only considers trophic links. While outside the brief of the study, the current research on birds and marine mammals should be encouraged to continue in addition to continuing liaison with the Port Phillip Bay Study.

5.3.7 General conclusions

There are major gaps in the present understanding of the ecology of Port Phillip Bay and of how the presence of toxicants and nutrients interacts with the ecology of the Bay. The broad biodiversity and large number of ecological processes involved in Port Phillip Bay precludes the identification of any small set of critical processes which could be studied in detail for predictions of the effect of changes in nutrient and toxicant inputs to the Bay.

Whilst there is existing information on commercial and recreational catches of seafood from the Bay, this information is similarly difficult to link to the inputs of nutrients and toxicants because of a lack of understanding of the processes of the valued species and their link to nutrients and toxicants. In this situation the only viable and practical option to manage the input of nutrients and toxicants more effectively is to construct a set of hypotheses about their impacts which can be tested in appropriate field and laboratory studies.

The most realistic way to develop these hypotheses is by examination of existing patterns in Port Phillip Bay seeking correlations between patterns in time and space and biological changes. These correlations can then form the basis for theoretical models about the impacts of nutrients and toxicants from which can flow testable hypotheses. As our understanding of the likely impact of nutrients and toxicants develops, selected areas of knowledge of processes may need to be studied in more detail to improve our existing knowledge of the likely impacts of nutrients and toxicants. Given the large number of interlinked processes operating within Port Phillip Bay, it is important to ensure that any process study is closely aligned with the Study Objectives.

5.4 Monitoring of biota

Long term monitoring is aimed at detecting a consistent change in particular parameters or specific indicator, against the natural variability which is characteristic of biological systems (Quinn and Keough 1991). For monitoring to be effective it is crucial that the selected indicators are known to respond directly or in some way be linked to changing levels of nutrient and/or toxicant inputs. Concomitant with the requirements of a management tool, long-term monitoring needs to sit within an hypothesis testing framework and the ability to test hypotheses implies the need for objective decision making. The reliability of these tests is based on established statistical procedures and therefore the need for a sound sampling design. These requirements also mean that indicators must be logistically easy to monitor and that there is adequate statistical power with the sampling programs to detect the required level of change (Ward and Jacoby 1992).

While it is important to establish baseline information from which to measure change in the health of Port Phillip Bay, it is important to recognise that monitoring techniques may develop and improve over time. It is therefore necessary to ensure that some level of standardisation/comparison of different methodologies occurs to ensure that time series data are comparable and that related studies in Port Phillip Bay are coordinated. The specialist reviews have highlighted many examples where due to differing methodologies (e.g. mesh/sieve size) data that were expensive to collect, while valuable in their own right, are virtually useless when employed to make comparisons between studies of different areas or times.

Finally it is important to recognise that while the need for standardisation and continuity in long-term monitoring is paramount, monitoring is also a dynamic process. Data are used to develop models about the nature of impacts and rates and directions of change. Any changes of management in the Bay need to become part of the model building process. The hypotheses and models created are only as good as the correlations that have been determined from existing patterns and these correlations and patterns are dependent upon the quality of data gathered from the long-term monitoring program.

5.5 Summary of Knowledge Gaps and Research Needs

Task E1: Phytoplankton

This Task which falls within Nutrient Tasks N2.1, 2.2. is of a high priority and should be commenced as soon as possible. Apart from investigating novel ways to monitor algal blooms (through Task G2), sampling should include both toxic and non toxic bloom species. The role of nanoplankton and picoplankton in Port Phillip Bay should be evaluated particularly in respect of their role in primary productivity and influence on chlorophyll *a* measures.

New task: The apparently major role played by zooplankton and planktivorous fish in grazing phytoplankton populations indicates that better information is needed on their respective grazing rates, and their spatial and temporal basis. A new task needs to be developed to review grazing rates of the dominant species.

Task E2: Light attenuation

This Task is high priority and should commenced as soon as possible. This Task falls within the Physical Task P7.

Task E3: Macroalgae and seagrasses

This Task is of high priority and should be commenced as soon as possible. Mapping of macroalgal beds and seagrass meadows will be undertaken through General Task G2, while measurements of productivity and nutrient fluxes will be addressed through Nutrient Tasks N2.3, 2.4.

Task E4: Benthic microalgae

This Task, which falls within Nutrient Tasks N2.3, 2.4, is of high priority and should be commenced as soon as possible.

Task E5: Benthic invertebrates

This Task is of medium priority and should be commenced in several stages. The first activity is to analyse and review existing data using modern multivariate (classification and ordination) and parametric statistical tools. Emphasis should be on determining the experimental design required for detecting change over space and time. The findings of this task will be used to design the subsequent monitoring program as discussed in the original Project Design. The review should be done by, or in collaboration with, Dr. G. Poore of the Victorian Museum.

New Task: The possible very major role for filter-feeding molluscs in the Bay suggests that the benthic invertebrate studies should initially include a review of the national and international literature on filter feeding and excretion rates, and assimilation efficiencies of filter feeding molluscs. This task should commence immediately. A small experimental study addressing issues specific to the modelling needs in General Task G8/9 may be necessary depending upon the findings of the literature review.

Task E6: Valued molluscs

This Task should be delayed pending the results of General Task G2 (habitat mapping).

Task E7: Reef invertebrates

This Task should be delayed pending the results of General Task G2 (habitat mapping).

Task E8: Fish productivity

Although this work is considered a medium priority, the funding of an additional 4 seasonal surveys to assist MSL to determine the distribution and abundance of demersal fish, independent of commercial fishing, can be carried out to fit in with MSL's research schedule.

Task E9: Toxic algal blooms

This Task, which falls within Nutrient Tasks N2.1, 2.2, is of high priority and should be commenced as soon as possible.

Task E10: Other biota and communities

This Task can be deleted.

Additional New E Tasks

As the modelling G8/9 task proceeds, a number of data requirements are expected to arise. These can not be predicted at this time, but would be needed to calibrate the model predictions and to test the robustness of possible optional management strategies. Most of these tasks are likely to be small, but can not be estimated at this time.

5.6 Acknowledgements

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Chapter 6. Integrated and management modelling

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6.1 Introduction

The development of integrated models within the Port Phillip Bay Environmental Study has been assigned to a single task, and integrated models are just one of a number of outputs of the study. However, because of the importance of models for integrating and interpreting field studies, and as management tools, modelling is represented specifically on the Technical Group overseeing the Study. Detailed planning of the integrated model task has started later than the other Study components because integrated modelling was not represented on the initial Study design team. A modelling workshop in November 1992 provided a first opportunity to address modelling objectives and conceptual design in detail. This chapter was prepared as a background document for that workshop, setting out some general principles for model development within the context of existing knowledge of Port Phillip Bay. A report of the workshop and its key outcomes is attached (Appendix 1).

6.2 General principles

6.2.1 Why model?

It may seem superfluous in the 1990s to begin with a justification of modelling, but it is true both that environmental modelling has an uneven record, and that there is still both misunderstanding and mistrust of modelling on the part of some scientists and managers. This has arisen in at least some cases from overambitious attempts at modelling, and/or a lack of appreciation of the strengths and limitations of environmental modelling. A very thoughtful and readable review of the state of the art in environmental coastal modelling can be found in GESAMP (1991). That review summarizes the scientific contributions models can make to environmental studies as: "to organize information and data for the system; investigate linkages and further understanding of interactions among ecosystem components; interpret and understand field observations; provide a means for comparison with other systems; and identify areas for further research". Other recent reviews and case histories can be found in Michaelis (1990).

The review goes on to make the critical point that when dealing with complex environmental systems it is a mistake to attempt to develop an all-purpose model that captures all aspects of the system. All models are caricatures of the real system, and they must be tailored to address specific questions. In applied projects such as the Port Phillip Bay Environmental Study, this means that models must be tailored to address specific management questions and issues. The attraction of models to managers is that they enable us to extrapolate from data and understanding derived from current and historical research to address questions about the likely outcome of future management actions and strategies.

The development of mathematical models forces us to be specific, both about the questions we ask, and the understanding and knowledge of the system on which answers are based. This can be both a blessing and a curse. For example, we cannot expect to build a model to address the "health of Port Phillip Bay". We will need to be much more specific about the properties and issues that define the health of the Bay. But this is arguably desirable in any case. The Study will continue to discuss and refine management issues as part of ongoing interaction with the Agency User Group.

6.2.2 **The scope of the Model.**

Model development needs to start from a set of management issues, preferably honed down to a set of variables of direct interest, and a set of potential management actions. There follows a set of choices which together determine the conceptual design. These choices decide:

a) the spatial and temporal scope of the model;
b) the spatial and temporal resolution of the model;
c) the key internal variables and processes;
d) the prescription of forcing variables;
e) the relative importance of process-based versus empirical submodels;
f) the extent to which the model is tuned or fitted to data;
g) whether a stochastic or deterministic approach is adopted.

These choices are made on the basis of the management questions asked, the current state of knowledge and understanding of the system, the availability of historical or new data, and the resources available for modelling. However, the general caution based on modelling experience is that models should be only as complex as is needed to address the question of concern. In particular, development of elaborate models with many processes in an attempt to mimic the complexity of the real system is generally counterproductive because the added processes are often poorly understood and/or poorly parameterized.

6.2.3 **Model evaluation.**

Evaluating a model's performance is not a simple matter. The GESAMP report distinguishes three stages. Verification involves checking the model for gross errors, either in the mathematical formulation or more likely in the numerical implementation. This may require consistency checks (e.g. mass conservation), or application of the model to artificial situations where its behaviour can be independently predicted. Calibration involves the comparison of the model predictions with observations. This may involve tuning of the model parameters, formally or informally, or choices between process formulations. In the case of empirical or inverse models, most of the information content in the model may be derived from the parameter estimation process. The third step is validation, which preferably involves testing of the calibrated model against an independent data set.

Validation raises another issue of critical importance to management applications of models: the transportability or robustness of models. A model is transportable to the extent that it can be applied to a different system (e.g. another bay) with a minimum of tuning of parameters and process assumptions. Similarly, a model is robust to the extent that it maintains its performance under conditions of forcing outside those for which it has been calibrated. While some management questions concern current or historical circumstances (e.g. was one source of toxicants more important than another in producing an observed outcome), most management questions are of the "what if" variety, and frequently pose questions outside the range of historical experience. We can only have confidence in answering these questions to the extent that we believe our models are robust; that is, based on assumptions which are unlikely to change under a change in forcing.

Another related topic is the issue of uncertainty in model predictions. Uncertainty in predictions can arise from the unpredictability of forcing variables such as climate and weather, ignorance about the appropriate form and values for model processes and parameters, and in some models from an intrinsic non-linear instability which leads to sensitivity to initial and boundary conditions such that variables are essentially unpredictable in a deterministic sense (i.e. chaos). The general advice on uncertainty is that it is not necessarily intrinsically bad (there are well established theories for decision making under uncertainty), but that it should be treated explicitly, and quantified as far as possible, by consideration of model formulation, by model experimentation and sensitivity analysis, and by statistical analysis of predictions and observations where appropriate.

6.3 **Integrated models for Port Phillip Bay.**

The large scale of the Port Phillip Bay Environmental Study is reflected in the diverse set of management issues, the scope of the historical data set, and the extensive field program planned. At the same time, as will become clear below, our understanding of processes is very uneven. Because there are more resources available, the modelling program can be more ambitious than it would be for a smaller more focused study. Following the principles outlined above, the diverse set of management issues raised will almost certainly require the development of a range of models, not just one model. At the same time, it must be recognised at the outset that modelling is not a panacea, and will only be able to address a subset of the management issues. Some indication of this subset is given below; its further definition will be a key goal of the November workshop and subsequent activity.

6.3.1 **Management issues**

The Study Design has of course been based on extensive consultation with management agencies and user groups, and the resulting identification of a set of key issues. As is clear from the Design, these issues revolve around the inputs of nutrients and toxicants into the Bay, their distribution and fate within the Bay, and their impacts on the Bay ecosystem. The set of nutrients and toxicants of concern is identified elsewhere in this report. As noted in the previous section, there is a need for modelling purposes to define management issues as clearly as possible. One way to do this is to try to identify a set of specific indicator variables of direct concern to management. This set then defines the "health" of the Bay.

For Port Phillip Bay, one can break this set of indicator variables into two classes. One consists of variables related to water quality: concentrations of nutrients, chlorophyll, dissolved oxygen, specific toxicants, pathogens and turbidity. One could extend this to include concentrations in sediments where appropriate. The other set includes ecosystem indicators such as toxic algal blooms, fish yields and fish kills, levels of toxicants in seafoods, abundance and physiological health of mammals and birds, distribution and state (density, productivity, diversity) of benthic communities such as seagrasses, macroalgae and invertebrate fauna, presence of rotting weed on beaches.

Note that the second set is generally of direct concern to user groups, whereas the first set, while it includes variables in this category, is of concern to managers because these variables are seen as mediating the impact on indicators in the second set. Thus for example, high nutrient concentrations are unlikely to be directly measured and noted by users: it is their potential results such as algal blooms, anoxia, fish kills, and rotting weed on beaches which are of direct concern. This distinction corresponds roughly to the distinction between Environmental Quality Standards and Environmental Quality Objectives (Orr *et al.*, 1990).

The philosophy of the Study Design also reflects this distinction. The Study has generally separated the problem of measuring and predicting water and sediment quality (nutrient and toxicant fate and distribution) from the measurement and prediction of ecosystem impacts. This separation also reflects the degree of process understanding and predictive capability associated with these components. Thus the approach to physics can be strongly process and theory based, the approach to nutrients and toxicants less so, requiring more empirical input, while the approach to ecosystem impacts is almost entirely empirical.

Consistent with the philosophy of the Study Design, the "integrated" modelling will in practice involve varying degrees of integration. Both the nutrient and toxicant budget models will be strongly integrated with the physics. It is not clear to what extent they will need to be integrated with each other. Ecosystem impacts will be linked to these integrated models as a series of simple, mostly empirical relationships, derived either from analysis of observations in Port Phillip Bay and other embayments, and/or from the literature. The modelling task is entirely reliant on the Ecological tasks to prescribe these impact links, and prediction of ecological impacts will depend on their ability to do so. In practice, the break between nutrient and toxicant fates and ecosystem impacts cannot be entirely clean, and the question of feedbacks of ecosystem-level impacts on nutrient and toxicant budgets will be a critical one for the Study to consider.

6.3.2 **Time and space**.

The management issues also determine the time and space bounds, and the time and space resolution, required of the model. Most of the envisaged scenarios cover periods of up to 50 years. Current research suggests that there will be additional uncertainty towards the end of this period due to predicted changes in climate. A key decision has been to limit the Study and the modelling to the Bay itself. The Project will not study or attempt to model the catchment and models will be designed to assess responses to prescribed changes in inputs at the Bay boundaries. Assessment of some management policies will require catchment models; these are being developed separately.

In terms of time and space resolution, a natural separation emerges from both management questions and current knowledge/understanding of the Bay between those questions related to the overall Bay budget and those related to local distribution and fates near inputs. The Phase 1 Study split the Bay into a Central Zone, an Exchange Zone controlling exchange with Bass Strait, and a Mixing Zone along the Bay shore, in which inputs are diluted to typical Bay concentrations. The distinction between Mixing Zone and Central Zone corresponds generally to the distinction between "near field" and "far field" as defined by the GESAMP report. While some of the questions for the Central Zone may be answered with simple box models resolving seasonal or annual time scales, it seems likely that questions about the Mixing Zone will require models which can, at least locally, resolve fine space scales and short (bloom or runoff event) time scales. However, final decisions about space and time scales will require a more careful progression from management issues through the ecosystem impact links to the knowledge required of water quality (e.g. response to average vs. peak exposure).

Questions of time and space resolution also have a bearing on the observational program. A general problem with modelling studies and the interpretation of field data is that observations are usually isolated points in space and time, while predicted variables represent implicit or explicit averages over space and time. This problem will be addressed at modelling workshops and in planned sample design workshops for particular field tasks.

6.3.3 **Previous modelling.**

The Phase 1 Report included an account of a primitive numerical circulation model, which was used to predict the distribution of phosphorous as a passive tracer. The Phase 1 Report projected that truly integrated models of the kind discussed here would be developed as part of Phase 2. Williams (1978) developed an integrated model of Port Phillip Bay, based on a 2-D depth-averaged finite difference numerical model, and a derived compartment model representing cycling of N and P through a phytoplankton, zooplankton, DON model. This model was, in its formulation, certainly state-of-the-art at the time of its development. The analysis was fairly limited, and the study appears to have been neglected by subsequent management and research publications. Newell (1990) developed a static nitrogen budget for the Bay treated as a single box.

6.4 **Integrated model components**

This section provides a more detailed account of modelling background and issues under sub-headings of physics, nutrients, toxicants and ecology.

6.4.1 **Physics.**

Physical modelling is discussed in detail in Chapter 2, so this subsection will simply highlight some of the issues related to the support for integrated models.

It is fairly clear from the Phase 1 Study that the turnover time of Port Phillip Bay as a whole due to tidal exchange with Bass Strait is in fact limited by mixing in the Exchange Zone rather than the tidal exchange volume.

It is clear that the techniques and computer power now exist to build much more sophisticated 3-D circulation models with high spatial and temporal resolution. It seems likely that the principal payoff may come not in modelling Bay-wide budgets, but in modelling transient small scale phenomena in the Mixing Zone adjacent to shore inputs.

Sediment dynamics in Port Phillip Bay have been virtually neglected in past studies, and are likely to play a key role in nutrient and toxicant models. There are a number of important aspects to sediment dynamics, including sediment transport as bed and suspended loads, the exchange of both dissolved and particulate fractions between water column and sediments, and the vertical transport due to diffusion, bioturbation etc within sediments. Models of sediment physics are in general more empirical, and likely to require more tuning, than water column circulation models. Sediment transport is complicated by questions of sediment composition and consolidation, as well as surface cover, surface morphology, etc. One can point to a number of potential feedbacks of ecosystem impact (e.g. seagrass cover) on sediment dynamics, which will need to be considered.

6.4.2 **Nutrients**

The nutrient cycle is perhaps the best studied aspect of Port Phillip Bay yet there has been no clear consensus on the overall nutrient budget or the factors controlling it. The nutrient data and their interpretation are discussed in detail in Chapter 3. While the N inputs to the Bay are reasonably well known, there is considerable uncertainty about the sinks. Newell (1990) presents a nutrient balance based on the Phase 1 Study (MMBW/FWD, 1973) suggesting that N inputs (*ca* 8000 tonnes N y^{-1}) are balanced primarily by a large flux of DON to Bass Strait, with a small loss (*ca* 1000 tonnes N y^{-1}) to denitrification. However, more recent data suggest a smaller pool of DON in the Bay, and, when account is taken of the background levels of DON in Bass Strait, the net export of DON could be as low as 2000 tonnes N y^{-1}. This leaves a major imbalance in the N budget, which could be balanced by N accumulation in sediments. The recent data suggest a large organic N pool (*ca* 20000 tonnes N) in surface (top 2 cm) sediments, but the net rate of N accumulation in sediments is not known.

There are additional ambiguities about the internal cycling of N within the Bay. In Chapter 3, calculations (based on very limited productivity data) suggest that annual phytoplankton production is equivalent to the uptake of *ca* 11000 tonnes of N p.a.. The annual input of DIN from terrestrial sources is estimated to be 3800 tonnes N p.a., and the efflux of DIN from sediments into the water column is estimated at 11800 tonnes N p.a., yielding a total of 15600 tonnes N p.a.. If the conclusion that phytoplankton are responsible for most primary productivity in the Bay is correct, then phytoplankton N uptake should at least approach this figure. If there is significant recycling of nitrogen within the water column, which seems likely, then the required N uptake by phytoplankton could be substantially higher.

The estimated standing stock of phytoplankton in the Bay is quite small (*ca* 200 tonnes N) so the turnover rate, even at the minimum production estimate, is quite high (*ca* 55 times p.a., equivalent to a net growth rate of 0.15 d^{-1}). This is still much lower than maximum (light and nutrient saturated) phytoplankton growth rates, but may be plausible as a depth and seasonal average. Average DIN levels, at least in the Bay centre, are typically less than 0.5 M, and phytoplankton growth there may well be N limited. At the same time, the Bay is relatively turbid, at least compared with oceanic or continental shelf environments, and light may be limiting in deeper sectors when these are well-mixed. The quantitative importance of light and nutrient limitation in controlling phytoplankton growth and production has not been established, but is amenable to modelling.

Models of nutrient cycling must address phytoplankton growth and uptake of inorganic nitrogen, losses of phytoplankton (to grazing by zooplankton, grazing by filter feeders and sedimentation), and the remineralization of N within the water column. They must also address the input of phytoplankton and detrital N to the sediment, the remineralization of N within sediments and the efflux of inorganic N and resuspension of particulate organic N from the sediments.

The picture created by the existing data and the nitrogen budgets is of Port Phillip Bay operating not as a standard phytoplankton chemostat, but as a community chemostat, analogous to an N-depleted oceanic mixed layer. However, it should be noted that the Phase 1 Study reported typical NH_4 values of about 3 M. Such levels are inconsistent with N limitation and would indicate a system delicately poised on the brink of turning eutrophic. However, methodological differences in the ammonia analysis in the early study throw some doubt on the reported high concentrations and more recent measurements are typically an order of magnitude lower.

In developing model equations for the Bay nutrient system, there is a long tradition of plankton ecosystem models, and ocean and coastal biogeochemical models,to draw on (Riley *et al.,* 1949; Steele, 1974; Kremer and Nixon, 1978; Longhurst, 1981; GESAMP, 1991). I won't discuss the details here, but I note that in using these models to predict the likely response of the Bay to increased N loading, one will rapidly come up against some classic problems in plankton modelling. What determines the current phytoplankton biomass levels in Port Phillip Bay? What fixes phytoplankton loss rates? What fixes the N regeneration rates? How close are current (N-limited) phytoplankton growth rates to maximum (light and temperature limited) growth rates? One will predict different responses to changes in N loads if one assumes grazers will keep phytoplankton levels fixed, as some models predict, or if one assumes phytoplankton loss rates will stay fixed (*cf* Steele and Henderson, 1992). These are questions that we need to address in the Study, both through modelling and appropriate field research.

Of course the Bay differs from an ocean mixed layer in that it is shallow, and N cycling through the sediments plays a key role. A significant amount of particulate production is likely to end up in the sediment, either through sedimentation or due to benthic suspension feeders. The processes controlling N recycling in sediments, any net N accumulation in sediments, and losses to denitrification must also be represented. There are a number of models of coastal and estuarine systems which establish good precedents.

The discussion so far has focused on the Bay-wide nutrient balance. The Study will also address the concentration and fluxes of N in the Mixing Zone near inputs. What is not clear here is the relative importance of N uptake by the highly productive benthic systems (benthic microalgae and macroalgae) in the Mixing Zone compared with dilution. However, limited sediment analysis off Werribee suggests that there is little accumulation of N in the sediments, so that any N trapped by these benthic systems is either remineralised and ultimately transported to the Bay centre as DIN, exported as PON, or lost through denitrification.

The water column in the Mixing Zone is characterised by very high and variable inorganic N concentrations, and occasional high chlorophyll levels. There is some evidence of a seasonal cycle with a winter build-up of inorganic N concentrations and a spring bloom. The high input and high local concentrations of inorganic N in this zone create conditions conducive to the development of intense blooms. The development of these blooms is likely to depend strongly on short term weather forcing, and the prediction of events such as toxic algal blooms, or intense blooms of unattached macroalgae, may be only possible in a statistical sense. That is, the predicted frequency of blooms will depend on weather patterns or climate, and, depending on the outcome of the ecological monitoring, we might expect to predict that as the frequency of intense blooms increases, the frequency of toxic blooms will also increase.

6.4.3 **Toxicants**

The historical data set for toxicants for Port Phillip Bay, reviewed in Chapter 4, appears to be much weaker than that for nutrients. At the same time the state of the art in toxicant modelling is more empirical, being based primarily on simple linear models and empirically adjusted partition coefficients. The physical processes of sedimentation and resuspension which couple the water column to the sediments are generally critical to the modelling of toxicant transport and fate. The relatively high clearance rates estimated for suspension feeders in Port Phillip Bay suggest that they may also play a significant role in particulate fluxes to the sediment.

A review of the key processes affecting fate and distribution of toxicants can be found in Ambrose *et al.* (1988). The processes of complexation, precipitation, ionization and sorption affect the chemical form of toxicants, their distribution between dissolved and particulate fractions and their bioavailability. Processes such as sorption are particularly important because of the role of sediments in storage and transport. Sorption reaction rates are usually rapid, and are assumed to reach equilibrium, with equilibrium concentrations determined by partition coefficients. However, laboratory determined coefficients may not be applicable in the field, and may in practice be replaced by empirically determined distribution coefficients which can vary in space and time.

Processes controlling sinks for toxicants include volatilization, hydrolysis, photolysis and biodegradation. These are usually modelled by first-order or linear kinetics, with empirical loss rates. Some of these processes occur at interfaces, and loss rates may also vary in space and time.

It seems likely that the general modelling approach for toxicants will need to be more empirical than for nutrients. A budget model can be developed based on simple process submodels, which will probably need to be tuned to observations. It is possible that inverse models based on predicted transport and applied to observed toxicant fields can be used to estimate sources and sinks without making process assumptions. There will again be a choice of Bay-wide and local models. The data appear to indicate that, for most toxicants, elevated concentrations approaching quality standards occur locally rather than Bay-wide.

Given this focus on empirical models as well as the shortage of historical data, there will need to be careful integration of field and modelling activities to ensure that the time and space scales of sampling are sufficient to support development of budgets, to apply inverse models or to estimate model parameters.

6.4.4 **Ecology**.

It is clear from the discussion above that some elements of the ecosystem, including certainly phytoplankton and possibly grazers and benthic flora and fauna, will need to be included explicitly or implicitly as part of the integrated nutrient and toxicant models. This subsection is concerned with the impacts of nutrients and toxicants on valued components of the ecosystem where feedback effects on the nutrient and toxicant budgets are perceived to be unimportant. As noted earlier, these impacts will be handled by simple empirical models which will be driven by the output of the integrated models for purposes of addressing management scenarios.

The Study has eschewed a process-based approach to ecosystem impacts in favour of a more or less completely empirical approach. The project will draw on empirical relationships established overseas, and correlative relationships between impacts and water and sediment quality variables based either on contrasts within Port Phillip Bay, or contrasts of Port Phillip Bay with other similar systems. It will only be possible to address these impacts through modelling to the extent that such relationships can be established, and subject to the uncertainties associated with those relationships.

The choice of space and time scales for nutrient and toxicant models will to some extent be determined by the needs of the ecosystem impact links. It is clear that there will need to be an ongoing exchange of information between the integrated modelling and the ecology tasks. It should be possible to make some progress up front by drawing on experience overseas, and the relevant literature.

6.5 Management applications

Aspects of model evaluation and application were discussed in a general sense in Section 6.2.3, and the key management issues or indicators were presented in Chapter 1 and discussed in Section 6.3.1. There is little point in attempting to be more specific here. The distribution of effort for the integrated modelling tasks has been altered from the initial Project Design so that there is a more uniform distribution of effort through the life of the Study. This is designed to allow both for more interaction between model development and the field tasks, and for ongoing interaction between modellers and managers. The intention is not to produce a single model and walk away, but to create a series of models and, through development of appropriate tools (graphics interfaces, etc) and a series of workshops, to transfer the understanding and predictive capability gained through the Study to managers.

One of the key issues to be addressed later in the modelling task will be the application of adaptive management techniques to ongoing management of Port Phillip Bay. The underlying concepts are presented by Holling (1978). It is likely that, even at the end of this Study, there will remain significant areas of uncertainty in predicting the response of the Bay to certain management actions. The adaptive management process attempts to identify and quantify these uncertainties in the context of appropriate system models, and to develop both monitoring and management strategies which will allow managers to learn about the system through management, and to maximise the chances of identifying unacceptable negative responses and taking corrective action before they become irreversible. This process provides a systematic approach to the design of long-term monitoring programs within the framework of management decision-making.

Chapter 7. References

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