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Environmental Impact Assessment and Review of Effluent Disposal Options for Eastern Treatment Plant

EMS FINAL REPORT 1999

Final Report
June 1999

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Prepared for **Melbourne Water Corporation**

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ISBN 0 643 06060 X

Limitations Statement

The sole purpose of this report and the associated services performed by CSIRO is to provide scientific knowledge for Melbourne Water Corporation to prepare an Effluent Management Strategy for Eastern Treatment Plant. Work was carried out in accordance with the scope of services identified in the agreement dated November 9 1996, between Melbourne Water Corporation ('the Client') and the Commonwealth Scientific and Industrial Research Organisation (CSIRO).

The findings and recommendations presented in this report are derived primarily from information and data supplied to CSIRO by the Client and from field investigations and research conducted by CSIRO, other agencies and consultants. The passage of time, manifestation of latent conditions or impacts of future events may require further exploration and subsequent data analysis, and re-evaluation of the findings, observations, conclusions, and recommendations expressed in this report.

This report has been prepared on behalf of and for the exclusive use of the Client, and is subject to and issued in connection with the provisions of the agreement between CSIRO and the Client. CSIRO accepts no liability or responsibility whatsoever for or in respect of any use of or reliance upon this report by any third party.

Foreword

I have pleasure in writing this foreword for this major environmental study managed by the CSIRO Environmental Projects Office.

This Effluent Management Study is the second major environmental study undertaken for Melbourne Water by CSIRO's Environmental Project Office. The knowledge and skills that were acquired during the Port Phillip Bay Environmental Study proved to be invaluable in developing a detailed understanding of the local ecosystem at Boags Rocks. However, unlike the Port Phillip Bay Environmental Study, the Effluent Management Study was concerned with addressing a known and specific environmental issue comprising two major themes: (i) impact assessment of ocean disposal of treated effluent; and (ii) investigation of feasible alternative disposal and treatment options.

During the course of this study, we brought together scientists from within CSIRO as well as experts from local universities, government research organisations, and private companies. They developed and executed a program of field work to address knowledge gaps and construct a detailed scientific understanding of the ecology and physical processes that dominate the hostile environment at Boags Rocks. Significant insights from these investigations have been obtained and these form the basis of this final report.

CSIRO acknowledges the contributions made by many individuals, research teams and organisations involved in this study. We are grateful to Melbourne Water who provided us with the resources needed to complete these investigations. We are equally appreciative of the open and constructive participation of officers of the Environment Protection Authority whose input at various stages of the study allowed us to identify and prioritise issues of concern.

Environmental issues are invariably complex and usually defy simple remedies. The situation at Boags Rocks is certainly no exception. Contained in the pages of this report are the results of detailed scientific investigations that shed considerable light on the environmental status of Boags Rocks, impacts on the local ecosystem, and alternative disposal options. The remaining challenge for Melbourne Water, the Environment Protection Authority and the public is to integrate the scientific outcomes presented here with other equally important considerations relating to social, economic, and environmental values to identify an appropriate management response to effluent disposal from Eastern Treatment Plant.

Dr. Nan Bray

Chief CSIRO Marine Research
(Chair of Environmental Projects Office Steering Committee)

Acknowledgments

This report draws heavily from and synthesises the results of a number of scientific investigations undertaken over the past two and half years. We are indebted to the critical role played by the Technical Group in ensuring the scientific integrity of the Study and of its outputs.

This Study could not have been undertaken without the significant contributions made by each of the contractors and CSIRO Divisions who undertook individual scientific tasks. The dedication and commitment at both an individual and organisational level to the delivery of quality scientific outcomes is reflected in the pages of this report.

The Victorian EPA provided valuable input throughout the Study. Their contributions at various stages of the project are greatly appreciated.

Finally, we are appreciative of the resources committed by Melbourne Water to enable this study. We are particularly appreciative of the assistance provided by Melbourne Water Project Manager, Margo Kozicki for making herself available to answer our many questions and provide the CSIRO team with the data and information they needed.

CSIRO Technical Group:

Dr Graeme Batley, Centre for Advanced Analytical Chemistry
Dr David Fox, Mathematical and Information Sciences
Dr Jeanette Gomboso, Land and Water
Dr John Parslow, Marine Research
Dr Sebastian Rainer, Marine Research
Dr Jenny Stauber, Centre for Advanced Analytical Chemistry
Dr Stephen Walker, Marine Research

Summary

Melbourne Water engaged the CSIRO Environmental Projects Office to manage and conduct an environmental impact assessment and review of land and marine effluent disposal options for the Eastern Treatment Plant (ETP).

The plant, situated on a 1000 ha site at Carrum, treats about 42% of Melbourne's sewage. The secondary treated effluent, is chlorinated prior to discharge into Bass Strait at Boags Rocks near Cape Schanck.

As part of the Study a series of scientific tasks was carried out between January 1997 and October 1998 to provide data to assess the nature and extent of the environmental impact of the existing ocean outfall, and to evaluate alternative disposal options. This report summarises the results of those tasks and provides a discussion of the scientific issues to consider when developing an Effluent Management Strategy.

Biological monitoring was conducted to assess the extent of the effluent's impact on the seabed communities offshore from the discharge point. This work included diver-assisted surveys to establish the distribution and abundance of plants and animals along the rocky reef that runs parallel to the shore. Other work in the offshore zone included sampling of the soft seabed to ascertain the nature of the infauna. To complete the biological monitoring, a series of seasonal surveys was undertaken to examine the distribution and abundance of macroalgae on intertidal rocky platforms at Boags Rocks and other locations between Cape Schanck and Point Nepean.

The rocky platform at the outfall site has been denuded of its original brown algal cover. Of note is the loss of *Hormosira banksii* (Neptune's necklace) and *Durvilleae potatorum* (Bull kelp). Several opportunistic green algae, invertebrates and a spionid worm *(Boccardia proboscidea)* have partially occupied the void. However, we were unable to detect any longitudinal gradient of effect against the diverse assemblages present, which have natural longshore variation on all examined rocky platforms. Many of the species present exhibit seasonal and year-to-year variation, which confounds the impact assessment.

An attempt was made to establish the extent of impact along the coast by identifying and counting fauna in samples of beach sands. However, the preliminary investigation of four beaches found that the fauna exhibited wide variation in species present and that they were present in low abundances. To assess outfall related impacts would require an excessively large sampling program, which was considered to be inappropriate. Instead, an offshore sampling program was conducted. This revealed that there was lower diversity of infauna within about 660 m of the outfall, which may be an effect of the discharge. Further afield, impacts were difficult to establish due to natural spatial and temporal variation in fauna and seabed conditions.

A survey of the offshore reef, which is about 600 to 800 m from and running roughly parallel to the shore, revealed abundant flora and fauna but also exhibited longitudinal variability, which masks any effluent effects. Based on the results of this single survey, it was postulated that effluent discharge was a factor affecting the biological characteristics of the reef to a distance of 1100 m from the line of the outfall, and that a lesser impact may occur out to about 1400 m.

With any effluent discharge there is concern over the potential contamination of seafood through toxicant bioaccumulation. For this Study, samples of abalone, wrasse (parrot fish) and sea squirts (cunjevoi) were collected just offshore from the discharge point and analysed for a suite of contaminants. The results were

similar to previous studies carried out by Melbourne Water, and suggested that contamination was insignificant and posed negligible risk to human health.

To obtain a direct measure of the potential impact of ETP effluent, bioassays were conducted using a battery of local test species including a bacterium, microalga, macroalgae, invertebrate and fish larvae. The results showed that the effluent discharged from ETP was non-toxic to bacteria, 1-3 day old fish larvae and macroalgal fertilisation. It was mildly toxic to 4-5 week old fish, and inhibited both diatom and macroalgal growth. The effluent was most toxic to scallop larvae. The ecotoxicology studies suggested that a 300 times dilution of effluent would satisfy internationally accepted guidelines for no hazard to 95% of species. Subsequent laboratory testing identified ammonia as the principal toxic agent, although the freshwater nature of the effluent has a synergistic effect.

Receiving water quality was measured by underway sampling and spot samples to assess the concentration of nutrients (ammonia, nitrate, nitrite, phosphate and silicate) and associated parameters (chlorophyll *a*, salinity, temperature and dissolved oxygen), and for the presence of toxicants. None of the heavy metals or common organic pollutants tested exceeded recommended water quality guidelines, however undissociated ammonia levels did exceed existing EPA limits. No algal blooms have been recorded in the area and the sampling for chlorophyll *a* indicated that it was not above normal coastal levels. The sampling also showed that dissolved oxygen in the water column was always above 90% saturation and often above100% saturation.

In a parallel project, Melbourne Water engaged Monash University to undertake a literature review on the health effects of ocean outfalls and to review the results from routine *E.coli* sampling by Melbourne Water, and additional sampling for *Enterococcus* spp. and total coliforms. Based on these data it was concluded that surfers appear to be at no additional risk of contracting disease from surfing in the area when compared to other beaches studied.

It is difficult to establish the level of change that has occurred in the biological communities without having reference to “before discharge” data. However, the changes that have been noted in both the near and far fields, are likely to be due to a combination of freshwater, ammonia toxicity, and nutrient load.

The second part of the Study was the review of land and marine effluent disposal options. This included evaluation of flow reduction (reuse) opportunities, treatment improvements and better dispersion through an outfall extension.

Some 14 options for volume reduction were assessed. These offered an array of management strategies for flow reductions ranging from less than 1% for industrial or greywater re-use to approximately 95% for indirect potable re-use. Costs ranged from 10c to $9.86 per kilolitre of flow reduction. The review suggested that a concerted program of effluent re-use could reduce effluent discharge significantly over time and, for the future, indirect potable reuse should be considered in preference to building new dams.

Sewage treatment engineers modelled options for ETP treatment process modifications. The results indicated that ammonia levels could be reduced by a factor of six and total nitrogen halved. Concomitant improvements in other properties of the effluent would also be achieved. Cost estimates largely depended on the degree of peak flow management adopted but were in the range $40 to $100 million.

The engineering feasibility and cost of extending the existing outfall from the current shoreline discharge to a point further offshore was investigated. Costs ranged from $26 million for a 1.3 km extension to $46 million for a 3.1 km extension. It was calculated that all options are practicable without additional pumping, provided that the final 10 km of the existing pipeline was sealed in order to increase the head pressure necessary to allow the effluent to discharge at a depth of 32 m below sea level. The outfall extension would consist of twin 1.5 m diameter steel pipes with terminal diffusers giving greater than 50:1 initial dilution.

The effectiveness of extending the outfall to increase dispersion was assessed through the development of a three-dimensional hydrodynamic model with a water quality module, which was applied to six hypothetical outfall configurations located from 1 to 3 km offshore. The results indicated that the average dilution at the shoreline would be increased significantly through offshore discharge, with greater performance from the longer outfall. This would reduce the impact due to both freshwater and ammonia toxicity and would allow measurable recovery of the ecosystem at Boags Rocks. However the nutrient load to the region, which contributes to far-field effects would not be reduced and the risk of algal blooms occurring in the region would not be significantly modified.

Treatment improvements would reduce the concentrations of ammonia and total nitrogen (nutrient load), which would allow the far-field impacts to the subtidal communities and platforms towards Cape Schanck to be reduced. However there would still be a level of toxic impact at Boags Rocks as a result of the freshwater continuing to be discharged. It should be noted that both freshwater and ammonia have a synergistic effect. If one were removed the toxic impact presently observed on the platforms adjacent to the outfall would be reduced.

In summary, treatment improvements and increased re-use provide a compromise between cost, sustainability and environmental improvement. The far-field impacts of the discharge seen on the rocky platforms towards Cape Schanck and the offshore seabed would be reduced, but we are unlikely to see full recovery at Boags Rocks as the freshwater impact (though significantly less than ammonia) will still be present. Extending the outfall would enable the biological communities at Boags Rocks to recover to some extent, but would not lead to a reduction in the overall nutrient load being discharged. Total re-use would eliminate the need for any discharge, therefore avoiding any impact to the marine environment.

Eastern Treatment Plant at Carrum, treats 42% of the city's sewage.

Contents

1 Introduction

Melbourne Water Corporation is licensed by the Victorian Environment Protection Authority (EPA) to discharge treated effluent from its Eastern Treatment Plant into the ocean at Boags Rocks and onto land in specified areas at the plant.

The Licence (EW 367) specifies the conditions under which effluent may be discharged and lists the minimum standard that the effluent must be treated to prior to discharge. It also specifies the scope of the monitoring program required to demonstrate environmental performance.

The Licence (Section 3.7) required Melbourne Water to undertake an investigation and consultation program to evaluate treatment methods, effluent re-use and offshore discharge in order to improve environmental performance.

To fulfill these requirements Melbourne Water engaged the services of the CSIRO Environmental Projects Office to manage and conduct an environmental impact assessment and review of land and marine effluent disposal options for the Eastern Treatment Plant. This project is also referred to as the Effluent Management Study for Eastern Treatment Plant.

1.1 The Study Objectives and Approach

The objective of the Study was to establish what impact the current method of effluent disposal is having on the environment in the vicinity of Boags Rocks and the adjacent waters of Bass Strait and what the benefits of alternative disposal options would be.

The results of the Study will assist with the evaluation of strategic options and development of an Effluent Management Strategy by Melbourne Water for Eastern Treatment Plant.

The Study was designed in two stages that related to the two key questions presenting themselves in the environmental impact assessment. These were:

1) What is the nature and magnitude of the environmental effect of the effluent, if any?

2) If an effect exists, how can it best be removed or adequately mitigated by improvements in treatment technology, better dispersion offshore or alternative disposal paths?

The first stage of the Study focused on assessing the environmental impact of the effluent discharge on the marine environment. It included: biological monitoring, bioaccumulation, toxicity assessment, and testing of receiving water quality.

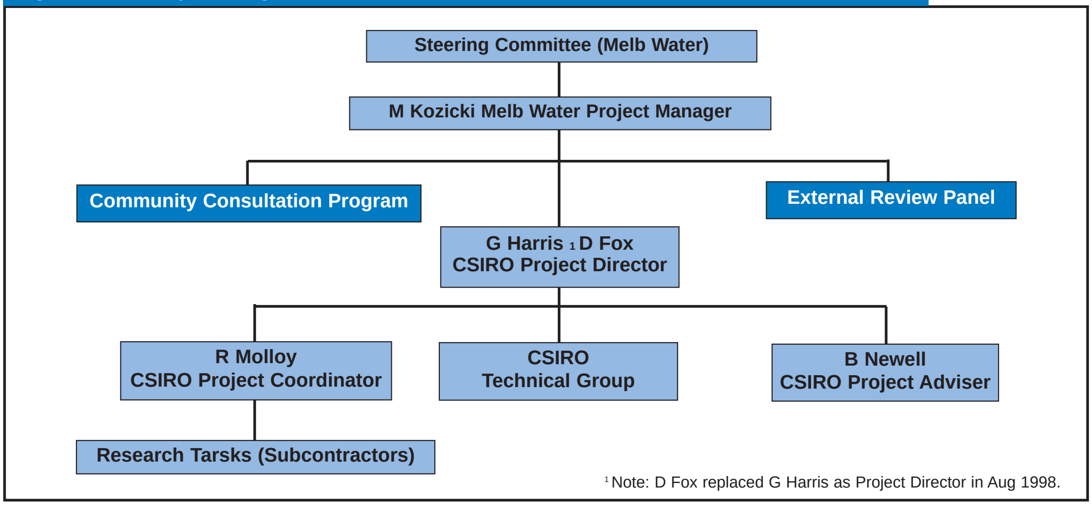
In the second stage alternative disposal options were assessed. This included a review of flow reduction and effluent re-use opportunities, treatment improvements, and modelling and design of various lengths of extended outfall to improve dispersion.

1.2 Study Management

Melbourne Water appointed a Project Manager and set up a Steering Committee to oversee the Study Program. Melbourne Water also established an external review panel to oversee the scientific output from the Study.

The reporting lines and management structure are shown in Figure 1.1

Figure 1.1 Study management structure



To monitor the individual components of the study program, CSIRO set up an internal peer review panel (Technical Group) which comprised scientists from a broad cross section of scientific fields.

They met and corresponded with the researchers throughout the course of the Study, to discuss and evaluate the results of the research.

Members of the CSIRO Technical Group and their area of expertise were:

Dr Graeme Batley, Centre for Advanced Analytical Chemistry	Toxicant chemistry
Dr David Fox, Mathematical and Information Sciences	Environmental statistics
Dr Jeanette Gomboso, Land and Water	Water resource management
Dr John Parslow, Marine Research	Water quality and modelling
Dr Sebastian Rainer, Marine Research	Marine biology
Dr Jenny Stauber, Centre for Advanced Analytical Chemistry	Toxicity testing
Dr Stephen Walker, Marine Research	Hydrodynamic modelling

The external review panel included:

Dr Des Lord, Des Lord & Assocs.	Oceanography and water quality
Dr Ian Law, CH2M Hill consultants, Sydney	Treatment, effluent re-use and outfall engineering
Dr Arthur McComb, Murdoch University, WA	Biology and water quality
Dr Jason Middleton, University of NSW	Hydrodynamic modelling
Dr Nancy Millis, University of Melbourne	Toxicity and biology

1.3 List of Consultants and Researchers

A combination of academic institutes, local consultants and CSIRO scientists carried out the various data collection and research activities for the Study (Table 1.1).

Table 1.1 List of study tasks and organisations responsible

	Major Contractors
Biological Monitoring	
Beach infauna feasibility study	• Museum of Victoria
Subtidal biological surveys (sandy seabed and reef)	• Consulting Environmental Engineers P/L
Shoreline platforms seasonal algal surveys	• Monash University, Biological Sciences
Bioaccumulation	
Seafood contaminant levels	• Marine & Freshwater Resources Institute
Toxicity Assessment	
Toxicity testing of effluent	• CSIRO Centre for Advanced Analytical Chemistry • Marine & Freshwater Resources Institute • RMIT University, Dept of Applied Biology • Victoria University, School of Life Sciences
Hormosira banksii bioassay research	• Monash University, Biological Sciences
Toxicity identification evaluation tests	• CSIRO Centre for Advanced Analytical Chemistry
Receiving Water Quality	
Underway analysis - Nutrients	• Marine & Freshwater Resources Institute
Point sampling - Toxicants	• Marine & Freshwater Resources Institute
Microbiological health risk assessment (managed by Melb Water)	• Monash University, Dept of Epidemiology and Preventive Medicine
Flow Reduction (Re-Use) Study	• CSIRO Land and Water • Gutterridge, Haskins and Davey P/L
Oceanographic Studies	
Measuring currents	• Marine & Freshwater Resources Institute
Monthly profiles (salinity, temperature and dissolved oxygen)	• Marine & Freshwater Resources Institute
Measuring winds at Gunnamatta	• TTS Systems P/L
Hydrodynamic and water quality modelling	• CSIRO Marine Research
Extended outfall design and costing	• Consulting Environmental Engineers P/L
Treatment Improvement Review (managed by Melb Water)	• CMPS&F Environmental P/L

1.4 Community Consultation Process

Melbourne Water implemented a community consultation program, which provided opportunity for interest groups, environmental groups, stakeholders and the wider community to have input into the Study and to be kept regularly informed of the progress and outcomes of the various tasks being undertaken. Regular briefings, newsletters, media releases, public displays and a website were all part of the extensive consultation process.

Some of the issues raised through the consultation process included:

- The loss of seaweed species from the immediate area around the outfall.
- The level of toxicity and cumulative effect of the effluent on the flora and fauna of the marine environment.
- Establishing whether the treated effluent has an impact on human health, particularly surfers who are in the water considerably longer than other beach users.
- Reducing the volume of effluent being disposed of at Boag's Rock through re-use options and community education.
- The long-term objective of eliminating ocean outfalls through alternative methods of disposal.
- Ensuring the Study tasks and outcomes meet the environmental objectives set by the EPA.
- The impact of the Study outcomes on long-term planning issues, particularly as they relate to sewerage management and the water retail companies.

As an outcome of the community consultation program, Melbourne Water conducted a series of oral history interviews with people identified by the reference groups. Each interview was recorded and the transcripts (Melbourne Water 1999) provided to CSIRO for reference.

Members of the Reference Groups were provided with tours of ETP

2 Background

2.1 Eastern Treatment Plant (ETP)

Melbourne Water is responsible for the transfer, treatment and disposal of sewage supplied by Melbourne's retail water businesses - City West Water, South East Water and Yarra Valley Water. Eastern Treatment Plant (ETP), which is owned and operated by Melbourne Water, treats and disposes of flow from two of these businesses (South East Water and Yarra Valley Water).

ETP is situated on a 1000 hectare site at Carrum (Fig 2.1). It treats about 42% of Melbourne's sewage. The average flow is about 370 megalitres per day (ML / day), and is delivered by the South-Eastern, Chelsea-Frankston and Dandenong Valley Trunk Sewers.

The activated sludge treatment plant produces a secondary treated effluent, which is then chlorinated prior to discharge. The treated effluent is transferred via an outfall pumping station through a ten kilometre rising main to the gravity-fed South Eastern Outfall. The outfall discharges to Bass Strait at Boags Rocks near Cape Schanck, 56 km from the Plant. The 2.75m diameter outfall pipe terminates in a concrete discharge structure below the low tide mark.

About 20 ML / day is added to the outfall from three local treatment plants (Mornington 12 ML / day, Rosebud 6 ML / day, and Hastings 2 ML / day) which are operated by South East Water. Each of these plants operates under separate EPA licences.

2.2 History

The following history has been compiled using the book by Tony Dingle and Carolyn Rasmussen, "Vital Connections", which covered the history of the Melbourne and Metropolitan Board of Works (1891 - 1991).

In 1891 the Melbourne and Metropolitan Board of Works was formed. Its charter was to build a sewerage system and take over operation of the water supply system. Funding for the new Board was provided through rates levied on property owners.

The new Board was presented with a sewerage system designed by an English consultant, James Mansergh. He recommended two sewage treatment farms be established at Mordialloc and Werribee. An alternative was an ocean outfall to Cape Schanck.

In 1892, work commenced on designing and constructing a network of gravity fed sewers that would service the inner ring of Melboune suburbs. The Board had opted for a single farm at Werribee (now known as the Western Treatment Plant), with a pumping station at Spotswood.

By the 1950s Melbourne had continued to develop its urban sprawl. The number of unsewered properties continued to increase and the sewerage system struggled to cope with wet weather flows. Five alternatives were presented to the Board; a secondary treatment plant at Carrum with an outfall to Port Phillip Bay, Western Port or Bass Strait; an outfall sewer to Bass Strait without treatment; and a primary treatment plant at Carrum with discharge to Western Port via a pondage system.

As part of the evaluation of a discharge to Bass Strait, the Board commissioned a study of ocean currents in the area. During 1953-54 radar was used to track floats released offshore from Boags Rocks. There had been a perception that the tides would take the sewage back into Port Phillip Bay, but the studies showed the predominant currents flowed to the east. Also as part of the investigation, Board surveyors did a levelling run from Boags Rocks to Keysborough. However the construction of the outfall was considered too costly, given the shortage of finances. Instead the capacity of the existing system was increased.

In 1963 there were 117,000 unsewered houses in the metropolitan area; 38,000 of them relied on septic tanks, while the remainder used the pan system. This inadequate situation had led to the gross pollution of metropolitan drains and creeks. Melbourne's residential expansion was being concentrated in the east and southeast, and it was becoming increasingly difficult to connect these areas to Werribee.

The Board decided in 1964 to build a major sewerage treatment plant at Carrum. Effluent disposal was to be either by an outfall pipeline into Port Phillip Bay (discharging 2km offshore from Carrum Beach) or to Boags Rocks. The latter option though twice the price was the preferred option, however by 1967 the continued financial uncertainty and the growing sewerage backlog forced a re-appraisal of the alternative discharge to Port Phillip Bay.

The impact to Port Phillip Bay of the increased discharge of effluent with a high nutrient load was unknown. This prompted the 1968-71 Phase One Environmental Study of Port Phillip Bay, for which the Board provided substantial funding. However, before the study was completed, key unions threatened to black ban the scheme if effluent went to the Bay and with an election looming, the State Government vetoed the scheme. The more expensive ocean outfall was to be funded by an increase in rates. Design work for the outfall was recommenced and the Board purchased the land abutting the foreshore reserve at Boags Rocks in July 1971.

The Eastern Treatment Plant (then known as the South Eastern Purification Plant) was commissioned on 19 September, 1975, and was the centrepiece of the entire $204 million sewerage system upgrade. It also included the 33 km South Eastern Trunk Sewer carrying sewage by gravity from Kew to Carrum; intercepting sewers diverting waste from existing main sewers into the trunk and the outfall to Bass Strait.

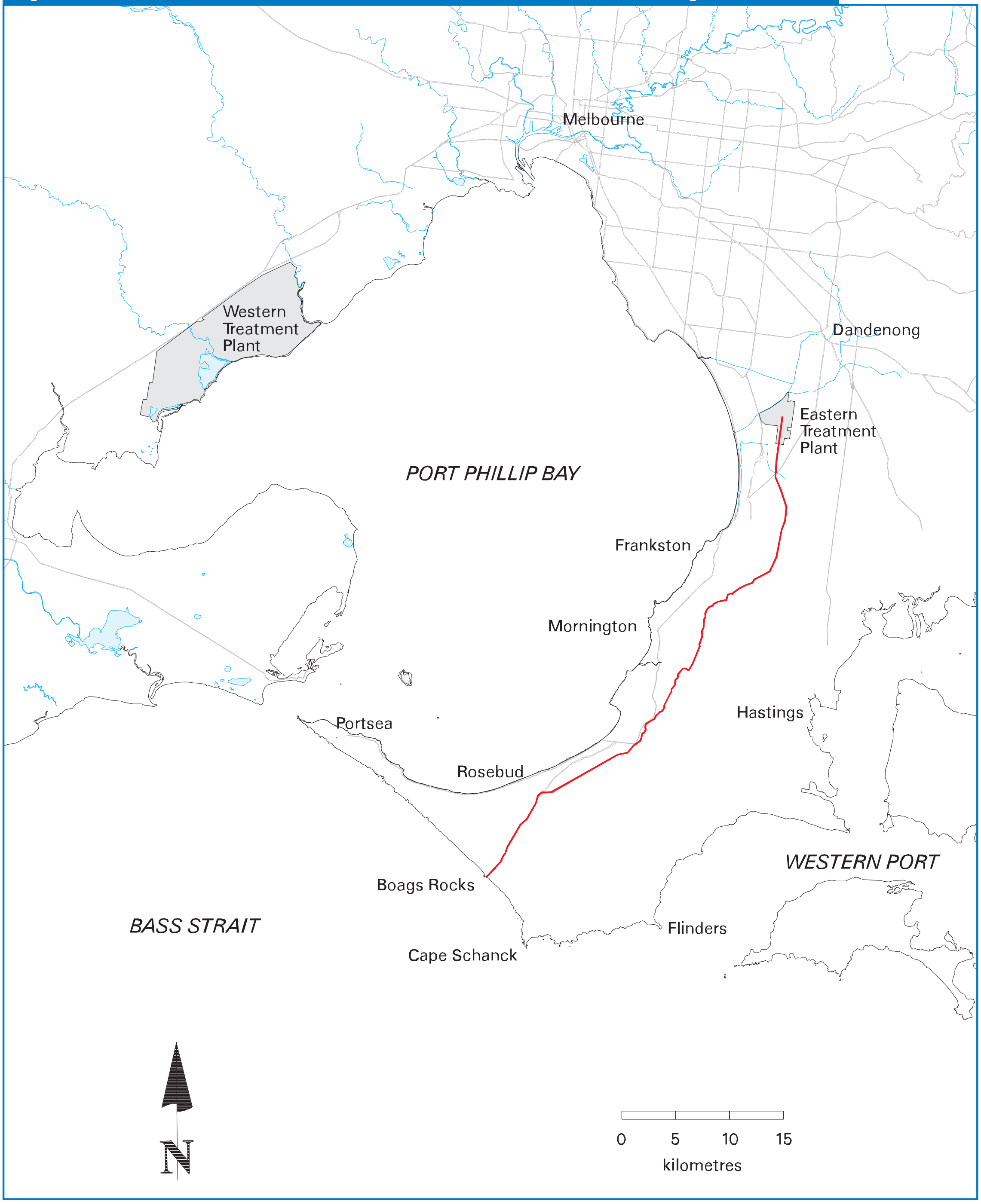
Apart from providing many areas in the outer southeastern suburbs with urgently needed sewerage facilities, the new system gave temporary respite to the Western Treatment Plant. It allowed flows from various overloaded sewers in Brighton, Caulfield, Moorabbin, Oakleigh and Sandringham to be diverted. It also relieved the load on the Braeside Purification Plant, which was ultimately phased out of operation.

Construction of the sewers and the outfall pipeline were mammoth tasks involving the use of tunnelling machines through rock and soft ground. The most challenging part of constructing the ocean outfall was at Boags Rocks, where pipes had to be laid underwater, and there were risks associated with the unpredictable behaviour of the surf.

With the aid of compressed air, a tunnel was driven beneath the shoreline cliffs and below sea level. The 2.75 m steel pipeline was assembled on rails within the tunnel, which was then flooded. At the same time, using a previously constructed jetty as a work platform, a submarine trench was dredged through the surf zone. Before excavation, piles were driven along both sides of the full length of the proposed trench. Sheet piling was then driven between the piles until both sides and the seaward end were closed in. A deck section was placed on top of the piles to allow an excavator to operate. After excavation, the rails were extended from the tunnel to the end of the trench and the final section of the outfall pipeline was secured and the trench backfilled with mass concrete topped with large boulders. The work platform and other construction equipment were removed.

Effluent discharge commenced in 1975, under licence from the EPA. Since then, Melbourne Water has carried out various monitoring of the effluent-receiving environment as part of those licence requirements

Figure 2.1. Location of Eastern Treatment Plant, with ocean outfall to Boags Rocks



2.3 Eastern Treatment Plant Discharge Licence

Under section 20 of the Environmental Protection Act 1970, Licence Number EW 367 (last amended February 21 1997) has been issued to Melbourne Water Corporation by the EPA. It allows Melbourne Water to discharge waste into waters, and onto land from Eastern Treatment Plant and into associated pipelines, subject to certain conditions being met.

Melbourne Water has the following objectives for the discharge of waste to the environment:

a) The management and operation of the treatment facility shall aim to optimise treated wastewater quality and minimise environmental impacts.

b) Future upgrades and/or augmentation of the treatment works shall aim to improve treated wastewater quality and further reduce environmental impacts.

c) The reduction in the size of the mixing zones shall be progressively achieved by the application of cost effective waste treatment technology, waste minimisation and sustainable re-use of wastewater.

d) The sustainable re-use of treated wastewater and sludge shall be maximised as much as practicable.

2.3.1 Discharges to Water

Melbourne Water may only discharge treated wastewater to Bass Strait at Boags Rocks, via its ocean outfall pipeline. The waste discharge must comply with the limits in Table 2.1.

The waste discharged via the Boags Rocks outfall must not cause the waters of Bass Strait (including the mixing zones) to exhibit visible floating oil, grease, litter or other objectionable floating matter; or generate objectionable odours.

The waste discharge must not cause the death of fish or other motile species; or the contamination of fish and crustaceans which causes them to be unacceptable in commercial markets or which causes them to exceed health standards as set out in the National Health and Medical Research Council's Food Standards Code outside the 200 metre mixing zone.

Outfall pipeline was commissioned in 1975.

Table 2.1: Discharge to water - performance limits

Performance Indicator, Unit	Median, Annual	90th Percentile, Annual	Maximum	Monitoring Frequency	Results 1995/96 median
Biochemical oxygen demand (mg/L)	20	40	NS	weekly	24
Suspended solids (mg/L)	30	60	NS	weekly	17.5
Total residual chlorine (mg/L)	NS	NS	1.0	weekly	< 0.1
E.coli bacteria (organisms/100 mL)	200	1000	NS	weekly	17
Ammonia as nitrogen (mg/L)	30	NS	40	monthly	25.7
Total combined nitrogen (mg/L)	NS	NS	NS	every 4 months	33.7
Total phosphorus (mg/L)	NS	15	NS	every 4 months	6.0
Dissolved oxygen (mg/L)	NS	NS	not < 6.0	monthly	7.9
pH range (pH units)	NS	NS	6.0 - 9.0	weekly	7.5
Anionic surfactants (mg/L)	0.4	NS	0.8	monthly	0.4
Phenol (µg/L)	NS	NS	100	monthly	< 2.0
Toluene (µg/L)	NS	NS	50	monthly	< 2.0
Benzene (µg/L)	NS	NS	25	monthly	< 1.0
PAHs total* (µg/L)	NS	NS	15	every 6 months	5.3
Mercury (mg/L)	NS	0.0005	0.001	monthly	< 0.0002
Chromium (mg/L)	NS	0.075	0.15	monthly	< 0.01
Lead (mg/L)	NS	0.05	0.10	monthly	< 0.01
Copper (mg/L)	NS	0.05	0.10	monthly	0.02
Cadmium (mg/L)	NS	0.005	0.01	monthly	< 0.001
PCCD/F** (µg/L)				every 2 years	
Broad spectrum toxicant analysis				annually	
Flow (ML/day)	540		770	daily	413

NS not stated

* includes all alkyl derivatives

** PCCD/F means polychlorinated dibenzo dioxins and furans as toxic equivalents of 2,3,7,8 tetrachloro-dibenzo-p-dioxin

2.3.2 Mixing Zones

Mixing Zones are areas adjacent to the discharge point (Fig. 2.2) where the receiving water quality objectives otherwise applicable under the State Environment Protection Policy (SEPP) (Waters of Victoria, Schedule B) do not apply with respect to certain indicators as specified in the licence. The following mixing zones are applicable for the specified water quality indicators:

a) The total dissolved solids objective for the coastal segment, as stated in the SEPP - Waters of Victoria, Schedule B shall not apply to the waters contained in a rectangular segment 900 metres off-shore to 1.7 kilometres long-shore to the west and 2.3 kilometres long-shore to the east of the discharge point.

b) The nutrient objective for the coastal segment, as stated in the SEPP - Waters of Victoria, Schedule B shall not apply to the waters contained within a 600 metre radius from the discharge point.

c) The chronic toxicant objective for the coastal segment, as stated in the SEPP - Waters of Victoria, Schedule B shall not apply to the waters contained within a 200 metre radius from the discharge point.

2.3.3 Water Quality Indicators and Objectives

Table 2.2 provides water quality indicators and objectives for coastal waters as set out in Schedule B of the State Environment Protection Policy for Waters of Victoria. Note that the discharge licence allows for variation to some of the parameters as discussed in the section on mixing zones.

Figure 2.2. Licenced mixing zones for Boags Rocks discharge

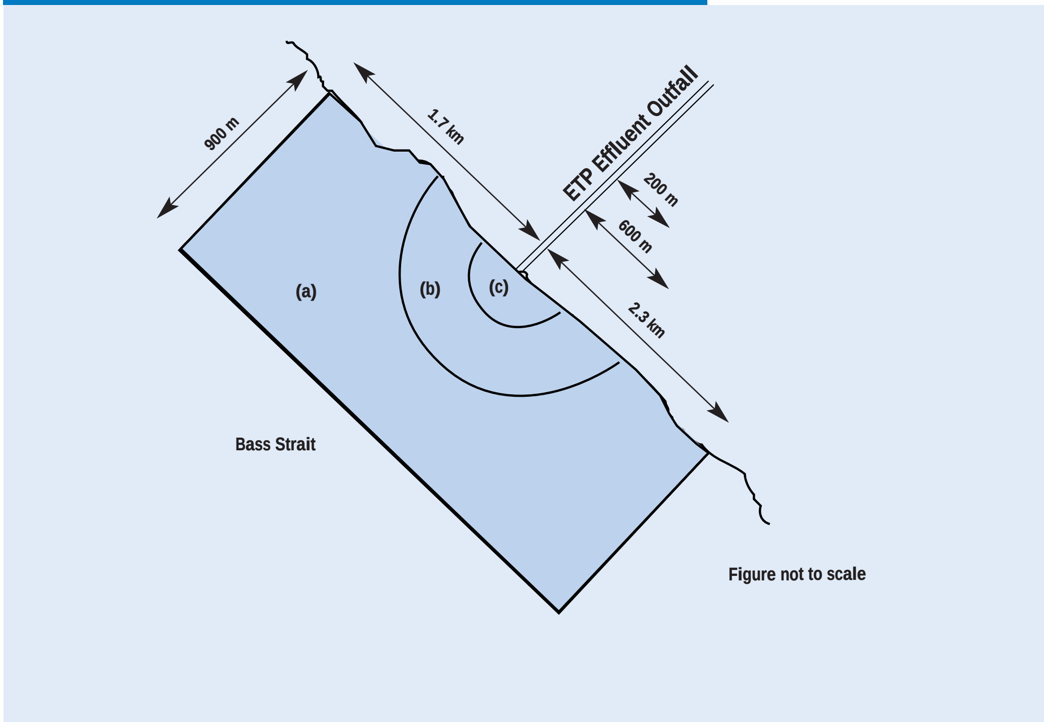


Table 2.2: Water quality indicators and objectives for coastal waters

Dissolved Oxygen	Dissolved Oxygen concentrations shall be sufficient to maintain beneficial uses and shall at all times be higher than 6.5 g/m^3, 85% saturation.
Bacteria *(E.coli).*	Within the area bounded by the high water mark and a line 300m offshore, the number of *E.Coli* shall at all times be less than 200 orgs/100mL (geometric mean based on not less than 5 samples taken over a period of not more than 42 days) and 400 orgs/100mL (80th percentile).
pH	Variation from seasonal background value shall at all times not be more than 0.5, and it should be within the range 7.5 to 8.5.
Water Temperature	Variation from background value shall not affect beneficial uses and shall not exceed ± 0.5˚C.
Light penetration	There shall be no reduction in light penetration to the detriment of beneficial uses.
Toxicants	Water shall be free of substances in concentrations which either individually or in combination, produce toxic effects or genetic damage to plants, animals, aquatic life or humans, as these relate to the beneficial uses. For the protection of ecosystems the concentrations of toxicants shall not exceed N + 0.5(T - N) where T is the threshold concentrations of chronic sublethal effects on aquatic life, and N is the natural background level of the toxicant. T may be obtained from Tables 14 and 15 of “Recommended Water Quality Criteria” EPA, Victoria 1983 (RWQC). For the protection of human health the concentrations of toxicants in water shall not exceed levels where bioaccumulation would cause fish, shellfish or crustacea to lose acceptability on commercial markets or under the food standards established by the Health Commission of Victoria, or exceed values given in Table 10a the RWQC.
Nutrients and Biostimulants	Waters shall be free of substances in concentration, which cause nuisance plant growth or changes in species composition to the detriment of the protected beneficial uses.
Total Dissolved Solids (TDS)	The level of TDS shall not vary from background levels by more than 5%. (ie. background salinity is 35, so objective is to achieve salinity of more than 33.25)
Suspended Solids	Suspended Solids levels shall not exceed the following levels in receiving waters: a) for 50th percentile 10g/m^3 b) for 90th percentile 25g/m^3
Aesthetic Characteristics	There shall be no objectionable colours or odours in waters, or objectionable taints in edible aquatic organisms. There shall be no visible floating foam, oil, grease, scum, litter or other objectionable matter. The concentration of chemical compounds found to cause the tainting of aquatic organisms shall not exceed those specified in Table 11 of the RWQC.
Settleable Matter	The level of settleable matter shall not result in deposits which adversely affect the recreation and ecosystem values of the surface waters as expressed in the beneficial uses.

2.4 Discharge Volume and Quality

The average annual volume discharged from ETP is about 136,000 ML (373 ML / day) of treated effluent (Table 2.3). The plant has a large storage buffer capacity for wet weather flows, which results in a more consistent output volume, compared to the significant variation in the volume of flows to the plant. During periods of wet weather flow to the plant increases dramatically, with maximum volumes reaching five times the average dry weather flow.

The design of the treatment plant and its operation enable a consistent quality of treated effluent to be produced. The wastewater flowing to the plant varies in quality as well as quantity. The variation in quality is evident in Fig 2.3, which shows the Biochemical Oxygen Demand (BOD) of untreated sewage in comparison to the BOD of the treated effluent. Fig 2.4 compares the level of suspended solids in the untreated sewage with that of treated effluent.

Graphing of fortnightly sampling results for various forms of nitrogen (Fig 2.5) provides a visual indication of the effluent quality. The existing licence states that nitrogen as ammonia is not to exceed a maximum of 40 mg / L, and the annual median should not exceed 30 mg / L. It should be noted that ammonia is the largest form of nitrogen in the effluent, and the impact of this is discussed later in the report.

Table 2.3. Annual average daily flows of treated effluent from ETP

Year	Average (ML/day)	Range (min - max) (ML/day)	Annual volume (ML)
1993	369	121 - 683	134,685
1994	359	233 - 657	131,035
1995	395	95 - 682	144,175
1996	399	86 - 665	145,635
1997	343	181 - 492	125,195
Average	373		136,145

Figure 2.3. BOD of the untreated sewage (higher line) and BOD of the treated effluent (lower line) for ETP.

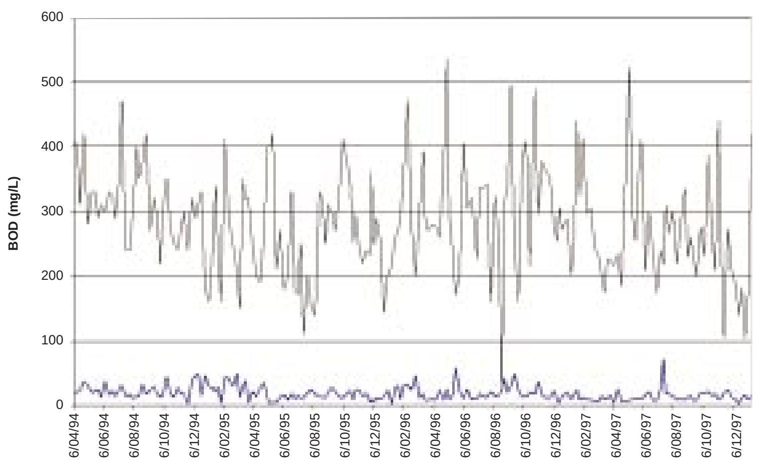


Figure 2.4. Suspended solids in the untreated sewage (higher line) and in the treated effluent (lower line) for ETP.

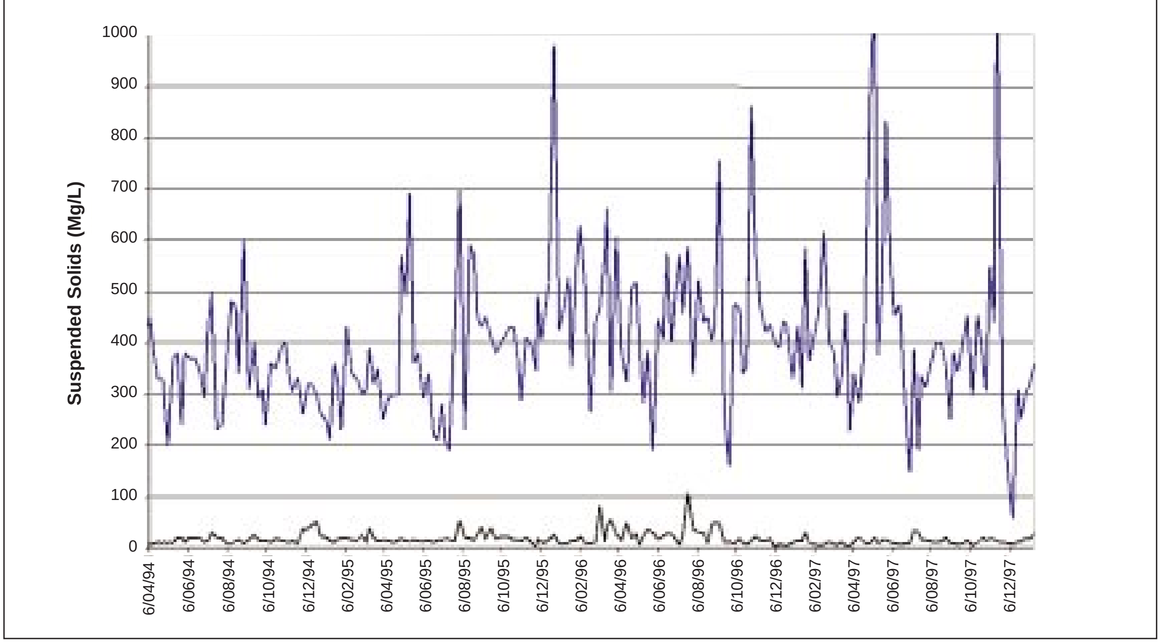
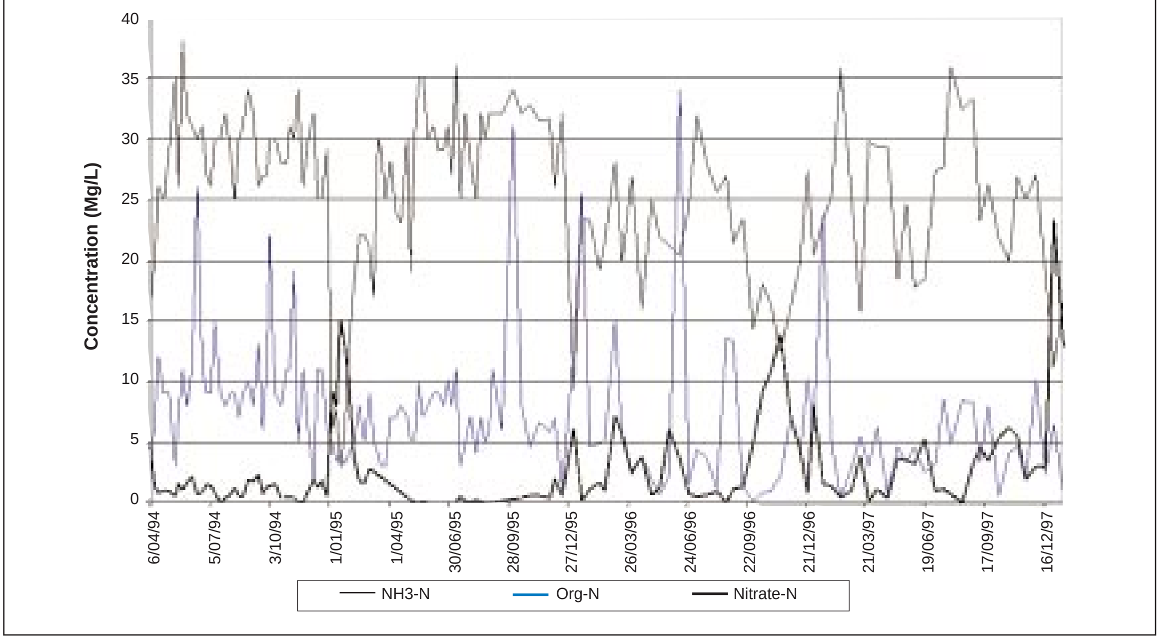


Figure 2.5. Nitrogen in the form of ammonia (higher line), organic nitrogen (middle line) and nitrate (lowest line) from fortnightly samples of treated ETP effluent.



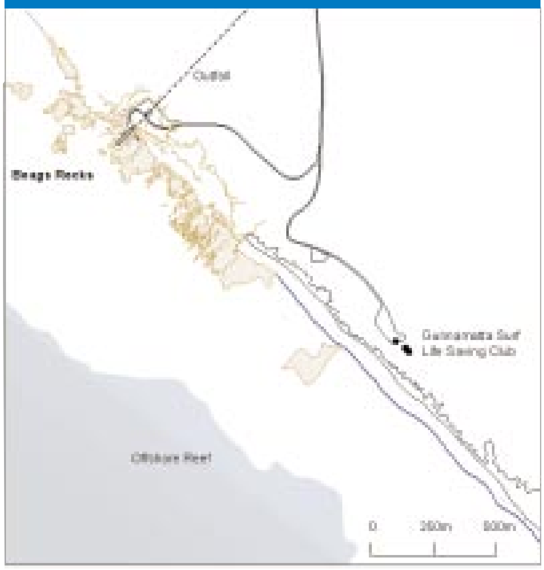
2.5 Site Description and Physical Environment

The following information, covering the site description and physical environment was extracted from the review of the Boags Rock Outfall conducted by D.A. Lord and Associates P/L in 1996 (Lord 1996) and has been supplemented by information collected during this study for the hydrodynamic modelling.

2.5.1 Boags Rocks

The area called Boags Rocks is part of a shoreline rock platform that lies between St. Andrews and Gunnamatta Beaches (Fig 2.6). The platform is composed of friable Pleistocene aeolianitic limestone. The 2.75 m diameter outfall pipeline discharges at the seaward edge of the platform, beneath the low tide mark.

Figure 2.6. Map of Boags Rocks and Gunnamatta Beach



Offshore from the platform the seabed gradient to a depth of 18 m is approximately 0.01 to 0.018. The surf zone adjacent to Boags Rocks is composed of bars and troughs interspersed with small limestone reef outcrops. Beyond about 500 m offshore, reefs have a considerably higher profile than the reef outcrops in the surf zone.

Broad stretches of sandy shoreline, interspersed with reef, extend either side of Boags Rocks. To the northwest the aeolianitic limestone extends as disjunct intertidal reef platform and is exposed at the shoreline in several places. This coastline exhibits an intermediate longshore-bar-trough morphology during moderate surf conditions with a width of 300 to 400 m. Rips generally occur on both sides of the outfall with an approximate spacing of 300 m. During large surf conditions the rip currents disappear and the surf zone becomes flat and dissipative with a width in excess of 400 m.

Sediment movement in the surf zone is very dynamic due to lateral shifting of the rip channels and onshore/offshore movement of the bars.

The coast adjacent to Boags Rocks is backed by a relatively stable parabolic dune system (with occasional blowouts) that overlay sections of the Pleistocene aeolianite.

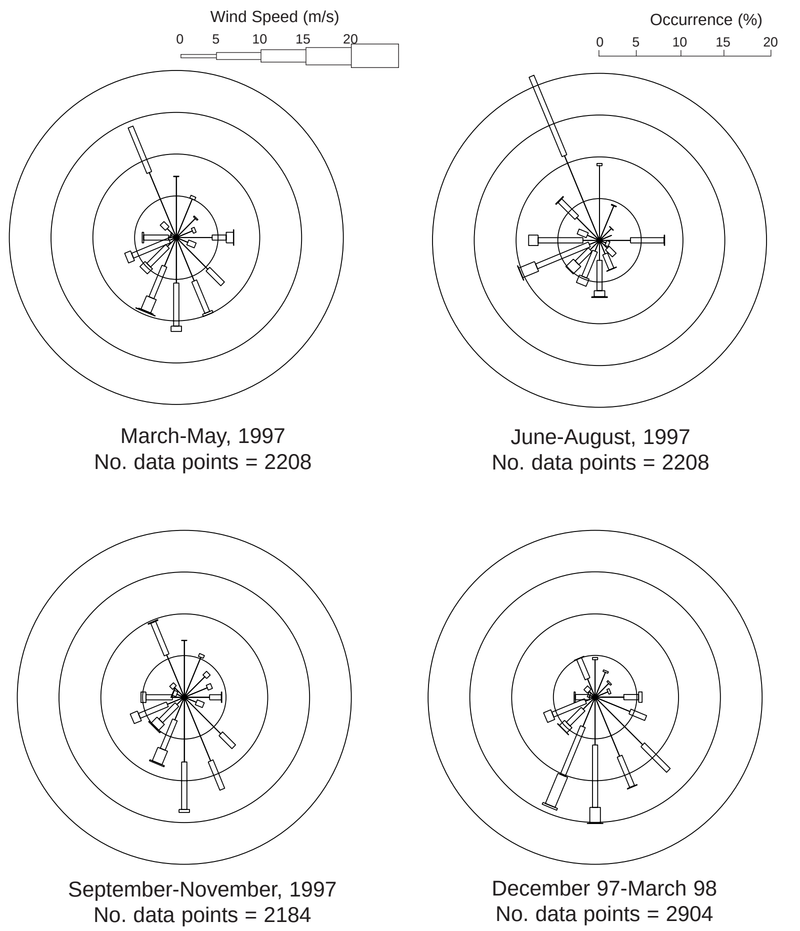
2.5.2 Winds

Sokolov and Black (1994) reported that there is a pronounced seasonal variation in wind patterns. During winter, the predominant wind is from the northwest with a maximum mean wind velocity of 7 m/s. In summer, southwest and easterly winds prevail with a maximum mean wind velocity of 6 m/s. Northwest and southwesterly winds prevail during spring and autumn, respectively.

During the Study, winds were measured at the Gunnamatta Surf Life Saving Club. A sensor was mounted on a mast located on the roof of the clubhouse. Analysis of the winds over four periods; March to May, June to August, September to November and December 1997 to March 1998, showed a seasonal pattern similar to that described by Sokolov and Black. Wind roses for the four modelling periods are shown in Fig 2.7.

Over the whole year the most prevalent single wind direction was from the north of northwest. Speeds were in the range 0 to 10 m/s and blew for 12% of the time. The strongest winds (10 to 20 m/s) were the onshore winds from the southwestern quadrant (5% of the time). Lighter winds from this quadrant (0 to 10 m/s) blew for approximately 18% of the time. Offshore winds (from the northeast) were both infrequent and very weak, blowing approximately 8% of the time with speeds mostly below 5 m/s.

Figure 2.7. Telescope plots of wind data measured at Gunnamatta Surf Life Saving Club for each of the modelling periods. The telescopes point towards the directions from which the wind is blowing, the width of the scope indicates wind speed and the length indicates percentage of time they occur (Andrewartha *et al.* 1998).



2.5.3 Tides

Boags Rocks experiences semi-diurnal tides with a significant diurnal inequality. The dominant tidal harmonic constant is the M2 (lunar semi-diurnal) component, which has a maximum range of 1.5 m at Boags Rocks. The tides at Boags Rocks are reasonably similar to those measured at Lorne approximately 80 km to the west, or Flinders Jetty, around Cape Schanck to the east. Although Point Lonsdale is also close (28 km to the northwest), tides at that site are substantially altered by the dynamics of the strong tidal flows through the Rip (at the entrance to Port Philip Bay).

2.5.4 Waves

Waves have been monitored at Boags Rocks (August 1971 to April 1972) and at Flinders (November 1990 to February 1991) (Sokolov and Black 1994). The maximum wave height recorded offshore from Boags Rocks was 7.16 m and the wave periods varied between 10 and 15 seconds. The median significant wave height recorded at Flinders was 1.3 m. The significant wave height exceeded 2 m for 10% of the time and remained below 0.7 m for 10% of the time. Directional data was not available from either of these stations. Due to the orientation of the shoreline, waves that can directly impinge on Boags Rocks must have a directional origin between 190° and 250°.

Wave orbital velocities were observed during this study at a site about 1 km offshore from the outfall. Analysis of these data over the period March to November 1997 gave mean (and median) wave height of 1.5 m, period of 11.2 seconds and direction of propagation towards 60°, consistent with the above results (Andrewartha *et al.* 1998).

2.5.5 Currents

The important currents at Boags Rocks include nearshore currents, tidal currents, coastal-trapped waves, wind-driven currents and thermohaline currents (Sokolov and Black 1994). Nearshore currents (inside the surf zone) are particularly important in controlling water movement in the area immediately adjacent to the existing outfall. These include longshore currents driven by obliquely incident waves, which vary in strength and direction with variation in wave approach. The other nearshore currents of importance are the rip currents, which operate to discharge water from the surf zone and may reach velocities up to 1 m/s.

Further offshore, measurements obtained during the Study showed that flows are dominated by the tides, with mainly long-shore oscillatory motions of about 20 cm/s. However lower frequency flows tend to determine the ultimate path taken by the effluent. These are also predominantly long-shore, with speeds rarely exceeding 10 cm/s (Andrewartha *et al.* 1998). Low frequency flows are significantly correlated both with low-pass filtered winds and low-pass filtered sea-level observed at Lorne. Sokolov and Black (1994) reported that mean flows at Boags Rocks (over a 35 day period) were 11 cm/s towards the southeast. However results from this Study indicated generally lower mean flows of 1 to 4 cm/s over the period March 1997 to March 1998 (the values varying with measurement site and time).

Surface drogue measurements taken offshore from Boags Rocks as part of earlier studies showed that the surface water movement is significantly influenced by the winds. It can occur in both directions parallel to the shore, though southeasterly currents are dominant giving a net longshore movement in that direction (Sokolov and Black 1994). The surface drogue measurements indicated that the net cross-shore movement of water associated with the turn of the tide could be up to 600 m and that the current velocity increased with distance offshore.

3 Research

3.1 Biological Monitoring

The objectives of the biological monitoring were to assess the extent of impact of the treated effluent on the rocky and sandy biological assemblages and to determine whether the extent of the impact was increasing or decreasing.

A standard approach to biological monitoring is to compare impacted sites with control sites that are of similar characteristics and are assumed to be unimpacted (Underwood 1989). An alternative approach is to make an assessment of the change at a specific site over time. Ideally these surveys commence prior to impact occurring.

Along this coast the high level of natural variability due to differing substrata and ocean dynamics means suitable control sites could not be identified for comparative purposes. The other confounding factor for Boags Rocks is that monitoring undertaken prior to commissioning of the outfall (Manning 1979) was insufficient to allow reliable 'before and after' type comparisons to be made.

These issues mean that assessing the extent of impact and determining whether it is increasing or decreasing is a difficult objective to meet. Both past and present results have produced ambiguous results and only the impact in the immediate vicinity of the outfall is irrefutable.

Broad stretches of sandy beaches, interspersed with reef, extend either side of Boags Rocks.

3.1.1 Intertidal Rocky Platforms

Considerable effort has been expended since 1975 in surveying the rocky platforms along the coast between Point Nepean and Cape Schanck. Manning (1979) undertook surveys at four sites - one at the outfall site itself, two at 700 m (Boags Rocks East) and 5 km (Fingals Beach) southeast respectively and one situated 16 km to the northwest (Sorrento). Two surveys were conducted before discharge began and six afterwards, from July 1975, to August 1976. Manning commented that before discharge the four sites displayed differences in algal and mollusc species composition with "considerable differences in the relative density of many common mollusc species". After discharge commenced, the most noticeable effect was a decline in abundance of brown algae at the outfall site with some attendant changes in other taxa. Manning also reported similar effects at Boags Rocks East.

From 1980 to 1994 numerous other surveys were conducted at a total of eleven sites, including some of those of Manning, together with supplementary sites. All were on the accessible rocky platforms. Seven reports were prepared, elaborating and refining the early findings of Manning.

Quinn and Haynes (1996) reviewed all the published material in a report to Melbourne Water. They posed the question "is the biota, including measures of temporal change, at the Boags Rocks sites within or outside the range of natural variation in biota between unimpacted sites?" They concluded that the statistical tests applied by the various workers were either inadequate or inappropriate to answer this question with the data available.

They also commented that Manning's pre-discharge surveys were too limited to overcome the marked temporal changes now known to occur in most taxa, especially *Hormosira banksii.* Their overall conclusion was that rocky platforms do not provide a useful substrate for assessing the extent of biological effects of the outfall even on the basis of comparing the outfall site with supposed "control" sites. They tested this latter conclusion by applying univariate analysis of variance (ANOVA) and non-metric multidimensional scaling (NMDS) to the data collected by Melbourne Water in 1993-4.

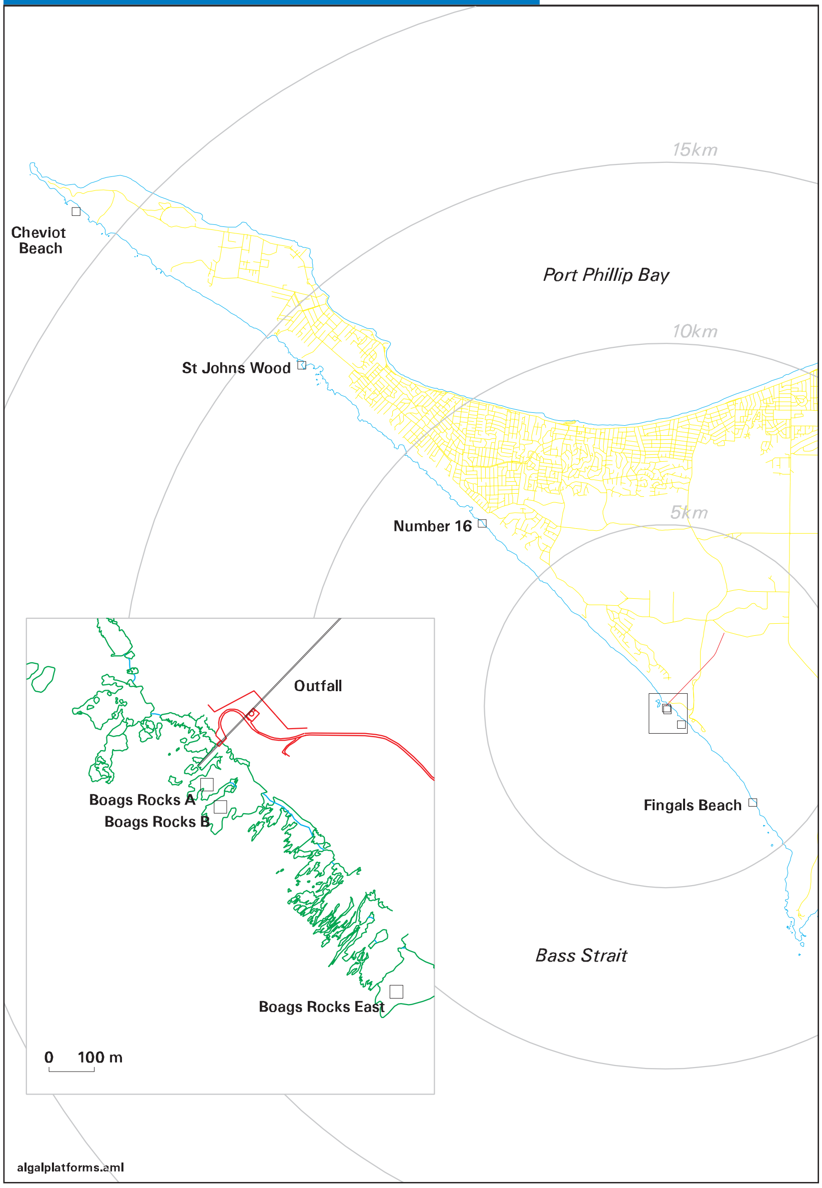
The ANOVA analysis showed only that the appearance of one opportunistic alga *(Ulva rigida)* could be considered to be due to effects from the outfall. The NMDS plots showed some evidence of separation of the Boags Rocks site from others on the basis of macroalgal percentage cover and faunal abundance but the distant control sites were also different from intermediate sites. Quinn and Haynes attribute differences mainly to changes in abundance of common taxa rather than presence / absence.

Quinn and Haynes recommended that more sophisticated statistical analysis be applied to the 20 year collection of data in the hope that significant patterns might emerge. All data and reports were therefore sent to CSIRO Mathematical and Information Sciences, Envirometrics Project. The resulting report (Shao 1997) reiterates many of Quinn and Haynes reservations about deficiencies in methodology. Based on a Correspondence Analysis (Greenacre 1984) of four macroalgae and nine invertebrates Shao concluded that:

- The largest differences are between Boags Rocks and the site at Number Sixteen (7km to the northwest).
- Boags Rocks East and Fingals Beach can be grouped together for some variables suggesting that they are quite similar in terms of time profiles
- Boags Rocks is more similar to Boags Rocks East than Fingals Beach.
- The similarity between Boags Rocks East and Fingals Beach is comparable with that between Fingals Beach and Number Sixteen but the latter two are clearly different.

The net result of these investigations suggests that there are effects of the effluent at the outfall site, but that longshore and temporal differences confound attempts to assess the extent of this effect or temporal trends on intertidal rocky platforms either side of the outfall. One solution would be closer interval sampling along the coast but sufficient suitable sites for sampling do not exist. Of course the existing data would provide an excellent "before" baseline if the volume or quality of the effluent were significantly changed or the outfall were to be extended.

Figure 3.1. Location of the seven sites used for seasonal algal surveys (Kevekordes 1998a)



It is generally acknowledged that at the outfall site important habitat forming brown algae (including *Hormosira banksii* and *Durvillea potatorum)* and some red algae have disappeared, together with their associated faunal communities. Their place has been taken by green algal turfs and high densities of a few species of limpets and gastropods. Furthermore, a tube-building spionid polychaete, *Boccardia proboscidea,* has colonised the rocks. Anecdotal evidence suggests that the above two brown algae have diminished in abundance or become absent on platforms to the east of the outfall (DKO Services 1998).

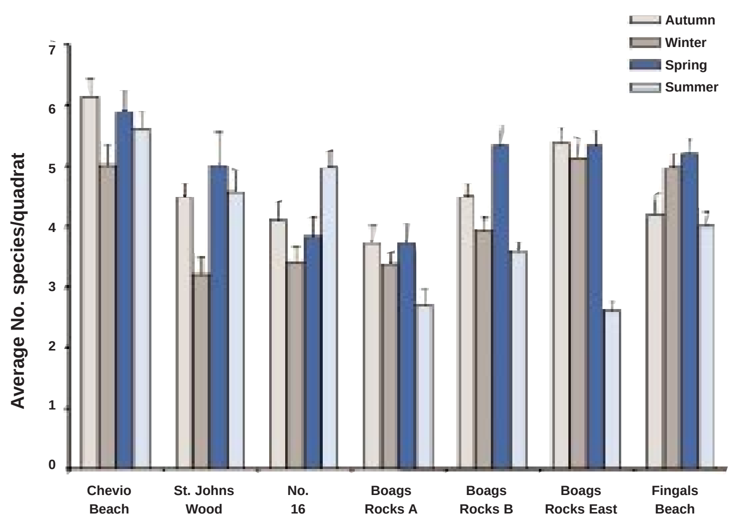
The confounding of effluent effects and natural variation has also been noted during rocky platform surveys of macroalgae at seven sites (Fig. 3.1) conducted by Kevekordes (1998a) as part of the present Study. Thirty 0.25 m^2 quadrats were haphazardly chosen within a 20 by 20 m area adjacent to the lower edge of the intertidal rock platform. The taxa within each quadrat were identified and their percentage cover measured. The surveys were conducted in autumn, winter and spring, 1997, and summer, 1998.

The results for average number of species per quadrat over the year are shown in Fig. 3.2. It can be seen that there is a diminution in species number at the outfall with recovery either side, although the differences are small.

The highest number of species overall was found to the northwest, with 21 species at St. Johns Wood and Number Sixteen, and 20 species at Cheviot Beach. Only 9 species occurred at Boags Rocks 'A' and 12 to 16 at the eastern sites.

Of course, the species present are not the same at all sites and Fig. 3.3 illustrates the distribution of six common macroalgal species averaged over the four surveys. The distribution varied slightly with season but the same pattern prevailed. *Corallina officinales* and *Laurencia filiformis* tended to be confined to the northwest whereas other species occurred mainly in the east. Note that the percentage cover differs markedly between species. Algal cover at the outfall sites is generally low but *Capreolia implexa, Ulva rigida, Cladophora subimplex* and *Ceramium flaccidum* predominate.

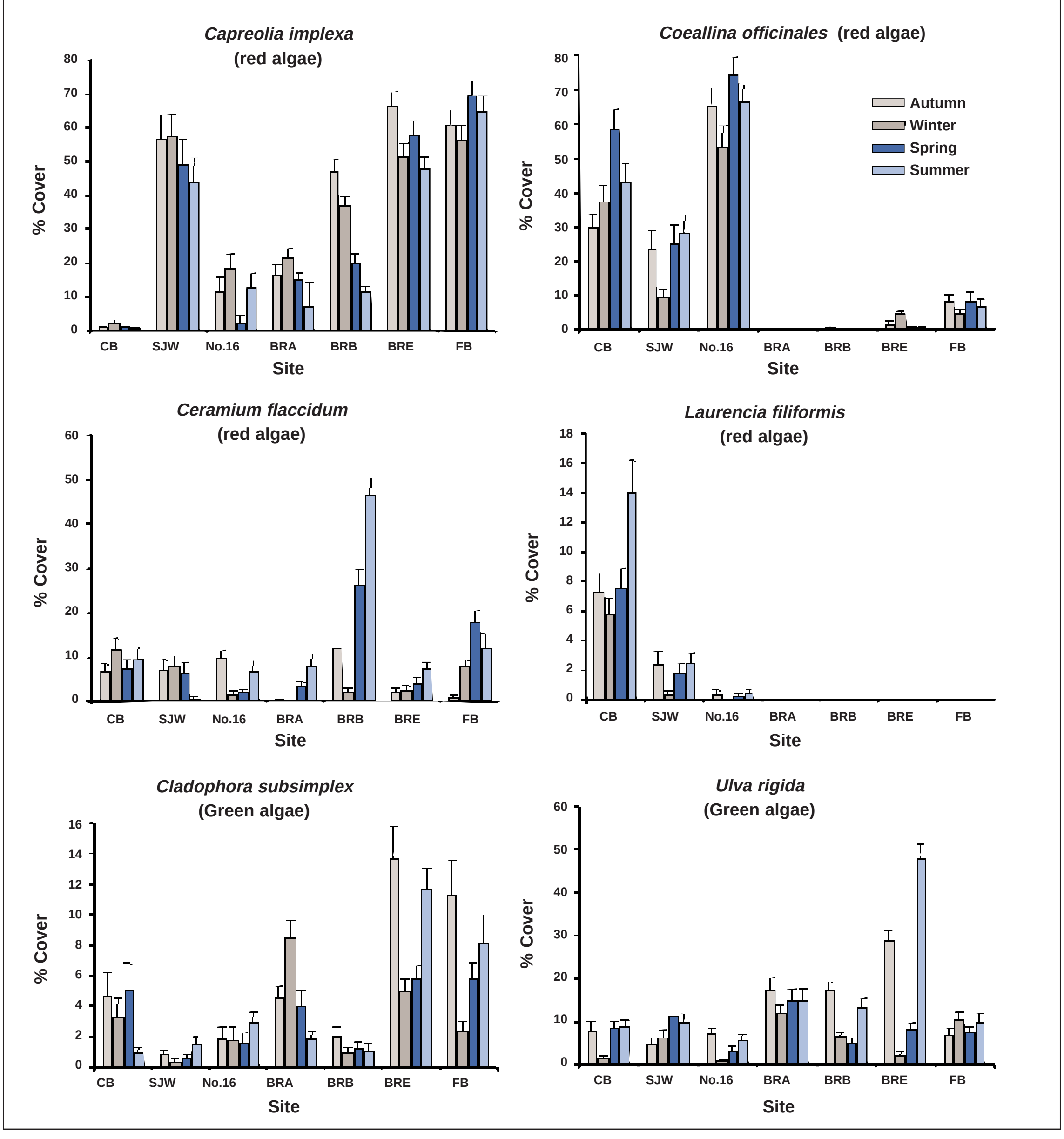
Figure 3.2. Average number and standard error (vertical line) of macroalgal species per quadrat from the four seasonal surveys. Sites listed from northwest to southeast.



Previous work has indicated that *Corallina officinales* and *Laurencia filiformis* tend to be found in low nutrient conditions, while *Ulva rigida* is typical of high nutrient conditions. This makes it more likely that the observed longshore distribution may reflect an effect of effluent nutrients on algal composition.

A comparison of algal assemblages at each site in each season using non-numerical multi dimensional scaling (NMDS) showed no

Figure 3.3. Average abundance (% cover) of common macroalgal species from the four seasonal surveys. Sites listed from northwest to southeast: Cheviot Beach (CB), St. Johns Wood (SJW), Number Sixteen Beach (No.16), 2 sites at Boags Rocks (BRA and BRB), Boags Rocks East (BRE), Fingals Beach (FB).



grouping or gradient, all sites being different from each other. The same pattern emerges as in previous work, namely, that of a definite signal of selection of species and poor cover at the outfall site against a complex natural longitudinal variation.

3.1.2 Intertidal Beach Sediments

Both Quinn and Haynes (1996) and Shao (1997) recommended alternative sampling methods. One such method was to examine intertidal beach sediments between the rocky platforms to provide a more continuous regular profile. To this end, the Museum of Victoria were engaged to sample intertidal beach sediments and report on the feasibility of using macrofauna to assess the extent of effluent impact (Heisler *et al.* 1996).

The Museum scientists collected five cores (150 mm diameter) to a depth of 150 mm at low tide and mid-tide level at four beaches (40 total). The beaches were situated 15 km (Sorrento), 4 km (Rye) and 50 m northwest of the outfall and 250 m southeast. The cores were washed through a 1 mm sieve and the fauna collected for examination.

Very few individuals or species were found. Only one isopod, three amphipod and three polychaete species occurred and these in very low numbers. Distribution was also very irregular with some cores yielding no organisms at all and the highest count was only five. As excessive quantities of sediment would be required to obtain enough organisms to assess potential effluent impacts on infauna, this method of assessment was considered inappropriate.

3.1.3 Subtidal Reefs

Three workshops were convened through 1996-97 to consider other alternative sampling regimes. These meetings were attended by various representatives of CSIRO, Melbourne Water, the EPA and Monash University, some of whom also held further discussions on special aspects of the monitoring problem. There was general consensus that intertidal rocky platform sampling was not suitable for establishing the extent of impact and that offshore subtidal surveys be attempted.

Consulting Environment Engineers P/L (CEE) were engaged to conduct a reconnaissance survey along the reef situated offshore from the coast and along the stretch of sand between the shore and the offshore reef.

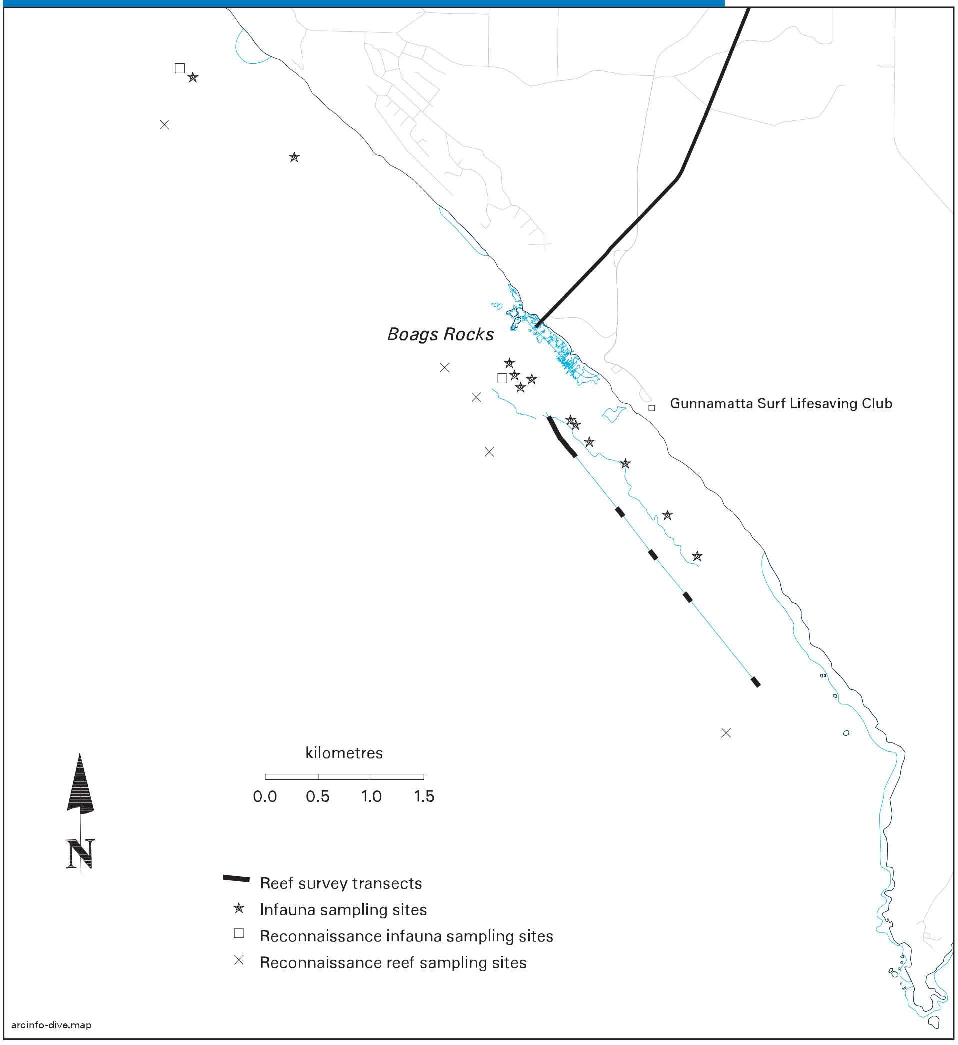
The limestone reef commences about 800 m from shore although the boundary is irregular and closer to shore in the east. The reef is of low relief with numerous fissures and has areas covered with sand. The reef reconnaissance used four sites situated at 4 km northwest of the outfall, off the outfall and 500 m and 4 km southeast of the outfall (Fig. 3.4). Distances from shore varied slightly but all sites were in 18 m depth of water. At each site a 100 m transect was laid down and a video record taken (video held by CEE). Organisms were identified and counted or cover estimated in 0.9 m^2 quadrats of which five were placed haphazardly at each 20 m interval along the transect.

The reef communities were characterised by the kelp *Ecklonia radiata*, which formed a patchy forest. Smaller red and green algal species covered most of the remaining reef, with a variety of fixed animal species also present including sea squirts (cunjevoi), sponges, seastars, bryozoans and molluscs such as periwinkles and abalone. It was concluded that the reef communities were moderately diverse with generally similar assemblages in both directions. With the limited data provided from the reconnaissance survey, it was not possible to determine whether the differences seen were an effluent effect or not. However more detailed sampling and analysis of reef community structure and abundance of common species may provide an indication of the effect of the effluent discharge.

A more substantial reef survey was carried out in January 1998 (Chidgey *et al.* 1998). Given that the objective was to determine the extent of impact, it was agreed that the main concern of the survey was to document biological changes as a function of distance from the outfall. As the reconnaissance had suggested possible effects to the southeast of the outfall, which coincides with knowledge that the effluent plume has a net drift in that direction, it was decided to repeat the transect technique, but closer to shore and to the southeast only (Fig. 3.4). The intervals were 5 x 100 m from 900 to 1400 m and 1 x 100 m at 2000, 2500, 3000 and 4000 m.

The cover of the dominant alga, *Ecklonia*, and epifauna was estimated by counting in 4 x 0.5 m strip quadrats along the transects giving a total of 195 quadrat counts. The algal understorey was examined by stripping all *Ecklonia* from a 0.2 m^2 area and photographing the area with a frame mounted camera (slides held by CEE). This was done at 6 m intervals along each transect.

Figure 3.4. Location of subtidal survey sites. The inner line of the offshore reef was plotted from aerial photographs where visible.



As in the reconnaissance, the reef was found to support a rich and diverse biological assemblage. *Ecklonia radiata* was the dominant macroalga, with its understorey space being inhabitated by smaller algae and numerous reef animals. The predominant animals included abalone, rock lobsters, seastars and molluscs. The fish included wrasse, hula fish, leatherjackets, magpie perch, sweep, boarfish and old wife. The most common benthic animals were filter feeding solitary ascidians (sea squirts) with *Herdmania* and *Cnemidocarpa* the most abundant.

The composition of the understorey is shown in Table 3.1 and the principal invertebrates present are shown in Table 3.2, which does not include the numerous fish species passing across the quadrats.

Table 3.1. Summary of abundances of Ecklonia and understorey plants and animals for 100 m transects. Abundances for *Ecklonia* are mean number per 2 m^2. Abundances for understorey algae and animals are mean substratum cover (%).

	Transect Section								
	900	**1000**	**1100**	**1200**	**1300**	**2000**	**2500**	**3000**	**4000**
Ecklonia radiata	11.8	8.4	11.8	16.0	13.4	12.8	6.4	6.1	31.1
Foliose reds	69.8	70.1	60.7	54.6	74.1	55.2	65.6	58.9	34.0
Turfing reds	1.9	3.0	8.6	8.1	7.5	6.7	7.2	7.2	6.3
Encrusting coralline	3.8	2.2	5.7	12.9	2.9	6.9	3.3	7.6	32.3
Erect coralline	1.7	2.6	0.9	4.1	3.0	2.6	1.0	0.7	1.6
Melanthalia	1.1	0.1	4.2	1.3	1.9	1.5			
Sonderopelta				0.6	0.5	0.6	0.4	4.9	0.5
Encrusting reds		0.6		0.2					
Blade reds		0.2	0.1	0.1	0.4		0.3		
Ecklonia (juv.)	5.3	4.8	5.7	3.4	3.2	3.6	5.4	1.4	13.4
Dictyotales	0.5	0.6	1.5			0.9	2.8	7.3	7.2
Carpoglossum						0.3			1.2
Other browns						0.6	7.2		2.8
Caulerpa brownii	4.7	6.2	2.7	2.7	0.3	5.9	0.8	8.5	
C. cactoides	0.2	1.1	0.2	0.1				0.1	
C. obscura	1.1	4.4	1.0	1.5	1.3	0.0	1.4		
Herdmania	0.9	0.2	0.5		0.3	0.4		0.2	
Cnemidocarpa	0.1	0.1	0.2					0.1	

Table 3.2. Summary of abundances of animals counted within 4 x 0.5 m strip quadrats for 100m transects. Abundances are mean numbers of organisms per 10 m^2

	Transect Section								
	900	1000	1100	1200	1300	2000	2500	3000	4000
Asidians									
Cnemidocarpa	18.1	14.0	9.8	9.8	5.6	8.1	1.4	10.0	1.0
Herdmania	8.8	45.8	12.6	16.2	31.6	48.6	7.9	16.6	1.6
Pyura australis	0.8	1.8	3.6	1.0	0.8	0.8	0.4		0.4
Other ascidians		0.8	1.0	0.4		0.3	0.4		
Abalone									
Haliotis rubra						0.3			1.4
Seastars									
Tosia australis	0.4	0.4							
Nectria ocellata					0.2			0.2	
N. macrobrachia		0.6			0.2	0.3		0.2	
Fromia polypora			0.4	0.2					
Uniophora		0.2							
Echinaster						0.3			

The abundances of the plants and animals measured along the transect showed a high degree of variation. This is evident in the plots of results for the large brown kelp *Ecklonia* and for the ascidian *Herdmania* (Figs. 3.5 and 3.6). For most species examined, patterns and trends in abundances were evident at a range of spatial scales. To examine patterns and trends at larger spatial scales, the 'noise' of smaller scale variation was smoothed using lowess (robust locally weighted regression). This method can be effective at elucidating patterns from within very noisy data, without underlying assumptions about the distribution and variance of the data (Ellison 1993).

Most of the transect was covered by a canopy of the large brown kelp *Ecklonia*, with densities mostly above 10 plants per 4 m^2. However, density was patchy, with large changes in density occurring over short distances. The average density of plants appeared to be reduced between 2500 m and 3100 m. Densities were consistently higher from 4000 m, with a minimum density of 36 plants per 4 m^2.

The ascidian *Herdmania* tended to occur in discrete clumps along the transect, with the density and spatial extent of the clumps being quite variable. Exceptionally high densities were present at 1076, 1084 and 2096 m from the outlet, with over 50 individuals per 4 m^2. The clumps generally had 10-20 individuals per 4 m^2 and extended 8 to 40 m along the transect. Average densities appeared to be lower at 900-1000 m, 2500-2600 m and 4000-4100 m.

Figure 3.5. Number of *Ecklonia* plants per 4m²

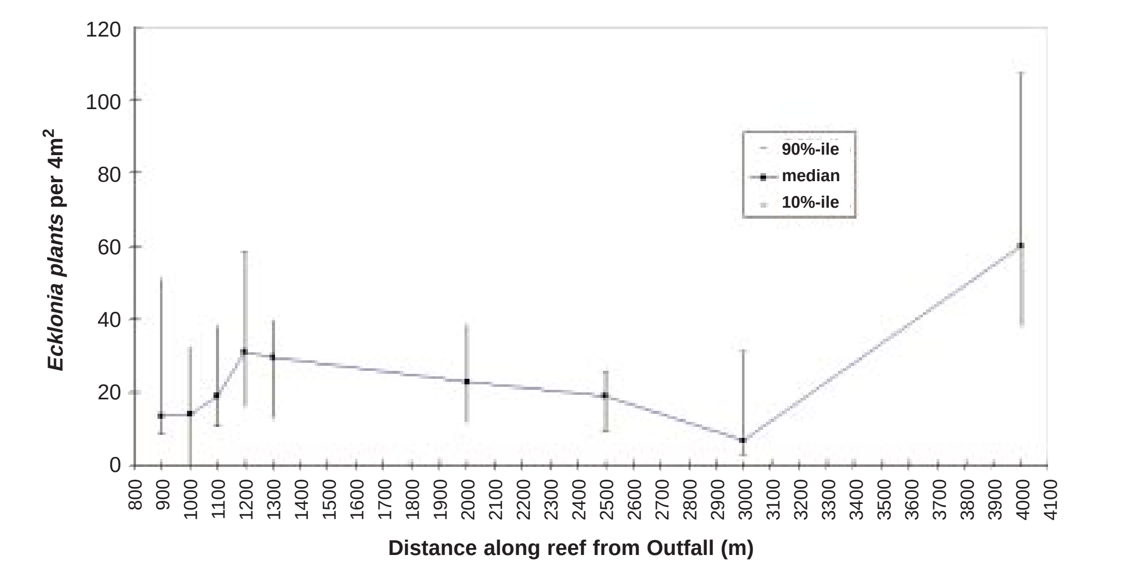
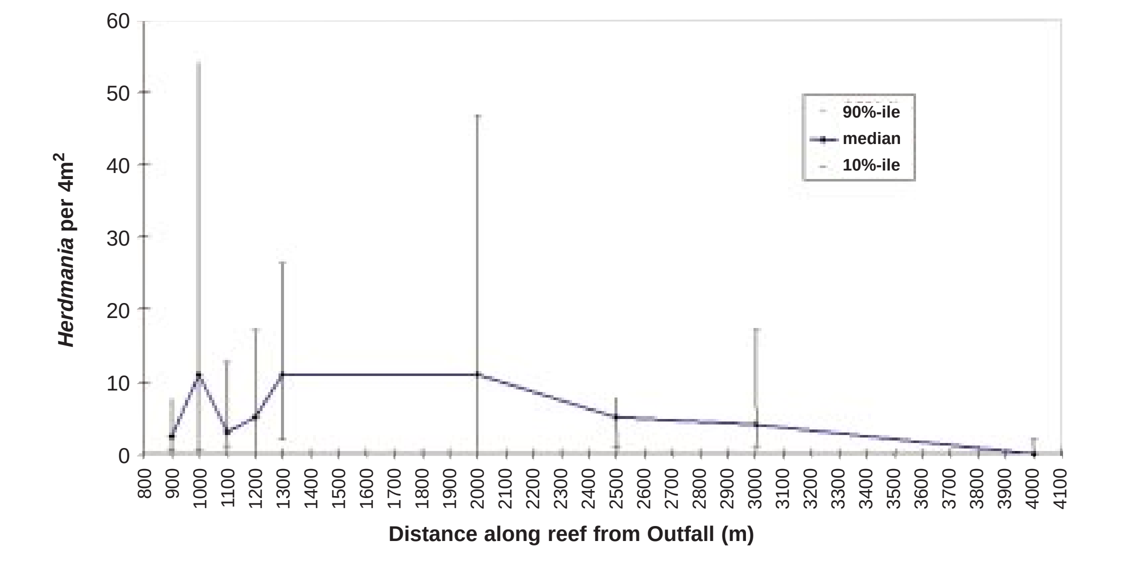


Figure 3.6. Number of *Herdmania* per 4m²



In addition an analysis of community structure was made. The 15 understorey algal groups were subjected to Bray-Curtis dissimilarity index calculation followed by non-metric multidimensional scaling and calculation of Kruskal stress. This analysis showed a clear separation of communities, that is a difference in floral composition - along the survey track from 900 to 3000 m.

If the effluent discharge has a biological impact, it is envisaged that the effects could also lead to spatial gradients and discontinuities in populations near the outfall. This was tested by use of spatial autocorrelation analysis (Mantel cross products tests and Mantel r correlograms) using the densities of *Ecklonia*, *Cnemidocarpa* and *Herdmania*, the percent cover of foliose red algae, turfing algae and encrusting coralline algae and the understorey algal community structure. The results confirmed the trends for *Ecklonia*, *Herdmania* and the foliose red algae. It also disclosed an increasing density away from the outfall for encrusting coralline algae and confirmed the gradual changes in understorey community structure.

It should be noted that while the observed trends would seem to show a gradient away from the outfall this may reflect natural factors of reef substrate and consequent competition. However, the following general conclusions may be drawn (Chidgey *et al.* 1998).

There are several spatial patterns in the distribution of biota on the reefs offshore from Boags Rocks to Cape Schanck. These patterns include:

- Changes over tens of metres due to biological patchiness and interaction, and habitat variation.
- Changes over hundreds of metres possibly due to effluent effects, biological interactions and habitat variation.
- Changes over kilometres possibly due to effluent effects and physical regional boundaries such as changes in reef topography and wave climate.

To capture these spatial patterns, four zones are proposed:

- **First Biological Zone < 1100 m from the outfall.** This zone had relatively low abundance of turfing red algae and encrusting coralline algae, low to medium abundance of *Ecklonia* kelp and *Herdmania* sea squirts, medium abundance of foliose red algae, and relatively high abundance of *Cnemidocarpa* sea squirts. It is considered that effluent is a factor affecting the biological characteristics in this zone.
- **Second Zone 1100 - 1400 m from the outfall.** This zone had high abundance of turfing red algae; erect coralline algae, *Ecklonia*, *Herdmania* and foliose red algae; and comparatively low coverage of encrusting coralline algae. It is considered that this zone is affected, but less so than the first zone.
- **Third Zone 2000 - 3100 m.** This zone had medium abundance of *Ecklonia*, turfing red algae, *Cnemidocarpa*, *Herdmania* and erect coralline and turfing red algae. It is considered that any effect of effluent within this zone would be largely indistinguishable from other factors affecting the biological variation documented at sites within this zone.
- **Fourth Zone > 4000 m.** An abrupt change in community structure from 3100 to 4000 m southeast, particularly an increase in abundance of *Ecklonia*, suggests a substantial change in factors affecting the community structure. While effluent exposure is expected to decrease slightly between these two locations, we consider more prominent changes in seabed structure and higher wave exposure may be major factors affecting the change in biological community structure between the third and fourth zones. This

discontinuity was considered an effect of natural processes rather than an effluent impact related pattern.

3.1.4 Subtidal Infauna

As part of the subtidal reconnaissance, infauna was investigated at two sites between the offshore reef and the shoreline. One site was 640 m offshore directly opposite the outfall, the other was a similar distance offshore but 4.2 km to the northwest (a “control” site) (Fig. 3.4). The seabed at both sites was hard packed white sand with wave ripples and resuspension by surge energy. No epifauna were observed at either site. The seabed at the site opposite the outfall was slightly grey in colour, which was assumed to be due to accumulation of sewage particles. However, later work (see below) casts doubt on this assumption.

Eight hand driven cores (100 mm diameter by 150 mm depth) were taken at the northwestern site but bad weather allowed only three cores to be collected at the outfall site. This latter site was revisited six weeks later and eight cores collected. This glitch in the program proved to be useful in demonstrating marked temporal variation in the composition of the infauna.

Ampeliscid amphipods were present in all samples and dominated the population but were significantly higher at the site directly opposite the outfall. Distribution was very patchy with numbers ranging from zero to 84 per core. Spionid polychaetes were the next most abundant taxon, especially at the outfall site, but in the second sampling they had disappeared. Other polychaete species were present but in very low numbers. All other biota were sparse and patchy.

A cumulative percent abundance plot of species at both sites showed the outfall site to be numerically dominated by fewer species than the control site (4.2km NW). It was evident that the high variability and sparse numbers of the infauna would require much larger samples before any conclusions could be drawn on likely effluent effects.

To provide further information a more extensive survey was conducted (Chidgey *et al.* 1998). Divers sampled twelve sites on April 5 1998. The sites were located between 4000 m northwest and 3000 m southeast of the outfall, along the 10 m depth contour (Fig. 3.4). Sediment cores 200 mm in diameter and 200 mm deep were obtained using a frame pushed into the sand. The sediment was collected into a 500 µM mesh bag with an airlift suction device. Five cores were collected from each site, with an additional sample for analysis of grain size and organic content.

Infauna were sorted, identified to the family taxonomic level and counted. Total wet-biomass was also determined for each sample. Grain size composition of the sediments was determined by sieving and total organic matter was determined by loss on ignition at 550°C.

The infauna were compared between sites for patterns which correspond with the location of the outfall. The data were examined for trends in abundance’s of common taxa as well as trends in community structure. Community structure was examined and compared using diversity statistics, dominance curves and multivariate ordination. Relationships between biological assemblages and the physical environment, particularly total organic matter and particle size composition were also examined.

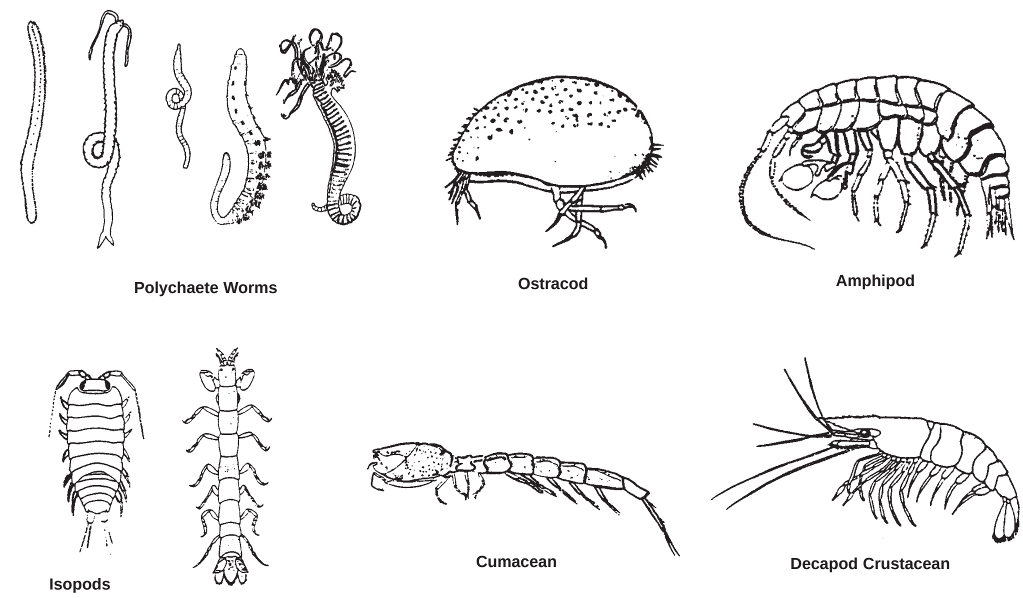
The infauna (Fig. 3.7) was generally composed of polychaete bristle worms, shell-like ostracod crustaceans, decapod shrimps and shrimp-like cumaceans, amphipods and isopods. The common polychaete worms included the deposit feeding spionids, which live in mucus lined tubes and have long palps to gather food particles. The lumberinid and sigalionid worms are burrowing predators with eversible, fang-like mandibles. Ostracods, cumaceans, isopods and amphipods are small crustaceans that burrow through the sediments for small organic particles. The decapod crustaceans were mostly callianassid (ghost) and alpheid (snapping) shrimps. These shrimps create burrows in the sediments and feed on organic particles.

Table 3.3. Abundances (counts) of some of the more common taxa at each sample site.

Site positions relative to the outfall are: (A) 4000 m NW; (B) 3000 m NW; (C) 540 m SE; (D) 550 m SE; (E) 600 m SE; (F) 660 m SE; (G) 900 m SE; (H) 1000 m SE; (I) 1200 m SE; (J) 1500 m SE; (K) 2000 m SE; (L) 3000 m SE.

Taxon	A	B	C	D	E	F	G	H	I	J	K	L
Annelida: Polychaeta (bristle worms)												
Lumbrineridae	4	14	10	16	6	10	8	4	2	2	2	3
Spionidae	4	1	35	5	22	49	3	0	0	0	0	1
Arthropoda												
Ostracoda (seed shrimps)	81	20	11	1	5	27	48	39	25	0	0	50
Malacostraca (higher crustaceans)												
Cumacea 11	24	12	2	7	5	2	2	0	0	2	2	
Amphipoda												
Ampeliscidae	295	387	660	377	647	313	1	0	7	3	3	8
Oedicerotidae	17	1	21	5	19	10	7	1	1	1	0	2
Phoxocephalidae	38	6	9	4	20	17	9	8	17	0	0	0
Urohaustoriidae	32	43	10	7	11	11	17	45	23	2	6	6
Isopoda												
Anthuridae	5	0	1	8	13	9	0	0	0	0	1	0
Cirolanidae	20	0	0	1		4	4	62	48	16	0	1
Decapoda	27	12	15	9	6	9	59	50	8	0	0	3

Figure 3.7. Illustrations of the types of animals present in sediments (infauna) in the Boags Rocks region (not to scale; Chidgey *et al.* 1998).



Some trends in the infauna were evident between the northwest and southeast sites. The most obvious trend was a high dominance of ampeliscid amphipods between 4000 m northwest and 660 m southeast of the outfall, with 295 to 660 animals collected per site. Other taxa with higher abundances to the northwest include lumbrinerid worms, cumaceans and lysianassid amphipods. In contrast, cirolanid isopods and decapod crustaceans were higher in abundance at the southeastern sites. These northwest-southeast trends were reflected in the total number and total biomass of infauna at each site, with both total numbers and biomass being higher to the northwest (Figs. 3.8 and 3.9). The total number of animals was particularly low at 1500 and 2000 m southeast, with 13 and 21 animals respectively (Fig 3.8).

Figure 3.8. Total number of animals (infauna) collected at each site (NW to SE of outfall).

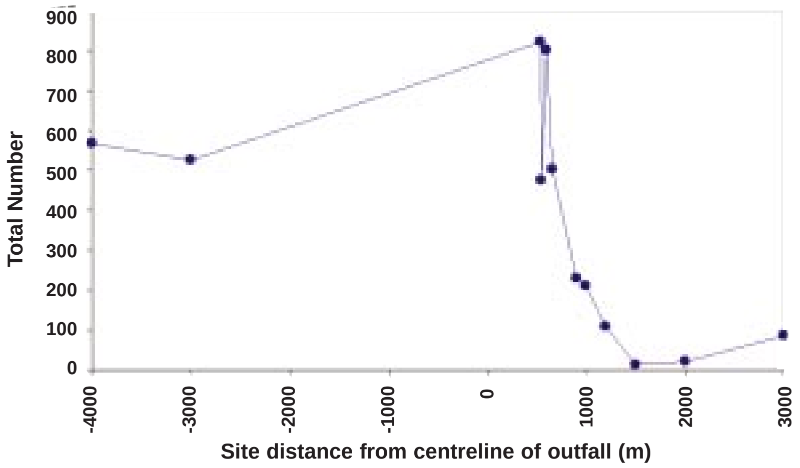
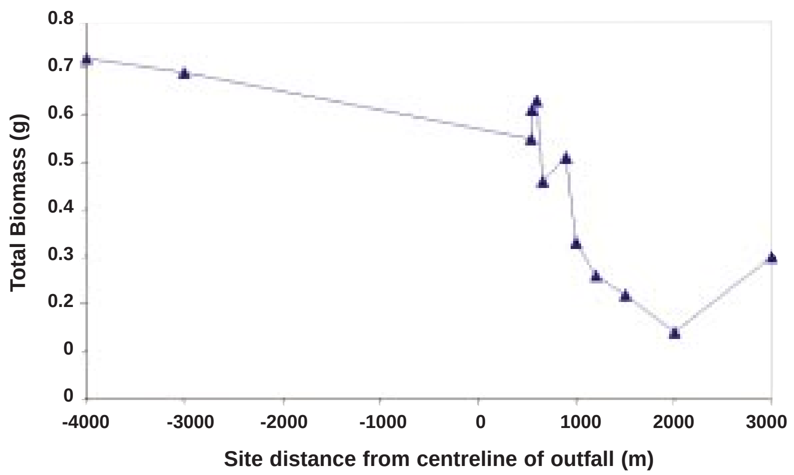


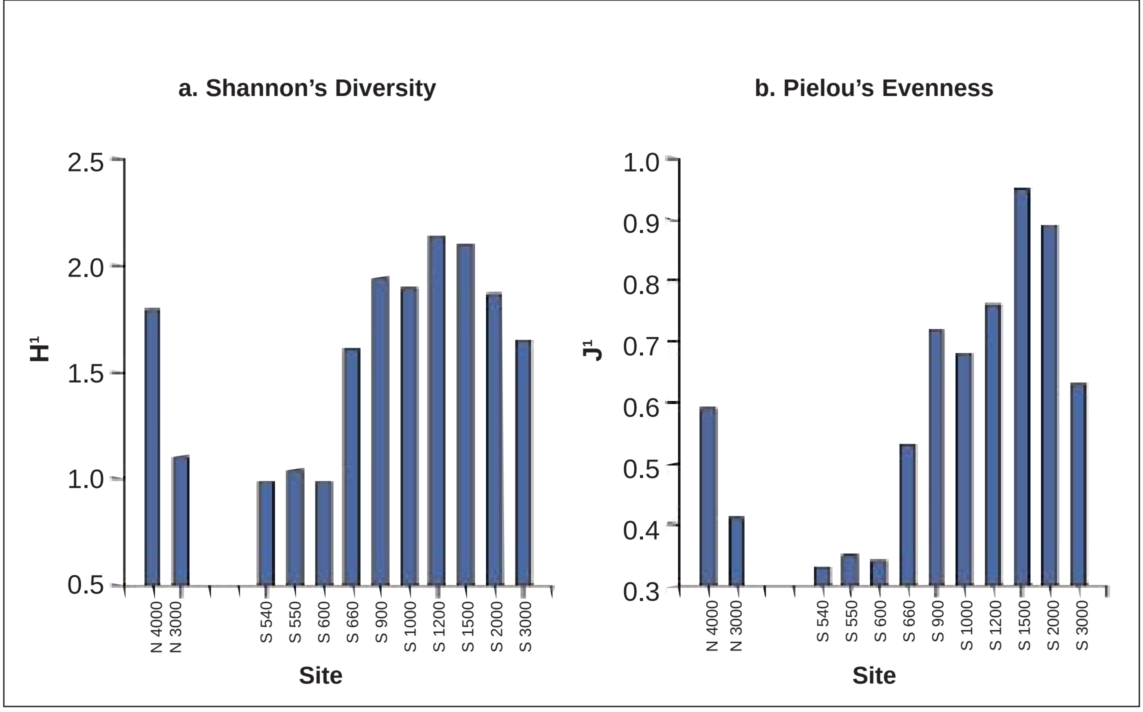
Figure 3.9. Total biomass of animals (infauna) collected at each site (NW to SE of outfall).



Species diversity and evenness is a measure of the distribution of individuals among species comprising a community. A community with higher species diversity has a more even distribution of individuals among species. These measures are often used as indicators of ecosystem health (biodiversity). In unhealthy systems, the overall biomass (living matter) is dominated by a limited number of species of a particular function group. As the number of species (biodiversity) and functional diversity increases, so to does biomass, productivity, nutrient retention and stability within the ecosystem. This also allows a natural resilience to perturbation to develop.

For the infauna, diversity and dominance were examined by calculation of the Shannon H' and Pielou's Evenness J' statistics. Diversity and evenness were least at the sites nearest the outfall (540, 550, 600 and 660 m south east) (Fig. 3.10).

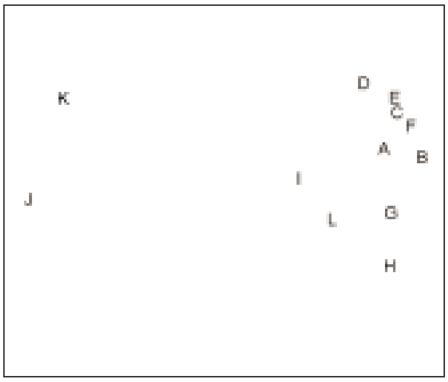
Figure 3.10. Comparison of infaunal diversity and evenness between sites (NW to SE of outfall).



When community structure was examined, with appropriate transformations (fourth-root transformed and use of Bray-Curtis dissimilarity index), the non-metric multidimensional scaling (MDS) plot in Fig. 3.11 was obtained. The community structures at 1500 m southeast and 2000 m southeast (Sites J & K) were quite different from the other sites. The sites closest to the outlet, 540 to 660 m southeast (Sites C-F), were similar in assemblage structure. Sites further southeast of the outlet (> 660 m; Sites G, H, I & L) had assemblages different to the sites near the outlet. The two distant northwest sites (A & B) had assemblages intermediate to the near (C-F) and far (G-L) southeast assemblage structures, clustering in the middle of these two groups. These differences in community structure do not show a clear relationship with the location of the outlet.

Figure 3.11. MDS plot comparing infaunal community structure between sites (Stress = 0.083).

Site positions relative to the outfall are: (A) 4000 m NW; (B) 3000 m NW; (C) 540 m SE; (D) 550 m SE; (E) 600 m SE; (F) 660 m SE; (G) 900 m SE; (H) 1000 m SE; (I) 1200 m SE; (J) 1500 m SE; (K) 2000 m SE; (L) 3000 m SE.



Some of the differences in community structure were related more strongly to organic matter content and grain size of the sediments giving a confounding effect between these factors and effect of effluent. However, the following conclusions may be drawn:

- There was a strong pattern of high abundance and high biomass of infauna close to and northwest of the outfall, which reduced rapidly with distance to the southeast.

In isolation, these gradients would be a strong indication of an impact close to the outfall, declining with distance from the outfall. However, grain size and organic content of the sediments also showed a strong trend from the outfall. The trend of increasingly coarse sand and lower organic content towards the southeast is not considered to be an effect of the discharge, but could be a major factor affecting infauna characteristics. The high organic content to 4000 m northwest of the outfall seems to preclude sewage origin.

- There was a lower diversity of infauna at the 4 sites within 660 m of the outfall. This is considered to be an effect of exposure to effluent.
- Certain species of infauna were more abundant at the 4 sites within 660 m of the outfall. Relatively high abundance of some species - notably spionid worms, corphid amphipods and eusirid amphipods - is considered to be a response to the effluent discharge.
- The infaunal community composition was different at the 4 sites within 660 m of the outfall relative to the 6 sites between 900 and 3000 m southeast of the outfall. This is considered to be an effect of exposure to effluent within 660 m of the outfall.

Overall, the effluent discharge appeared to be a factor affecting infaunal community structure within 660 m of the Boags Rocks outfall at the time of the survey.

3.2 Bioaccumulation

Whilst there are various aspects to deleterious effects of treated effluent on the marine environment, one of the first likely to come to public attention would be contamination of seafood. Previous analyses of marine organisms near Boags Rocks have shown tissue toxicant contents to be below guideline values. However these analyses were conducted some years ago and there was a need to update knowledge and take advantage of developments in improved analytical techniques.

In 1986, the (then) MMBW arranged for samples of wrasse flesh and livers and abalone flesh to be collected from near Boags Rocks and analysed for a suite of 10 organochlorine pesticides. The levels found in the flesh of both species were about two orders of magnitude lower than the levels found in the livers of the wrasse, which themselves were well below EPA Standards for Human Consumption.

A more extensive program was conducted in 1991, when samples of limpets, abalone and the periwinkle (Turbo) were collected at Boags Rocks, Boags Rocks East and West and Number Sixteen sites. Samples of the horseshoe leatherjacket were taken at Boags Rocks and Number Sixteen only. All samples were analysed for four organochlorine pesticides (Lindane, Aldrin, DDE and Dieldrin), PCBs and dioxins and furans. All levels were well below EPA Standards, even when the four pesticides were summed under the EPA combination rule.

Levels were, however, extremely variable within samples and between sites, which confounded attempts to identify suitable samples sizes for further monitoring.
In an attempt to resolve this question the data was provided to Monash University for statistical analysis (Quinn 1996). Quinn suggested that the sample sizes required to estimate the true mean concentrations with 90% confidence were impractical. For example, to estimate the true mean concentration of Aldrin, ±10%, a sample size of 456 abalone and 641 wrasse were required. To reliably detect a site difference of even ±100% for Dieldrin in abalone, 27 sites would be required.

This level of sampling is both inappropriate and (for ecological reasons) impracticable. In consultation with the EPA it was decided to restrict the sample sizes to ten of each species. We thus regard the results of these analyses as descriptive rather than inferential. The data nevertheless provides useful information that is indicative of overall levels of contaminant concentrations.

The organisms collected were the blue throated wrasse, because it is commonly taken by anglers and is territorial, and the black-lip abalone, a commercial species. Also taken were individual specimens of the cunjevoi ascidian (sea squirt) *Pyura stolonifera.* This organism is a filter feeder and was likely to show effects of uptake of suspended matter within the effluent.

Samples of axial muscle were taken from each fish for analysis. From the abalone, the foot, which is usually consumed, was used. From the cunjevoi, the fleshy contents of the leathery sack were used. All fish were taken within 500 m of the outfall and all other specimens from a submerged reef less than 200 m from the outfall.

The question of which pollutants to analyse was resolved by examining results from analyses of ETP effluent that spanned over 130 elements and compounds. A few pollutants are sometimes detected and of these, the representative heavy metals lead, copper, chromium and nickel were chosen. Toluene is sometimes found, as are the phthalate esters, so these were also included.

The phthalate esters measured were: diethyl phthalate (DEP), di-n-butyl phthalate (DBP) and di-(2-ethylhexyl) phthalate (DEHP). Phthalate esters are used in the production of plastic to reduce brittleness and increase flexibility. They are widely used in the production of packaging, building materials, furnishings, clothing and kitchenware. Although not especially toxic they are on the list of endocrine disruptors and their impact in the marine environment is being investigated.

Whilst the polychlorinated dibenzo-p-dioxins (PCDDs) and polychlorinated dibenzofurans (PCDFs) are not detected in the effluent, they were included in this exercise because even at extremely low levels they can accumulate in fatty tissues and are commonly seen as "typical" industrial pollutants.

Cunjevoi samples were analysed for heavy metals, as only limited research has been carried out on this organism giving a limited understanding of their metabolism.

The sampling and analyses were conducted by the Marine and Freshwater Resources Institute (MAFRI). For those analyses for which MAFRI did not have NATA certification, samples were sent to laboratories that were certified. The results are given in Tables 3.4 and 3.5 (Brady and Fabris 1997a).

It can be seen that heavy metal levels in the fish and abalone were mainly low (sometimes undetectable) and are mostly below the National Food Authority Standard and EPA Water Quality Guidelines given in the Table 3.4. Nickel levels in abalone showed the most variability; with levels ranging from undetectable to 12.6 µg/g. There are no National Food Authority Standards for nickel in abalone. However, the US Food and Drug Administration suggest that the danger level for a person consuming shellfish on a continuous daily basis would be 80 µg/g, nearly seven times the maximum level identified in the samples taken.

The cunjevoi copper levels were compared to analyses made on specimens collected off the NSW coast (Florence *et al.* 1997). The mean of 4.45 µg/g is at the mid range found in the latter which varied from 1.6 in a pristine environment to 13.1 µg/g in a bay close to a primary treated sewage outfall. The range of values in the 10 replicates was fairly small.

Of the organic pollutants, only DEHP was detectable in all fish, DEP and DEHP only in one abalone, DEP in two fish and DBP in one fish. All levels were close to the detection limit.

Phthalate esters vary in their chemical structure, but all readily biodegrade and hence do not accumulate in the food chain. phthalate esters are also removed from the environment by photooxidation and microbial activity. It was concluded that the levels of phthalate esters found were unliklely to lead to endocrine disruption in organisms present.

The dioxin-furan levels were three orders of magnitude lower than the international standard in fish and over 300 times lower than the standard in abalone.

It was concluded that bioaccumulation off Boags Rocks is too negligible to present a human health hazard. The scientists conducting the assessment, also suggested that if a long term monitoring program is initiated, that a well studied filter feeder such as the common mussel be used in place of the fish and invertebrates studied here.

Table 3.4. Results for heavy metals (inorganic toxicants)

Sample	No.	Size mm	Chromium	Copper µg/g wet weight	Nickel	Lead	wet wt/dry wt ratio	sex
Wrasse	1	310	<0.2	<0.2	<0.2	<0.5	3.9	F
(Notolabrus tetricus)	2	375	<0.2	<0.2	<0.2	<0.5	4.0	M
	3	265	0.3	<0.2	<0.2	<0.5	2.6	F
	4	325	<0.2	<0.2	<0.2	<0.5	3.4	F
	5	345	0.3	<0.2	<0.2	<0.5	3.7	M
	6	255	0.4	<0.2	<0.2	<0.5	3.4	F
	7	250	0.3	<0.2	<0.2	<0.5	3.6	F
	8	385	<0.2	<0.2	<0.2	<0.5	4.5	M
	9	200	0.9	<0.2	<0.2	<0.5	3.4	Undetermined
	10	230	<0.2	<0.2	<0.2	<0.5	3.8	M
#NFA Standard			NL	10.0	NL	0.5		
##RWQC			5.5		5.5			
Abalone	1	110	0.3	3.5	2.4	<0.5	4.0	
(Haliotis rubra)	2	125	0.4	3.2	1.2	<0.5	4.4	
	3	110	0.4	3.2	0.9	<0.5	4.1	
	4	100	0.2	3.2	2.1	<0.5	3.9	
	5	110	0.3	3.3	4.6	<0.5	3.8	
	6	115	<0.2	2.2	0.3	<0.5	4.2	
	7	115	<0.2	1.6	<0.2	<0.5	3.9	
	8	100	<0.2	2.2	4.3	<0.5	4.2	
	9	100	0.5	5.4	12.6	<0.5	3.8	
	10	110	0.5	3.2	4.3	<0.5	4.1	
#NFA Standard			NL	70.0	NL	0.5		
##RWQC			5.5		5.5			
Cunjevoi	1		0.6	3.6	0.2	<0.5	8.6	
(Pyura stolonifera)	2		0.8	4.2	0.4	<0.5	9.0	
	3		0.5	5.5	0.3	<0.5	8.5	
	4		0.7	5.1	0.3	<0.5	10.1	
	5		0.7	4.4	0.4	<0.5	10.4	
	6		0.5	4.4	0.4	<0.5	9.6	
	7		2.5	6.3	0.5	<0.5	7.5	
	8		1.0	2.3	0.3	<0.5	9.2	
	9		0.4	3.3	0.2	<0.5	12.6	
	10		1.0	5.4	0.4	<0.5	8.2	
#NFA Standard			NL	10.0	NL	0.5		
##RWQC			5.5		5.5			

NL Not listed, therefore RWQC standard used
National Food Authority Standard (1996)
From EPA Recommended Water Quality Criteria (1983)

Table 3.5. Results for organic toxicants

Sample	No.	Toluene	DEP	DBP	DEHP	PCDD/PCDF Congeners
		µg/g wet weight				Total I-TEQ pg/g wet weight
Wrasse	1	<0.5	0.3	0.3	0.2	
Notolabrus tetricus	2	<0.5	<0.2	<0.2	0.4	
	3	<0.5	<0.2	<0.2	0.3	
	4	<0.5	<0.2	<0.2	0.2	
	5	<0.5	<0.2	<0.2	0.2	
	6	<0.5	0.2	<0.2	0.2	
	7	<0.5	<0.2	<0.2	0.2	
	8	<0.5	<0.2	<0.2	0.2	
	9	<0.5	<0.2	<0.2	0.2	
	10	<0.5	<0.2	<0.2	0.2	
Composite	1					0.034
	2					0.020
	3					0.018
Abalone	1	<0.5	0.5	<0.2	0.3	
Haliotis rubra	2	<0.5	<0.2	<0.2	<0.2	
	3	<0.5	<0.2	<0.2	<0.2	
	4	<0.5	<0.2	<0.2	<0.2	
	5	<0.5	<0.2	<0.2	<0.2	
	6	<0.5	<0.2	<0.2	<0.2	
	7	<0.5	<0.2	<0.2	<0.2	
	8	<0.5	<0.2	<0.2	<0.2	
	9	<0.5	<0.2	<0.2	<0.2	
	10	<0.5	<0.2	<0.2	<0.2	
Composite	1					0.049
	2					0.099
	3					0.035

DEP - diethyl phthalate, DBP - di-n-butyl phthalate and DEHP - di-(2-ethylhexyl) phthalate represent the three main phthalate esters. The National Food Authority Standard (1996) and the EPA Recommended Water Quality Criteria (1983) Table 10a, do not list objectives for these toxicants.

Total I-TEQ is the international toxic equivalent for summing dioxin-furan congeners. For the above analyses, congeners lower than the detection limit have been assigned a value of 0.5 times the detection limit. The Department of the Environment, Ottawa, Ontario, Guidelines for Water Quality Objectives and Standards list the standard as 20.0.

3.3 Toxicity Assessment

The extensive program of bioassays undertaken in this Study was conducted for two reasons. Firstly Melbourne Water is required to monitor the toxicity of the effluent. Secondly there has been difficulty in determining the extent and trend of changes in biological assemblages along the coast that may be effluent related. This problem was discussed in Section 3.1 of this report.

It was reasoned that if several test species of common marine flora and fauna were subjected to effluent exposure under controlled conditions and the effluent concentrations having no toxic effect were determined, then the extent of toxic effect could be estimated from the known dilution pattern of the effluent plume.

CSIRO's Centre for Advanced Analytical Chemistry (CAAC) coordinated and reported (Stauber *et al.* 1998) on the assessment program, which involved three other institutions.

The respective roles were: -

CSIRO CAAC:	Coordination and reporting / Microtox® test / Microalgal growth bioassay
MAFRI:	Sample preparation, analysis and dispatch / Doughboy scallop larval development bioassay
RMIT University:	Fish larval mortality bioassay
Monash University:	Macroalgal fertilisation and growth bioassay

The choice of test organisms for the toxicity-testing program was based on their seasonal availability; the availability of established bioassays with indigenous species and the need for both acute and chronic tests. Acute tests essentially measures short-term effects (both lethal and sub-lethal) whilst the chronic tests indicate long term detrimental effects or impacts on early life stages.

The acute tests were performed on *Vibrio fischeri*, a bacterium whose luminescence decreases in response to toxicants and larval fish, either Australian Bass *(Macquaria novemaculata)* or estuarine perch *(Macquaria colonorum).*

The chronic tests were carried out with *Nitzschia closterium* a unicellular marine alga (diatom) common in Australian waters, larvae of the doughboy scallop *(Chlamys asperrima)* and early life stages of the brown macroalga *Hormosira banksii* common on intertidal rock platforms along the Victorian coast. This latter organism provided three test stages - fertilisation of the eggs, germination at 48 hours and cell division (growth) at 72 hours.

The protocol of sample preparation and experimental procedures is very precise and standardised so that bioassays on the same organisms may be compared at different places or times. A full description of procedures is given in the report by Stauber *et al.* (1998). Because the large volumes of effluent required (100 litres) precluded sampling at the outfall, samples were collected from the rising main leaving ETP. This excludes a small contribution (20 ML / day or about 5% of total discharge) of treated effluent from three local sewage plants into the pipeline between ETP and the outfall. However, a comparison of routine chemical analyses of effluent at ETP and Boags Rocks does not show any obvious differences in effluent composition.

The organisms were subjected to various concentrations of effluent, produced by diluting effluent (pre-adjusted to 33 ‰ salinity) with filtered natural seawater or artificial seawater. This was done to separate the effects of toxicants from the impact on marine organisms of salinity changes caused by the discharge of large volumes of freshwater effluent. This latter effect has been studied at Monash University and Victoria University of Technology (VUT) and some results of this will also be discussed.

The toxicity results are expressed as LC50 / EC50 %, LOEC and NOEC. The first measure gives the percentage concentration of effluent at which the test organism shows only 50% of the control response. For example, in the case of the microalga *Nitzschia,* where growth is measured over 72 hours (3 to 4 doublings of cell numbers in seawater controls), the EC50 is that concentration of effluent in which only half this growth rate is achieved. Similarly, in the fish bioassays, the LC50 is the concentration of effluent in which half the fish have died after 96 hours. The LOEC is the Lowest Observable Effect Concentration where statistical testing of the results indicates the smallest significant difference from the controls. NOEC is the No Observable Effect Concentration where no significant toxic effects are detectable.

A summary of the results over three samplings (June, September and November 1997) is given in Table 3.6. The NOEC values only are given since this is the level at which organisms are unaffected and which may be used to calculate "safe" dilution of the effluent.

Table 3.6. Comparison of no observed effect concentrations (NOEC) to test species for the three sampling events

Test Species	Endpoint	NOEC (%)		
		June	September	November
Nitzschia closterium (microalga)	72-h Growth	25	12.5	12.5
Hormosira banksii (macroalga)	Fertilisation	>100	>100	>100
	48-h Germination	6.25	6.25	6.25
	72-h Cell Division	6.25	6.25	6.25
Chlamys asperrima (scallop larvae)	48-h Development	6.25	0.5	0.5
Vibrio fischeri (bacteria)	15-min Light Output	>90	>90	>90
Macquaria spp (fish larvae)	96-h Mortality	50	50	6.25

NB: For the November fish larvae test, 4-5 week old larvae were used rather than the preferred 1-3 day old larvae, which were seasonally unavailable.

It can be seen that bacterial luminescence and *Hormosira* fertilisation are not affected by even undiluted effluent. Fish larvae 1-3 days old are also little affected. They all survived in effluent of 50% dilution. In November when 4-5 week old bass larvae were used, the results indicated that the effluent was more toxic. However, this may not have been a true indication of increased toxicity. It was observed that the older larvae, having exhausted their normal food sources, consumed the sewage particles. Such particles were likely to have many of the contaminants adsorbed to them and thus uptake of chemicals would have been via the gut as well as the gills. Some substances would thus be made more bioavailable and possibly more toxic (such as lipophilic ones that would not be in the water column). It is likely that in open waters the fish would be more selective in their choice of food.

The germination of *Hormosira* and subsequent cell division were affected but only in concentrations of effluent above 6.25%, i.e. dilutions of less than 16 times. Growth of the diatom, *Nitzschia closterium*, was inhibited unless at least an 8 times dilution was used. Most sensitive were the scallop larvae where some abnormal development occurred unless the effluent was diluted 16 times in June and 200 times in September and November.

We have previously mentioned that VUT conducted toxicity testing on ETP effluent and their results were made available to this Study (Shir and Burridge 1998). The VUT work used three species of macroalgae - *Hormosira banksii, Macrocystis angustifolia* and *Phyllospora comosa.* For the first two, the criteria of 48 hour germination, 96 hour embryo mortality and 1 and 2 week embryo growth were used. In the case of *Phyllospora,* zoospore germination, 48 hour germ tube growth, 19 day sporophyte production and 20 day sporophyte growth were used. Samples of primary treated effluent and secondary treated effluent before and after chlorination, were tested over a two year period. Samples were tested with and without salinity adjustment to determine the effects of both chemical toxicants and salinity reduction.

The results show that primary effluent is about 5 times more toxic than secondary effluent suggesting that most toxicants are destroyed in the secondary treatment process or locked up in the sludge. Chlorination slightly increased toxicity in some cases. Adjusting effluent to seawater salinity decreases toxicity although this was not as significant for primary treated effluent. Comparison of the VUT and Monash results on *Hormosira* is problematical because of differences in procedures but EC50, LOEC and NOEC levels for the Monash work were within the range determined by VUT.

As mentioned earlier, one objective of the toxicity assessment program was to determine whether effluent concentrations along the coast either side of Boags Rocks were likely to produce loss of biological assemblages, and if so, to what extent. We now have a total of 32 bioassay results (including VUT results) to draw on for such an estimate. The question arises as to which risk assessment methodology should be used.

The most sensitive organism tested in the program was the scallop larva, which gave a NOEC of 0.5% or 200 times dilution. However, some authorities believe that a further safety factor is desirable. Stauber *et al.* (1998) used two additional approaches, one depending on the range of EC50 results from chronic tests and the other on NOEC results from all tests. The former gives a required dilution of up to 333. The latter gives a dilution of 250 for protection of 95% of species with 50% confidence. Based on experimental data, they suggest a compromise figure of 300 times dilution for no significant measurable effect.

The hydrodynamic model of plume dispersion was used to generate contours of effluent concentration from the present outfall (Andrewartha *et al.* 1998). The shape of the plume is highly variable, depending on local meteorological and oceanographic conditions, sometimes moving longshore as a discrete band, other times moving offshore. With onshore winds the effluent tends to 'pool' along the coast.

Taking two simulations made for March and June 1997, validated with MAFRI underway surveys, it would seem that the 1:300 dilution contour at the coast occurred about 3km northwest and 4km southeast of the outfall on 18th March and about 2km northwest and 6km southeast on 20th June. Model simulations commonly show exposure to effluent dilutions less than 300:1 over a distance of about 4 - 5km either side of the outfall along the coast, with some bias towards the south-east. Lower dilutions (higher effluent concentrations) and more frequent exposure occur closer to the outfall.

To gain a better understanding of what substance (or substances) present in the effluent is causing the toxicity, Monash University and CSIRO carried out further experiments. Monash University researchers conducted *Hormosira banksii* reference toxicant experiments using ammonia, phenol, chlorine and a surfactant to explain the measured toxicity. From their limited results it was clear that ammonia was one of the principal substances creating a toxic impact.

The US EPA has recently pioneered marine Toxicity Identification Evaluation (TIE) tests that can provide clearer understanding of what causes toxicity. TIE is a three phase approach that combines toxicity tests with chemical/physical manipulations to identify specific toxicants in complex effluents. Based on the US EPA procedures, CSIRO have developed TIE tests using the diatom, *Nitzschia closterium*.

In September 1998, TIE tests were carried out using ETP effluent. The results (Stauber *et al.* 1999) indicate that the majority of toxicity is caused by ammonia, however an unidentified non-polar organic may also be involved. Toxicity was not due to particulate matter, volatiles, hydrogen sulfide, metals, surfactants, oxidants such as chlorine, phthalates, PAHs, organochlorine or organophosphate pesticides.

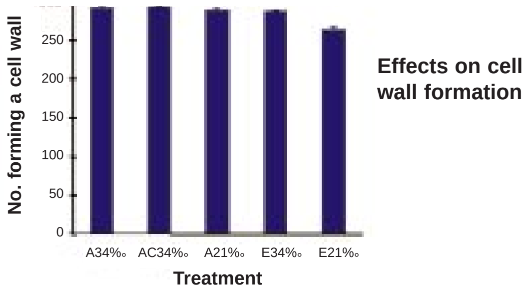
Reference has been made to the freshwater impact of the effluent. Considerable work has been carried out on this aspect at Monash University and was continued under this Study. There is a very profound synergism between salinity and effluent toxicity as shown in Fig. 3.12. The findings from the spring series of tests were consistent for all three bioassays used and provided similar results to those obtained during winter, summer and autumn (Kevekordes 1998b).

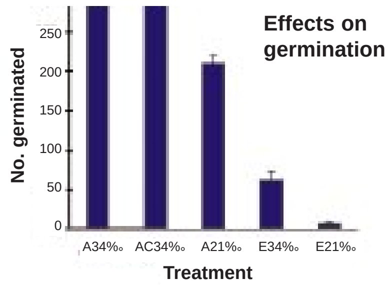
Fig. 3.12 shows results from exposure of *Hormosira banksii* fertilised eggs to five solutions of varying mixes of seawater and effluent. SC34 and S34 are control solutions made from artificial seawater and having no effluent. S21 is also made from artificial seawater but it is diluted with UHQ water to a salinity of 21‰. This level of salinity approximates that found immediately adjacent to the existing outfall, which is a result of a 40% effluent and 60% seawater mix. E34 is 40% effluent mixed with 60% seawater with salt added to reach normal seawater salinity of 34‰. E21 is 40% effluent mixed with 60% seawater giving the solution a salinity of 21‰.

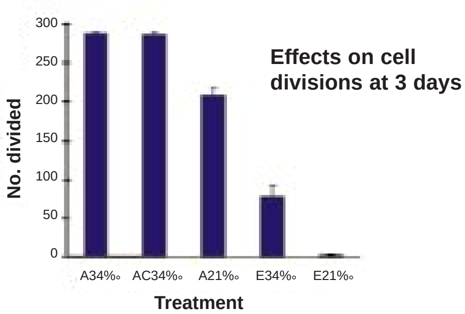
Alone, salinity reduction reduced macroalgal germination and growth by about a third. Effluent adjusted to seawater salinity (34‰) was even more toxic, reducing macroalgal germination and growth by about three quarters. This reduction was due to chemical toxicity of the effluent e.g ammonia. A combination of effluent and low salinity completely depressed macroalgal germination and growth. The results show that a combination of effluent and low salinity has the most depressing effect on macroalgal germination and growth.

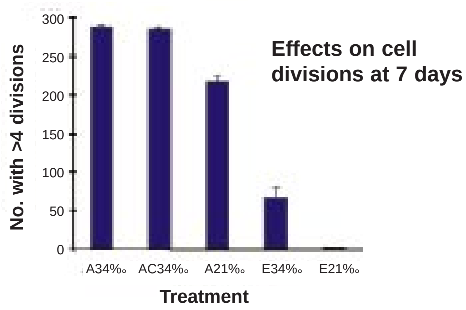
Figure 3.12. Results from the Spring 1997 *Hormosira banksii* Postfertilization bioassay:- (Kevekordes 1998b).

Effect of treatments on the number of viable embryos that have formed a cell wall by 24 hours, germinated by 48 hours, undergone cell division by 72 hours and have more than 4 cell cleavages by 7 days



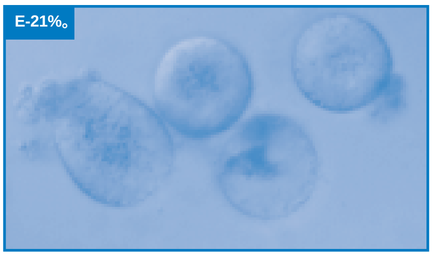






Abbreviation	Treatment Solutions	Salinity
S-34‰	Artificial seawater	34‰
SC-3‰	Artificial seawater (alternative method)	34‰
S-21‰	Artificial seawater	21‰
E-34‰	40% Effluent, 60% seawater with salt added	34‰
E-21‰	40% Effluent, 60% seawater	21‰

Microscopic photograph of cells taken at 72 hours postfertilization for various solutions.



3.4 Receiving Water Quality

3.4.1 Toxicants

Water samples offshore from Boags Rocks were collected in April 1997 and analysed for selected toxicants to access whether the concentrations found were below the EPA guidelines (Brady and Fabris 1997b).

Samples were taken as near as practicable to the boundary of the 200 m mixing zone and analysed for the same toxicants measured in the bioaccumulation project described earlier. Dioxins and furans were not included in this because the levels found in fish and abalone tissue were so low that it seemed unlikely that any of these substances would be detectable in water with the methods available. The results obtained are given in Table 3.7 and 3.8 (Brady and Fabris 1997b).

All heavy metals were present at levels well below the Victorian EPA Recommended Water Quality Criteria (RWQC) and in fact chromium was below detection limits. Samples 1 to 5, which were in the effluent plume, were higher than ocean background but Sample 6 (1.3 km from outfall) levels were close to background values.

Table 3.7. Inorganic toxicant concentrations (16/4/97)

Site	Lead µg/L	Nickel µg/L	Copper µ.g/L	Chromium µg/L	NH_3 Ammonia µg/L
1	0.17	1.56	0.83	< 0.5	48.0
2	0.12	1.12	0.66	< 0.5	33.5
3	0.18	1.60	0.83	< 0.5	47.7
4	0.10	0.96	0.56	< 0.5	25.6
5	0.15	1.51	0.76	< 0.5	45.7
6 (control)	0.05	0.46	0.28	< 0.5	7.9
RWQC	5.0	15.0	5.0	5.0	4.0

Table 3.8. Organic toxicant concentrations levels (16/4/97)

Site	Toluene µg/L	DEP µg/L	DBP µg/L	DEHP µg/L
1	< 1.0	< 0.5	1.1	0.48
2	< 1.0	< 0.5	1.5	< 0.5
3	< 1.0	< 0.5	0.66	0.71
4	< 1.0	< 0.5	0.52	< 0.5
5	< 1.0	< 0.5	0.67	< 0.5
6 (control)	< 1.0	< 0.5	< 0.5	< 0.5
RWQC	1.0	0.1	2.0	0.15

RWQC Recommended Water Quality Criteria (Vic EPA 1983)

Phthalate esters:
DEP Diethyl phthalate
DBP Di-n-butyl phthalate
DEHP Di-(2-ethylhexyl) phthalate

Of the organic toxicants, toluene and diethyl phthalate (DEP) were undetectable. Di-n-butyl phthalate (DBP) was present in Samples 1 to 5 but at levels below RWQC. The ester di-(2-ethylhexyl) phthalate (DEHP) was present at many times RWQC at sites 1 and 3, but undetectable at the other sites. This anomaly could be a reflection of the innate variability in measuring poorly soluble organic substances, which sorb onto particles and hence may give anomalous results. It should also be noted that the RWQC for DEP and DEHP are below detection limits which makes them rather meaningless. Canadian Government RWQC for DBP and DEHP are 19 and 16 ppb respectively, much higher than EPA criteria.

Undissociated ammonia (NH_3) levels were well above RWQC, a reflection of the ammonia nitrogen content of the effluent. It should be explained that most ammonia nitrogen in natural waters exists as the ammonium ion NH_4^+. This is the form of nitrogen taken up by plants and is in fact often supplied as, for example ammonium sulfate fertiliser. At the pH and temperature of seawater, about 3 percent of total ammonia nitrogen exists as NH_3, which is toxic.

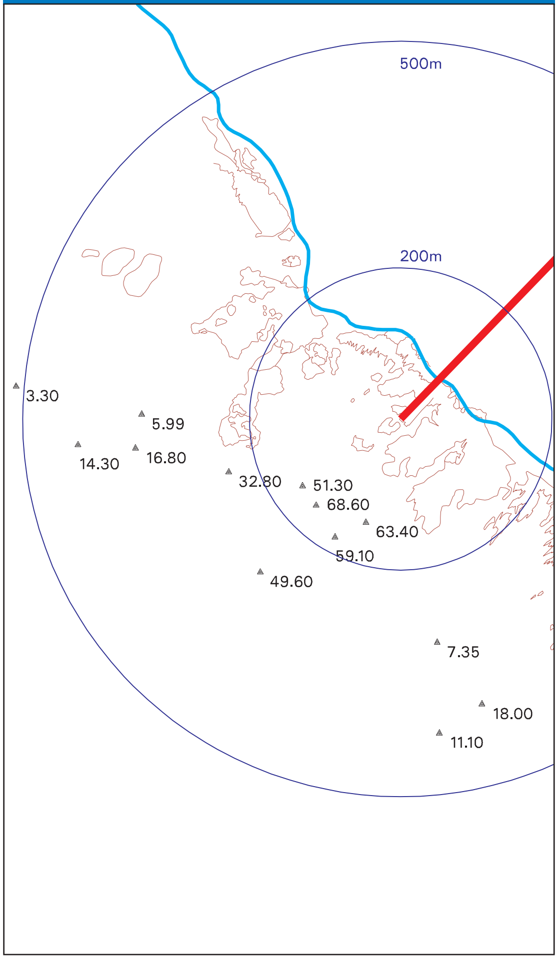
The ammonia concentration was calculated from the measured ammonium concentration and estimated salinity and pH. The concentrations derived were above the guidelines, and since some of the parameters used to calculate the concentrations had been estimated it was considered prudent to undertake a second sampling cruise to confirm the results.

The confirmation cruise (September 1997) for ammonia took 16 samples (Brady and Fabris 1997b). Of these, 13 were within 500 m of the outfall (Fig. 3.13) whilst 3 samples were taken some 2.5 km to the southwest. On this occasion the salinity and pH of each sample was measured to ensure that calculation of undissociated ammonia was based on measured parameters.

The results confirmed the previous survey. Near the outfall, undissociated ammonia (NH_3) levels varied from 3.3 ppb to 68.6 ppb. The NH_3 values were inversely proportional to salinity i.e. as the effluent mixed with seawater the NH_3 was diluted. The samples from 2.5 km away were 0.65, 0.84 and 2.89 ppb.

The results indicate that apart from ammonia (NH_3), toxicant levels in waters adjacent to Boags Rocks are not significant.

Figure 3.13. Water sampling sites, with ammonia values shown. Circles (200 and 500m radius) are centred on the end of the outfall.



3.4.2 Nutrients

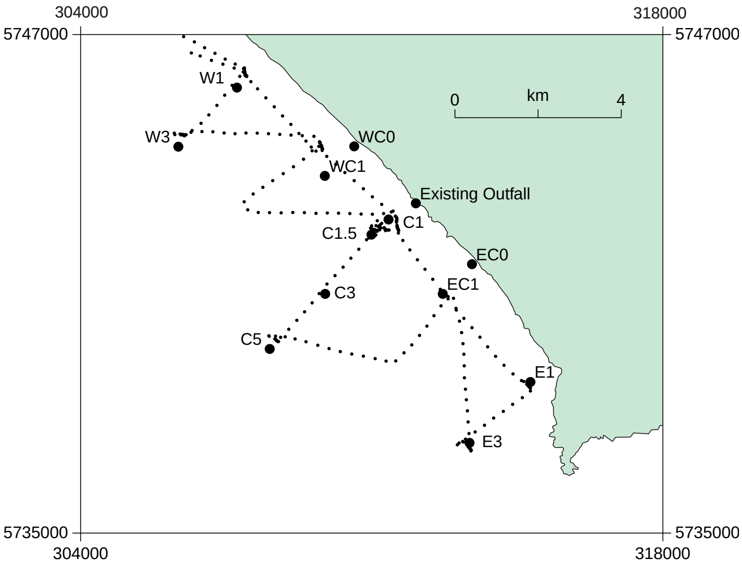
The discharge licence for ETP stipulates that beyond 600 m from the outfall, nutrient levels shall not cause nuisance plant growth or changes in species composition to the detriment of the protected beneficial uses. In effect this means that the nitrogen and phosphate in the effluent should not stimulate algal blooms. Whilst such blooms have not been recorded off Boags Rocks (by the EPA or Melbourne Water) the Study included a program of underway water sampling and the development of a water quality model to access the likelihood of bloom formation.

The underway water quality sampling cruises were carried out by scientists from MAFRI on four occasions, March, June and September 1997, and January 1998 (Longmore *et al.* 1998). Ammonium, oxidised nitrogen, phosphate, silicate, salinity, temperature, dissolved oxygen and chlorophyll in surface waters were measured by continuous sampling over an area 5 km east and west of the outfall and 5 km offshore (see Fig. 3.14, September cruise path).

MAFRI also measured the vertical distribution of physical parameters at seven sites spaced in a grid offshore covering the same area as the underway sampling (Fig. 3.14). These were primarily conducted to provide calibration data for the hydrodynamic model but included vertical profiles of dissolved oxygen at monthly intervals from March 1997 to February 1998. Samples of surface and bottom waters were also taken for analysis of nutrients. The per cent oxygen saturation levels fluctuated widely with time and depth but were always greater than 90% so that the EPA criterion of 85% saturation was always satisfied (Longmore *et al.* 1998).

The MAFRI underway sampling indicated that the chlorophyll levels found show oligotrophic to submesotrophic conditions. In June and September particularly, the highest chlorophyll levels did not approach 1 microgram/litre (µg/L). This is background level for most southern Australian waters. In summer there was slightly more phytoplankton production with chlorophyll levels rising to 2.5 µg/L in March and about 4 µg/L in January.

Figure 3.14. Track from the underway sampling cruise, September 1997. temperature, salinity and dissolved oxgen profiles and near bottom water nutrient samples were taken at sites W1, W3, C1, C1.5, C5, E1 and E3.



The zone off Boags Rocks behaves somewhat like an estuary with a high nutrient freshwater input mixing conservatively with a low nutrient high salinity receiving water. If ammonium, nitrate, phosphate or silicate values are plotted against salinity for any of the four surveys, a linear (straight line) relationship is observed. The relationship of chlorophyll to salinity is linear at low salinities indicating little or no primary production during mixing. Once the high background salinities are reached, however, chlorophyll values show considerable variation suggesting patchy primary production.

The MMBW carried out nutrient monitoring longshore from 1974 to 1979 with further monitoring in 1987 - 88. The Marine Chemistry Unit of the Ministry for Conservation also measured nutrients and chlorophyll in 1976. Most of this material was reviewed by the Victorian Institute of Marine Science (Black and Hatton 1994) and by Camp, Scott Furphy and Consulting Environment Engineers (1992). Their purpose, however, was mainly to delineate plume mixing patterns. No one has yet examined secular trends so that we do not know how the increase in volume, from 140 ML / day to the present 390 ML / day or changes in effluent composition, have affected the shape of the plume or contours of nutrient concentration.

Plots of salinity contours show that the licence requirement that total dissolved solids (TDS) levels should not deviate by more than 5% from background beyond 900m offshore, 1.7km to the west and 2.3km to the east cannot be met exactly because the plume does not diffuse uniformly through such an envelope. Rather, a core of lower salinity water moves back and forth mixing laterally and occupying less area than the above envelope.

A water quality model based on the hydrodynamic model (Murray and Parslow 1998) predicts elevated concentrations of dissolved inorganic nitrogen (ammonium and nitrate), exceeding 5 µM, over a region extending up to several kilometres either side of, and offshore from, the outfall. These nutrient concentrations are sufficient to potentially support dense phytoplankton blooms, and / or to modify benthic plant communities. Because nutrients behave approximately conservatively at high concentrations, the predicted concentrations (supported by MAFRI's observations) depend only on the dilution factors predicted by the hydrodynamic model.

The hydrodynamic model showed that the effective flushing time of the extended region around the outfall varies with wind conditions, but periods of low flushing, sufficient to allow phytoplankton blooms to develop, occur on a semi-regular basis. The water quality model can only reproduce the low chlorophyll concentrations observed by MAFRI (Longmore *et al.* 1998) by assuming tight coupling of zooplankton grazing with phytoplankton growth, suggesting that phytoplankton may be dominated by smaller cells susceptible to grazing. Diatom blooms may not occur because silicate concentrations in the effluent are low relative to nitrogen and phosphorous. The observed spatial distribution of chlorophyll also suggests a delay or inhibition of phytoplankton growth in the vicinity of the outfall.

Because these elevated nutrient concentrations occur over a region extending several kilometres from the outfall (out to dilutions of about 400:1), the risk of algal blooms would not be significantly modified by outfall extensions in the order of 1 to 3 km in length. Extending the outfall further out to sea and to a deeper depth would be a substantially larger project than the one discussed later in this report.

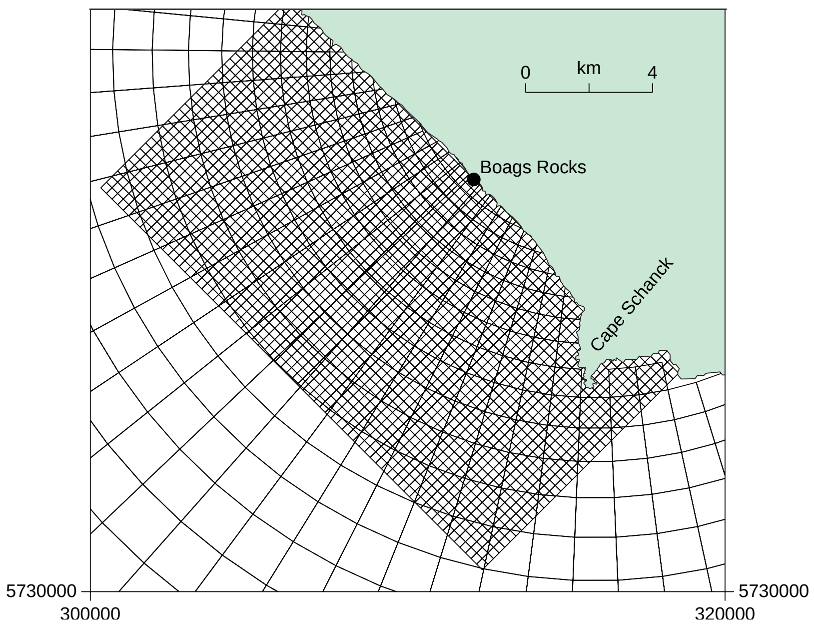
3.5 The Hydrodynamic Model

Scientists at CSIRO Marine Research in Hobart developed a three-dimensional hydrodynamic model to simulate the dispersion patterns of effluent if it were discharged from points further offshore than the existing shoreline outfall. The modelling assumed that if an extended outfall (1, 2 or 3 km in length) was to be built it would consist of a diffuser of some length (range of 250m to 1000m), located normal to the coastline on the seabed. CSIRO Marine Research provide a comprehensive description of the modelling and results in a series of technical reports (Andrewartha *et al.* 1998).

The model consisted of a rectangular grid covering waters near Boags Rocks, nested inside a polar grid centred near Boags Rocks and covering Bass Strait (Fig. 3.15). The outer polar grid was forced by sea-level, winds and atmospheric pressure recorded at a number of sites in and near Bass Strait.

The inner rectangular grid was forced by sea-level obtained from the polar grid output, as well as from winds measured at Gunnamatta Surf Life Saving Club, and freshwater (effluent) input from the relevant outfall being simulated. The rectangular grid had a horizontal resolution of 250 m and a vertical resolution near the surface of 2 m. The model included a finite difference tracer advection-diffusion equation as well as a particle tracking module. Both techniques were used to simulate the effluent dispersion, with somewhat different results, which reflect inherent numerical differences between the two approaches.

Figure 3.15. Rectangular grid overlayed on the polar grid covering Bass Strait.



The model was fully calibrated and validated by the program of field studies (March 1997 to March 1998) in the coastal waters off Boags Rocks. This program provided continuous measurement of local wind and in situ currents at two sites (1 and 1.6 km offshore). This was supplemented by contemporaneous tide and sea-level data. In addition, monthly depth profiles of salinity and temperature were taken at seven locations in a grid offshore (Fig. 3.14). Lastly, underway sampling for all common oceanographic parameters was undertaken quarterly over an area extending 5 km either side of the present outfall and 5 km out to sea (Longmore *et al.* 1998).

The model hindcasts measured tides, sea levels and currents very well so that we have a high degree of confidence in its ability to describe the oceanographic regime off Boags Rocks. In addition, the underway sampling provided calibration of the model depictions of plume behaviour in terms of salinity and ammonia concentration contours.

Of course, the complexity of plume behaviour in response to tides, currents and wind makes any attempt to present a “typical” pattern very difficult. However, since we are concerned here with the response of biological assemblages to exposure to effluent, we may take the average and 95th percentile effluent concentration contours as an indication of exposure.

Modelled surface dilution levels of effluent for the present outfall and three notional extensions to 1, 2 and 3 km are shown in Table 3.9. The first value is the average dilution and the bracketed value is the 5th dilution percentile (95th percentile for concentration) in the relevant (250 by 250 m) cell of the model. The actual dilution immediately adjacent to the existing outfall is less than that shown for ‘C0’, because the value provided is for the entire cell and dilution occurs rapidly as the effluent moves away.

Table 3.9. Average and 95th percentile (bracketed) dilution levels of effluent for the existing outfall and three notional extensions to 1, 2 and 3km offshore. (Andrewartha *et al.* 1998).

Cell (250 x250m) where dilution is modelled	Outfall extension lengths			
	Existing	1km	2km	3km
C0 (centreline of outfall, at shore)	23 (14)	114 (49)	253 (102)	442 (165)
C1 (centreline of outfall, 1 km offshore)	149 (47)	38 (25)	153 (72)	355 (143)
C3 (centreline of outfall, 3 km offshore)	860 (169)	536 (121)	249 (84)	90 (61)
C5 (centreline of outfall, 5 km offshore)	3630 (551)	2384 (361)	1286 (217)	734 (170)
EC0 (2km southeast, at shore)	67 (31)	144 (62)	276 (112)	460 (168)
EC1 (2km southeast, 1km offshore)	216 (71)	162 (70)	252 (110)	418 (168)
E1 (5km southeast, 1km offshore)	218 (81)	249 (96)	384 (141)	547 (194)
E3 (5km southeast, 3km offshore)	1720 (388)	1370 (314)	1067 (250)	914 (230)
WC0 (2km northwest, at shore)	105 (35)	190 (66)	303 (113)	491 (179)
WC1 (2km northwest, 1km offshore)	205 (62)	169 (59)	266 (101)	465 (172)
W1 (5km northwest, 1km offshore)	434 (113)	441 (119)	549 (175)	794 (255)
W3 (5km northwest, 3km offshore)	1006 (224)	732 (181)	632 (181)	678 (203)

Note: The results presented were derived using the tracer method, and are surface dilutions for outfall configurations: existing, a, c, and f from Andrewartha *et al.* (1998).

3.6 An Extended Ocean Outfall

The second part of the investigation of an extended outfall was an engineering feasibility study of various outfall designs and their cost. This had to take into consideration that the 56 km pipeline from ETP to Boags Rocks, whilst over-designed in capacity has only a small gravity pressure head over the final length. The costs for various lengths are summarised in Table 3.10, with more detail of the various options provided in the consultant's report (CEE 1998).

Table 3.10. Range of outfall lengths and corresponding costs, based on a common design.

Short Outfall	1.3 km	$26 million
Medium Outfall	1.8 km 2.2 km	$32 million $36 million
Long Outfall	2.7 km 3.1 km	$42 million $46 million

The Boags Rocks site faces one of the most hostile wave climates in Australia with very few days having consistently low wave heights, and sudden increases in wave height to 4 m or more as a result of storms do occur. Hence only large ocean-going vessels can be used in construction, and an outfall must be anchored to the seabed to resist the forces from waves and currents.

The consultant recommended that the outfall be constructed as twin 1.5 m diameter steel pipes, which will convey the design peak flow of 8 m^3/s with good security against movement, overturning or sliding. It was calculated that an extension out to 3.1 km was practicable without additional pumping, provided that the final 10 km of the existing pipeline was sealed in order to increase the head pressure necessary to allow the effluent to discharge at a depth of 32 m below sea level.

The pipeline would extend offshore perpendicular to the depth contours. It is suggested that it be buried across the shoreline and to at least the 5 m depth contour. Further offshore it would utilise a natural break in the offshore reef for its alignment (Fig. 3.16).

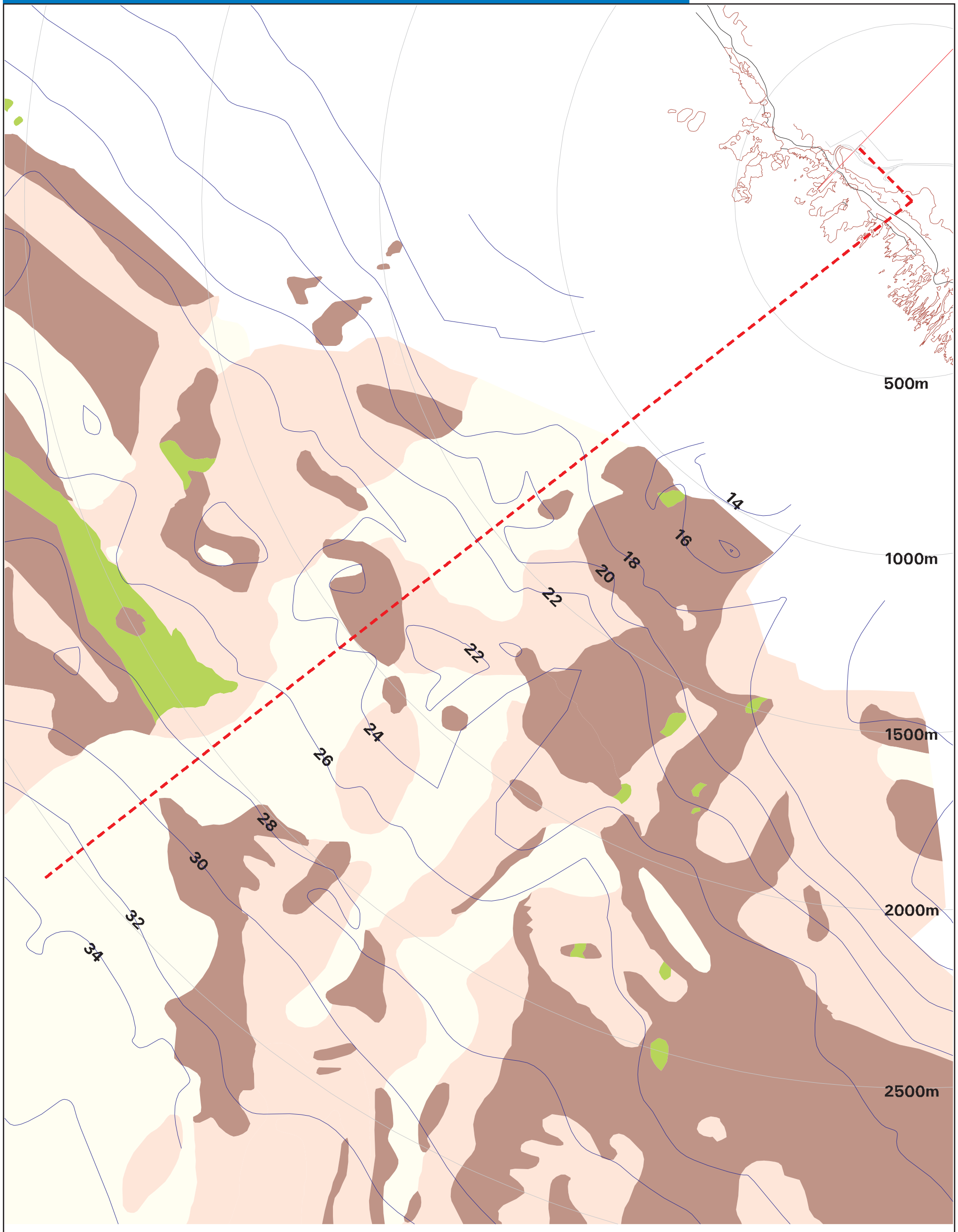
The length of outfall required depends on the extent to which the environmental impact presently exerted at Boags Rocks is to be ameliorated. We can now make some reasonably well-informed estimates of this because of the toxicity tests conducted (Stauber et al. 1998) and the hydrodynamic modelling (Andrewartha *et al.* 1998).

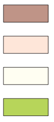
We may usefully compare the exposure (average dilutions) that potential outfall extensions provide in Table 4.9 with the measured responses of several organisms in the bioassay tests conducted by Stauber *et al.* (1998) on ETP whole effluent. A summary of the bioassay results provided in Table 3.11, gives the dilution of effluent at which no observable effect (NOEC) was found for seven taxa.

It can be seen that except for the scallop larvae, no toxicity effect of effluent could be found at dilutions less than those present outside the first few hundred metres from the existing outfall. However the elevated far-field nutrient concentrations produced by the outfall at dilutions up to 400:1 and greater would be expected to affect benthic community composition by changing competitive outcomes, although they would not be considered toxic.

The deleterious effects observed at the Boags Rocks platform are likely to be due to exposure to essentially undiluted effluent in some

Figure 3.16. The path proposed by CEE for extending the outfall.





Notes:
the Bathymetric and side Scan Survey was carried out by Arrowsmith Muir & Assocs, November 1997. Contours were derived from soundings reduced to Chart Datum Port Phillip Heads (add 0.875 mm for AHD) The proposed alignment of the long outfall extension is shown dashed.

Table 3.11. Measured responses in bioassay tests on ETP effluent (Stauber *et al.* 1998).

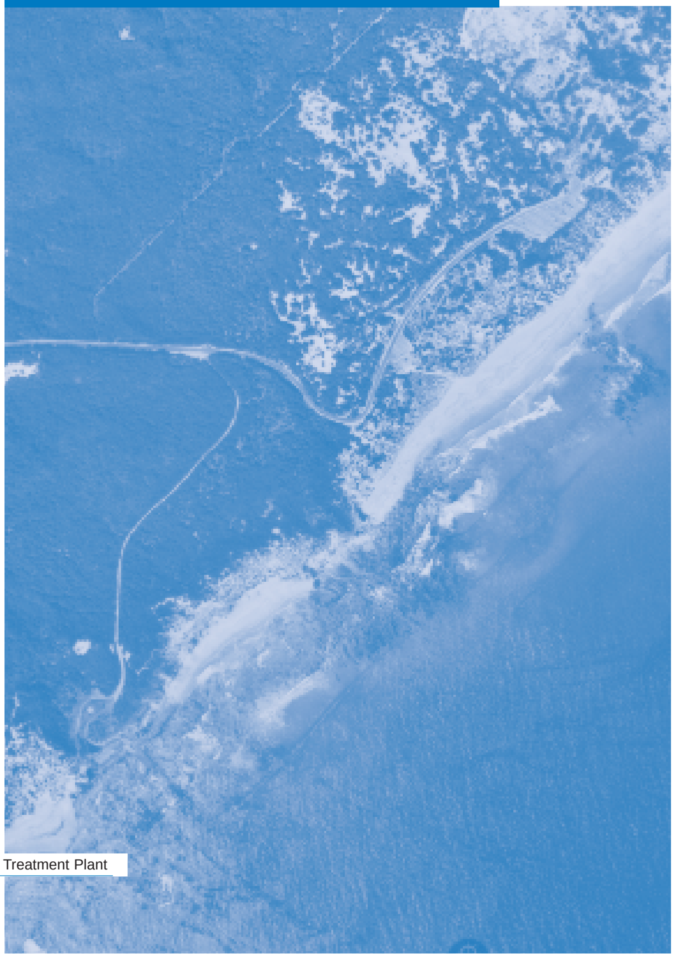
Organism and test	NOEC dilution*
Vibrio (bacteria) (Microtox®)	0
Hormosira (macroalgae) fertilisation	0
Fish larvae mortality	2
Nitszchia (microalgae) growth	8
Hormosira (macroalgae) growth	16
Fish fingerlings mortality	16
Scallop larval development	200

* the dilution of effluent at which there was no observable effect (NOEC)

weather conditions and poorly diluted effluent most of the time. Elsewhere, communities are subjected only to shorter term inundations of more dilute effluent. The test organisms for the bioassays, of course, were subjected to effluent solutions continuously for the duration of the test, usually a few days and the recommended "safe" dilution of 300:1 was based on this. It could be useful to quantify exposure by some combination of average dilution and contact time.

It is clear from Table 3.9 that extension of the outfall produces improvements in dilution at the sites near the outfall (C0, WC0 and EC0), but makes less difference to the already high dilutions at sites W1 and E1, which are 5 km either side of the outfall and about 1 km offshore. As well, the dilutions at or near the termination of the extended outfall would be greater than those presently existing at the end of the current pipe, because the use of a diffuser produces an initial dilution of greater than 50:1 and the buoyancy of the effluent facilitates further mixing.

Aerial perspective of existing outfall taken from 2400 metres (15/3/1996)



3.7 Effluent Flow Reduction (Re-use) Study

CSIRO Land and Water in conjunction with Gutteridge, Haskins and Davey P/L., undertook an investigation of 14 possible approaches to volume reduction (Gomboso *et al.* 1998). Of these, 6 were concerned with reduced input to ETP and 8 with alternative disposal paths for treated effluent from the plant.

The investigation covered engineering feasibility, cost, environmental impact, social acceptance, commercial aspects, government policy and quantitative contribution.

The fourteen options could be ranked in order of this latter aspect as percentage reduction in flow or in order of cost but this rather simplistic approach concealed other important factors listed above. In the end it proved most effective to list options in order of cost, since this largely governs practicability, and then examine each option in the light of the other factors. The options investigated are listed in Table 3.12.

In order to derive a comparative cost basis, all capital, operating, maintenance and/or replacement costs were calculated over a 30 year period. Total costs were then discounted to the beginning of year 1 using a real discount rate of 7% to give a Net Present Value (NPV) for each option. The NPV was then amortised to give an equivalent annuity over the 30 year period again using a 7% discount rate. This annuity was then divided by the flow reduction achieved to give relative cost per kilolitre.

Table 3.12. Effluent re-use and influent flow reduction and re-use options investigated and costed (Gomboso *et al.* 1998).

Option	Cost* ($/kL)	Volume reduction (%)
Water demand management	0.10	1-12
Industrial use of reclaimed water	0.16	< 1
Land Irrigation	0.20	8-10
Aquifer storage (Bridgewater)	0.31	10-20
Diversion to Western Treatment Plant	0.34	1-12
Indirect potable re-use (Cardinia Reservoir)	0.39	≈ 95
Woodlots irrigation	0.42	2
Constructed wetlands	0.59	2-2.5
Detention/sewer mining with local re-use	0.66	0.2
Untreated greywater use	0.72	< 1
Non-potable new lots (third pipe)	0.99	5
Sewer inflow/infiltration reduction	1.93	6
Non-potable retrofit (third pipe)	1.99	5
Treated greywater use	9.86	< 1

*Costs were discounted to the beginning of year 1 (7% discount rate) to give NPV, which was then amortised to give equivalent annuity over 30 year period (7% discount rate) then divided by flow reduction to give cost/kL.

Water demand management, whilst it would reduce input volumes to ETP, would not reduce actual load (ie. the amount of waste products carried in the water would not be reduced). However, reduction in water consumption has environmental and economic benefits in deferring the need for new headworks (eg. new reservoirs or diversions). It is also simple to implement, is favoured by the community and is presently in progress assisted by new pricing structures.

Experience in demand management effect varies. In Britain (UK), subsidised measures produced only a 1% saving whereas in Lismore, NSW a 25% saving was achieved with similar results in Kalgoorlie, WA. (Gomboso *et al.* 1998).

Industrial use of reclaimed water for cooling, cleaning, steam generation and so on offers only a small reduction in flow. The scheme investigated by Gomboso *et al.* (1998) was for supply of treated effluent to the industrial region at Hastings. However it is considered that the demand may be minimal as industry prefer to purchase mains water for critical operations (to minimise corrosion and other problems) or for less critical operations use stored rainwater or their own treated wastewater.

Irrigation of agricultural lands and urban watering is presently practised by both Melbourne Water and the water retailers. Less than 1% of ETP effluent was diverted to irrigators in 1996/97 but there are possibilities for extension if a determined program is implemented taking overall diversion to 10% of flow. This is not only a significant reduction in effluent volume but also offers concomitant benefits in reduced mains water consumption and enhanced agricultural and horticultural production. Unfortunately there are also obstacles related to salinisation, public perception, commercial acceptability of produce and rigorous government regulation, but these could be resolved with time.

Disposal to the Bridgewater aquifer through injection wells offers a means of disposing of a large volume (10-20%) of effluent. The effluent would diffuse through the aquifer sands and enter southern Port Phillip Bay and Bass Strait, as an attenuated front over a very large area. No attempt to assess the impact was made, however it is likely it would result in a large nutrient load entering a fairly pristine area of Port Phillip Bay. While the area is relatively well-flushed, the nutrient would be supplied through sediment via groundwater, and may modify benthic communities. However the objective of the EPA groundwater policy is to restore the quality of the aquifer to beneficial use standard. This means that effluent would need to be tertiary treated prior to injection. The effluent would then become more usable elsewhere.

Diversion to the Western Treatment Plant (WTP), whilst it offers major diversion of effluent, is very capital intensive and would transfer impact to Port Phillip Bay. An increase in the load of nutrients to the Bay is contrary to the recommendations of the Port Phillip Bay Environmental Study (Harris *et al.* 1996). However, improvements in treatment presently in train at WTP are likely to reduce the current load. This reduction may then enable increased loads to be handled without increasing the net impact on Port Phillip Bay.

Gomboso *et al.* (1998) reviewed the proposal to further treat effluent prior to transfer to Cardinia Reservoir as part of an indirect potable re-use scheme. This could provide a total solution, which would eventually lead to no ocean discharge. The disadvantages of this option are public acceptance and disposal of the concentrated residues. Education and a gradual implementation should increase public acceptance. The disposal of the concentrated residues presents an engineering problem but there are solutions to be drawn from treatment of mining waste by, for example, evaporative reduction and burial. The potential for accumulation of total dissolved solids would need to be addressed. The high capital cost of treatment to potable standard and transport costs (approx. $500 million) must be set against the high cost of new water storages and any associated infrastructure. The benefit of this proposal is that any capital expended is there for the long term and also expenditure would occur over a long time frame, perhaps thirty years.

The irrigation of woodlots with a total area of 250 ha could offer about a 2% volume reduction. However at present opportunities along the pipetrack are limited and the high cost of land on the Mornington Peninsula makes this a very hypothetical option. Other considerations are the potential of groundwater contamination, and that the amount of effluent used is inversely proportional to rainfall.

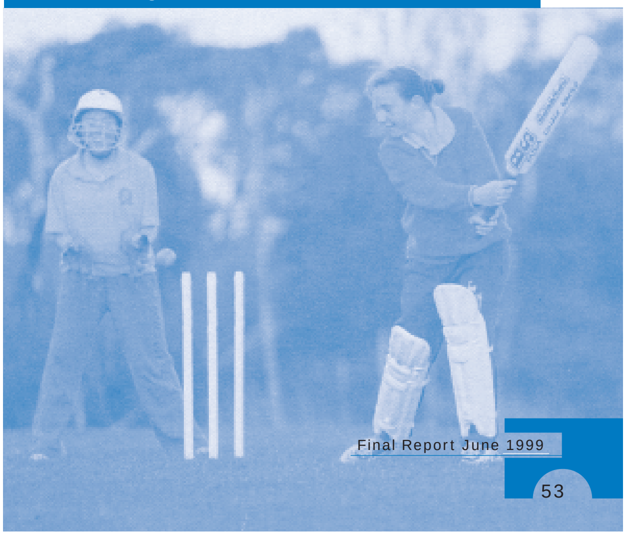
The installation of a third pipe system within existing and developing urban areas, for non-potable re-use was investigated. The water would be used for garden watering, surface washdown and optionally toilet flushing. Gomboso *et al.* reviewed costs of installing the third pipe system as part of the backlog sewer program on the Peninsula and as part of land releases in the Cranbourne area. The cost is moderate and the volume reduction is potentially high but there are concerns about health hazards. More stringent disinfecting might be required to address this latter factor. However this option could be gradually implemented with appropriate safeguards.

Provision of effluent to constructed or reconstituted wetlands has an initial attraction as a means of restoring natural habitat but it carries problems of odour and health risks and the probability that storm events would flush nutrients into Port Phillip Bay. If there were public access to the wetlands the effluent would require further treatment to meet EPA standards. The justification for wetland disposal would be restoration rather than effluent volume reduction and there would need to be flexibility to take account of seasonal factors.

Flow control detention storages retrofitted with small tertiary treatment plants provide the opportunity for local re-use. Gomboso *et al.* considered that they offered very small reductions in volume as they are being installed as a function of sewerage flow management rather than to satisfy demand for effluent re-use. Similarly programs for inflow / infiltration reduction are also being implemented for management reasons. These sewer renewal programs have the potential to reduce flow to ETP by about 6 %, but are expensive and thus are proceeding slowly.

The final option reviewed was greywater re-use. As the potential uptake of this option is very low the scope for volume reduction is less than 1 % in total. The use of greywater also raises potential health problems unless it is disinfected, which increases the cost dramatically. It is unlikely to be attractive to householders who would be better served by their own rainwater tanks or access to a third pipe system.

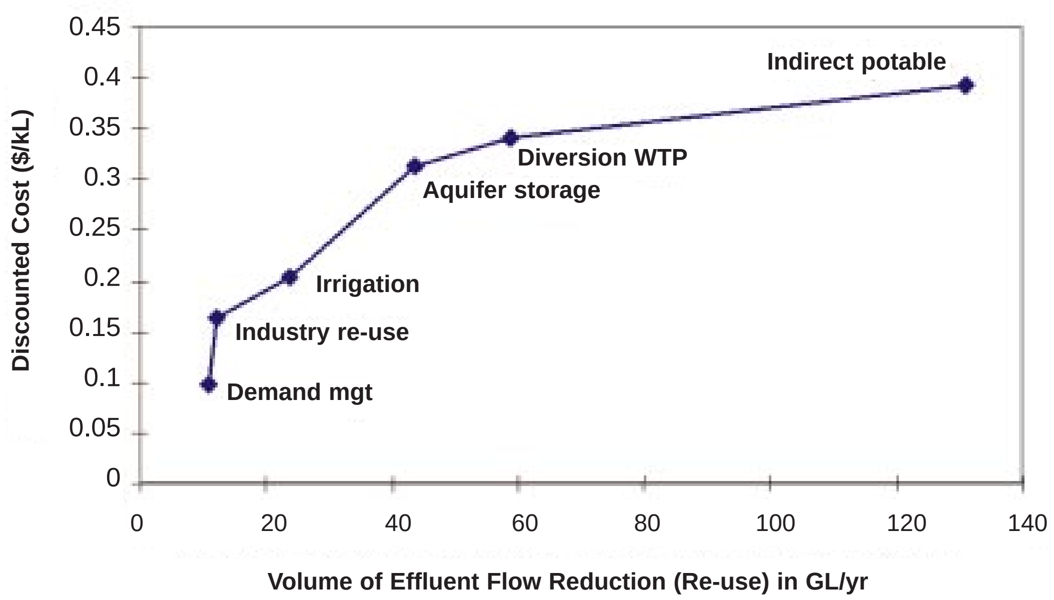
ETP effluent is reused for irrigation for orchards, market gardens and sports ovals like this one at Padua College, Rosebud.



The cumulative effect of the six cheapest options in reducing the volume of ocean discharge is shown in Fig. 3.17. Without indirect potable reuse, the objective of removing total annual flow to the ocean is unlikely. The immediately practicable options of water demand management, extended irrigation and non-potable reticulation offer a total potential volume reduction of 20 - 32%. If inflow / infiltration reduction is continued the total comes to 38% over time. This is a considerable diminution of effluent volume. It should also be considered that acceptance of irrigation and non-potable uses may well depend upon further treatment of ETP effluent for health reasons and this provides a precursor to tertiary treatment for indirect potable re-use.

As well as the social, environmental and financial considerations discussed above there are several important institutional factors to take into account such as pricing arrangements and respective roles between Melbourne Water and the water retailers. Since these are partly determined politically, discussion of them lies outside the scope of this report.

Figure 3.17. The cost curve and cumulative effect of six cheapest options in reducing ocean discharge to zero. (Gomboso *et al.* 1998).



3.8 Treatment Improvement Options

Eastern Treatment Plant (ETP) employs a non-nitrifying activated sludge process, which uses physical and biological processes to treat sewage to secondary quality. A common method of improving this type of treatment process is to implement biological nitrification / denitrification.

Biological nitrification / denitrification involves several groups of microorganisms, metabolising a large number of chemical components in two distinct regimes - aerobic and anaerobic (Fig. 3.18). The introduction of this stage of treatment results in greater nutrient and carbonaceous removal from the sewage stream. In combination with the treatment process currently employed at ETP, nitrification / denitrification would result in quasi-tertiary treatment level.

Melbourne Water retained CMPS&F P / L., consulting engineers, to examine potential process modifications at ETP with the objective of introducing a biological nitrification / denitrification process to reduce nitrogen compounds in the effluent under dry and wet weather conditions.

The feasibility of a nitrification / denitrification process was investigated using two computer models - the BioWin Model (the IAWQ Activated Sludge Model 2) and the Reid Crowther Clarifier Model. Both models were calibrated using the operational data on raw sewage characteristics and sewage flow projections to the plant. The modelling simulated the configuration of the plant to determine whether the existing infrastructure could provide an oxygenated environment and settling capacity to enable nitrification / denitrification to occur and be adequately controlled. The results indicated that the current configuration of the plant was unable to meet the oxygen demand imposed by nitrification / denitrification.

Consequently the consultants produced six options recommending necessary modifications at ETP to make nitrification / denitrification feasible. Three of these were low cost retrofit options, which required attenuation and subsequent treatment of the peak flows for the duration of wet weather events. The other three options included full treatment of peak flows during wet weather events. All options were capable of reducing ammonia from its present value of 26 ppm to 4 ppm and total nitrogen of 34 ppm to 15 ppm.

Figure 3.18. Chemical process for nitrification/denitrification

nitrification | denitrification

$$NH_4^+ \rightarrow NO_2^- \rightarrow NO_3^- \rightarrow NO_2^- \rightarrow N_2$$

ammonium | nitrite | nitrate | nitrite | nitrogen gas

aerobic | anaerobic

The estimated cost of retrofit options ranged from $5 million to $7 million, which mostly reflected the augmentation cost of the existing aeration system. The cost of peak flow management options varied from $35 million to $94 million and in addition to the aeration system upgrade included augmentation of the hydraulic capacity of the plant through constructing additional aeration and sedimentation tanks.

It should be noted that the low cost options, although attractive economically, pose higher risk from an operational and environmental compliance point of view due to the necessity of periodical storage of primary treated sewage. Therefore, feasibility of these options should be verified through a field trial to confirm consistency of plant performance and to explore probable economic benefits.

3.9 Microbiological Health Risk Assessment

Monash University, Department of Epidemiology and Preventive Medicine carried out a microbiological Health Risk Assessment for the waters adjacent to the Outfall (Fairly and Sinclair 1999). The objective was to indicate whether there is a likelihood of disease occurring from swimming and or surfing at these beaches.

Assessment of the risk of disease from swimming in polluted beaches can be derived from either epidemiological studies or risk assessment modelling. Both approaches have their limitations and in the present context, epidemiological studies, which do not try to identify the microorganism responsible for disease, were considered more relevant.

Overall, epidemiological studies conducted elsewhere suggest an increased risk of disease associated with swimming in waters contaminated by sewerage or storm water, although levels of statistical significance were not always reached. The diseases associated with swimming in these contaminated waters are varied and derive from direct contact with the water (e.g. ear, eye and skin conditions) or from ingestion or inhalation of water (e.g. gastroenteritis and respiratory disease). Most studies have concentrated on gastroenteritis due to the primary concern over enteric pathogens in sewage.

There is no general agreement on the indicator organisms that are best able to assess the likelihood of disease although *E.coli, enterococci* or faecal *streptococci* have been most closely correlated with an increase in disease risk. The absolute levels of these indicator organisms found to be associated with disease varies in different studies.

As part of the licence requirements, Melbourne Water carry out routine sampling for *E.coli* at beaches either side of the discharge point every six days. The nature of the relationship between *E.coli* levels and the risk to swimmers is not a simple and direct one, as *E.coli* only indicates the likely presence of other potentially harmful organisms. Consequently the EPA has adopted conservative water quality objectives. Data from the monitoring indicate that the *E.coli* levels are below the standards set by the EPA.

Additional water samples taken during January and February 1998 were tested for *enterococcus* and total coliform counts in addition to *E.coli* at sites where surfers generally swim (i.e. offshore). This data also supported the conclusion that it is very unlikely that swimmers or surfers at Gunnamatta Beach would be at increased risk of illness due to faecal microorganisms in the treated effluent being discharged at Boags Rocks, compared to swimmers or surfers at other ocean beaches (Fairly and Sinclair 1999).

4 Conclusions

The investigation and consultation program began in January 1997, and is now complete. The first part of the Study provided an assessment of the environmental impact and the current environmental performance of the ETP effluent discharge as it affects the receiving waters and their associated ecosystems. The framework of these projects was based on the requirements of the current EPA discharge licence.

The results show that the licence requirements are being met except in two regards. One is exceedance of the permitted levels of undissociated ammonia outside the allowed mixing zone. The other is a failure to determine whether the impact of the treated wastewater discharge on the rocky and sandy intertidal biological assemblages of Bass Strait is increasing or decreasing.

As Table 4.1 shows, measurable impacts of effluent discharge over 23 years are limited. There have been dramatic ecological changes on the intertidal rocky platform (Boags Rocks) at the point of discharge. However, the spread of effect diminishes with distance from the outfall. This is perhaps not surprising in view of the only mild toxicity of the effluent and its rapid mixing with coastal seawater.

Table 4.1. Summary of impact assessment

Criterion	Impact
Biological assemblages	The rocky platform at the discharge point has been denuded of its original brown algal cover (includes loss of *Hormosira banksii* and *Durvilleae potatorum*). A spionid worm *(Boccardia proboscidea)* and several opportunistic green algae and invertebrates have partially occupied the void. No longitudinal gradient of effect has been detected against the diverse assemblages present, which have natural longshore variation on all examined rocky platforms. No temporal trend has been detected on rocky platform biological assemblages because taxa present exhibit high seasonal and year-to-year variation. Intertidal beach sands having low abundance of fauna make longshore comparisons inappropriate. Offshore sand deposits exhibit extreme variability of infauna but provide some evidence of likely effluent effects within 660 m from the point of discharge. The offshore reef displays an abundant flora and fauna but this exhibits longitudinal variability, which confounds effluent effects. The discharge may be a factor affecting the biological characteristics of the offshore reef to a distance of 1100 m, and that a lesser impact may occur out to 1400 m.
Bioaccumulation	No significant bioaccumulation of pollutants was found.
Toxicity assessment	The effluent was non-toxic to bacteria, 1-3 day old fish larvae and macroalgal fertilisation. It was mildly toxic to 4-5 week old fish, and inhibited both diatom and macroalgal growth. It was toxic to scallop larvae. The major cause of effluent toxicity was ammonia. A 300 times dilution of effluent would satisfy internationally accepted guidelines for no hazard to 95% of species. The low salinity of the effluent appears to exacerbate the observed toxic effects.
Receiving water quality	
• toxicants	No heavy metals or common organic pollutants were found above RWQC. Undissociated ammonia levels outside the mixing zone exceeded EPA limits.
• Nutrients	No algal blooms have been recorded - chlorophyll *a* was at normal coastal levels. However, elevated levels of dissolved inorganic nitrogen, sufficient to potentially support phytoplankton blooms and/or to modify benthic plant communities, do at times extend some kilometres from the outfall.
• Oxygen	Dissolved oxygen in the water column was above 90% saturation and often above100% saturation.
• TDS (salinity)	The plume less than 5% from background salinity at times extends offshore beyond the mixing zone (2.3km SE and 1.7km NW by 900m offshore), but never occupies the whole of this area.

The major cause of effluent toxicity was ammonia, however the freshwater nature of the effluent also has some deleterious effect on some taxa. Field studies suggest that mixing of the effluent with Bass Strait water is fairly rapid and salinities low enough to affect macroalgae are present only in the near vicinity of the outfall. The residual chlorine from disinfection of the effluent is vanishingly small (less than 0.1ppm) and comparison of effluent toxicity to three macroalgal species showed only a slight increase in toxicity between chlorinated and un-chlorinated effluent.

With regard to nutrient loads from the outfall, the Study was not able to make any definitive assessment of impacts in the receiving environment due to elevated nutrient levels. The water quality modelling indicated that productivity in the region was likely to be influenced by the elevated nutrient levels, although it is unclear as to how this relates to changes identified in benthic communities.

On the intertidal rock platform at Boags Rocks loss and reduction in original biological assemblages (eg. loss of *Hormosira* and *Durvilleae*) and replacement with opportunistic species (eg. *Boccardia*) is likely to be caused by a combination of low salinities, increased organic load and the toxic effect of high ammonia concentrations. However, the changes in macroalgal assemblages (reduction in *Hormosira* and loss of *Durvilleae*) towards Cape Schanck (Fingals Beach) is likely to be either effects of nutrient loads, ammonia toxicity, freshwater input or a combination of these.

It has been argued that lack of prior knowledge of the coastal ecosystem makes “before-and-after” comparison impossible so that we will never know the full extent of effluent impact. The issue of not being able to identify suitable control sites with which to compare the extent of impact also confounded the situation. Against this however, it may be argued that we have clear evidence of impact at the outfall site but that this impact is much less discernible either side of the outfall, especially to the west. Furthermore, 20 years of rocky platform monitoring shows marked temporal and spatial variability in both algal and invertebrate communities along the coast with no clear trends to indicate the extent of impact and whether it is increasing.

As the effects of the effluent on the near-field environment are partly due to its freshwater nature, reductions in discharge through increased water conservation, effluent reuse or recycling of wastewater should reduce the impact incrementally as the volume is reduced. Reductions in the discharge volume will also reduce the nutrient load therefore reducing the far-field impacts.

Ammonia was shown to be a major cause of toxicity, and is suspected to be a major factor in the near-field impact. Changes in treatment processes that lead to a reduction in volume of ammonia being discharged also have the benefit of reducing the overall nutrient load.

Offshore discharge through an extended outfall with a diffuser providing initial dilution of at least 50:1 would result in lower concentrations of effluent reaching the shoreline. This would lead to an immediate improvement in aesthetic qualities (sight and odour) and should further reduce any risk to human health. However, evidence for health effects is presently largely anecdotal. It should also be noted that the dilution of effluent reaching the shore from an extended outfall would vary depending on the weather. Under onshore winds the plume would be pushed towards the shore.

The water quality modelling indicated that elevated concentrations of nutrients necessary to increase productivity would at times remain within the area of impact even if an extended outfall of 3 km in length were constructed. The Study did not consider the environmental impact the engineering works would have on either the foreshore or the marine environment. Nor was the specific impact on the ecosystem along the diffuser considered.

5 Recommendations

These recommendations have been made for the benefit of the client to help their assessment of various management options. Our recommendations are based upon and relate to specific investigations undertaken and knowledge gained during the course of this study. CSIRO fully appreciates that the ultimate management response to ETP effluent disposal will integrate many social, political, economic and environmental values and issues that are outside the terms of reference for this Study. Thus our advice should be seen as one component of the overall decision-making process.

Table 4.1 of the previous section and Table 5.1 succinctly identify the environmental issues at Boags Rocks and identify management actions that are expected to improve one or more components of the affected ecosystem. It is evident that no single strategy alone will completely ameliorate the cumulative effects of over 20 years of effluent disposal and indeed, each of the management options indicated in Table 5.1 has a potential downside. We therefore believe an appropriate management response is one which attempts to provide a level of environmental restoration and improvement which is acceptable to Melbourne Water, the Victorian EPA and the general community.

Table 5.1. Summary of environmental advantages and disadvantages of disposal options

Options	Advantages	Disadvantages
Outfall Extension with diffuser Approximate costs - $26M for 1.3 km to $46M for 3.1 km extension - level of improvement and rate of recovery likely to be proportional to increased length. .	• Reduces aesthetic impacts (colour, odour) • Reduces impact from freshwater discharge/ ammonia on nearshore rocky platforms • Recovery should be measurable within relatively short space of time (< 5 years)	• Damage to foreshore and reefs from engineering works • No reduction in nutrient load and risk of algal blooms will not be significantly modified • Far-field impacts likely to remain and may increase impacts to offshore subtidal communities • Does not address sustainable water use (is short-term solution only)
Treatment Improvements Approximate costs - $40 to $100M, dependent on flow management adopted, plus additional running costs of $0.2 M pa.	• Will reduce ammonia (the main toxicant) and overall nutrient loads • Far-field impacts reduced - may lead to recovery of rocky platforms southeast of Boags Rocks • Will reduce BOD and suspended solids, facilitating greater potential for re-use options	• Unlikely to see full recovery at Boags Rocks as the freshwater impact (though significantly less than ammonia) will still be present • If ammonia is the major inhibitory factor for phytoplankton growth, its reduction may increase the risk of algal blooms
20% Reduction in flow through re-use initiatives Approximate costs - $100 to 200M, plus up to $10M annually, dependent on initiatives.	• Will reduce both volumes and loads (freshwater, toxicants and nutrients) by 20% • Far-field impacts reduced - may allow some recovery at rocky platforms southeast of Boags Rocks • Consistent with sustainable water use practices	• Only a 20% reduction in nutrient load. • Unlikely to see improvement at Boags Rocks as volume being discharged still significant • Extent of impacts would only decrease with increasing volume reduction.
Indirect Potable Re-Use Approximate costs - $500 M, plus $50 M annually	• Eliminates need for marine discharge of ETP effluent • Marine impacts eliminated and full recovery at rocky platforms southeast of Boags Rocks likely • Consistent with sustainable water use practices	• Disposal of local STP effluent not addressed • Residue from treatment to potable standard creates disposal issues
"Do Nothing"	• under the assumption that the environmental changes have stabilised, then status quo likely to be preserved if effluent quality and volumes remain unchanged.	• Provides no basis for environmental improvement in the receiving environment • Does not address sustainable water use issues

Note: Detailed costing of all options was not provided as part of the study. However, cost estimates have been made, to allow a limited degree of economic comparison.

We cannot anticipate the relative weightings each of these stakeholders places on the components of Table 4.1 and thus we cannot recommend a single preferred option.

To address the environmental concerns identified in Table 4.1 would require a management response that integrated the individual actions identified in Table 5.1. We therefore propose that the following activities be considered:

Outfall Extension - offshore discharge would ameliorate effects inshore to a degree varying with outfall length. Presently, both the subtidal infauna and the reef epibiota show complex patterns of richness and / or diversity with minimal impact from the outfall. It is unclear how the offshore discharge would effect these subtidal communities. Extending the outfall is one disposal strategy that should reduce the current environmental impacts close to the present discharge point.

Treatment Improvements - As ammonia is a significant contributor to toxicity and nutrient load, an ammonia reduction process change at ETP with concomitant peak flow handling changes should be considered. This would have accompanying advantages of reducing BOD and suspended solids thereby facilitating greater potential for re-use. The process improvements are also compatible with future system upgrades necessary to achieve potable reuse.

Flow Reductions - As freshwater is known to be a part contributor to the environmental effects of the effluent, along with the volume of nutrients and concentration of toxicants, a gradual implementation of flow reduction should be pursued. Impact to the marine environment would be avoided altogether if total recycling of wastewater was achieved, but this is unlikely in the short-term. The option of recycling the effluent for potable use should be put on the public agenda for discussion and appraisal as a long-term goal. Important processes for flow reductions include:

- Continuation and enhancement of programs of water demand management.
- Provision of effluent for all forms of irrigation (agriculture, urban watering and wood lots).
- Investigation of the potential for reticulation of effluent for outdoor use on residential, commercial, industrial and recreational premises.
- Continuation of the present programs of inflow / infiltration reduction.
- Monitoring of initiatives elsewhere with respect to potable re-use.

In considering strategies for effluent disposal, it should be recognised that the costs of treatment improvement and re-use schemes will be partly offset by economic return on re-use of water. However, expenditure on an outfall does not provide this advantage.

Whatever action is taken, a monitoring program should be designed and implemented to assess immediate and on-going changes in environmental condition as a result of the management response. The monitoring activity should ensure sufficient spatial and temporal resolution to provide both an 'early-warning' capability of dramatic change over short time and space scales coupled with an ability to detect longer-term trends that can be isolated with confidence from background variation. The program will need to be a combination of monitoring activities, each focussing on a specific habitat and known impact. The design should be such that any impact as a result of alterations in the nature and volume of effluent can be quantified and measured against set objectives.

6 Monitoring

In order to assess whether the changes are providing the desired results (improvement) it will be necessary to have a well designed monitoring program in place. The program should have two focuses. The first should be internal monitoring of the treatment process and environmental impact associated with the operations at ETP. The second should be a program of monitoring that focuses on the coastal impact. The coastal monitoring should be implemented in conjunction with other Statewide coastal monitoring initiatives.

The internal monitoring should ensure that ETP is operating within the constraints of its operating licence. As part of the program the volume and composition of effluent being discharged to the outfall should be analysed regularly. The suite of parameters should be chosen in consultation with the EPA, but should include ammonia on at least a weekly basis. Ideally if ammonia could be measured continuously this would provide a clearer picture of the variability of the effluent and associated toxicity.

A suggested coastal monitoring program is provided in Table 6.1, as an indicative guide only. The final program, designed in consultation with the EPA, would be dependent on any changes proposed as part of the Effluent Management Strategy for ETP.

The subtidal sands and reef were surveyed for the first time as part of this Study, which means they provided only a single snapshot on which to base an assessment of impact. In order to support that initial assessment and allow continued change to be identified, further subtidal surveys are required. Initially the frequency should be at least annual, however this could be reduced dependent on results found. Ideally the offshore monitoring program should be dovetailed into a Victorian coastal monitoring program. The combined results would provide a greater knowledge base upon which to assess background (natural) variation.

The numerous intertidal rocky platform surveys that have been carried out over the years provide a significant historical background. However the difficulty of site comparisons and the ability to use these types of surveys to assess the extent of impact has been well documented by Quinn and Haynes (1996) and Shao (1997). Rather than using platforms to assess the extent of impact, they should be used to assess the rate of change, and whether that change is related to the discharge of effluent. An alternative to the sampling surveys for assessing the level of biological impact would be to use the life form of a single algal species, which exhibits measurable ranges of effluent impact.

The issue of bioaccumulation of toxicants in seafood is of continuing concern to the community. Though the Study concluded that bioaccumulation was insignificant, it was acknowledged that further studies of this nature may be necessary to satisfy public concern over risk to the quality of seafood. One program to be considered, would be the use of the cultured mussel (*Mytilus edulis*), which is a well studied filter feeder often used to assess the risk of bioaccumulation. The results could also be compared to other “mussel watch programs” for outfalls. It should be noted, however, that the existing licence requires one commercially and one recreationally harvested species from near the outfall to be assessed for bioaccumulation.

The rapid dilution of effluent from the outfall is most relevant to the adjacent beach users. To assess that appropriate mixing is occurring, water sampling in conjunction with the beach bacteriological monitoring program should be considered. The existing program consists of water samples at beaches either side of the outfall (2 northwest and 4 southeast) being tested for *E.coli* every six days. Consideration should be given to including the measurement of ammonium, salinity and pH in each sample.

Of course the dilution at fixed points along the coast is dependent on the volume of discharge, the tide and the weather conditions at the time of sampling. However, over an extended period, the data would provide a means for establishing the level of dilution being achieved along the coast.

To monitor water quality offshore from Boags Rocks involves logistical problems. Sampling by boat is weather dependent, infrequent and unlikely to coincide with times of increased phytoplankton productivity (key criteria of water quality). The alternative is to moor automated measuring devices at strategic points offshore from the outfall. These devices would measure temperature, salinity, and chlorophyll fluorescence (indicator for phytoplankton biomass) within the water column. The deployments would need to remain in place for extended periods of time, in order to obtain a significant time series of data. The results would also provide another indication of the efficiency of effluent mixing in the region. For this reason it may be appropriate to place one on the boundary of the nutrient mixing zone, and one about 2 km to the southeast.

Table 6.1. Indicative coastal monitoring program

Area of concern	Possible Method	Frequency
Biological Monitoring		
Survey of intertidal rocky platforms to measure temporal variation on each platform surveyed.	Survey of algal species (relative abundance and percentage cover) and macroinvertebrates (abundance) in continuous one metre wide transects crossing the platform from high water mark to low. As an alternative or in addition low level aerial photography may provide the means for mapping habitat change on a platform wide scale.	Annual
Subtidal offshore reef survey	Surveys of animals and plants along a series of 100m transects. (Based on same technique used for this Study.)	Annual
Subtidal infauna survey	Sampling of soft sediment between shoreline and offshore reef, for macroinvertebrates. Grain size and TOM to be included. (Based on same technique used for this Study.)	Annual
Bioaccumulation		
Bioaccumulation in receiving waters	Study using transplanted cultured mussels *(Mytilus edulis)* to assess the risk of bioaccumulation of toxicants.	3 years
Receiving Water Quality		
Bacteriological quality of bathing waters adjacent to point of discharge	*E.coli* (or other indicator) monitoring program - water samples at series of beaches either side of outfall	Every 6 days
Assessment of longshore mixing	In combination with the bacteriological monitoring, water samples be analysed for ammonium, salinity, temperature and pH.	Every 6 days
Assessment of water quality	Deployment of automated instruments measuring S, T, and fluorescence at two sites offshore from Boags Rocks for two consecutive periods of 6 weeks.	3 years (summer)

7 Glossary

Acute: Severe and short lived.

Aerobic: Presence of free oxygen in chemical process.

Algae: Large group of non-flowering plants, many microscopic, generally containing chlorophyll. Most algae are aquatic.

Algal bloom: Microalgae occurring in dense numbers in a water body, as a result of favourable conditions (ie. nutrient enrichment).

Ammonia: Compound consisting of a single nitrogen atom coupled with three hydrogen atoms. It is a nitrogen source for algae.

Ammonium: The positively charged cation formed when ammonia is neutralised. It is a nitrogen source for algae.

Anaerobic: A process conducted in the absence of free oxygen

ANOVA: Analysis of Variance. Statistical technique for testing the equality of several population means.

Anoxic: Devoid of oxygen.

Autocorrelation: Statistical measure of the degree of association between pairs of points separated by some fixed increment of time.

Benthic: Belonging to the sea floor.

Benthos: Organisms living on or in association with the sea floor.

Bioaccumulation: Concentration of substances (especially toxicants) in the tissues of plants and animals.

Biochemical: Chemical reactions occurring in living organisms.

Biodiversity: Measure of the number of species inhabiting a given area.

Biomass: The living weight of animal or plant populations or communities.

Biota: All living organisms of a region.

BOD: Biochemical Oxygen Demand. Measure of the amount of oxygen required by bacteria and other microorganisms engaged in breaking down organic matter.

Chlorophyll: Green pigments of plants, which capture and use the energy from the sun to drive the photosynthesis process.

Chronic: Over a long portion of the organism's life span. Less severe and generally over a longer time span than "acute".

Conductivity: Electrical conductivity - the capacity of water to conduct electrical current; used to measure level of salinity.

Denitrification: Conversion of bound nitrogen to elemental (gaseous) form.

Detection limit: Minimum level of quantification for a particular analytical method.

Diatom: Variety of microalga that has a siliceous skeleton.

Dioxins, furans: Toxic compounds, which are byproducts of the manufacturing process of herbicides and disinfectants, but also derived from other industrial processes.

Diversity: In statistical terms, a measure of the distribution of items across a set of categories

E.coli (Escherichia coli): Bacteria/bacterium found in the stomachs of mammals (eg. humans) and used as an indicator of recent faecal contamination.

Ecotoxicology: Study of the fate and adverse effects of chemicals on the ecosystem.

Endocrine disruptor: Substance which has the capacity to disrupt the normal functioning of the endocrine glands in animals.

EPA: In this report the EPA refers to the Environment Protection Authority, which is a statutory body established under an Act of the Victorian Parliament.

Epidemiology: Science of statistically evaluating and dealing with diseases in populations.

ETP: Eastern Treatment Plant (located at Carrum, Victoria).

Eutrophic: Having an unnaturally high content of algae due to excess nutrients.

Evenness: In statistical terms, a measure of the relative distribution of items across a set of categories; 'bunching' of items in a few categories yields low evenness.

Grazing: Eating of plants by animals. In water the term is associated with zooplankton grazing on phytoplankton.

Guideline values: Values (often concentrations) thought to represent safe conditions and chosen as a result of available research.

Heavy metals: General term for cadmium, chromium, copper, iron, mercury, nickel, manganese, lead, zinc, arsenic and selenium.

Indicator species: Animals or plants which indicate an effect or impact. eg. pollution or loss of habitat.

Infauna: Fauna that lives within benthic sediment.

Invertebrate: Animals without a backbone.

Macroalgae: Large algae, in this report used to describe kelps and larger seaweeds.

Mesotrophic: Water body that has moderate nutrient and algal levels.

Microalgae: Single-celled plants. Less than $\frac{1}{10}$th millimetre in length or diameter.

Model: Mathematical equation or series of equations that provides simplified description of system or situation devised to facilitate calculations or predictions. With use of a computer, can provide simulation of large scale environmental processes, eg. hydrodynamic model.

Multi-Dimensional Scaling (MDS): multivariate statistical technique that attempts to portray relationships in high-dimensional space using much fewer (typically two or three) dimensions.

Multivariate: Consisting of several varying factors.

Nitrate: The NO_3 anion.

Nitrification: Formation of nitrate from reduced forms of nitrogen.

Nitrite: The NO_2 anion.

Nutrients: Substances (eg. nitrogen and phosphorus in various forms) required for the growth of plants (like fertiliser).

Oligotrophic: Water body that has low nutrient and algal levels.

Organochlorines: Complex organic molecules with chlorine atoms attached (eg. many pesticides).

Organophosphates: Group of pesticides chemicals containing phosphorus, which are intended to kill insects.

Oxic: Having oxygen present.

Particulates: Particles suspended in water.

pH: The negative logarithim of the hydrogen ion concentration; an index of acidity or alkalinity.

Photooxidation: Breakdown of a chemical by oxidation in the presence of sunlight

Photosynthesis: Transformation of carbon dioxide and water to organic matter and oxygen by means of light energy.

Phthalate esters: Substance used in the manufacturer of plastics to give flexibility. Given time they leach from the plastic, and become an environmental pollutant.

Phytoplankton: Microalgae that live in the water column.

Polychaete: General term for a class of segmented worms with several seta (bristles) per segment, which are widespread in the marine environment.

Productivity: Creation of living matter out of inorganic or inanimate matter. In this report, primary productivity in the ocean is the making of organic matter from carbon dioxide, nutrients and water by algae living in the marine environment.

Receiving waters: Waters that receive effluent from a particular source.

RWQC: Recommended Water Quality Criteria as set by the Victorian EPA

Salinity: The salt content of seawater.

Sediment: Any solid material in water that sinks to the bottom.

Sewage: Strictly speaking household waste but loosely applied to any waste sent to a treatment plant.

Sewage Treatment Plant: A place where human and industrial wastes are treated before disposal to land or water.

> **Primary treatment:** Screening the solids from the water and allowing organic matter and suspended solids to settle. This treatment typically removes one third of the BOD and two third of suspended solids.
>
> **Secondary treatment:** Achieves stabilisation of biodegradable material through biological degradation. This treatment typically removes 85-90 % of the BOD.
>
> **Tertiary treatment:** Typically removes nutrients (nitrogen and phosphorus) and the remaining small volume of organic matter and organisms.

Similarity Index: Statistical measure of the degree of 'closeness' of two groups which have been measured on a number of attributes.

Standard error: Statistical measure of the uncertainty associated with an estimate of a true value.

Stress: Statistical measure of the 'goodness of fit' between the placement of groups in a multi-dimensional scaling (MDS) with a given number of dimensions; lowering the number of available dimensions increases stress; stress values range from 0 to near 1

Suspended solids: See particulates.

Suspension feeder: Animal which lives by filtering particles from the water.

Synergistic interaction: Interaction between two or more factors that influence a process in the same direction; their combined effects may be equal to or greater than the sum of the two effects alone.

Taxa (taxon): General term for a class, order, species etc. of animals or plants.

Toxic: Poisonous.

Toxicant: A poison.

Toxicity: Level or concentration of toxicant to create a toxic response.

Trophic: Related to food chains and food webs.

US EPA: United States Environment Protection Agency.

Vertebrates: Animals with backbones.

WTP: Western Treatment Plant (located at Werribee, Victoria).

Zoobenthos: Animals living on the sea floor.

Zooplankton: Small animals living in the water column, usually drifting with the water.

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This is the final report for a major environmental study managed and coordinated by the CSIRO Environmental Projects Office. The study commenced in January 1997, cost $1.3 million dollars, and incorporated research undertaken by scientists and engineers in CSIRO as well as local universities, government agencies and private consultants. The study was part of the Eastern Treatment Plant Effluent Management Study and was funded by Melbourne Water Corporation.

The study investigated the impact of effluent disposal from the Eastern Treatment Plant on the coastal environment in the area of Boags Rocks, near Cape Schanck. It included biological monitoring, bioaccumulation assessment, toxicology, evaluation of water quality and the development of integrated hydrodynamic and nutrient models. In addition the study evaluated effluent re-use opportunities and benefits of extending the existing shoreline outfall to a point further offshore.

This reports draws together the knowledge gained from the individual research tasks, and provides input for the development of an Effluent Management Strategy for Eastern Treatment Plant.

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