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Groundwater Nutrient and Toxicant Inputs to Port Phillip Bay

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EXECUTIVE SUMMARY

The Port Phillip Bay Environmental Study was announced in 1991 to address the need to manage the recreational, commercial and environmental impacts on the Bay in a sustainable manner. HydroTechnology was contracted by the study team to investigate the inputs of groundwater nutrients and toxicants to Port Phillip Bay.

The groundwater component of the Port Phillip Bay Environmental Study is to be investigated in two phases. These comprise:

(1) Determination of the quality and volume of groundwater flowing into the Bay from the northern Werribee Delta, Nepean Peninsula, Western Suburbs and Geelong-North Shore areas. This involves hydrogeological and land-use characterisation and groundwater flow and solute calculations; and,

(2) Determination of the existing and likely future levels of nutrient and/or toxicant contamination of all groundwater systems discharging directly, or via streams, to the Bay. This stage will be undertaken if it is considered, by the study team, that the risk to the Bay identified from the four nominated areas in phase 1 is significant.

This report contains details of the phase 1 investigation only.

Previous studies of the hydrogeology of Port Phillip Bay focussed largely on the groundwater resources and extent of contamination of the aquifer systems. The only known study to provide estimates of nutrient or toxicant loads into Port Phillip Bay was that commenced by Anderson (1988) and concluded by CPI (1992), which quantified the level of nutrients discharging to the Bay via groundwater contaminated by the Werribee Treatment Farm (Otto, 1992).

The approach taken in the first stage of this study was one of delineation of potential contamination areas, collation of data, characterisation of the hydrogeology of the four areas, land-use assessment, modelling of the groundwater flows and contaminant loads and overall assessment of the results, with a comparison to other known inputs to the Bay.

Available data has allowed estimates of nutrient or toxicant loads to be calculated for three of these study zones (northern Werribee Delta, Nepean Peninsula and Western Suburbs) (Summary Table 1). Insufficient data has precluded the calculation of reliable toxicant loads for the Geelong-North Shore area.

Summary Table 1 shows that, in terms of nutrients, groundwater appears to play only a minor role in total contributions to Port Phillip Bay. However, heavy metal values, in particular cadmium and lead, appear to be high when compared to estimated inputs from surface water systems (Summary Table 2). The high groundwater loads may be a function of the non-representativeness of some of the groundwater analyses in the western suburbs. The bores have historically tended to be sited near known polluting activities, yielding a biased result. The Geelong-North Shore area would be expected to yield similar values for toxicant loads as those from the Western suburbs. This assumption is reinforced by the prominence of Cowies Creek as a source of surface water toxicants (NSR, 1993). Groundwater discharge to Cowies Creek may explain the high toxicant levels encountered.

Summary Table 1. Summary of groundwater nutrient and toxicant loads to Port Phillip Bay

	Nutrients		Toxicants			
	N (t/yr)	P (t/yr)	Cd (t/yr)	Pb (t/yr)	Fe (t/yr)	Phenols (t/yr)
North-east Werribee Delta (Groundwater)	10 - 24	1 - 3				
South-west Werribee Delta (Groundwater)	20	3				
Nepean Peninsula (Groundwater)	18.7	3				
Western Suburbs (Groundwater)	12.2 - 16.9	1.7 - 2.4	0.2 - 0.3	1.0 - 1.4	136.3 - 187.9	6.1 - 8.4

Summary Table 2. Summary of surface nutrient and toxicant loads to Port Phillip Bay. [From NSR, 1993 (*) and CPI, 1992 (#)]

	Nutrients		Toxicants		
	N (t/yr)	P (t/yr)	Cd (t/yr)	Pb (t/yr)	Fe (t/yr)
Patterson River *	516	333	0.08	7.13	1073.33
Mordialloc Main Drain *	332	269	0.03	3.29	445.13
Mordialloc Settlement Drain *	223	113	0.07	7.65	870.07
Kororoit Creek *	103	167	0.02	0.49	62.91
Werribee Treatment Complex #	4000	1200			

An issue of concern is the amount of groundwater estimated to flow from the western suburbs Newer Volcanics aquifer into rivers and streams such as the Maribyrnong River and Kororoit Creek. Only approximately 2% to 12% of groundwater flows directly into the Bay from the aquifer in this location, the remainder flowing to the Bay via surface streams. The total loads, including the baseflow component to streams, is reported above. The small percentage of groundwater flow directly to the Bay may indicate that surface water monitoring of the streams may detect a majority of the toxicant loads entering the Bay. However, to be effective, this monitoring needs to be as close to the stream mouth as practicable. Physical effects from the estuary, the Bay or tides may make this monitoring extremely complex.

Nevertheless, a number of factors render the toxicant loads important in the overall input of contaminants to the Bay. These are:

- Rate of migration. From numerical and analytical modelling of hydrodynamic dispersion and solute transport (Section 3.6), it appears that the solute can migrate at rates between 500 and 1000 m per year. This means that there is the possibility of a large "contaminant front" discharging to the Bay in the future. To further quantify the rate of movement and likely extent of contamination with time, the migration of solutes in the Newer Volcanics aquifer needs further investigation.
- Sufficiency of data. The Geelong-North Shore area suffers from a lack of hard data with which to estimate the toxicant risk to the Bay. Establishment of monitoring bores and field testing may be warranted. There is a significant volume of groundwater, soils and geotechnical data in existence which has been collected for site assessments of local industry in the western suburbs and Geelong-North Shore areas. Some of this information has been accessible for this study, but it is suspected that there are many more investigations which are not known to the EPA or the study team.
- Surface water monitoring. Monitoring of toxicant and nutrient loads to Port Phillip Bay from streams must be sited to also pick up the groundwater baseflow which contributes to the streams. This is particularly important in the western suburbs and Geelong-North Shore areas. Approximately 90% of groundwater discharge to the Bay from the Newer Volcanics aquifer of the western suburbs has been found to occur via initial discharge to streams. Existing stream monitoring sites would miss the important groundwater baseflow component.
- Representativeness of monitoring bores. The groundwater data collated from the western suburbs was originally collected for studies of known polluting areas or for specialist geotechnical studies. The issue of the extent of a large scale, combined, contaminant plume was not identified as one worthy of monitoring. As a result, given the heterogeneity of the aquifer, it is not possible to accurately estimate the extent and path of any large scale contaminant plumes.

Recommendations for future works to further quantify the toxicant loads are suggested below:

- Surface water monitoring needs to be established in the streams traversing the western suburbs and Geelong-North Shore areas to pick up the important groundwater baseflow component. Sampling of the stream estuary at various depths will detect toxicant loads discharging to the Bay.

- Data needs to be collected to more accurately characterise the interaction of the Newer Volcanics aquifer with the streams of the western suburbs. A number of trial sites should be established adjacent to streams or between streams and known contaminating sites, monitoring hydraulic gradients, groundwater velocity, groundwater quality and dispersion parameters. A site may also need to be established along the coastal margin of the western suburbs (Altona - Williamstown) to trace the bulk water quality changes and the subsequent discharge to the Bay.

- The migration of solutes in the Newer Volcanics Basalt aquifer needs further investigation. More detailed solute transport modelling of the abovementioned trial sites, in conjunction with field observations, is required. This is suggested to enable more realistic estimates of toxicant migration rates, salt water interfaces and solute flow paths. Such a study would focus on the trial site(s) in the highest priority area, the western suburbs. The data produced may also be of relevance to the basalt areas of the Geelong-North Shore area.

- Groundwater monitoring sites need to be established in the Geelong-North Shore area to define the aquifer morphology, aquifer parameters and to provide a means of monitoring variations in chemistry and water level. Suggested sites would be along the coastal margin.

1.0 INTRODUCTION

Port Phillip Bay provides a range of recreational and commercial activities to the people of Melbourne, Geelong and other communities. Urban, agricultural and industrial activity within the Port Phillip catchment has contributed a variety of chemical contaminants to the Bay via streams, drains, urban and agricultural runoff, the atmosphere, groundwater and point-source outlets.

Inputs to Port Phillip Bay must be managed to ensure that beneficial uses are maximised and the Bay environment is not further degraded. The Port Phillip Bay Environmental Study was announced in 1991 to address the need to manage the recreational, commercial and environmental interests in a sustainable manner. HydroTechnology was contracted by the managers of the Port Phillip Bay Environmental Study, CSIRO, to investigate the inputs of groundwater nutrients and toxicants to part of Port Phillip Bay.

1.1 Objectives

The objectives of the groundwater component of the Port Phillip Bay Environmental Study are:

1. To utilise existing data and information to determine the quality and volume of groundwater flowing into the Bay from the following areas (refer Figure 1.1):

Northern Werribee Delta	Nutrients
Nepean Peninsula	Nutrients
Western Suburbs	Toxicants
Geelong-North Shore	Toxicants

including that component which may be the result of groundwater discharge to adjacent streams.

2. To determine the existing and likely future levels of nutrient and/or toxicant contamination to Port Phillip Bay from all groundwater systems discharging directly to the Bay, or via streams.

Stage 1 of the study aims to evaluate the effect on the Bay of contaminated groundwater from the sources stated in the first objective. The second objective will be addressed if it is considered, by the study team, that the risk to the Bay from the four nominated areas is considerable.

It must be stressed that stage 1 of the study is based solely on existing data and information. Additional field work, including sampling and analysis, may be undertaken if warranted from the results of stage 1.

This component of the Port Phillip Bay Environmental Study addresses tasks N1.4 and T4 as identified by the study team (Otto, 1992).

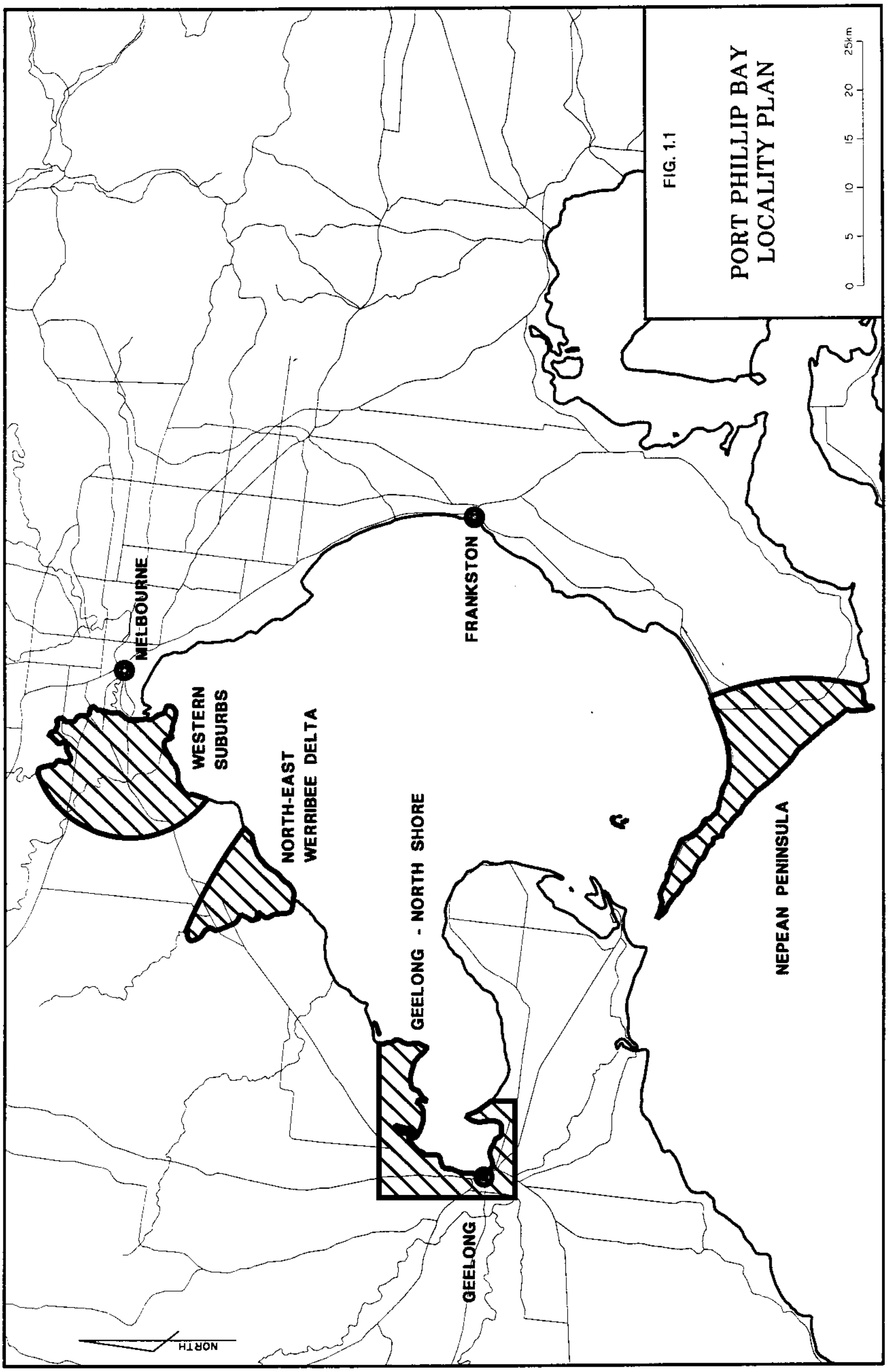


Figure 1.1 Port Phillip Bay Locality Plan

1.2 Previous Work

The hydrogeology of the Port Phillip Bay area has been studied to varying degrees by different workers over the last 30 years. Riha (1971a, 1971b, 1975, 1976) and Riha and Kenley (1978) studied the hydrogeology and contamination potential of the Newer Volcanics aquifer to the west of Melbourne. Shugg (1975, 1976, 1981, 1986, 1987a, 1987b, 1987c, 1989) has conducted numerous studies on waste disposal and pollution of aquifers in the region, including reviews of the Sandridge site at Port Melbourne, the Mornington Peninsula and waste disposal sites around the metropolitan area. Walker (1990) undertook an investigation into interactions between the Carrum wetlands and the groundwater system. Thompson and Harris (1972) conducted an early investigation into the hydrogeology of the south-eastern suburbs of Melbourne, while Anderson (1988), CPI (1992) and Stewart (1988) investigated different portions of the Werribee Delta aquifer and environs. Hydrogeological overviews have been produced by Leonard (1979, 1992) which provide an insight to the broad geology and hydrogeology of the Port Phillip area. Due to its high level of development, a need for groundwater supplies, waste disposal sites and the presence of contaminated sites, the area has also been the target of numerous hydrogeological studies by groundwater and environmental consultants. The only known study to provide estimates of nutrient or toxicant loads into Port Phillip Bay is that commenced by Anderson (1988) and concluded by CPI (1992), which quantified the level of nutrients discharging to the Bay via groundwater contaminated by the Werribee Treatment Farm (Otto, 1992).

1.3 Methodology

The approach taken in the first stage of this study by HydroTechnology comprises the following steps:

- Delineation of areas, based on existing information, where it is considered that the greatest risk to the Bay from contaminated groundwater discharge may occur. This process was built into the original contract proposal.

- Extraction, collection and collation of hydrogeological data on the four areas. Sources of data have included the Rural Water Corporation (RWC) Statewide Groundwater Database, Melbourne Water, Coffey Partners International (CPI), Environment Protection Authority (EPA) and private sources.

- Characterisation of the hydrogeology of the four areas. An assessment of the main aquifers at risk, their typical water quality, hydraulic conductivity, storativity, relationship to the Bay and groundwater flow directions was undertaken.

- Assessment of land use and potential pollutants to each aquifer. Evaluative techniques such as the LeGrand system (1980) have been employed in this study

- Analytical modelling, where sufficient data was available, combining potentiometric details, aquifer thickness, discharge segments to the Bay and water quality was undertaken to quantify the nutrient or toxicant loads discharging to Port Phillip Bay.

- Discussion of the results, placing them in a context which reflects the relative importance of the discharging loads to the Bay. Additional calculations and sensitivity analysis have been referred to.

Finally, conclusions of the results from the first stage of the study have been drawn in this report. Recommendations regarding contamination control strategies, areas at risk and areas for further investigation are made.

2.0 NUTRIENT INPUTS - NORTHERN WERRIBEE DELTA

2.1 Location

The Werribee Delta is located at the mouth of the Werribee River, on the north western side of Port Phillip Bay. It covers an area of approximately 117 km^2 (Leonard, 1992) and receives an average annual rainfall of 518 mm. The Delta is bisected by the Werribee River (Figure 2.1).

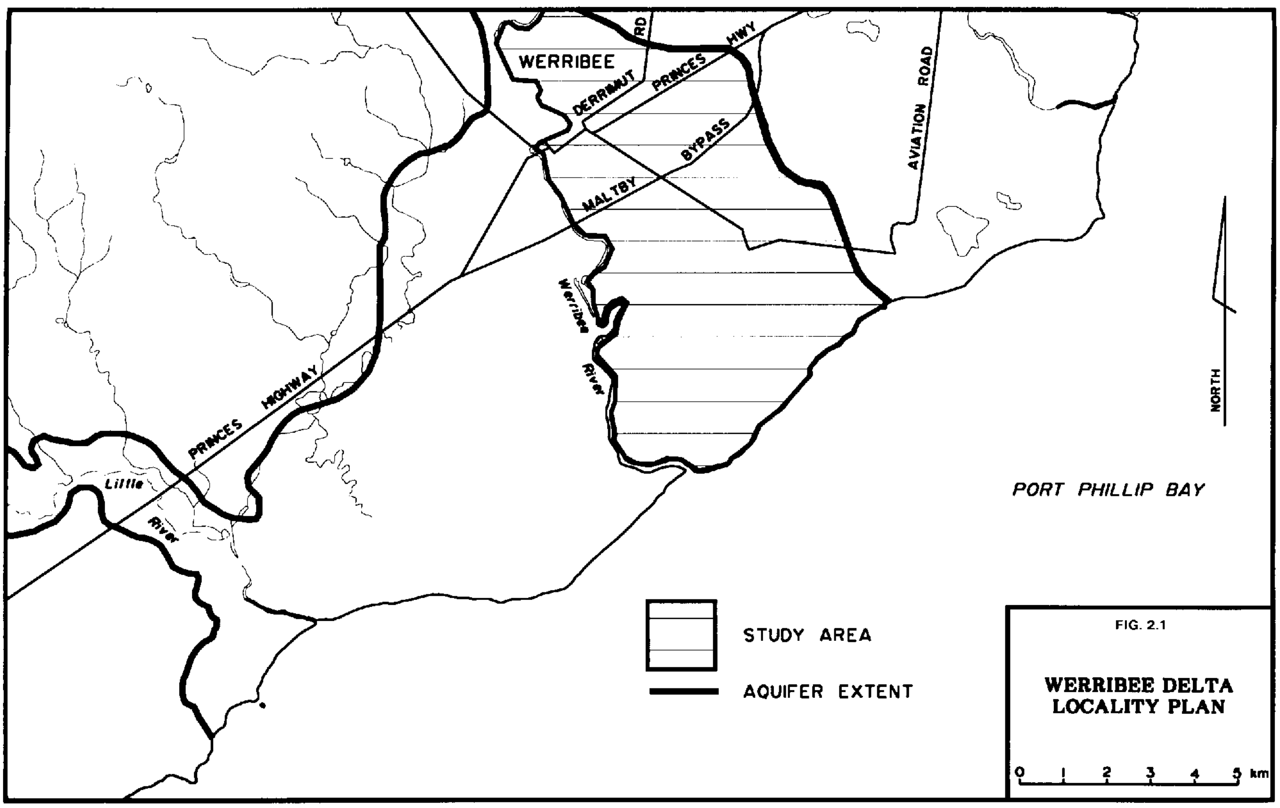


FIG. 2.1
WERRIBEE DELTA LOCALITY PLAN

2.2 Land Use

The area to the southwest of the Werribee River is largely occupied by the Werribee Treatment Complex (WTC), where urban sewage is treated and processed water is discharged, via drains, to Port Phillip Bay. Anderson (1988) and CPI (1992) quantified groundwater flow and nutrient loads to Port Phillip Bay from this section of the Werribee Delta. Results from these studies are later described and compared with results from the north-eastern sector of the delta.

The area to the north-east of the Werribee River is largely devoted to agricultural land uses. Pasture and horticulture predominate, with a significant number of landholders extracting groundwater for irrigation purposes from the Werribee Delta aquifer.

2.3 Hydrogeology

The Werribee Delta aquifer and the Newer Volcanics basalt aquifer are to be addressed in this section. These aquifers overlie older, deeper aquifers which are not considered important in transmitting groundwater to the Bay. The aquifers present in the Werribee Delta area are dealt with in detail in Leonard (1979 and 1992) and are summarised briefly in Table 2.1.

The two uppermost aquifers (Figure 2.2), being the conduits for groundwater recharge and agricultural nutrient infiltration are considered to be the aquifers of prime importance in nutrient transport to the Bay in this area. The deeper aquifer of the Brighton Group is believed to contribute relatively little nutrient load to the Bay because the unit is confined above by clays, silt and the chilled base of the overlying basalt (Leonard 1992). Vertical connection between the overlying Newer Volcanics and the Werribee Delta sediments does occur, which is evidenced in the hydrographs of bores in the study area (Appendix A). This vertical connection which occurs between the two uppermost aquifers results in mixing of groundwater from the Newer Volcanics aquifer and the Werribee Delta sediments and supplies a pathway through which nutrient rich groundwaters may flow.

Figure 2.2 represents an east-west cross section along the coastal margin of the delta.

TABLE 2.1. Hydrogeological summary of the aquifers in the Werribee Delta area (After Leonard, 1992).

Aquifer Name	Lithology	Aquifer Type	Water Salinity (mg/L)	Nutrient Status
Werribee Delta (Uppermost)	Silt, sand, gravel, clay; silty sand in lower parts of deposit; minor gravel and sand levees.	Unconfined to semi-confined porous media aquifer of shoestring sands and gravels interbedded with low permeability clays.	500 - 6000	High
Newer Volcanics	Basalt, scoria, pyroclastics.	Multi-layered fractured rock aquifer system with sheet-like basalt aquifers separated by thin clay layers of low hydraulic conductivity; Upper aquifer is unconfined, lower aquifers are semi-confined to confined.	500 - 25000	High
Brighton Group	Sand, clayey sand, sandy clay, gravel, quartzite, sandy limestone.	Sand and silt sheet-like aquifer; Confined to semi-unconfined to the west of Melbourne.	2500 - 8000	Low
Werribee Formation	Sand, gravel, clay, coal, lignite.	Semi-confined to confined aquifer of sand and gravel; sheet-like formation; lignite and coal beds impart anisotropy in hydraulic conductivity.	2500 - 11000	Low
Silurian Metasediments (Bedrock)	Folded mudstones, siltstones, sandstones and shales, affected to varying degrees of regional and contact metamorphism.	Fractured rock aquifers. Semi-confined to confined. Physical properties of aquifer varies according to degree of fracturing and weathering.	1000 - 5000	Low

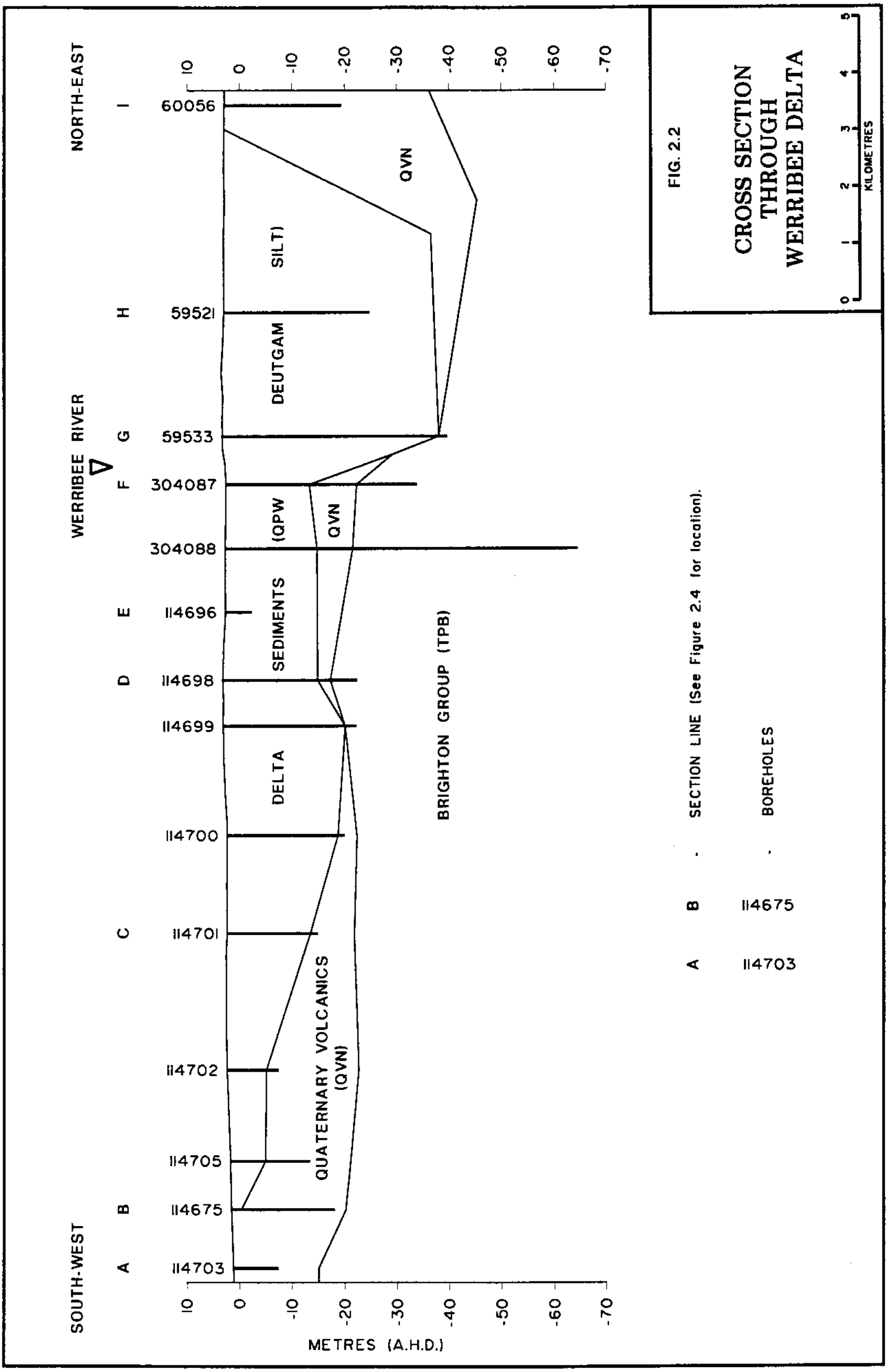


Figure 2.2 Cross Section Through Werribee Delta

2.3.1 Werribee Delta Aquifer

The Werribee Delta aquifer is predominantly comprised of sand and gravel lenses situated within clays and silts, deposited in a deltaic sedimentary environment. The lensing of the coarser sediments within a finer matrix is typical of deltaic depositional systems, where near channel coarse sediments grade laterally to finer sediments and where the currently active river channel continually meanders, cross-cuts and re-works older sediments. The source of the delta sediments are Palaeozoic rocks eroded from the upthrown blocks of the Rowsley and Coimadai faults and deposited in the lower reaches of the Werribee River (VandenBerg, 1973).

Due to the wide ranging lithologies encountered, the Werribee Delta aquifer is both heterogenous and anisotropic. The hydraulic conductivity and specific yield of the coarser lenses range from 10 to 15 m/day and 0.15 to 0.20, respectively, whilst the overall values of hydraulic conductivity and specific yield are considered to be approximately 5 m/day and 0.1, respectively (Leonard,1992). However, results from field tests by Anderson (1988) and CPI (1992) yielded hydraulic conductivity values ranging from 0.001 to 1.5 m/day, indicating the presence of much finer sediments. Nutrient load calculations utilising the Leonard (1992), Anderson (1988) and CPI (1992) values for hydraulic conductivity and hydraulic gradient are presented in Section 2.5.

Groundwater quality varies from 500 mg/L to 6000 mg/L Total Dissolved Salts (TDS) while typical bore yields range from 1 to 15 litres/second. Data from private and government boreholes in the area indicates that nitrate concentrations range from 17 to 23 mg/L (expressed as N) and phosphorous concentrations range from 1.5 to 4.5 mg/L (expressed as P).

The aquifer reaches a maximum thickness of 40 m in the area along the coast immediately east of the present course of the Werribee River (Figure 2.3) where the ancestral river course has cut through and entirely eroded the Newer Volcanics basalt. To the north-east of the Werribee River the aquifer thins until it wedges out some 6 km west of Point Cook, and in the west the limit of deposition of the aquifer occurs just west of the present course of the Little River.

Groundwater in the Werribee Delta sediments is unconfined to semi-unconfined. The watertable in the deltaic sediments in the eastern section of the Werribee Delta tends to be at a depth of about 7 m from natural surface and becomes shallower towards the coast (Figure 2.4). In the western section of the delta, however, the water table generally occurs at a depth of less than 2 m due to mounding effects created by the lagoons and irrigated areas of the Werribee Treatment Complex.

Recharge to the Werribee Delta aquifer occurs mainly via the direct infiltration of rainfall and irrigation water (especially in the western area of the Delta near the Werribee Treatment Complex). Where the Newer Volcanics aquifer underlies the delta the hydraulic pressure is in some places greater than that which occurs in the shallowest aquifer. This would suggest that the deltaic aquifer may receive a component of recharge via upward leakage from the underlying basalt aquifer. The extent and magnitude of this component of recharge is difficult to quantify, owing to a lack of data.

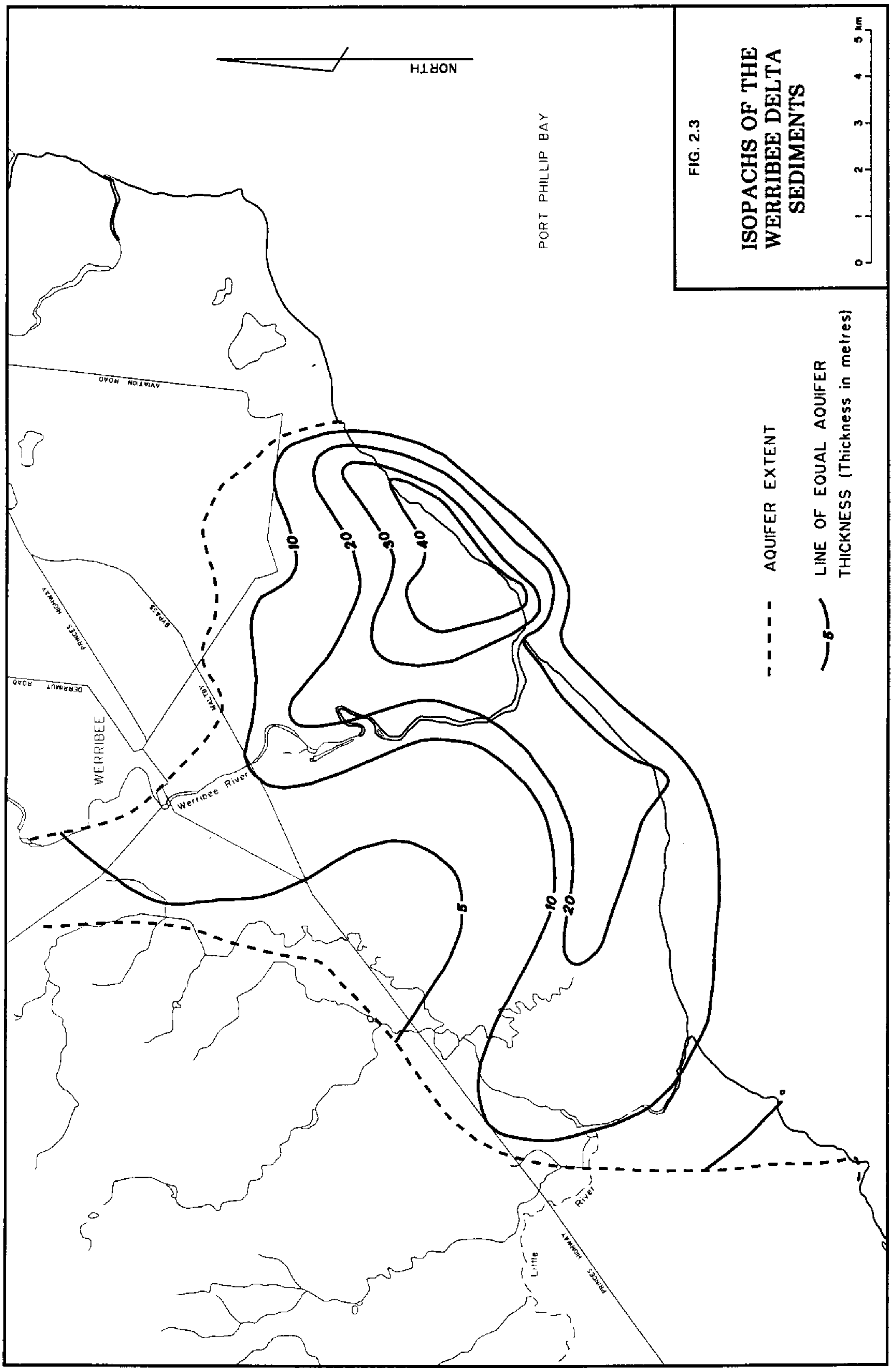


Figure 2.3 Isopachs of the Werribee Delta Sediments

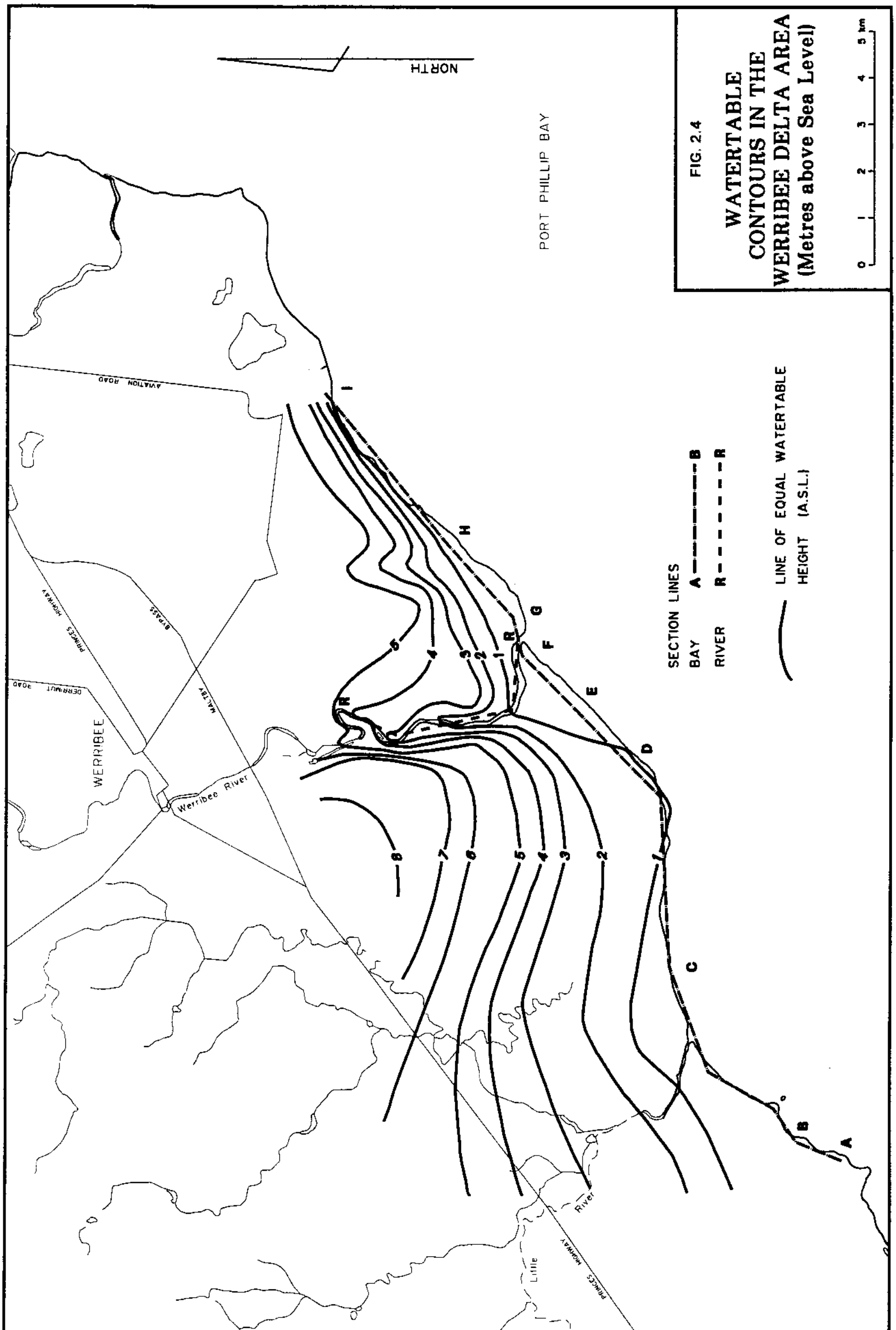


Figure 2.4 Watertable Contours in the Werribee Delta Area (metres above sea level)

2.3.2 Newer Volcanics Basalt Aquifer

The Newer Volcanics basalt aquifer occurs as a subsurface feature over almost all of the Werribee Delta area. The aquifer consists of up to four separate phases of volcanic activity and as such the aquifer is heterogeneous and anisotropic. The basalt is thickest in the northern area of the study region - occasionally exceeding 30 m thick - and progressively thins towards the Bay where it becomes absent just east of the Werribee River. At that point the Werribee Delta sediments directly overlie the sediments of the Brighton Group.

The movement of groundwater through the Newer Volcanics aquifer system is complex. Groundwater flow paths tend to be highly irregular and in some cases may vary considerably from the overall regional flow directions. Preferred pathways for groundwater flow may be via open rock joints, fractures, vesicular openings and in the contact zone between the different flows. Hence the hydraulic conductivity and transmissivity of the aquifer is highly variable as it largely depends on these rock defects. An average conductivity (K) of 1 m/day has been adopted for this study based on CPI (1992).

The nature of the potentiometric surface in this region is such that the water level pressures are rising slightly (bore 59526 - Appendix A). The hydraulic connection between the basalt and the overlying deltaic sediments can be seen in the hydrographs of the piezometers 59534/59535 and 59525/59526 (Appendix A).

Recharge to the Newer Volcanics aquifer system is mainly via infiltration of rainfall in the up basin areas where the aquifer outcrops, via discharge from streams where pressure levels are below the stream stage height, or from infiltration through the delta sediments.

2.4 Groundwater Contamination

The main objective of the study in this area is to determine the existing and likely future levels of nutrient contamination to the groundwater systems discharging into Port Phillip Bay.

A survey of RWC, EPA, Melbourne Water and private industry data was conducted in order to identify any existing pollution hazards. Analysis of this data led to the identification of a zone of groundwater discharge to the Bay with shallow groundwater salinities in the range 2,000 to 10,000 mg/L.

The Werribee Treatment Complex is located in this region on the western side of the Werribee Delta, whilst the eastern side supports extensive market gardening. These particular changes in land use have had a significant effect on the hydrogeology of the area resulting in rising watertables. This effect can be directly attributable to the infiltration of nutrient rich waters from the Treatment Complex and the irrigation occurring on the eastern side of the River, into the aquifer systems below. Where irrigation induced groundwater mounds have developed due to large groundwater accessions, the hydraulic gradients in the Werribee Delta aquifer system have increased resulting in a greater amount of throughflow via the delta sediments and into Port Phillip Bay. It also appears that there is upward groundwater flow from the Brighton Group to the Newer Volcanics aquifer.

However, groundwater in the Brighton Group aquifer does not show any anomalous nitrogen (N) and phosphorous (P) readings, hence nutrient contamination is only expected to occur in the top two aquifers. Consequently as stated earlier, the modelling of this nutrient environment only incorporates the top two aquifers.

2.5 Modelling

2.5.1 Discharge to the Bay

The area was modelled using an analytical approach, where the two shallow aquifers of the Werribee Delta and Newer Volcanics were considered. Data from a variety of different sources was used, including the RWC Groundwater database, and was compared to previous work by Leonard (1992), CPI (1992), Anderson (1988) and Stewart (1988).

Total groundwater nutrient transfers into the Bay were obtained using calculations of groundwater throughflow and then applying a suitable mean nutrient concentration to give an estimate of the total nutrient load into the Bay.

A cross section was constructed using both Melbourne Water and RWC bores to determine the geometry of the aquifer along the shoreline (Figure 2.2). The section was then divided into subsections depending on the data distribution and the aquifer morphology. For each subsection the hydraulic gradient was calculated from Figure 2.4, and the aquifer areas were also calculated, enabling a Darcian computation to be made for each subsection. Darcy's Law can be described in the form below:

$$Q = K\,A\,i$$

Where:

Q = groundwater flow in m^3/day;
K = hydraulic conductivity in m/day;
A = Cross-sectional area of the discharge face in m^2;
i = hydraulic gradient in m/m

Initially a hydraulic conductivity of 5 m/day and 1 m/day was used for the Werribee Delta and the Newer Volcanics aquifer systems respectively (taken from Leonard, 1992). Other values of K were later used, taken from CPI (1992), to enable comparisons and sensitivity analysis to be applied. Applying Darcy's Law, the total outflow from these two upper aquifers resulted in an estimate of the total groundwater flow into Port Phillip Bay. An average value for the concentration of nitrogen (N) and phosphorous (P) was then applied to each throughflow calculation in order to obtain the nutrient loads (in terms of N and P) into Port Phillip Bay.

In order to be able to employ comparative studies and to undertake a sensitivity analysis the area was divided into 2 main sections, the area west of the Werribee River (subsections A, B, C, D) and that area east of the Werribee River (subsections E, F, G, H). The location of these subsections can be seen on Figures 2.2 and 2.4. In addition, each section of the Delta area, i.e. east and west, has been divided into two layers - the upper aquifer (the Werribee Delta sediments) and the lower aquifer (the Newer Volcanics aquifer) - in order to determine the nutrient loads to Port Phillip Bay from each of the separate aquifers as well as conjunctively. Other workers (Anderson, 1988; CPI, 1992; Leonard, 1992) have undertaken similar calculations for parts of the region, and those results are also presented in order to provide a benchmark against which other studies can be compared. The areas investigated in each of these studies is delineated in Figure 2.5. As the CPI (1992) results for the Delta sediments relate only to the upper sandy unit, it was necessary to further break the Werribee Delta aquifer into an upper sandy unit (D) and a lower silty/clayey unit (S) where little or no flow occurs. This study has not distinguished between upper and lower units on the north-eastern side of the Werribee River, as there is insufficient data on this side. On the south-western side, the presence of the relatively impermeable clayey unit is taken into account by assigning a lower value of the horizontal hydraulic conductivity (K) to the Delta aquifer when compared to the other subsections.

HydroTechnology has calculated flow and load figures using a combination of compiled field data and the conductivity and hydraulic gradient values of Leonard (1992) and CPI (1992). Using the parameter values adopted by Leonard (1992), i.e., K = 5 m/day and i = 0.002 (Appendix B), the HydroTechnology results compare favourably with the results of Leonard (1992).

However, if the parameters used by CPI (1992) are adopted (Appendix C), where K < 2 m/day and i < 0.002 the throughflow calculations are significantly different. In the eastern section the total flow obtained for the upper aquifer, using CPI (1992) figures, is 450 ML/yr, some 670 ML/yr less than the flow calculated using Leonard's (1992) parameters. On the western side of the Delta however, the calculated flow is 900 ML/yr, which is some 390 ML/yr more than the flow calculated using Leonard's approach.

The two different sets of results highlight the effect that a change in hydraulic gradient and conductivity values have on the outcome. CPI (1992) however, calculated the throughflow in the upper aquifer for the western section as being 1118 ML/yr which compares well with the 900 ML/yr figure of HydroTechnology. The difference of some 220 ML/yr being mainly attributable to the differing aquifer areas used in the calculations. CPI (1992) makes no attempt at calculating throughflow for the eastern side of the Delta and hence no comparisons can be made here. The estimated total throughflow for the upper aquifer in the eastern and western sections ranges from 450 ML/yr to 1120 ML/yr and from 410 ML/yr to 900 ML/yr, respectively (see Table 2.2).

The total throughflow for the Newer Volcanics aquifer system on the eastern and western sides of the Delta are 12 and 43 ML/yr respectively, calculated using Leonard's (1992) parameters. When CPI's (1992) parameters are adopted for the HydroTechnology calculations, the results are 12 ML/yr for the eastern side and 105 ML/yr for the western side. The results in the east compare favourably, however a difference of some 60 ML/yr occurs in the west, possibly due to the values used for the aquifer cross-sectional areas. Anderson (1988) calculated throughflow from the west at 103 ML/yr which is very similar to the HydroTechnology estimate derived using similar parameters. A major assumption with the calculations undertaken in this section, and also in the works of Leonard (1992) and CPI (1992), is that horizontal Darcian flow occurs.

The next stage in the modelling exercise was to calculate the nutrient loads (in terms of nitrogen, and phosphorous) in tonnes/yr into Port Phillip Bay. Previous studies by Anderson (1988) and CPI (1992) utilised the chemical analyses from more than 30 bores located along the shoreline of the Werribee Treatment Complex and adjacent to the western bank of the Werribee River.

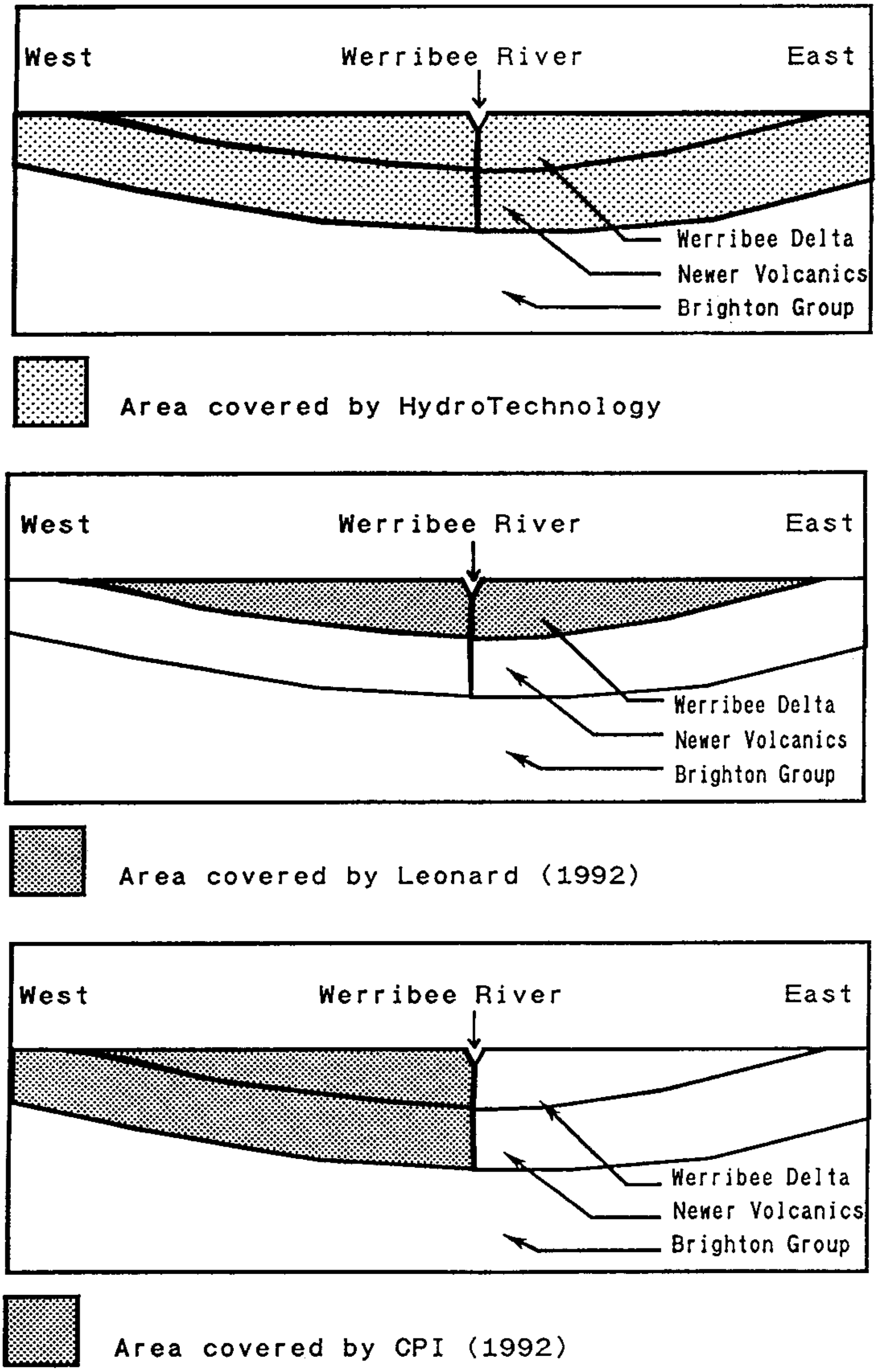


Figure 2.5 Schematic Diagram of Werribee Delta

Discrete values are available for the various sections of the south-western sector of the Werribee Delta. In the north-eastern sector of the Werribee Delta, reliable groundwater nutrient data does not exist to the same extent as that collected in the south-west. As such, average values of nitrogen and phosphorous were calculated from the available bores. The results of the nutrient load calculations are depicted in Table 2.2. From this table it is apparent that the majority of the groundwater nutrient load into this part of the Bay comes from the Werribee Delta aquifer. Loads from the Newer Volcanics aquifer system in the east are negligible. However, in the west, the nutrient loads from this aquifer are not so insignificant, and nitrogen loads may be more than 1.0 tonnes/yr. Phosphorous loads to the Bay, however, are low.

TABLE 2.2. Summary of groundwater flow and nutrient load from Werribee Delta area.

		EAST			WES T			COMBINED TOTAL (East and West)
		QPW*	QVN*	TOTAL	QPW*	QVN*	TOTAL	
HYDRO-TECHNOLOGY (parameters after Leonard, 1992)	Flow (ML/yr)	1120	12	1132	410	43	453	1585
	N load tonnes/yr	23.0	0.013	23.013	7.0	0.2	7.2	30.213
	P load tonnes/yr	3.0	0.04	3.04	0.9	0.02	0.92	3.96
HYDRO-TECHNOLOGY (parameters after CPI, 1992)	Flow ML/yr	450	12	462	900	105	1005	1467
	N load tonnes/yr	9.1	0.013	9.113	18.2	0.8	19.0	28.113
	P load tonnes/yr	1.3	0.004	1.304	2.9	0.1	3.0	4.304
LEONARD (1992)	Flow ML/yr	912.0	-	912.0	511.0	-	511.0	1423.0
	N load tonnes/yr	-	-	-	-	-	-	-
	P load tonnes/yr	-	-	-	-	-	-	-
CPI (1992)	Flow ML/yr	-	-	-	1118.0	103.0	1221.0	1221.0
	N load tonnes/yr	-	-	-	22.0	1.2	23.2	23.2
	P load tonnes/yr	-	-	-	3.0	0.1	3.1	3.1

N.B. QPW* = Werribee Delta Aquifer
QVN* = Newer Volcanics Aquifer

In addition to these calculations, the location of the saltwater interface for the watertable aquifers was calculated (Table 2.3 and Appendix D) so that the location and extent of groundwater discharge to the Bay could be determined. The calculation involves the Werribee Delta aquifer where present, and the Newer Volcanics aquifer where the Werribee Delta aquifer is absent (western margin of the study area, subsection A), thus considering the watertable aquifer only. Parameters used were adopted from the nutrient load calculations and the density parameters from Stewart (1988). From the results (Table 2.3) it can be seen that in the west where the hydraulic gradient is large the saltwater interface does not penetrate very far landwards. Where the gradient is less steep however (subsections E, F, G and H), the location of the interface is found much further inland (Figure 2.6). Consequently, it can be seen that where the volume of flow through the section is driven by a steep hydraulic gradient, discharge occurs further out into the Bay. However, where the gradient is less steep, the discharge occurs closer to shore, being controlled by the salt water wedge. It can be seen from the calculations that the groundwater flows and the nutrient loads are considerably affected by variations in hydrogeological parameters. The nutrient loads presented are what could feasibly be expected from this area, given the existing data. Further field investigations are warranted should it be desired to more accurately define the nutrient loads.

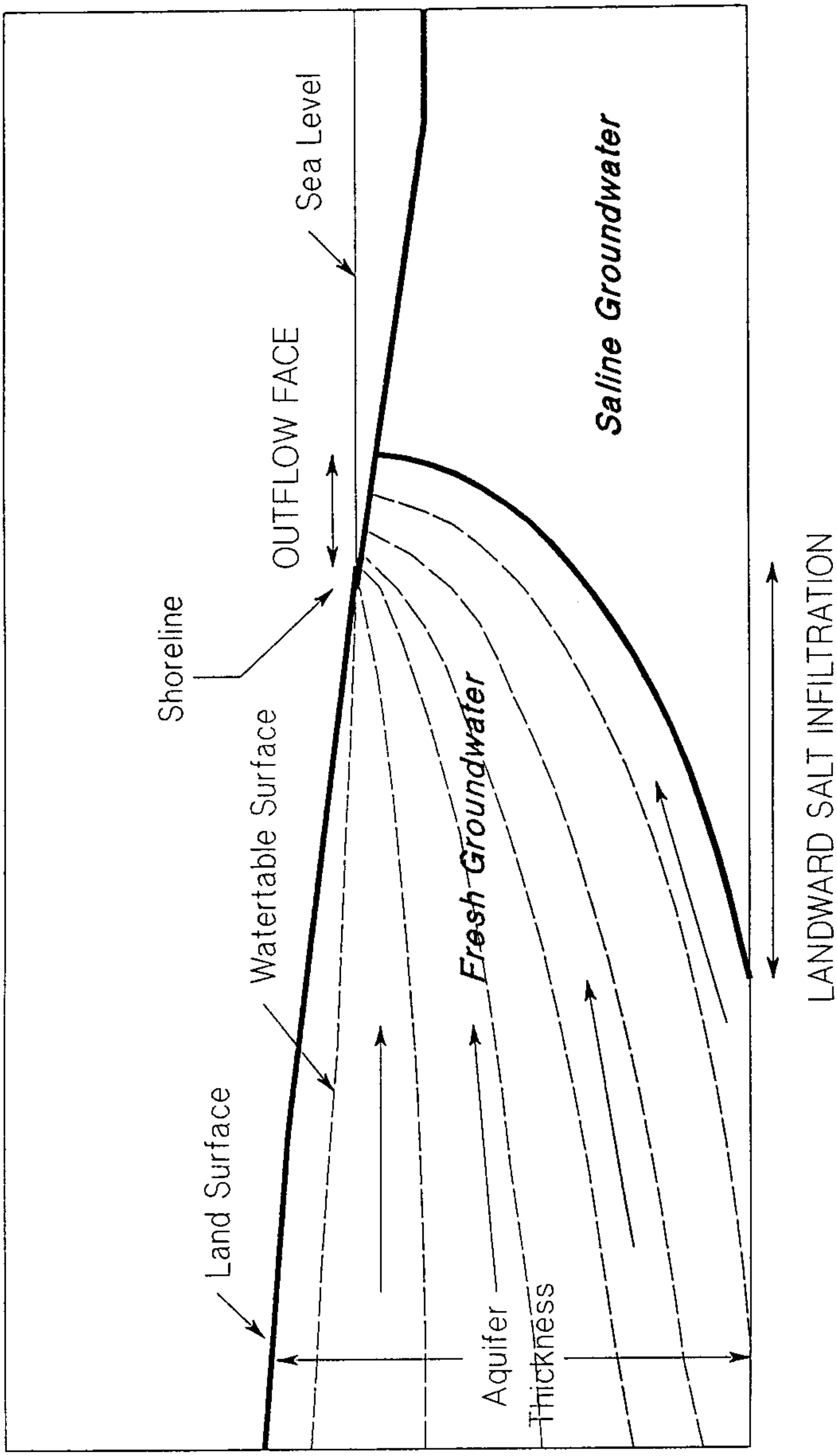


Figure 2.6 Location of Salt Water Interface

2.5.2 Discharge to Werribee River

The constructed potentiometric surface of the Werribee Delta aquifer (Figure 2.4) indicates that groundwater flow occurs into the Werribee River. Flow into the western bank of the Werribee River has been previously calculated along a 5 kilometre stretch by CPI (1992). An estimated load of 0.825 tonnes/yr of N for the western bank was calculated. This calculation assumed that the average thickness of the discharge face to the Werribee River for each section was equivalent to the height of the watertable above sea level in the upgradient borehole. In dealing with the eastern bank of the Werribee River, a similar approach has been adopted to ensure consistency. Assuming a 5 kilometre discharge face (Figure 2.1), average hydraulic conductivity values of 1 m/day (as against the higher 5 m/day values of Leonard [1992]), a hydraulic gradient of 0.0022 and a cross-sectional area of 17,500 m^2, Darcy's Law can be applied to calculate the groundwater discharge to Werribee River.

Table 2.3. Salt water interface model for watertable aquifers

Werribee Delta (A -F) and Newer Volcanics (G) *		
Average Density of fresh water 1.003 mg/mL		
Average Density of salt water 1.035 mg/mL		
Density Constant 31.344		
SECTION	**WIDTH OF OUTFLOW FACE (m)**	**MAXIMUM LANDWARD SEA WATER INFILTRATION (m)**
A	-5.36	2.47
B	-5.47	0.05
C	-23.45	0.01
D	-25.86	0.01
E	-0.47	105.11
F	-1.25	116.38
G	-1.80	167.05
H	-1.22	113.47

***** See Figure 2.4 for location of cross sections

Results for the eastern bank of the Werribee River indicate approximately 38 m^3/day or 14.1 ML/yr of groundwater discharge (Table 2.4). HydroTechnology has also calculated the estimated groundwater flow from the western bank of the Werribee River, for a comparison with the results of CPI (1992). A comparison of the calculated groundwater flow rates and associated nutrient (N and P) loads is tabulated below. Average N and P concentrations from groundwater bores of 14 mg/L and 0.3 mg/L, respectively, have been utilised in the HydroTechnology calculations for the eastern bank. It is assumed that the concentrations of N and P are representative, however, further sampling may provide further data on which to base the load calculations.

Table 2.4. Groundwater discharge and nutrient loads to the Werribee River

Source	Location	Groundwater Discharge (m^3/d)	Groundwater Discharge (ML/yr)	N (t/yr)	P (t/yr)
HydroTechnology	Eastern Bank	38	14.1	0.2	0.019
HydroTechnology	Western Bank	175	64	0.9	0.004
CPI (1992)	Western Bank	173 (back-calculated from CPI, [1992])	63 (back-calculated from CPI, [1992])	0.8	N/A

2.6 Discussion

The data and calculations indicate that the north-eastern Werribee Delta area may contribute from 10 to 24 tonnes/yr of N and from 1 to 3 tonnes/yr of P to the Bay (including that discharged to the Werribee River). The variations in the values arise from the use of different hydraulic parameters from Leonard (1992) and CPI (1992). It is considered that the nutrient levels would tend towards the lower end of the range. The contribution from the western side of the Delta is up to twice that from the east, utilising CPI (1992) parameters, mainly due to the presence of the Werribee Treatment Complex in this area. The total nutrient loads entering the Bay from the eastern and western sections have been calculated to range between 31 and 28 tonnes/yr of N and between 3 and 4 tonnes/yr of P. These nutrient loads contribute 0.8% of N and 0.3% of P when compared to the amount of nutrients that are discharged from treatment lagoons and drains at the Werribee Treatment Farm to Port Phillip Bay; i.e. in excess of 4,000 tonnes/yr N and 1,240 tonnes/yr P (Otto, 1992).

3.0 TOXICANT INPUTS - WESTERN SUBURBS

3.1 Introduction and Location

The area analysed in this section of the study encompasses about 4,680 hectares. The area is bounded by the Maribyrnong River to the east and the coastline in the south and includes the parishes of Cut Paw Paw, Maribyrnong and Truganina (Figure 3.1). In the north-west of the study area the land is rural and the groundwater is used mainly for stock watering. East of Kororoit Creek however, the use of the groundwater has been restricted due to contamination from urbanisation and the disposal of industrial wastes to quarries and bores in this area. In the following sections, the affect of the processes of long-term urbanisation and waste disposal will be assessed in relation to the beneficial use of the groundwater in the upper aquifers.

3.1.1 Topography

The study area forms part of the Basalt Plains of Victoria which extend from the Yarra Valley in Melbourne westward to the Rowsley Fault (Douglas and Ferguson, 1976). The plains were formed by coalescing lava flows and consequently have a gently undulating surface that is dotted with low volcanic cones. The major streams draining the plains have deeply incised youthful valleys as they cut into the basalt formations. Table 3.1 shows some characteristics for these streams.

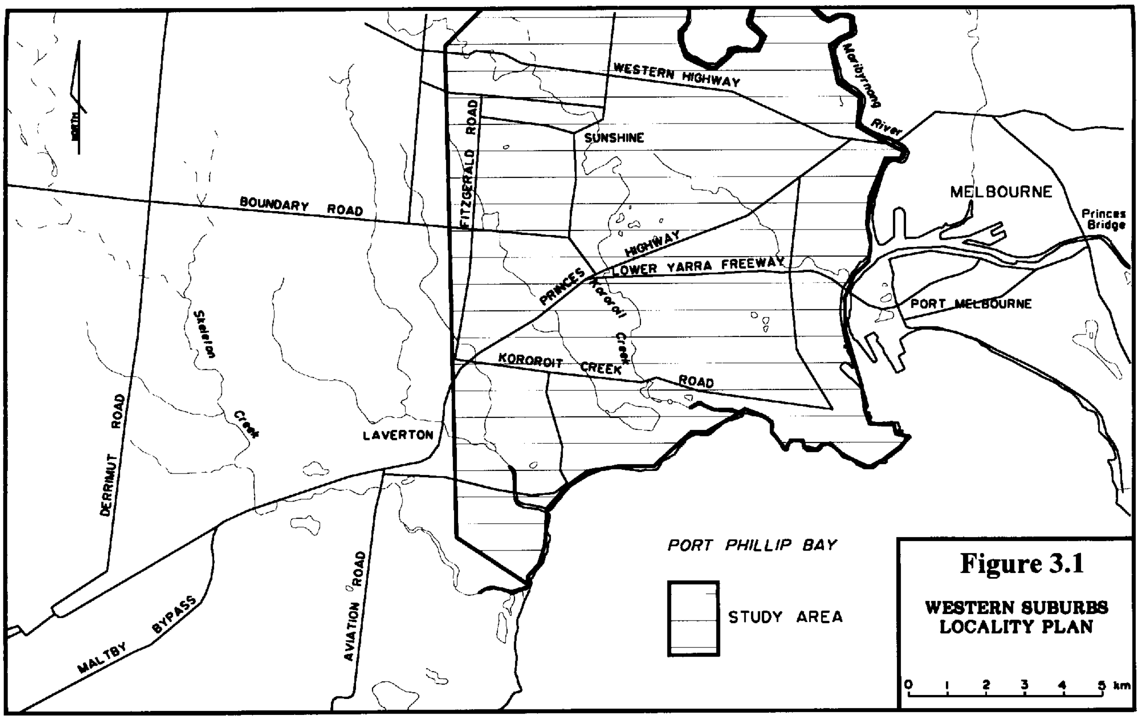


TABLE 3.1. Major streams which dissect the northern and western plains of Melbourne (After Leonard, 1992).

Catchment	Area(km^2)	Streamflow(10^3 ML/yr)	Rainfall (10^3 ML/yr)
Skeleton Creek	147	12	75
Kororoit Creek	323	36	170
Maribyrnong River	1471	177	1020
Moonee Ponds Creek	127	21	80

These streams are generally consequent streams and flow south-easterly, but in detail have repeatedly adopted new courses after blocking by basalt flows from one or more centres of eruption.

3.2 Landuse

Landuse is considered briefly here within a framework of two broad categories: zoned industrial land (both occupied and unoccupied), and land that is zoned non-industrial such as agricultural and/or residential uses. In terms of this study the main influence on both the quality and quantity of the groundwater is the high concentration of zoned industrial land in the region, consequently this landuse will be analysed in detail. Almost any landuse has the potential to contaminate land yet it is important to restrict the study to those uses that are considered to be high-risk. The EPA has prioritised the sites most likely to result in soil and groundwater contamination in Victoria and they are listed below in order of significance:

- former industrial sites
- mining/smelting industries
- agricultural chemical plants
- waste disposal sites (landfills)
- fill areas
- ongoing industries
- sites containing leaking underground storage tanks

Table 3.2 outlines the zoned landuses in the municipalities of Altona, Footscray, Sunshine and Williamstown, which cover the area studied in this section.

In all four municipalities the area zoned as industrial occupies a significant proportion of the total area. Consequently the risk of contamination from industrial use is high. This aspect will be assessed in greater detail in section 3.4.

TABLE 3.2. Industrial and other land use areas in the western suburbs (source: Department of Planning and Housing, 1993)

LOCAL GOVERNMENT AREA	DATE	TOTAL AREA (ha)	NON INDUSTRIAL LAND (ha)	INDUSTRIAL LAND (ha)		
				TOTAL	OCCUPIED	UNOCCUPIED
ALTONA	Jan 1983	4018	2658 (66.2%)	1360.1 (33.8%)	536.6 (13.3%)	823.5 (20.5%)
FOOTSCRAY	Jan 1983	1710	1404 (82.1%)	305.8 (17.9%)	272.5 (16.0%)	33.3 (1.9%)
SUNSHINE	April 1983	8006	5059 (63.2%)	2946.4 (36.8%)	1482.7 (18.5%)	1463.6 (18.3%)
WILLIAMSTOWN	Dec 1982	1450	1197 (82.6%)	252.8 (17.4%)	189.7 (13.1%)	63.2 (4.3%)

3.3 Hydrogeology

The hydrogeology of the western suburbs of Melbourne is relatively complex, but can be considered in two separate stages. Palaeozoic sediments and igneous intrusives form the basement of the area and Cainozoic sediments and igneous extrusives overlie the basement material (Leonard, 1979). In general, the basement rocks form very poor aquifers and it is the overlying Cainozoic sediments and intrusives which provide for valuable sources of groundwater. The dominant aquifers of the region are discussed below.

3.3.1 Werribee Formation

The Werribee Formation is a multilayered aquifer consisting of up to 100 m of sand, brown coal, gravel and minor amounts of carbonaceous clay (Riha and Kenley, 1978). The top of the aquifer occurs generally at depths greater than 60 m. Basalts of the Older Volcanics are interfingered with the Werribee Formation sediments on a local scale. Groundwater in the formation is confined by the overlying Altona Coal Seam and/or the Fyansford Formation, both of which are of low hydraulic conductivity. The formation wedges out rapidly to the north-east and does not outcrop.

Bore yields in the Werribee Formation range from 0.6 to 53.0 L/sec due to the variable nature of the aquifer. Salinities of the aquifer range from 2,400 to 5,000 mg/L TDS which consequently limits the purposes for which it can be used. The hydrochemistry of the aquifer system shows a high calcium bicarbonate content with a pH of more than 7, which is atypical of groundwater that is in direct contact with a coal seam (Riha and Kenley, 1978). This may be a result of natural recharge through the overlying Newer Volcanics aquifer system in the north where the two formations may be in direct contact and the intervening coal seams are locally absent. Water from the Werribee Formation is confined by the Fyansford Formation over the entire Bay area, which causes discharge to occur well out to sea and possibly into Bass Strait (Leonard, 1979).

3.3.2 Brighton Group

The Brighton Group sediments in the study area consist of clays, silts, sands and gravels with a maximum thickness of approximately 30 m. This aquifer is generally confined by clays or silts in the upper part of the formation and below by the nearly impermeable clays, marls and silty clays of the Fyansford Formation. While the aquifer is fully saturated, the high percentage of clays and marls do not provide adequate yields for productive use. Salinities are generally high, in the range 4,000 mg/L to 7,000 mg/L TDS.

The analyses of groundwater chemistry from the Brighton Group aquifer are similar to those of the overlying Newer Volcanics suggesting that recharge is through the basalt aquifer, possibly to the north and north-west (Riha and Kenley, 1978), with the groundwater characteristics of the basalt aquifer strongly influencing the water composition in the deeper aquifers. Discharge from the Brighton Group aquifer is into surface streams where these streams have cut down into the sediments, or offshore into Port Phillip Bay (Leonard, 1992).

3.3.3 Newer Volcanics

The basalts of the Newer Volcanics regime are olivine basalts with alkaline derivatives of limburgite, scoria and tuff (Shugg, 1972). The basalts are usually dark bluish grey to black and are relatively vesicular. Although their texture and chemistry is quite varied their primary mineralogy is reasonably constant. They form an extensive overlay of basalt over the entire study area and are characterised by an undulating surface scattered with numerous volcanic cones. The activity of the Newer Volcanics was that of numerous short-lived eruptions from relatively small vents forming a radial distribution around the point of eruption building up a series of overlapping lobes to form the basalt unit. In general these lava flows are thin, but form on top of one another resulting in a relatively thick unit. An isopach map (map of aquifer thickness) of the Newer Volcanics is represented in Figure 3.2 and shows a maximum thickness of up to 60 m in the north-west of the region while the unit thins considerably towards the coast.

The Newer Volcanics basalt is the principal aquifer in the study region, occurring as stony rises which overly basalt flows. The basalts infilled the ancient valleys incised into the land surface. The groundwater regime of the Newer Volcanics aquifer system is very complex with water occurring in the fractures, joints, vesicular openings and in the contact zone between the different flows. The storage capacity and the transmissivity of the aquifer depends on the size of the individual openings, their density and geometry, continuity and degree of interconnection. The aquifer is thus anisotropic and

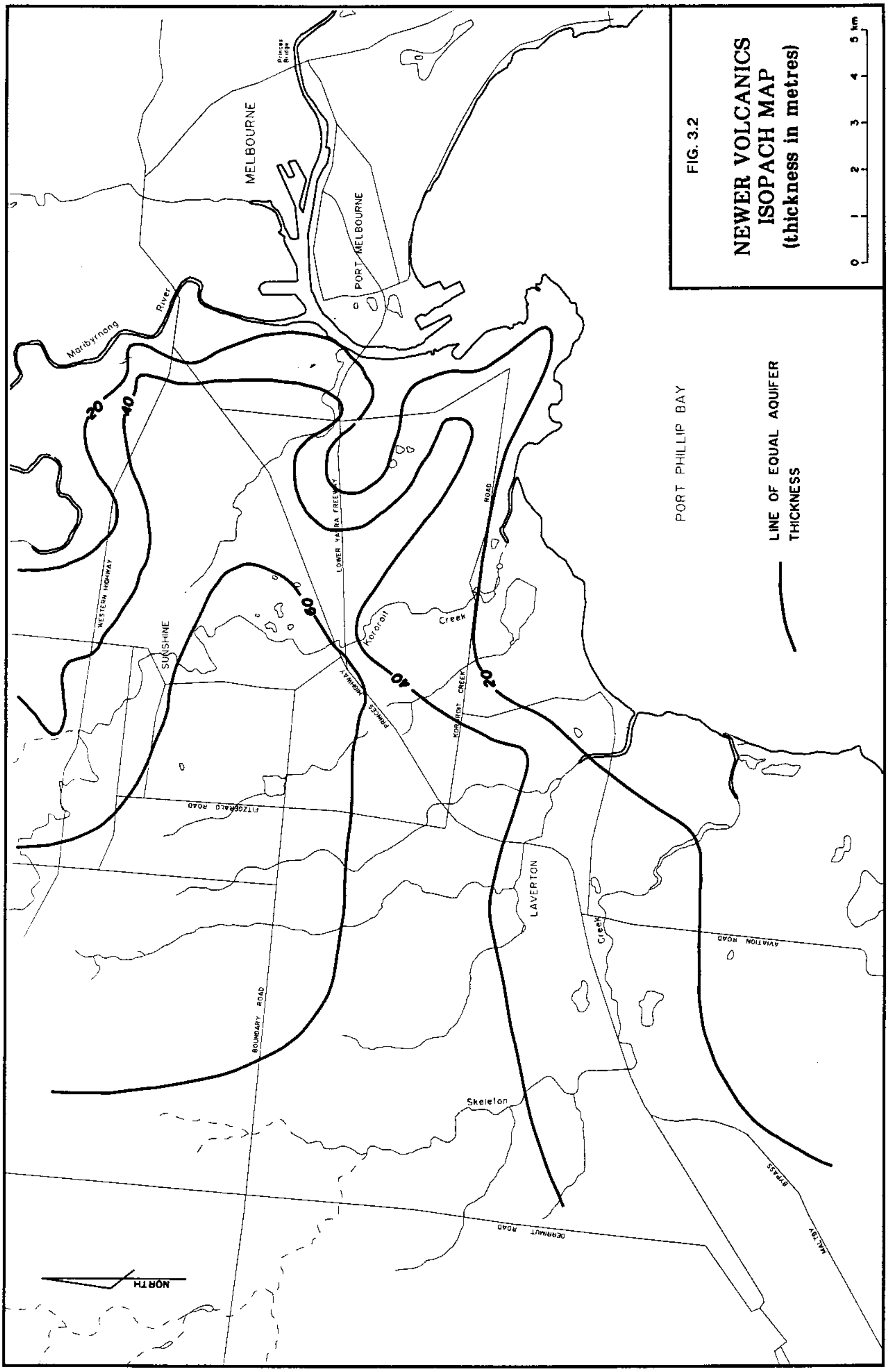


Figure 3.2 Newer Volcanics Isopach Map (thickness in metres)

heterogenous, implying that the hydraulic conductivity varies in both space and direction. The fractured basalt aquifer does not, at the micro scale, behave like a porous medium. Fluid flow within fractured rock can be dealt with in either a continuum approach or a discrete approach (Domenico and Schwartz, 1990). In dealing with bulk groundwater flows and solute loads, a continuum approach has been adopted.

The vertical hydraulic conductivity is usually much less than the horizontal hydraulic conductivity as the groundwater moves relatively freely through tabular jointing parallel to the flow layers but movement normal to the plane of lava flow is limited despite the evidence of vertical columnar jointing. The complexity of this aquifer system is further enhanced as several lava flows will form the actual aquifer, resulting in localised aquitards or aquicludes forming a multiple system of leaky aquifers as a result of the depositional environment. These minor aquitards are usually relict soil horizons or deposits of clay and silt. In some cases the low permeability layers consist of dense basalts that are associated with some degree of chilling at the base of the lava flow (Leonard, 1979).

The Newer Volcanics to the west of Melbourne contain at least two separate aquifers (upper and lower) which are vertically interconnected at a local scale. The upper basaltic aquifer is unconfined with the water table at a relatively shallow depth. Groundwater in the lower aquifer is confined or semi-confined and has limited vertical continuity with the upper aquifer. Consequently the most important aquifer in terms of pollution studies is the upper aquifer due to its shallow nature and the presence of the unconfined water table.

The potentiometric surface of the basalt aquifer is shown in Figure 3.3. The main direction of groundwater flow is south-westerly with localised mounds developing where waste percolation and other artificial recharge sources are evident.

The water quality of the natural groundwater in the Newer Volcanics varies significantly due to the variable nature of the aquifer. The salinity of the groundwater varies from 1,000 mg/L TDS in zones of recharge, to more than 10,000 mg/L TDS as one moves along the flow lines (Figure 3.4). The main anions present in the water are chloride, sulphate and bicarbonate while magnesium and calcium are the major cations (Riha and Kenley, 1978). The pH is typically in the range of 7 to 9. The groundwaters of this region are, therefore, seen as being typical of basalt groundwaters found throughout the world. The state of the water quality is, however, dependent on the state of the weathering of the basalts where changes in groundwater composition have been identified as one moves down the weathering profile. For example, total dissolved solids increase rapidly with depth (Riha, 1976).

Groundwater in the Newer Volcanics aquifer system in this area is seriously affected by pollution (Leonard, 1992). The basalt aquifer is recharged naturally by direct infiltration of rainfall and downward leakage from surface streams. The aquifer has also been artificially recharged by injection of effluent into bores (Riha, 1975) and by infiltration of effluent or leachate from extractive industry sites (Shugg,1981). It is these artificial recharge sites that have given rise to the many pollution problems in the area which will be discussed in detail at a later stage.

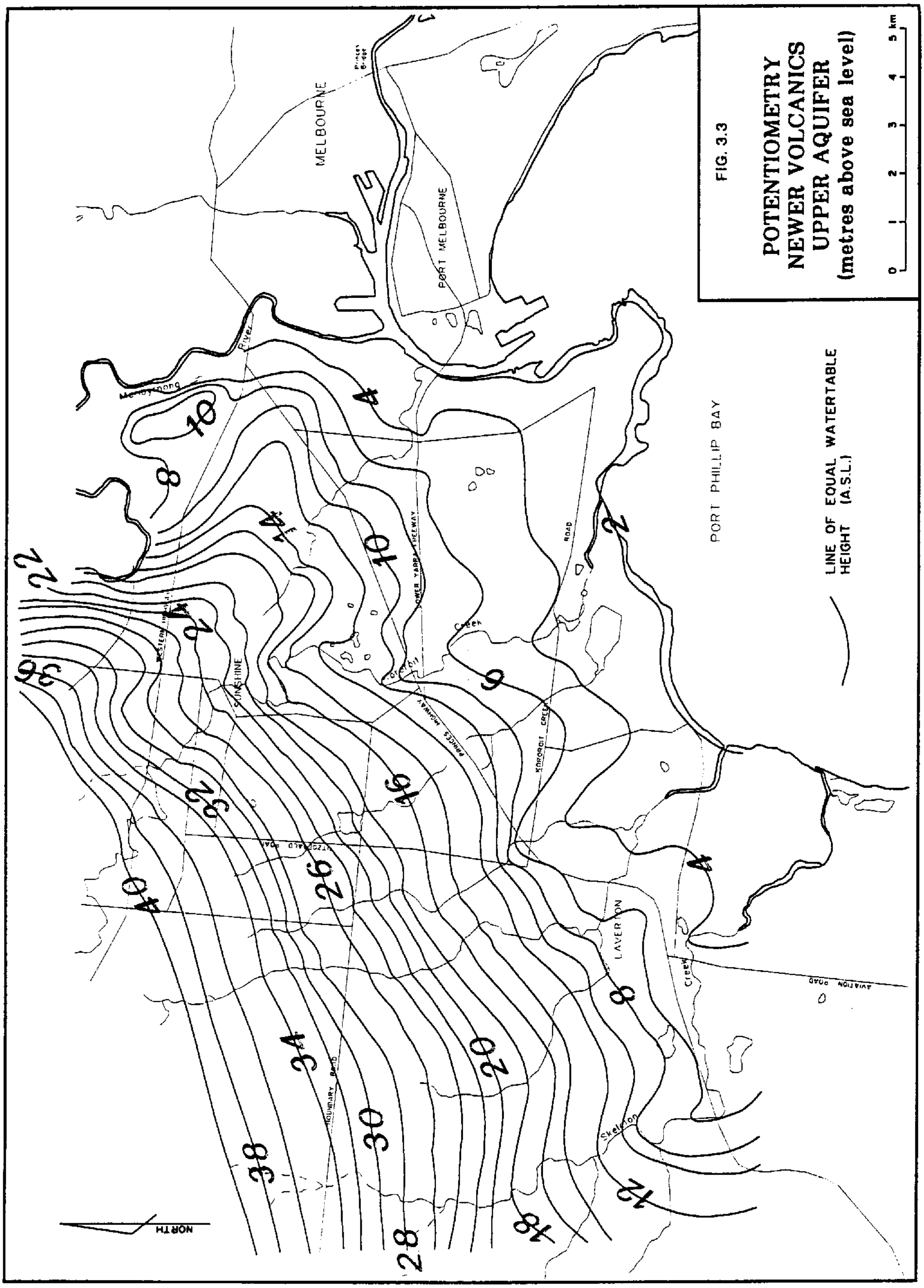


Figure 3.3 Potentiometry Newer Volcanics Upper Aquifer (metres above sea level)

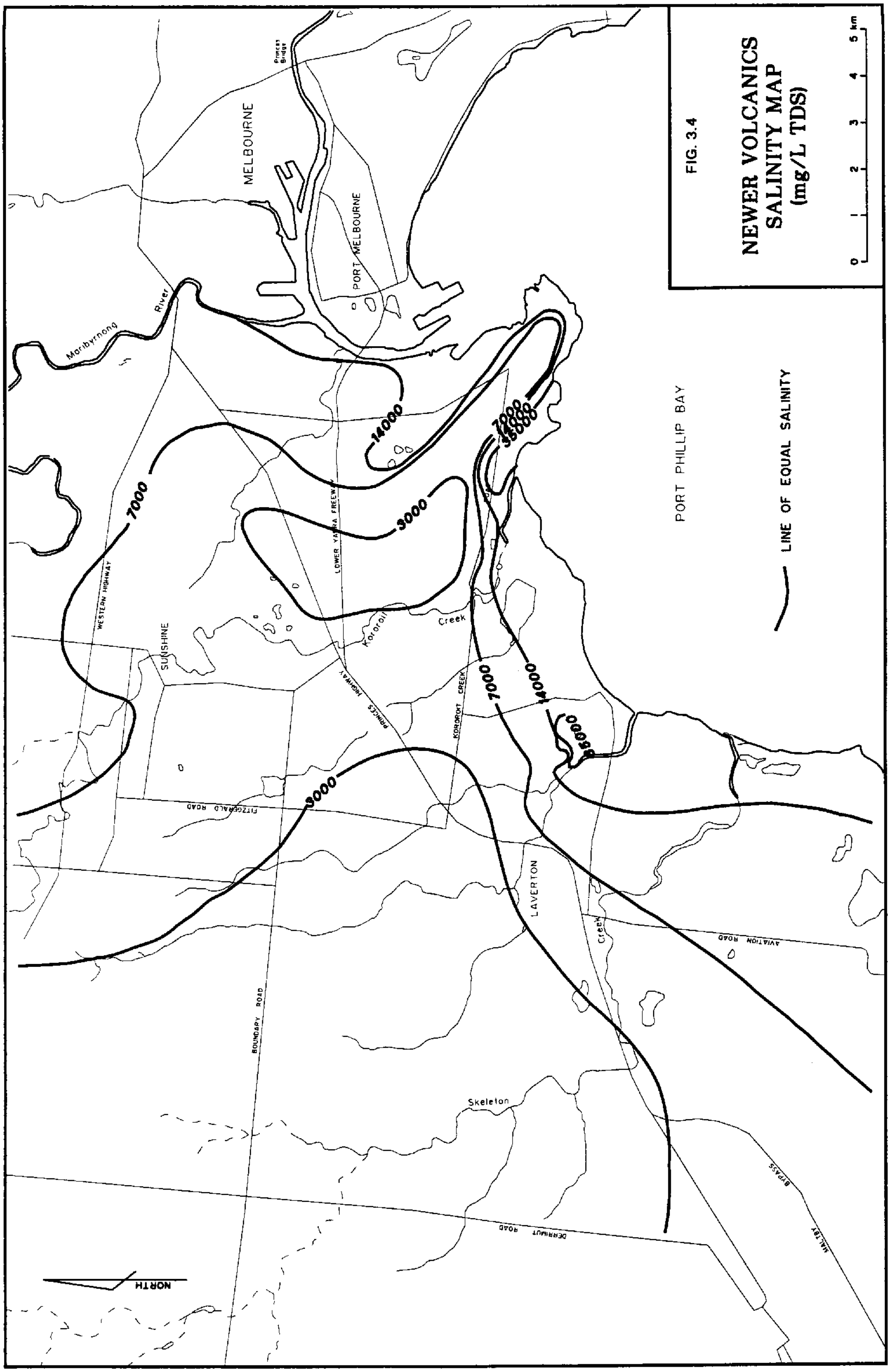


Figure 3.4 Newer Volcanics Salinity Map (mg/L TDS)

3.4 Surface and Groundwater Contamination

Groundwater pollution may be defined as any alteration of the chemical, physical or biological quality of the water which results in an unacceptable depreciation of its utility or environmental value (Leonard, 1979). This definition implies that the change must be detrimental or cause a significant loss of its applications.

Pollution of the groundwater arises from a number of activities including waste-water disposal, land development, agricultural practices, groundwater withdrawals, chemical spillage and leakage. The types of contaminants that may enter the groundwater are virtually limitless and are generally a function of the source. The subsequent fate of the contaminants depends on many parameters such as the type, nature and concentration of the contaminant, and on the aquifer material and the dominant flow conditions within the aquifer.

In this study the Newer Volcanics aquifer system was studied in detail in terms of assessing the groundwater pollution and the potential for groundwater pollution that ultimately enters Port Phillip Bay. Contamination of other aquifers in the western suburbs area is insignificant in comparison with the Newer Volcanics aquifer system (Leonard, 1992). The level of contamination in the Newer Volcanics aquifer has caused considerable concern to industry, the environment and local residents. Hydrogeological investigations have been undertaken by many workers to assess the extent of contamination from individual industries, to evaluate the potential of the aquifer as a waste disposal medium, and to assess the interaction of the aquifer with other aquifers and surface water bodies (Riha, 1971a, 1971b, 1975, 1976; Riha and Kenley, 1978; Shugg, 1987a, 1987b, 1987c; Leonard, 1979, 1992).

Within the study area contamination has been observed in the uppermost basalt aquifer. While contamination has also been observed in the lower aquifer, it is not as significant as that which occurs in the upper aquifer. The majority of the polluted areas are found between the Maribyrnong River and Kororoit Creek in the municipalities of Williamstown, Altona and the eastern side of Sunshine. However there exists a large number of potentially polluted areas west of the Kororoit Creek. The majority of the pollution centres in the study area are a result of discontinued disposal practices or industrial operations which are usually still functioning.

An inventory of known and potential contaminated sites was compiled from EPA, RWC and other data. Appendix E lists all potential contamination sites which occur within the municipalities of Williamstown, Altona, Footscray and Sunshine, and provides an indication of the degree of pollution in that municipality. For example, the City of Sunshine has 18 sites which may be contributing (or are actually contributing) to the groundwater pollution problem over a relatively small area. These four Local Government Areas (LGA) analysed were chosen due to the large amount of land zoned for industrial use (Table 3.2). The contamination of the upper basalt aquifer is from a variety of different parameters; those which have been analysed include phenolics, surfactants, sulphate, nitrate, dissolved solids, zinc, fluoride, manganese, lead, arsenic, copper, cadmium and mercury.

The contaminant sources in the western suburbs of Melbourne originate from a variety of activities, the main types being the petroleum and manufacturing industries, waste disposal practices and the chemical industry as shown in Table 3.3.

Table 3.3. Potential contaminant sources in the western suburbs of Melbourne

Contaminant Source	Number of Sites in Western Suburbs	Percentage of Total (55 sites)
Petroleum Industry	9	16%
Manufacturing Industry	18	33%
Waste Disposal (e.g. landfills)	10	18%
Chemical Industry	13	24%
Other (includes Shipyards, Extrusion Industry, RAAF base, Blacksmith, Rifle Range)	5	9%

While the 55 sites documented in Appendix E and the above table may not necessarily result in the contamination of groundwater they are identified as being potential sources of groundwater pollution based on the land use type and the proximity to the Bay and the closest surface water body (via groundwater flow paths). Some sites are known to be contaminated and those sites have been identified in the EPA Register of Contaminated Sites. From the table it is apparent that the majority of the potential groundwater polluters belong to the manufacturing industry (33%), while the chemical and petroleum industries each may make up to 24% and 16%, respectively, of the possible groundwater polluters in the western suburbs. Waste disposal practices may also potentially cause contamination problems as up to 18% of possible groundwater polluters fall into this category.

Once the sites which may potentially cause groundwater contamination had been identified, the next stage was to determine the degree to which each of these sites might contaminate the groundwater. The LeGrand (1980) system for evaluating waste disposal sites was used to standardise the potential sites and to identify those sites which might cause unacceptable groundwater contamination. This system is a numerical rating system used for evaluating the potential of groundwater contamination mainly from waste disposal sites, but has been applied to all sites identified as being possible polluters based on the type of land use. The system focuses on weighting four key geologic and hydrologic characteristics in the vicinity of the contaminant source. The key parameters used are:

- distance to the nearest surface water body
- depth to the watertable
- the hydraulic gradient of the watertable
- permeability-sorption of the soil and aquifer (depends on the geologic setting)

Four digits and four additional letter codes concisely describe characteristics of a site for standardised interpretation and indicate the relative contamination potential (LeGrand, 1980). The hydrogeological conditions of any particular site are identified on a scale of various aquifer sensitivities to contaminant spread and the particular waste's toxicity, concentration, volume and persistence and are identified on a scale which measures the severity of potential contamination by that particular product. This measure of severity is plotted on a standardised table (LeGrand, 1980) in order to determine the hazard potential of the site which then leads to the determination of the degree of seriousness. This refers to the way in which the aquifer sensitivity and the contaminant severity interact. The degree of seriousness ranges from A, which represents a relatively low degree of seriousness (due to a small area or quantity of product or where the contamination would be of slight consequence), through to I (where the area in question is large and/or the quantity of contaminant is large and the nature of the contamination would be of great consequence). The probability of contamination can then be evaluated by correlating a site's numerical description with the degree of seriousness.

The results of this analysis are set out below in Table 3.4 and in Appendix F.

Table 3.4. Results of LeGrand Classification of potentially contaminated sites

Probability of Contamination	Total number of sites	Number of sites	Percentage (%)
Very probable	58	21	36.2
Probable	58	31	53.4
Unsure	58	5	8.6
Improbable	58	1	1.7

From the table above it is apparent that the majority (53.4%) of the 58 sites identified as potential sources for groundwater contamination can quite probably cause groundwater contamination, while a further 36.2% result in very probable groundwater contamination. Only a small number (9.3%) of the sites analysed are unlikely to result in contamination of the groundwater. Consequently it can be seen that in the western suburbs area, the potential for groundwater contamination is very high, especially in areas where the land use tends towards industrial and manufacturing operations.

By analysing the results of the RWC's groundwater quality monitoring programme in the western suburbs, the groundwater contamination in the area can be more accurately defined. Pollution and "B" class (major ions) analyses exist for certain bores in the area, having been monitored several times per year from the 1970's to the early 1990's (Brinkley, 1992). Additional data from Melbourne Water and the EPA has also proven useful. Up to 400 chemical analyses are available for the area, ranging from Pollution to "B" class. Pollution analysis consists of:

Ph alkalinity SO_4 Mg K NO_X TSS Cl Ca
Na Fe EC Si Mn Cr Pb As TOC
F Cd Cu Fe Zn NH_3 phenols

Up to 30 bores in the area have Pollution analyses, while the remainder have "B" class or some equivalent. " B" class analysis consists of:

pH alkalinity SO_4 Mg K NO_X TSS
Cl Ca Na Fe EC Si

More than 1500 Melbourne Water bores also cover this area, drilled mainly for geotechnical purposes. The analyses for water from these bores consist of :

Cd	Cu	Fe	Pb	pH	phenols	SO_4	Hg
Zn	TDS	Mg	Cr	Ni	As		

N.B.

EC	*- Electrical conductivity*
TSS	*- Total Soluble Salts (equivalent to TDS - Total Dissolved Solids)*
TOC	*- Total Organic Carbon*
phenols	*- Phenols are a common groundwater contaminant. Phenols are based on a hydroxyl radical bonded to a benzene ring. Phenols are water soluble and are known to generate taste and odour at concentrations as low as 0.01 mg/L. Chlorophenols have one or more chloride ions on the benzene ring of a phenol and are used widely in industry and manufacturing. Chlorophenols are also found in groundwater where chlorine is added to drinking water and reacts with naturally occurring phenols. Chlorophenols give rise to objectionable tastes and odours at concentrations as low as 0.001 mg/L (Kostecki and Calabrese, 1991)*

These analyses have been collected over the past twenty years, as part of hydrogeological and geotechnical investigations conducted by government agencies. The samples were subject to the quality control procedures of the time. All metal analyses were for total metals and not filtered samples.

Both sets of data have been analysed in this study and the variability that exists between parameters is displayed in Table 3.5. Differences in the results for some parameters (e.g. phenols) between the Melbourne Water and RWC (ex Dept. of Minerals and Energy bores) data are probably due to the fact that those analyses taken from Melbourne Water bores were taken as part of extensive geotechnical investigation programmes, whilst the old DME bores were specifically sited to monitor polluting activities. However, the two data sets have been combined to obtain the values that are used in the toxicant load calculations presented in Section 3.6 of this report.

A major problem encountered was the time variability of sampling, that is, no complete data set exists for any one time period which includes all relevant parameters. For the year 1992, Pb and TSS have been analysed (Figures 3.5 and 3.6) and phenols are considered for the year 1985 (Figure 3.6). In addition, 20 bores containing Pollution analyses were chosen and the chemical time series data was analysed (Representative analyses in Appendix G).

When contaminants are discharged into a porous medium their movement is governed by the hydraulic conductivity, effective porosity, hydraulic gradient, dispersion and adsorption characteristics of the aquifer. A "plume" of contaminant will flow away from its source, elongated in the direction of groundwater flow. If the contaminants are discharged into the groundwater at a steady rate, the contaminant concentration is expected to decrease along the length of the plume (by way of adsorption and dispersion - Section 3.6). However, time series data in Appendix G does not suggest this, possibly due to the highly complex aquifer system and the presence of additional pollution sources further down the flow lines.

	Cd	Cu	Fe	Pb	pH	Phenols	SO_4	NO_x	PO_4	Zn	TDS	Mg	Cr	Ni	Ar	Hg
HydroTechnology																
Max.	0.02	0.17	100.00	7.60	11.10	240.00	6,000	23.0		1.04	12,947					
Min.	0.00	0.00	0.00	0.00	4.40	0.001	93	0.0		0.00	2,192					
Mean	0.01	0.02	9.49	0.13	7.61	2.07	695	2.7		0.07	2,767					
Std Dev.	0.00	0.02	18.15	0.70	0.77	22.19	935	4.6		0.11	2,561					
Melbourne Water																
Max.	0.10	1.30	260.00	1.00		0.32		16.9	4.5	11.00	60,900	3.20	0.84	1.30	0.05	0.0034
Min.	0.001	0.01	0.30	0.03		0.001		0.02	0.01	0.02	540	0.05	0.01	0.01	0.001	0.0001
Mean	0.06	0.14	37.28	0.22		0.03		1.5	0.3	0.53	9,863	0.66	0.15	0.18	0.01	0.0005
Std Dev.	0.48	0.18	58.99	0.19		0.05		2.4	0.5	1.55	9,683	0.90	0.20	0.29	0.01	0.0006
Combined																
Max.	0.10	1.30	260.00	7.60		240.00		23.0		11.00	60,900					
Min.	0.00	0.00	0.00	0.00		0.001		0.0		0.00	540					
Mean	0.037	0.08	23.4	0.18		1.05		2.1		0.30	6,315					
Std Dev.								3.5								

Table 3.5 Statistical Analysis of Chemical Parameters from upper Newer Volcanics Aquifer (Sources: RWC Groundwater Database and Melbourne Water Data) (All units in mg/L)

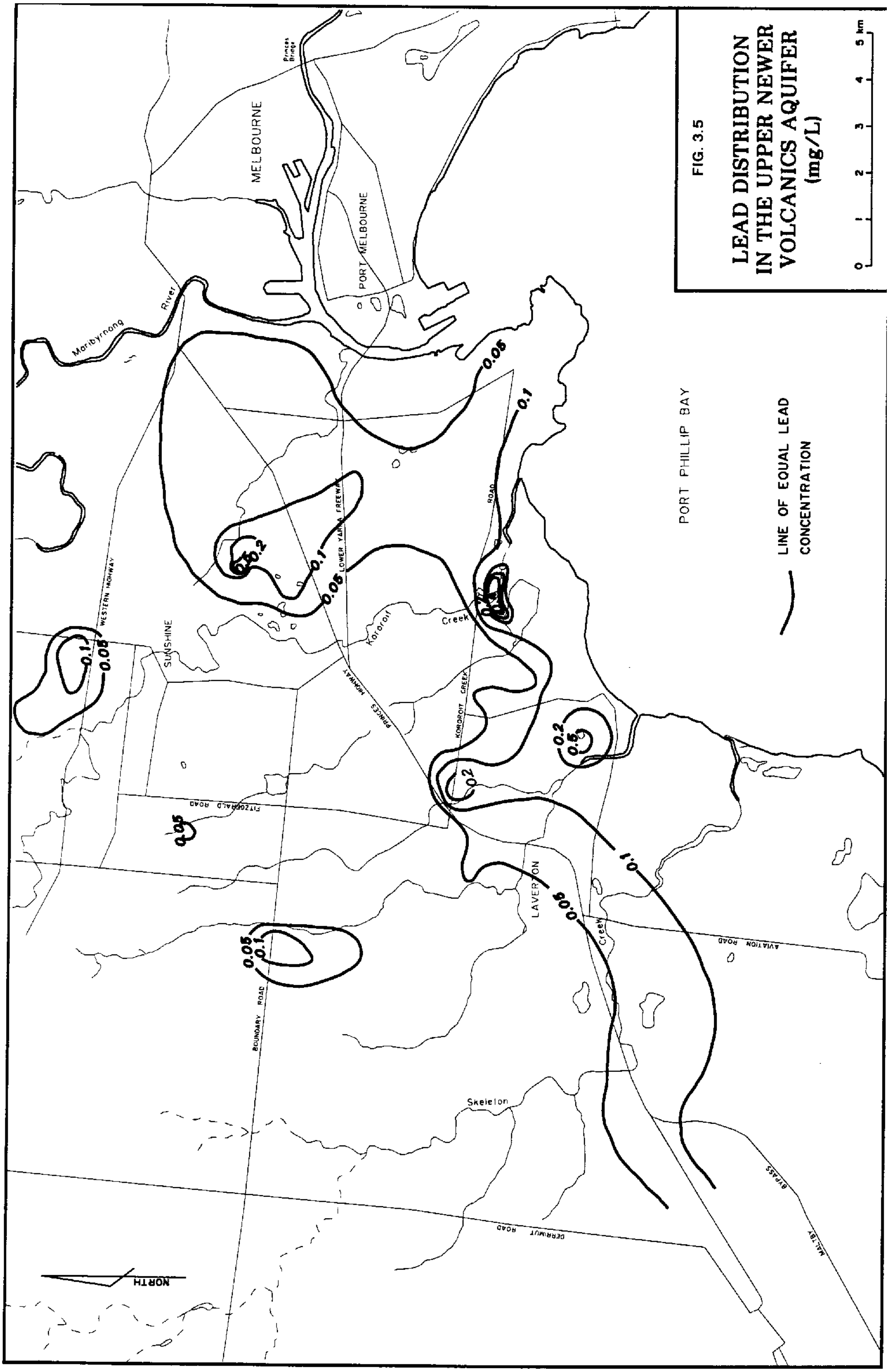


Figure 3.5 Lead Distribution in the Upper Newer Volcanics Aquifer (mg/L)

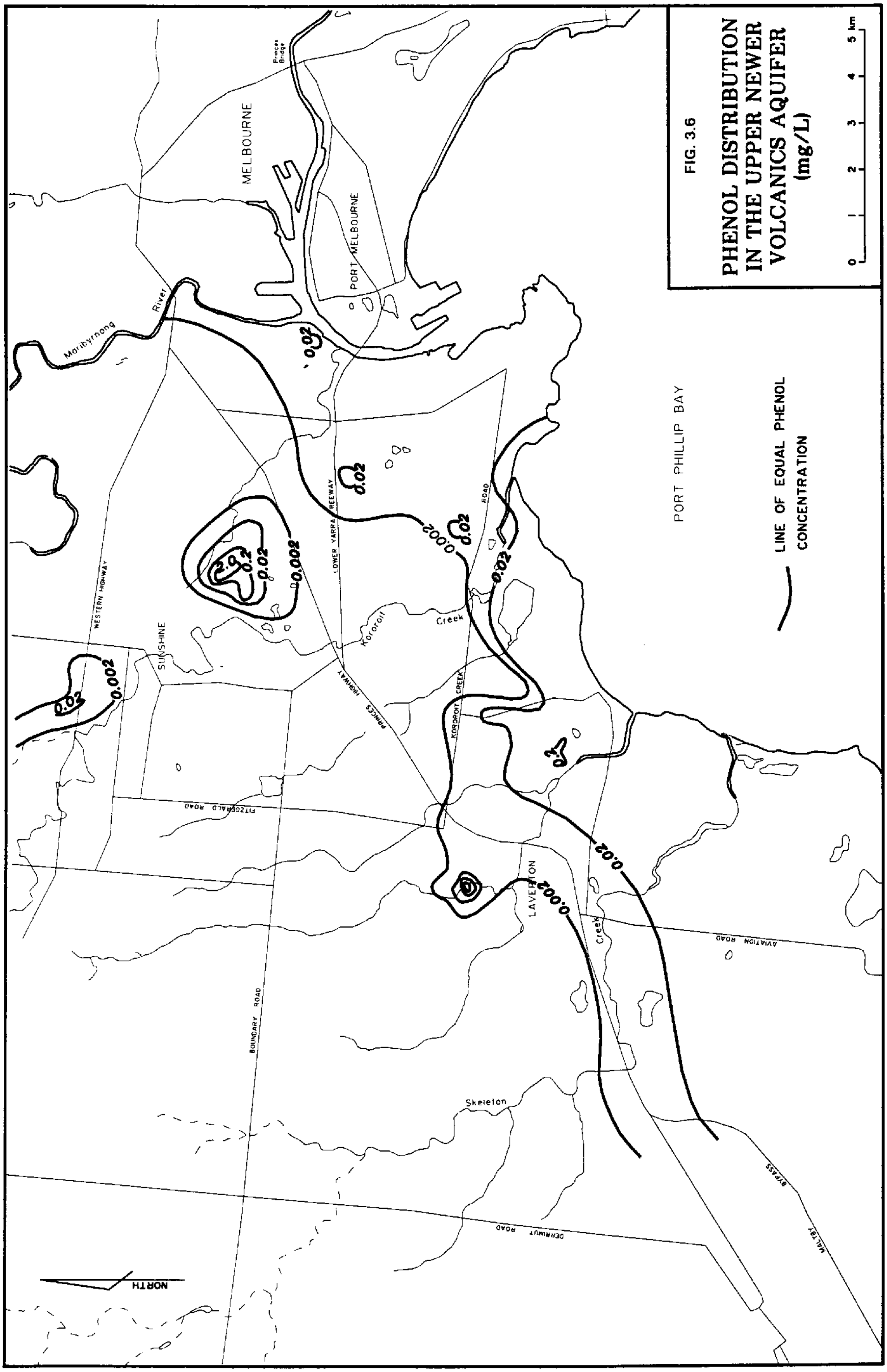


Figure 3.6 Phenol Distribution in the Upper Newer Volcanics Aquifer (mg/L)

In general however, a decrease in contaminant concentration would be the result of dispersion and attenuation processes occurring in the aquifer system. The degree of attenuation will depend on the type of contaminant and the aquifer material. Basalt aquifers, especially where weathered, tend to have a reasonably high capacity to attenuate metals either by cation exchange or by precipitation (Leonard, 1992). Consequently, heavy metal contamination is generally localised near the pollution source. This is further evidenced by the decrease in Pb concentration as one moves downgradient (seaward) (Figure 3.5). However, the salinity in the basalt aquifer system increases down gradient from 7,000 mg/L to 16,000 mg/L (Figure 3.4). Phenolic compounds are attenuated mainly by dilution and dispersion, but as they have a low solubility in water, they tend to float on the groundwater and these processes can often be ineffective. This results in significant levels of phenols being detected further downgradient (Figure 3.6). The overall trend in phenol concentrations has however, decreased over time due to a change in waste disposal practices and the introduction of an above-ground treatment and disposal facility of phenolic substances in West Footscray (Riha, 1975).

The effect of groundwater contamination on surface water bodies is a significant factor in this area as approximately 90% of groundwater flow is into the streams and creeks which drain the region (see section 3.5).

3.5 Analytical Modelling - Groundwater Flow to Bay and Streams

In an effort to further understand the extent and nature of the groundwater pollution entering Port Phillip Bay an analytical model was designed to quantitatively differentiate the amount of groundwater that enters surface streams and creeks and that which enters the Bay.

The first step taken in this exercise was to generate a water table elevation map (Figure 3.3) of the upper Newer Volcanics aquifer and to construct on this the groundwater flow directions. Flow lines, which are perpendicular to the contour lines joining places of equal watertable elevation, were drawn. The flow lines define the direction of groundwater flow (Figure 3.7). This product is not strictly a true flow-net, as the flow cells are not curvilinear squares. Curvilinear squares could not be constructed on this surface as the aquifer is of varying thickness. The flow lines are used to define the direction of groundwater flow. Flux calculations employ the Darcy approach, rather than addition of flow tubes. It is assumed in this exercise that the aquifer is homogenous and isotropic. The direction of groundwater flow is always perpendicular to the contours. It can be seen on Figure 3.7 that some of the flow lines, and more specifically the areas between them, disappear and regenerate across different areas of the map. The areas where the flow lines increase in number (e.g. point A, Figure 3.7) define approximate areas of groundwater recharge, while the areas where the flow lines disappear (e.g. point B, Figure 3.7) define groundwater discharge areas. The groundwater discharge areas usually coincide with zones around the incised streams, implying that the streams have a significant groundwater baseflow component.

Throughout the study area the general direction of groundwater flow is to the south-east (Figure 3.7). Once the direction of groundwater flow has been established a Darcian calculation, assuming the continuum approach (that is, at a suitably large scale, the fractured rock aquifer is assumed to behave like a porous medium), can be applied to the aquifer system in order to determine the flow entering the Bay or the creeks and streams.

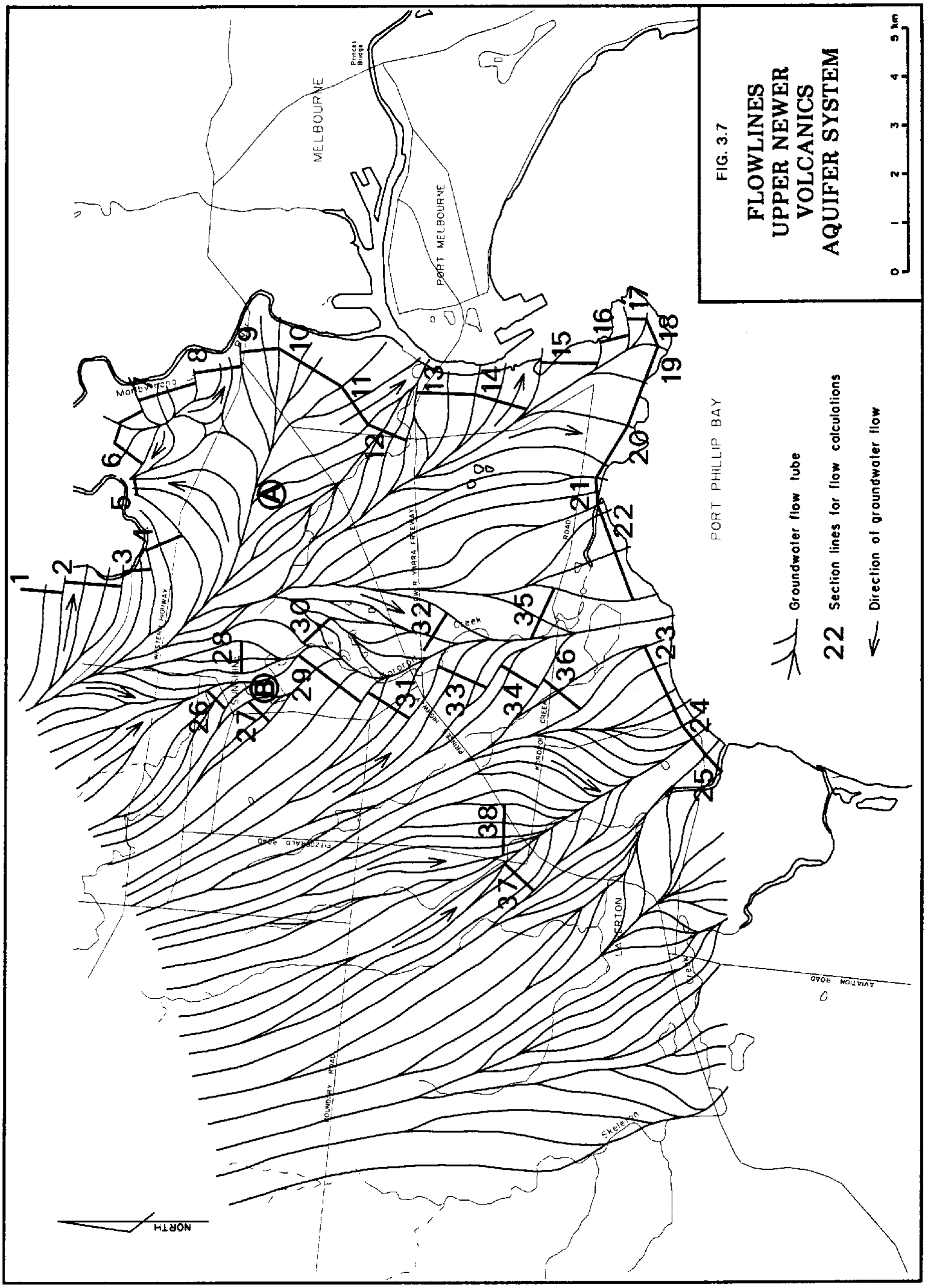


Figure 3.7 Flowlines Upper Newer Volcanics Aquifer System

Existing data and information from reports have been compiled to provide information on representative aquifer parameters of the Newer Volcanics. Table 3.6 contains the values of aquifer parameters that were used in this modelling exercise.

Table 3.6. Newer Volcanics aquifer parameters (after Riha and Kenley, 1978)

SECTION) (refer Figure 3.8	BORE NUMBER		HYDRAULIC CONDUCTIVITY (m/day) from Riha and Kenley (1978)
	DME NUMBER	RWC NUMBER	
1 to 4	Maribyrnong 35	109583	k=17
5 to 12	Cut Paw Paw 261	109502	k=5
13 to 18	Cut Paw Paw 246	109474	k=1.5
19 to 25	Cut Paw Paw 247	109476	k=1
26 to 30	Cut Paw Paw 254	109488	k=4
31 to 36	Cut Paw Paw 247	109476	k=1
37 to 38	Truganina 17	109709	k=1

All values represented in the above table were obtained from pumping tests designed to take into account the special nature of these basalts (Riha, 1971a).

Hydraulic gradients represented in the model were calculated from the potentiometry of the upper basalt aquifer (Figure 3.3).

The final parameter needed is the area through which the groundwater is flowing and this depends on the thickness of the aquifer (b_i). The thickness of the aquifer along the discharge segments was estimated using the isopach map for the upper aquifer (Figure 3.2). This, however, may not be a realistic representation of the system as the streams and creeks that exist in the area do not fully penetrate the aquifer, thus introducing some error into the calculation. If, however, it is assumed that the effective aquifer base (b_e) is 2 m below sea level, then a more realistic estimate of the discharge is obtained. Both approaches, however, were used in the model, yielding a flow based on the full aquifer isopach (Q_i) and a flow based on the effective aquifer thickness (Q_e). The results have been expressed as a range, using values of b_i and b_e.

Appendix H and Table 3.7 represent the results of the modelling exercise.

Table 3.7. Groundwater flow to Bay and streams utilising hydraulic conductivity values of Riha and Kenley (1978)

Surface Water Destination	Qi (m^3/day)	Percentage of total flow to Bay (Minimum)	Qe (m^3/day)	Percentage of total flow to Bay (Maximum)
Maribyrnong River	15930	72.3	13541	84.3
Yarra River	108	0.5	92	0.6
Kororoit Creek	5056	23	1890	12
Skeleton Creek	356	2	196	1
Port Phillip Bay	514	2	249	1.5

From this calculation the majority of the groundwater discharge is into surface streams and creeks rather than directly into the Bay. As the hydraulic conductivity is variable, and governed by local rock defects, anomalous values may be recorded from bores in high or low defect areas. The large volume of groundwater flow into the Maribyrnong River is essentially an artefact of the anomalously high value of K from the nearest bore which has undergone a pumping test. As a means of verification, a uniform K of 4 m/day has also been applied to all sections so that the proportion of groundwater discharge can be more accurately estimated. The results are presented in Appendix H and table 3.8.

Table 3.8. Groundwater flow to Bay and streams utilising uniform hydraulic conductivity value of 4 m/day

Surface Water Destination	Qi (m^3/day)	Percentage of total flow to Bay (Minimum)	Qe (m^3/day)	Percentage of total flow to Bay (Maximum)
Maribyrnong River	6312	38.1	5365	53.0
Yarra River	288	1.8	245	2.4
Kororoit Creek	6364	38.4	2477	24.3
Skeleton Creek	1424	8.6	783	7.7
Port Phillip Bay	1987	12.0	936	9.2

Utilising the spatially varying conductivities (Riha and Kenley, 1978) and the average conductivities in tables 3.7 and 3.8, approximately 2% - 12% of total groundwater discharge is directly into Port Phillip Bay. This compares favourably with Riha and Kenley's (1978) estimate of 20% of total groundwater flow discharging directly into the Bay. This is an important process to understand as it will affect the monitoring of these surface streams in the future. Appendix E identifies those sites that may be adversely affecting nearby surface water streams and creeks by reference to their proximity to the Bay and to the nearest stream, via the flow lines represented in Figure 3.7.

3.6 Contaminant Transport - Fate of Contaminants in the Sub-Surface

Contaminant transport in groundwater is essentially a function of the aquifer type, aquifer mineralogy, permeability and groundwater flow velocity. The key elements of transport for conservative (non-reactive) constituents are:

- Advection - the component of solute movement attributed to transport by flowing groundwater;
- Dispersion - the mixing of solute with non-contaminated water via flow through the porous medium. Mixing that occurs along the groundwater flow lines is called longitudinal dispersion. Mixing that occurs perpendicular to the groundwater flow lines is lateral dispersion (Fetter, 1988).
- Diffusion - the movement of solute of a higher concentration to an area of lower concentration. Diffusion is possible even in an aquifer where the hydraulic gradient is zero, as the concentration potential drives the mixing process.

In reactive constituents, a fourth process termed adsorption or attenuation, can be added. Reactive constituents can undergo chemical, biological or radioactive change that will reduce the concentration of the solute (Fetter, 1988). Adsorption-desorption and cation exchange reactions between solutes and the solid matrix are very important for hazardous wastes (Fetter, 1988). Heavy metals are readily adsorbed onto solid surfaces or trapped in clay lattices through ion exchange. If a solute is reactive, it will travel at a slower rate than the groundwater, due to adsorption. The rate of solute movement depends on the extent of retardation of the solute, a parameter which is determined experimentally, on a site by site basis.

The first three processes of advection, dispersion and diffusion cannot readily be separated in flowing groundwater. A factor, combining all three processes, can be employed, termed the coefficient of hydrodynamic dispersion. Hydrodynamic dispersion describes the process of movement of a non-reactive solute in flowing groundwater. The one dimensional equation for hydrodynamic dispersion (Freeze and Cherry, 1979) is given by:

$$D_L \frac{\delta^2 C}{\delta x^2} - \nu_x \frac{\delta C}{\delta x} = \frac{\delta C}{\delta t}$$

Where:

D_L	is the longitudinal dispersion coefficient;
C	is the solute concentration;
ν_x	is the average groundwater velocity in the x direction;
t	is the time since start of solute invasion; and
x	is the distance along the groundwater flow direction.

This equation can be reworked to give a means of calculating the concentration, C, at some distance, L, from the source of contamination, C_0, at time, t:

$$C = \frac{C_0}{2}\left[erfc\left(\frac{L - \nu_x t}{2\sqrt{D_L t}}\right) + \exp\left(\frac{\nu_x L}{D_L}\right) erfc\left(\frac{L + \nu_x t}{2\sqrt{D_L t}}\right)\right]$$

where *erfc* is the complimentary error function (Freeze and Cherry, 1979).

3.6.1 Analytical Contaminant Transport Modelling

The above equation has been utilised in an analytical model to estimate the concentrations of solutes through porous media aquifers. This is designed to provide an understanding of the levels of contamination which may travel down gradient of an input site. Parameters which are representative of the aquifer have been utilised. The calculations are in one dimension only, and represent the movement parallel to the groundwater flow direction. Lateral dispersion is ignored. The results are presented in Figure 3.8 and Appendix I. The graph (Figure 3.8) and spreadsheets depict the changes in concentration which occur through migration of the contaminant. The calculations assume that the contaminant is from a discrete, continuous source. The results are useful in estimating the likely concentration of a solute along groundwater flow lines and when the "contaminant front" is likely to reach an area of significance (e.g. a stream or the Bay).

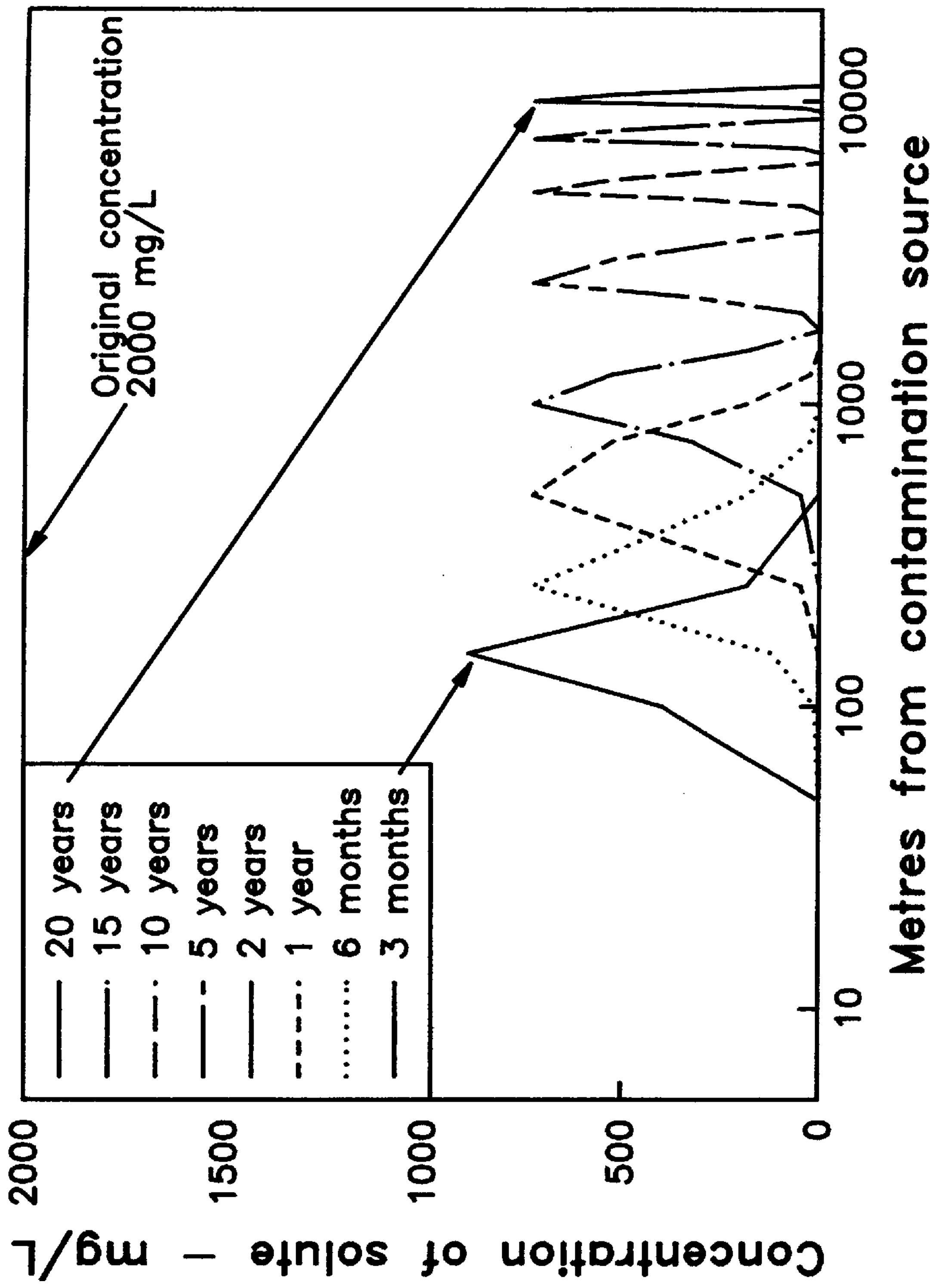


Figure 3.8 Hydrodynamic dispersion model of solute movement over time.

As a rough guide, using the parameters in Appendix I, a "contaminant front" can be expected to flow 1,000 m (to a level of approximately 25% of the original concentration) after approximately 2 years. The peak concentration of approximately 50% of the original concentration (as dispersion and advection have spread the concentration longitudinally) has travelled 10,000 m after approximately 20 years (Figure 3.8). These simulations are heavily dependent on hydraulic gradient and effective porosity (utilising the results from Riha, 1975 and Riha and Kenley, 1978) and also ignore the fourth parameter, adsorption. Therefore, the above estimates are conservative and present a "worst-case" scenario, as adsorption of the solute would be expected to retard the migration rate and concentration of the contaminant.

3.6.2 Numerical Contaminant Transport Modelling

The equation of hydrodynamic dispersion has been applied analytically in the above case, assuming a continuum (porous media) approach. A more indicative approach, using a numerical solute transport groundwater model, has also been undertaken to provide a greater perspective on the extent of potential contamination in the western suburbs area.

The SHE model (Systeme Hydrologique Europeane) is a hydrological model developed by the Danish Hydrological Institute. It is a finite difference model, containing saturated and unsaturated groundwater flow components into which an advection-dispersion module can be linked to simulate the movement of a solute in the groundwater flow field. It has been employed in this study to obtain an indicative picture of the extent and rate of solute movement in the Upper Newer Volcanics aquifer. It simulates the impact of a constant contaminant input to the aquifer and has utilised existing hydraulic parameters of conductivity, potentiometry and aquifer morphology. Additional parameters for the advection-dispersion module include horizontal longitudinal dispersivity (2500 m), horizontal transverse dispersivity (25 m), recharge flux to each flow cell (based on 15% recharge from 450 mm annual rainfall) and input solute concentration (250 mg/L). The input solute can be assumed to be any aqueous species, but for this example an input representative of typical values of phenol has been used.

The model has been conceptualised as a two-layer system, modelling the upper and lower Newer Volcanics aquifers, with a small degree of interconnection between the aquifers. This is consistent with the hydrogeological assessment of the degree of aquifer interconnection made earlier in this report. The modelling area is depicted in Figure 3.9. A finite difference grid covering 20 rows and 20 columns has been constructed over the study area. A contaminant input site, covering three model cells and corresponding to a munitions depot, has been nominated at the northern boundary of the model area.

The movement of solute was simulated over a ten year period and the results appear in Figures 3.9 and 3.10. Important observations from this indicative modelling include:

- The dispersion of the solute in the immediate vicinity of the input site, such that the solute concentration drops from 250 mg/L to a maximum of 100 mg/L;

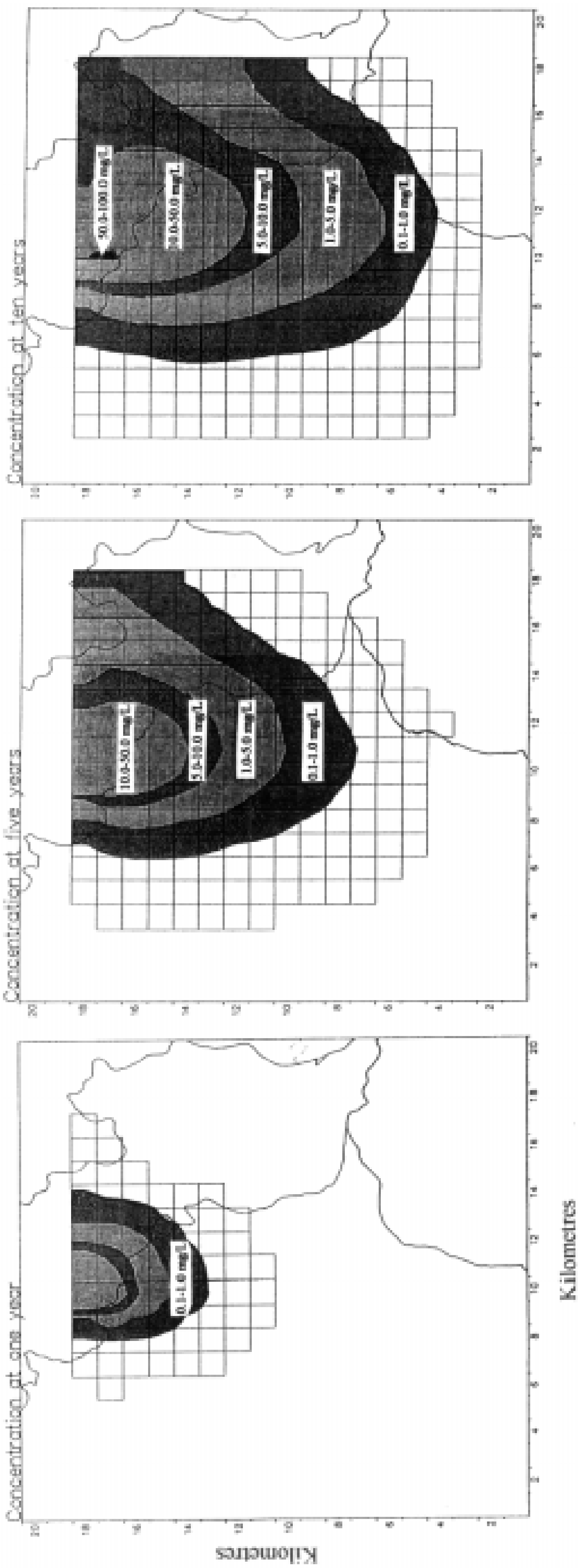


Figure 3.9 Simulated solute movement through Upper Newer Volcanics Aquifer

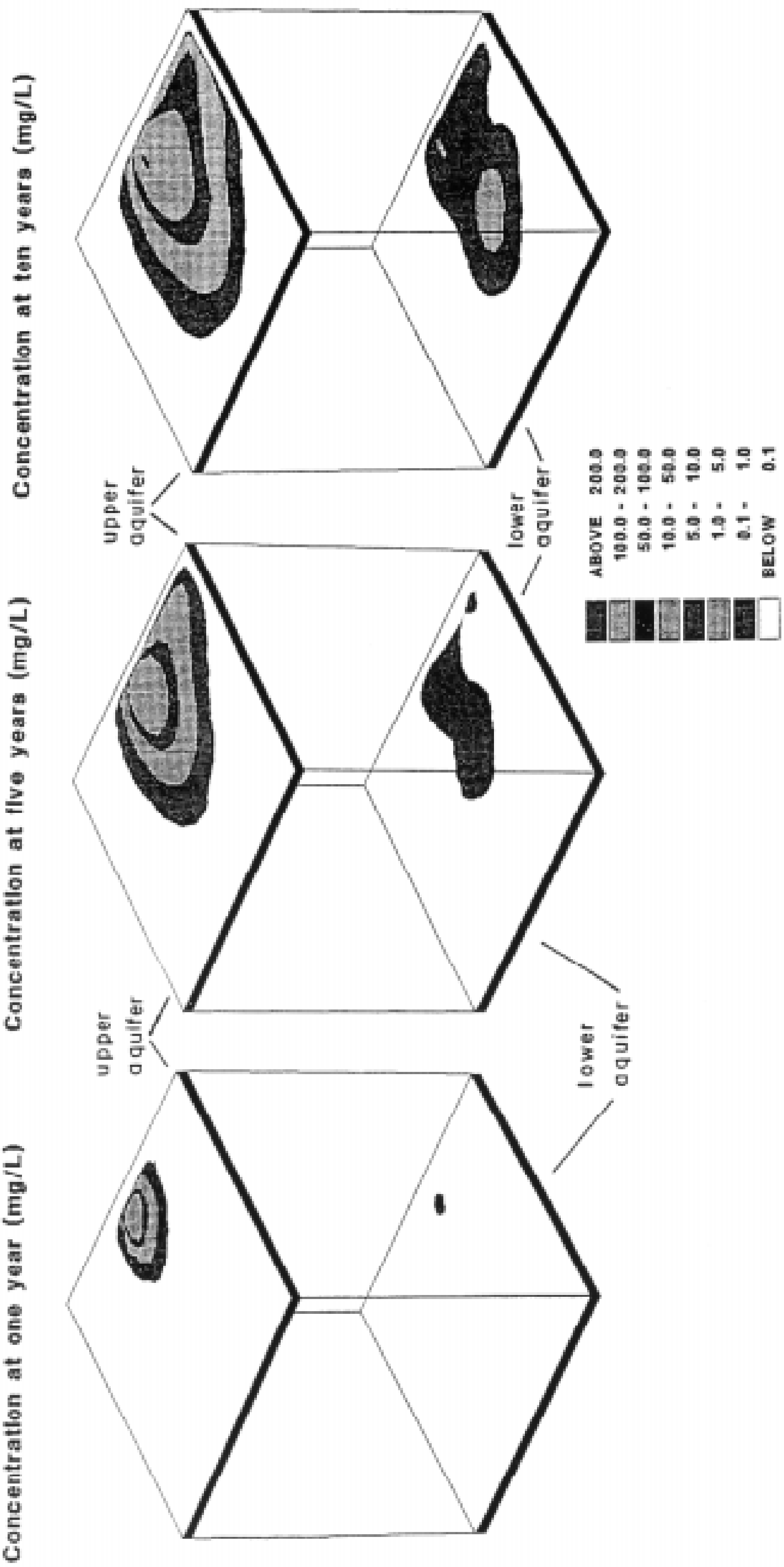


Figure 3.10 3-Dimensional diagram of simulated solute movement through Newer Volcanics Aquifer

- The movement of the solute with respect to time. After 1 year the 1-5 mg/L zone of contamination has migrated down gradient approximately 2 kilometres. After 5 years this has increased to 6 kilometres and after 10 years the contaminant zone has extended approximately 8 to 10 kilometres. This concurs well with the one-dimensional hydrodynamic dispersion modelling carried out in the preceding section, which estimated the rate of migration to be in the order of 500 m per year.

- The contaminant plume is very close to streams. The transverse dispersion of the aquifer will promote the lateral spreading of the plume across groundwater flow tubes and enhance the rate of contaminant flow into the surface water system. The proximity of contaminant sites to the streams and the Bay is tabulated in Appendices E and F.

It must be noted that this modelling is indicative only and simulates the movement of a conservative solute (which ignores any retardation of the solute due to reaction of the chemical species with the aquifer). As such, the modelling presents a "worse case" scenario of the possible effects of aquifer contamination on proximal streams and the Bay.

More accurate modelling of solute transport involves additional field sampling to provide real values of dispersion, adsorption and hydraulic parameters and calibration with observed data. This type of detailed field/laboratory investigation is beyond the scope of this study.

3.6.3 Toxicant Load Modelling

The previous contaminant modelling results serve to provide a picture of the rate of flow and direction of contaminant plumes. However, the coarseness of the modelling precludes the accurate calculation of current toxicant loads to the Bay. The previous calculations of groundwater flow (Section 3.5) can, however, be combined with the chemical composition statistics to provide an estimate of the toxicant load entering the Bay and streams at the present time. Using the average concentrations for the upper Newer Volcanics aquifer (in Table 3.5) and the calculations of groundwater discharge, an estimate of the approximate loads can be presented (Table 3.9 and Appendix J). Regression analysis was also conducted in an attempt to identify a relationship between certain chemical parameters of the aquifer (Appendix K) which may have provided a means of characterising different polluted regions. No relationship was identified however, and consequently the toxicant loads were calculated using the mean toxicant concentrations as shown in Table 3.5. The toxicant load calculations are shown in Appendix J.

Toxicant loads into the Maribyrnong River via groundwater baseflow from the Newer Volcanics aquifer reaches an estimated maximum level of more than 4,000 tonnes/yr. Sulphate constitutes almost 95% of this load, with other notable toxicants comprising phenols, iron and nitrates. Groundwater loads of toxicants into the Maribyrnong River is by far the greatest of the surface water bodies analysed here, followed by the Kororoit Creek where a maximum of more than 1,300 tonnes/yr of toxicant load enters the creek. The majority of this consists of sulphate and iron. Port Phillip Bay directly receives a maximum of 136 tonnes/yr of toxicant load from the groundwater system and similarly this consists mainly of sulphate and iron loads. Skeleton Creek and the Yarra River receive relatively small amounts of toxicant load via the groundwater system with a maximum of 94 and 29 tonnes/yr respectively. Cherry Lake receives approximately 14 tonnes/yr of toxicants from groundwater discharge.

Table 3.9. Toxicant loads to Port Phillip Bay

TOXICANT LOADS (tonnes/yr)							
Parameter**		**Maribyrnong River**	**Yarra River**	**Kororoit Creek**	**Skeleton Creek**	**Port Phillip Bay**	**Cherry Lake**
Cd	Maximum	0.2145	0.0015	0.0681	0.0048	0.0069	0.007
conc.=0.4	Minimum	0.1823	0.0012	0.0254	0.0026	0.0034	0.000
Cu	Maximum	0.4884	0.0033	0.155	0.0109	0.0157	0.0017
conc.=0.08	Minimum	0.4151	0.0028	0.0579	0.006	0.0076	0.000
Fe	Maximum	135.983	0.9219	43.1593	3.0389	4.390	0.4609
conc.=23.4	Minimum	115.5851	0.7836	16.1318	1.6714	2.124	0.000
Pb	Maximum	1.0279	0.007	0.3263	0.0229	0.033	0.0035
conc.=0.18	Minimum	0.8738	0.0059	0.1219	0.0126	0.016	0.000
Zn	Maximum	1.7443	0.0118	0.5536	0.0389	0.0563	0.0059
conc.=0.3	Minimum	1.4827	0.0101	0.2069	0.0214	0.027	0.000
Mg	Maximum	3.8627	0.0262	1.2259	0.0863	0.125	0.0131
conc.=0.66	Minimum	3.2833	0.0223	0.4582	0.0475	0.060	0.000
Cr	Maximum	0.8964	0.0061	0.2845	0.0201	0.029	0.003
conc.=0.15	Minimum	0.7619	0.0052	0.1063	0.0111	0.014	0.000
Ni	Maximum	1.0708	0.0073	0.3399	0.0239	0.035	0.0036
conc.=0.18	Minimum	0.9102	0.0062	0.127	0.0132	0.017	0.000
As	Maximum	0.0653	0.0004	0.0207	0.0015	0.0021	0.0002
conc.=0.01	Minimum	0.0555	0.0004	0.0078	0.0008	0.0011	0.000
Hg	Maximum	0.0028	0.0001	0.0009	0.0001	0.0001	<0.0001
conc.=0.0005	Minimum	0.0024	0.0001	0.0003	<0.0001	<0.0001	0.000
Phenol	Maximum	6.0989	0.0413	1.9357	0.1363	0.1969	0.02
conc.=1.05	Minimum	5.1840	0.0351	0.7235	0.0750	0.0953	0.00
Sulphate	Maximum	4 043.7	27.4148	1283.4	90.4	130.5	13.7
conc.=695	Minimum	3 437.1	23.3025	479.7	49.7	63.17	0.00
Total N	Maximum	12.2	0.0827	3.9	0.27	0.39	0.04
conc.=2.1	Minimum	10.3788	0.0703	1.4	0.15	0.19	0.00
Total P	Maximum	1.7443	0.0118	0.55	0.039	0.06	0.006
conc.=0.3	Minimum	1.4826	0.0101	0.2	0.02	0.03	0.00
TOTALS	Maximum	4264	29	1356	94	136	14
(tonnes/yr)	Minimum	3624	24	499	52	66	0

****** all concentrations (conc.) expressed as mg/L

Due to the lack of correlateable spatial and time series water chemistry data, the calculated toxicant loads have utilised the average concentrations of groundwater from the upper Newer Volcanics aquifer. This method may tend to under- or over-estimate the results in a particular area, depending on the real level of toxicants in the aquifer at that particular location. This method, by implication, also assumes that the toxicant concentrations are evenly distributed throughout the aquifer; and ignores the possible effect of an advancing "contaminant front" or spreading toxicant "plume". If the areas of high toxicant concentrations (e.g. Figures 3.5 and 3.6) can be assumed to move downgradient at a rate approaching that calculated previously (i.e. up to 1,000 m over 2 years), then there is the possibility of a higher toxicant load discharging to the streams or the Bay within the next 10 years. At the scale of this investigation, and using existing data, it is not possible to discern whether the toxicant "highs" from each contaminant source have coalesced into a larger plume. However it may be assumed from the inventories of the known and potential contaminant sites that the aquifer contains a "cocktail" of different contaminants, forming an overlapping series of migrating plumes.

3.7 Discussion

The magnitude of toxicant loads discharged to streams and the Bay from groundwater in the western suburbs is cause for concern, particularly as the toxicant loads may be increasing due to contaminant plume migration. The groundwater toxicant loads are of a similar order of magnitude to the toxicant loads from surface water bodies which are described by NSR (1993). The current study has highlighted the major input of groundwater to the Bay via the streams. Approximately 90% of groundwater flow to the Bay comes via discharge to the Maribyrnong and Yarra Rivers and Kororoit and Skeleton Creeks. This suggests that effective stream monitoring (i.e. in the estuarine environment) may detect the bulk discharge of toxicants from both surface and groundwater sources. Another major area of concern is the possibility that the toxicant discharges may have not yet peaked, but may be increasing as the result of migration of overlapping contaminant plumes.

The toxicant loads represent point source concentrations, and consequently are expected to fluctuate with time. Point readings in an isotropic, heterogenous fractured basalt aquifer are, by nature, non-representative. However, the results presented here give an indication of the expected maximum level of contamination of the aquifer at the present time. Without additional data it is very difficult to further estimate the location, migration rate and peak concentration of the contaminant plumes. More intensive field observations and contaminant transport modelling would be able to address these issues more accurately.

4.0 NUTRIENT INPUTS - NEPEAN PENINSULA

4.1 Location

The Nepean Peninsula covers the western portion of the larger Mornington Peninsula. It extends from the Selwyn Fault in the east, which defines the boundary between the uplands of the Mornington Peninsula and the lower ground of the Nepean Peninsula itself, to Point Nepean in the west (Figure 1.1).

4.2 Topography

Overall relief of the Nepean Peninsula is low, reaching a maximum of 100 m (A.S.L.) in the east where the peninsula abuts the Selwyn Fault. The peninsula is typically covered with an undulating dune terrain, generally reaching an elevation of 30 m (Shugg, 1986).

4.3 Land Use

The coastal fringe of the Nepean Peninsula bordering Port Phillip Bay is densely settled with permanent and holiday residences. Further inland, in the central region known as "The Cups", grazing, market gardens and recreational areas predominate (Foley, 1992). The intensive urban settlement of the Peninsula by holiday makers and permanent residents commenced in the post-war years, and led to the area developing faster than services could be supplied. As a result, large areas were not sewered or supplied with reticulated water supplies. Water connections followed some years after initial development and some properties are still being connected to sewerage services. Previously, septic tanks and night soil depots were the main method of sewerage treatment in the area.

Nutrients (mainly nitrates) from sewerage disposal activities associated with the intensive coastal development along Port Phillip Bay are considered to pose a potential risk to the Bay.

4.4 Hydrogeology

In the Nepean Peninsula area, Palaeozoic age sediments and granites are overlain with up to 900 m of Tertiary and Quaternary sediments. The main hydrogeological units have been summarised in Table 4.1 and are described in more detail in Leonard (1992). The aquifer most at risk from nutrient contamination and most likely to provide a substantial conduit for groundwater flow to the Bay is the upper unit, the Bridgewater Formation aquifer. Leonard (1992) describes the Bridgewater Formation as consisting of calcareous and quartzose sand which overlies sand and clay of the Wannaeue Formation and Brighton Group. The three aquifers are in hydraulic connection and have been termed the Wannaeue-Bridgewater Formation Aquifer by Leonard (1992). This composite aquifer covers the entire Nepean Peninsula and achieves a maximum thickness of 200 m in the Sorrento area.

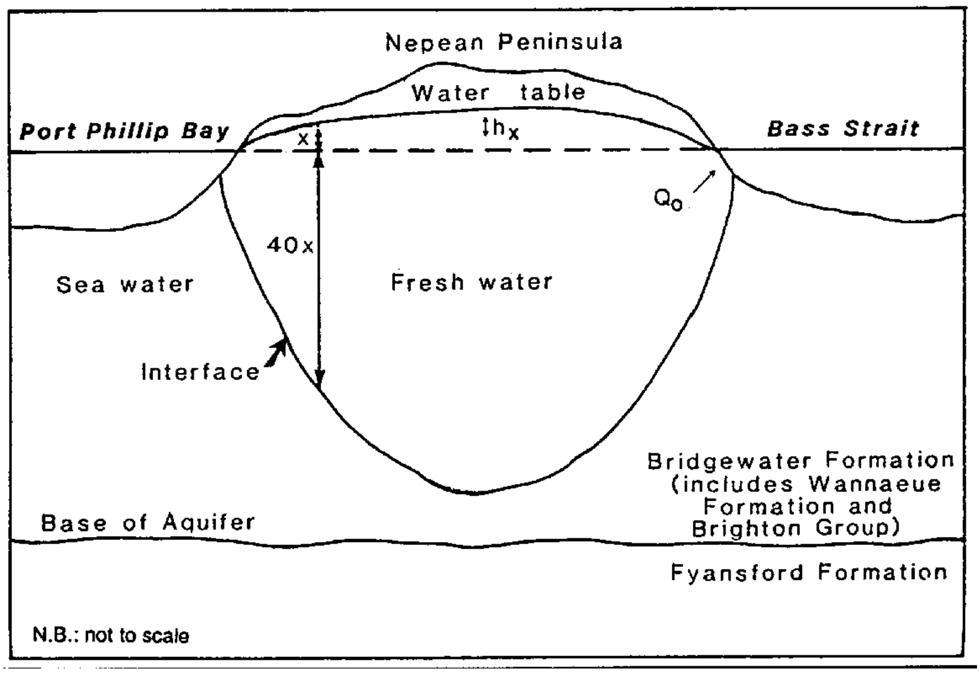
Hydraulic conductivity of the Bridgewater Formation aquifer is estimated to be in the order of 30 m/day (Shugg, 1986), whilst Leonard (1992) suggests that 20 m/day is more representative of the whole aquifer. Storativity for uncemented material ranges from 0.25 to 0.35, emphasising the coarseness of the aquifer (O'Rourke, 1993). This may reduce to 0.20 or less due to localised carbonate cementation in places (Leonard, 1992). Water quality generally ranges from 300 to 1200 mg/L TDS.

The groundwater salinity increases rapidly with depth, as the Bridgewater Formation aquifer is penetrated by a saline wedge due to its contact with sea water. A lens of freshwater floats on top of this salt water wedge. The salt water wedge dictates the nature of discharge of groundwater into the Bay and the Ghyben-Herzberg relationship describes the form of the freshwater-saltwater interface (See Figure 4.1).

TABLE 4.1. Principal hydrogeological units in the Nepean Peninsula area (After Leonard, 1992).

Aquifer Name	Lithology	Aquifer Type	Water Salinity (mg/L)	Nutrient Status
Bridgewater Formation	Sand, sandy calcarenite, shelly sand, clay.	Unconfined sand aquifer, sheet-like form.	300 - 1200 Approaching sea water salinity towards base	High
Wannaeue Formation	Sand, silt, clay.	Semi-confined to confined sheet-like aquifer.	500 - 25000	Low
Brighton Group	Sand, gravel, silt, clay, shelly sand, calcarenite limestone.	Sand and silt sheet-like aquifer; Confined to semi-confined.	2500 - 8000	Low
Werribee Formation	Sand, gravel, clay, coal, lignite.	Semi-confined to confined aquifer of sand and gravel; sheet-like form; lignite and coal beds contribute to marked anisotropy in hydraulic conductivity.	> 4000	Low
Bedrock	Folded mudstones, siltstones, sandstones and shales, affected to varying degrees of regional and contact metamorphism.	Fractured rock aquifers. Semi-confined to confined. Physical properties of aquifer varies according to degree of fracturing and weathering.	1000 - 5000	Low

Figure 4.1. Salt water interface and fresh water lens in the Nepean Peninsula (After Leonard, 1992)



4.5 Groundwater Contamination

The principal known groundwater contaminant in this area is nitrate. This is primarily due to the occurrence of both active and inactive septic tanks, night soil depots, waste disposal sites; and also from nitrogenous fertilisers and animal waste from agricultural land (Foley, 1992). The possibility of heavy metal, inorganic and organic contamination to the groundwater systems may also exist, but it is considered negligible as the extent of industrial land is small. There is also insufficient data, by way of chemical analyses, for these contaminant types.

There are two forms of nitrogen of major concern in groundwater contamination. These are nitrate (NO_3^-) and ammonium (NH_4^+) (Foley, 1992). The fate and transport of nitrate ions is determined by the movement of groundwater, uptake in plants or crops and denitrification of nitrates to nitrogen gas (N_2). Once in the groundwater, nitrate is very mobile and moves with minimal transformation because of its solubility and anionic form. If the subsurface materials are porous, nitrate can travel long distances and the only condition which can affect this process is a decline in the redox potential of the groundwater which may result in denitrification (Canter and Knox, 1988). Effluent from septic tanks situated in sandy soils can be expected to undergo aerobic reactions and nitrates will be formed by nitrification (oxidation) of ammonium (Canter and Knox, 1988). With the sandy aquifers of the Nepean Peninsula, the majority of ammonium from septic tank effluent and other sources is likely to be converted to nitrate and transported in the groundwater.

Nitrate values (as nitrogen) obtained from chemical analyses stored on the RWC's Statewide Groundwater Database range from less than 0.15 to a maximum of 37 mg/L N, with an average concentration of 3.2 mg/L N (Foley, 1992). These nitrate figures are expressed as units of nitrogen (a multiplication factor of 4.43 converts between nitrate as N and nitrate as NO_3). World Health Organisation (WHO) guidelines for drinking water quality set a maximum limit of 10 mg/L nitrate (as N) (Van der Leeden *et al*., 1990).

4.6 Modelling

The Bridgewater Formation aquifer, due to it being in contact with sea water on its northern and southern sides, can be conceptualised as an infinite-strip oceanic island (Figure 4.2). The fresh water takes the form of a lens floating on more dense salty water.

Fetter (1988) describes groundwater outflow, per unit width, as equal to the recharge rate times the half-width of the conceptualised "island":

$$q' = \omega \frac{a}{2}$$

where q' = discharge per unit width;
ω = areal recharge rate; and
a = width of "island"

Another equation describes the head of the water table, h, at any distance, x, from the shoreline (Fetter, 1988):

$$h^2 = \frac{\omega[a^2 - (a - \chi)^2]}{K(1 + G)}$$

where h = head of aquifer (m);
K = hydraulic conductivity (m/d);
G = density constant;
χ = distance from shoreline (m);
ω = areal recharge rate (m/d); and
a = width of "island" (m)

These equations were applied to two sections bisecting the Nepean Peninsula (Figure 4.2). Section A, extending from Point Nepean to Rye, has a cross-sectional length of 16,000 m and an average width of 1600 m. Section B, extending from Rye to the Selwyn Fault, has a length of 10,500 m and an average width of 5000 m.

Recharge has been assumed to be 20% of the average annual rainfall (20% of 750 mm/yr), after Leonard (1992). Total calculated groundwater flows are depicted in Table 4.2.

TABLE 4.2. Groundwater flows to Port Phillip Bay from Bridgewater Formation

	SECTION A	SECTION B	TOTAL
FLOW (m^3/day)	5,250	10,760	16,010
FLOW (ML/year)	1,920	3,930	5,850

These results compare favourably with those of Leonard (1992), who calculated a total groundwater outflow from this aquifer to the Bay of 6,450 ML/yr.

Nutrient loads, in terms of N and P, can be calculated based on average concentrations. A concentration of 3.2 mg/L of N was utilised, which is the average of recent values (Foley, 1992). No analyses for phosphorous exist in the area. Purely as a comparison, however, the P values from the Werribee Delta area (2.9 mg/L) were used to provide an indicative, maximum, expected P load. The nutrient load calculations for groundwater flow into Port Phillip Bay are depicted in Table 4.3.

TABLE 4.3. Nutrient loads to Port Phillip Bay from Bridgewater Formation

FLOW (ML/year)	N (mg/L)	N LOAD (t/year)	P (mg/L)**	P LOAD (t/year)	P LOAD (realistic estimate refer to Section 4.7 - t/year)
5844	3.2	18.7	2.9	16.9	3

**** Using average P concentration from Werribee Delta.**

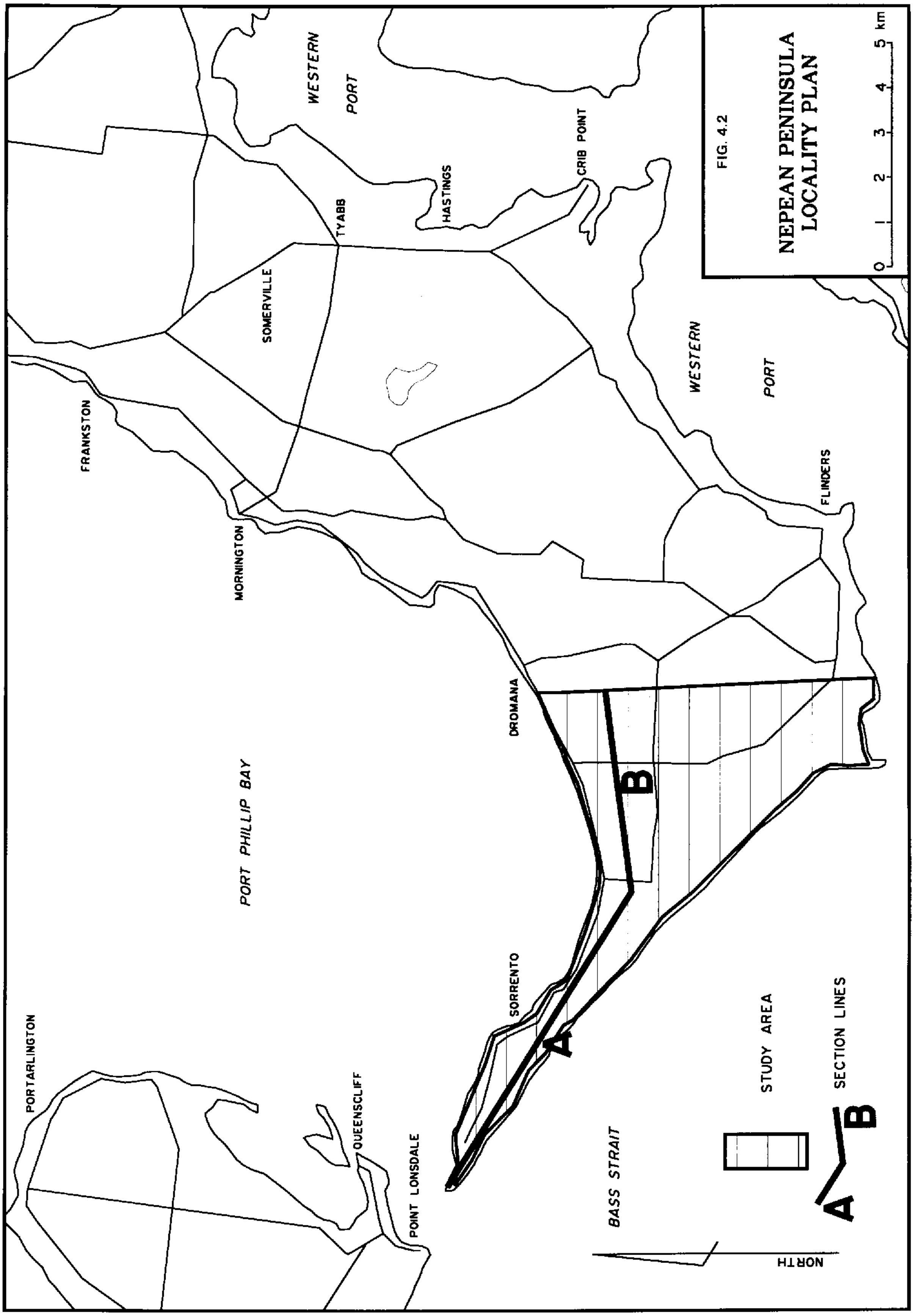


Figure 4.2 Nepean Peninsula Locality Plan

4.7 Discussion

As can be seen from the above table, the total estimated nutrient loads appear to be relatively low. The figures calculated for P, based on the concentration from the Werribee Delta area provide an unrealistically high P load, when compared to the calculated N load. The P load is an expected maximum, and a more realistic figure would be 2-3 tonnes/yr, if one applies a similar ratio which exists between N and P in the Werribee Delta area (i.e. P load is approximately 10% - 15% of the N load).

The N load calculations are based on analyses for nitrates only. It may be necessary to enlarge the groundwater sampling programme to analyse for other nutrient species, including NH_3 and Total Kjeldahl Nitrogen, especially if nitrification has not occurred as quickly as expected. Phosphorous should also be analysed.

5.0 TOXICANT INPUTS - GEELONG-NORTH SHORE

5.1 Location

This section of the Port Phillip Bay Environmental Study deals with the potential groundwater contamination discharging into Corio Bay, the western arm of Port Phillip Bay. The hydrogeological province under investigation encompasses the city of Geelong and extends from Batesford in the west to Point Wilson in the east (Figures 1.1 and 5.1).

5.2 Topography

The topography of the Geelong-North Shore region varies from a maximum 150 m (A.H.D) located at the south-west boundary of the study area at Barrabool Hills to sea level along the shoreline of the Bay. Apart from the Barrabool Hills, the Dog Rock granites form another regional high in the area.

In general, the topography governs the direction of groundwater flow, resulting in Corio Bay potentially receiving groundwater discharge on all three sides.

5.3 Land Use

The region is zoned into a number of different land use categories including:

- Residential
- Industrial (Manufacturing)
- Industrial (Extractive)
- Agricultural

Residential activity for the Geelong-North Shore region includes mainly the primary development of the Geelong district which comprises of a population in excess of 100 000 people with an anticipated growth rate of 1.1% per year (Leonard, 1992).

Manufacturing industries around the Geelong district include approximately 300 factories (Leonard, 1992). Established manufacturing industries are summarised in Table 5.1.

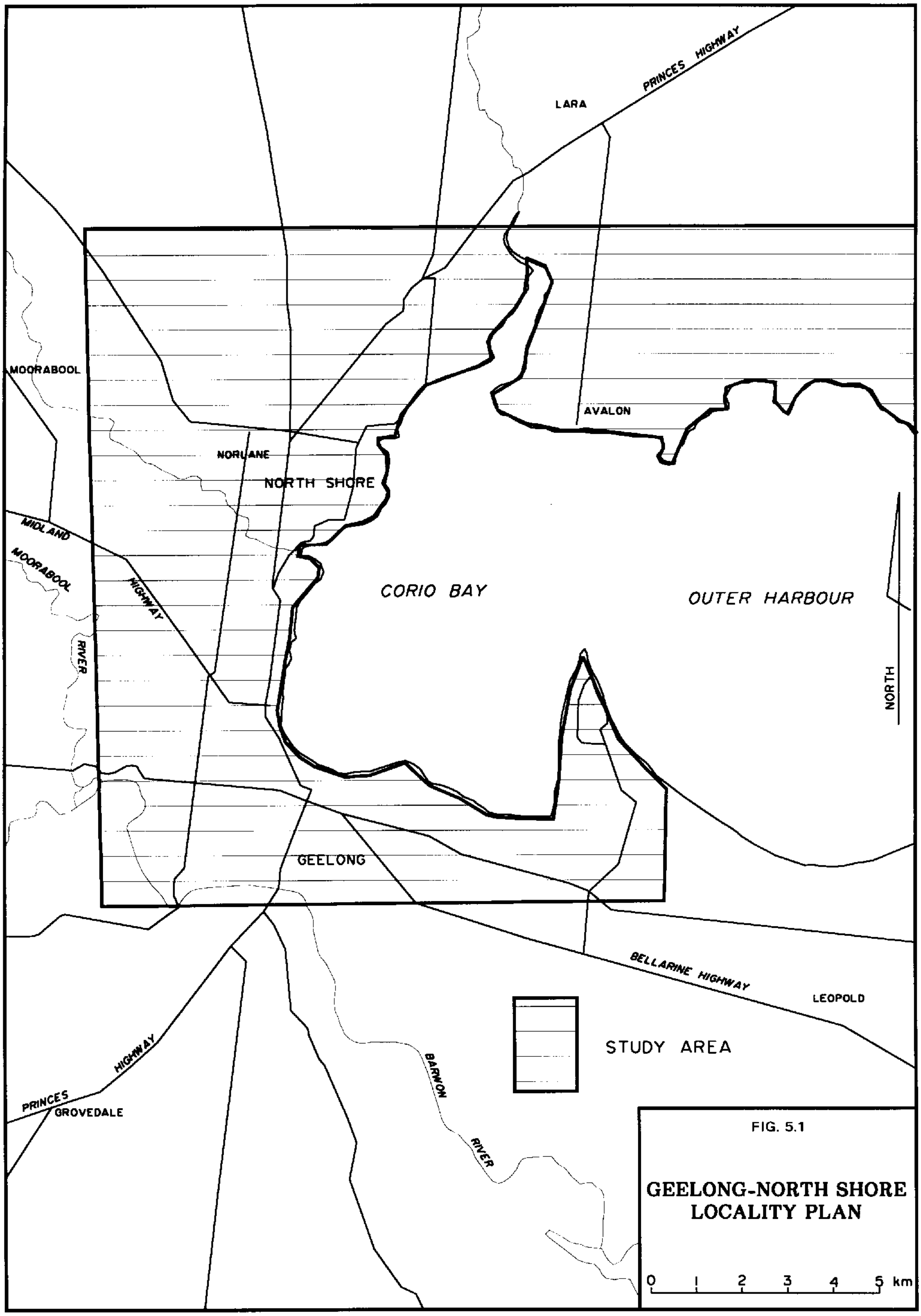


Figure 5.1 Geelong-North Shore Locality Plan

Industry	*Location*	*East*	*North*	*Possible Contaminants*	*Proximity to Bay (km)*	*Proximity to Stream (km)*
Shell Petrochemical Refinery	Corio	270000	5782000	Hydrocarbons Nutrients Suspended solids Sulfide *E. coli*	0.9	
Terminals Pty Ltd	Corio	270500	5781200	Organics Hydrocarbons	0.2	
Corio Container Terminal	North Shore	270500	5780500	Hydrocarbons	0.2	
Australian Wire Industries Pty Ltd	North Shore	269700	5780500		0.6	
Phosphate Co-op Co. of Australia Ltd	North Shore	269700	5780100	Fluoride compounds Suspended solids Total Phosphorous	0.5	
Ford Geelong Casting Plant	North Shore	269500	5781000	Heavy Metals	0.9	
Golden Fleece Service Station	Corio	270100	5783200	Hydrocarbons	1.3	
T. Costa and Co.	Corio	269800	5783150		1.4	
Stamping Plant and Toolroom	Norlane	268200	5779500	Heavy Metals	0.9	
Ford Engineering Centre	Norlane	267800	5779800	Hydrocarbons Heavy Metals	1.4	
Australian Portland Cement Ltd	Fyansford	264500	5775200		3.8	0.2 Moorabool R.
Malting Plant	Geelong North	268200	5778100	Heavy Metals	0.2	
Grain Elevator Board Facilities & Bulk Grain Pier	Geelong North	268500	5778400		0.3	
Chassis and Misc. Compounds Manuf. Plant	Geelong North	267900	5778800	Heavy Metals Hydrocarbons	1	
Cheetham Salt Works	Newcomb	271500	5773500	Inorganics Salinity	0.1	
Dunlop Pulsar Batteries	Breakwater	270200	5770100	Heavy Metals Lead Inorganics	4	1.1 Barwon R.
Quarry	Fyansford	263800	5774700		5	0.2 Moorabool R.
Industrial Areas	Moolap	273500	5772200		1.6	
Shire of Corio Rubbish Tip	North Shore	271800	5784000	Inorganics Organics Toxins	1.1	
Harbourside Industrial Complex	North Shore	26900	5779400		0.2	
Former Abbattoirs	Geelong West	267200	5776500	Organics Hydrocarbons	1.1	
Geelong Terminals	Geelong	268100	5774600	Organics Hydrocarbons	0.4	
ALCOA	Point Henry	274500	5776000	Suspended Solids Fluoride Phosphorous Chromium	1	
Former Tip Site	Point Henry	274200	5776100	Disposal of 'Potliner' - Carbon & Fluorine & Cyanide	1	
Commonwealth Explosives Area	Avalon	274100	5781500	Organics Heavy Metals		

Table 5.1 Contamination Potential from Land Use in Geelong-North Shore Area.

Extractive industries can be defined as any activity which requires the removal of material comprising part of an aquifer system. Such industries in the Corio Bay district include many small basalt quarries located near Geelong, Batesford and Fyansford. The Moorabool Viaduct Sand (member of the Brighton Group) is occasionally mined and used as foundry sand.

5.4 Hydrogeology

The four main groundwater systems of the Corio Bay area comprise the surficial sands, clays and beach deposits, Newer Volcanics Basalts, Moorabool Viaduct Sands (representing the Geelong equivalent of the Brighton Group) and the Torquay Group including the Fyansford Clays and the Batesford Limestone. The main hydrogeological characteristics of the different aquifer systems existing in the Geelong-North Shore district are summarised in Table 5.2 and are described in more detail in Leonard (1992). Table 5.3 summarises all the hydrochemical analyses available for the Geelong-North Shore region.

The uppermost aquifers are those which are at highest risk from contamination due to industrial and other land uses. In this study, the aquifers considered at highest risk are:

- Undifferentiated Sands, Clays and Beach Deposits
    - South-east Geelong
    - Point Henry
    - Limeburners Bay
    - Point Wilson
- Pleistocene Newer Volcanics basalts
    - Western edge of Corio Bay
- Pliocene Brighton Group (Moorabool Viaduct Sands)
    - Western edge of Corio Bay

5.4.1 Undifferentiated sands, clays and beach deposits

The undifferentiated sands, clays and beach deposit unit is exposed south-east of Geelong and also at Point Henry, Limeburners Bay and Point Wilson. The unit includes river alluvium, swamp deposits, dunes and beach deposits.

Aquifer parameters (transmissivity and hydraulic conductivity) for this unit are not known. Yield estimations are based on data from just one bore, indicating a pumping rate in the order of 3.5 L/sec, located near Avalon.

The groundwater salinity for the Quaternary sediments around the Corio Bay tends to be poor with TDS readings in the order of 45,000 mg/L TDS. The high TDS readings tend to reflect the location of the two bores where the samples were taken, being along the northern and southern boundaries of the Bay. As this water salinity is greater than that of sea water (approximately 36,000 mg/L TDS) it may be postulated that the higher water quality from these two bores is the result of groundwater discharge and salt accumulation through evaporation of water. It is considered that the salinity of the groundwater for the Undifferentiated unit improves in areas away from the shore boundaries of Corio Bay. Recharge activity for the Quaternary sediments is by direct infiltration of rainfall or discharge from localised streams.

Aquifer System	Lithological System	Occurrence	Aquifer Thickness (m)	Groundwater Quality				Aquifer Parameters		
				TDS (mg/l)	pH (mg/l)	N (mg/l)	Fe (mg/l)	K (m/day)	T (m2/day	Yield (l/sec)
Undifferentiated sands, clays and beach deposits	• Sands, clays and beach deposits	• South-east of Geelong • Point Henry • Limeburners Bay • Point Wilson	0.5 - 6.0	45,000	7.4	3.6	5.0	**	**	3.5
Newer Volcanics	• Bluish grey to black separated by clay and silt sediments • Aquifer tends to be mainly confined	• Exposed at the northern and western edges of Corio Bay. The occurrence of the unit is predominantly around the north-western portion of the Geelong-North Shore study area	0.5 - 20.0 *	8,000 - 20,000	7.9	0.2 - 7.9	33.0 - 60.0	1.0 *** (poorly fractured) 6.0 *** (Highly fractured)	50.0 *** 300.0 ***	0.7 - 5.0
Brighton Group ***Moorabool Viaduct Sands***	• Calcareous sand, clayey sand, quartzite and sand and gravel • Largely unconfined	• Exposed at the western and south-eastern edge of Corio Bay. The occurrence of the unit covers most of the study region with the exceptions of Barrabool Hill and at Dog Rock	5.0 - 30.0 *	6,000	7.5	63.0	28.0	**** 2.5 - 15.0	75.0	0.5-15.0
Torquay Group ***Batesford Limestone***	• Limestone, sandy fossilliferous • Confined by Fyansford Formation	• There is limited outcropping of this unit in the study area. It underlies the majority of the Geelong - North Shore region, excluding Barrabool Hill area	25.0 - 75.0 *	3,000	8.4	0.45	1.0 - 17.0	13.5	37.0	5.0

NOTE: The data gathered are from bore information available in the Geelong - North Shore district. Some estimates are based on one or tw bores due to the lack of available data

* Data taken from Leonard, 1992
** Parameters are not known
*** Riha and Kenley, 1978
**** Estimated using T = Kb relationship

Table 5.2 Principal hydrogeological units in the North Shore Area

Bore Number	Monitored Aquifer	Sample Number	Sample Date	Sample From	Sample To	TSS (mg/L)	CL (mg/L)	CO_3 (mg/L)	HCO_3 (mg/L)	TOTAL ALK (mg/L)	SO_4 (mg/L)	N (mg/L)	CA (mg/L)	MG (mg/L)	NA (mg/L)	K (mg/L)	FE (mg/L)	HARD	pH	EC
48857	?	30061	06/01/83			438	100		171	140	32	5.30	4	5	130	5	7.10	31	8.6	720
64297	Batesford Limes	6696	01/04/74	58.00	88.40	3841	2145		140		188	1.10	147	202	995	13	14.00	1197	7.58	6610
64302	Batesford Limes	6698	15/05/81	18.00	56.00	3640	1808	39	201		259	0.20	18	149	1092	30	1.00	659	8.90	6200
64303	Batesford Limes	6699	21/05/81	13.00	55.00	2815	1918	30	383		149	0.20	144	197	954	27		1171	8.20	6500
64304	Batesford Limes	6700	01/03/81	40.70	60.50	862	356	71	95		30	0.20	68	38	191	4		326	8.20	1600
64305	Batesford Limes	6701	16/03/81	43.80	56.00	1085	254		320		59	0.20	75	46	210	5		375	7.70	1850
64306	Batesford Limes	6702	23/03/81	37.60	56.00	1190	425		310		70	0.20	78	77	196	7		511	8.10	2000
64307	Batesford Limes	6703	22/04/81	46.80	49.40	1135						1.10							7.90	1940
64309	Batesford Limes	6704	06/03/81	37.50	46.70	690	219	12	170		63	0.20	31	32	150	2		209	8.60	1200
80122	New Volcanics	30654	05/05/83			12126	6900		537	440	690	0.20	300	500	3800	88		2829	7.80	21000
81337	Batesford Limes	12994	01/04/74	48.80	54.90	3996	1864		561		193	1.80	119	163	1032	37	1.50	967	7.84	6580
81338	Batesford Limes	12995	17/01/75	48.80	53.30	4090	2178	21			297		503	1	995	45	17.00	1260	11.40	7590
		12996	18/01/75	64.00	67.10	3874	2203		92		201		171	201	960	15	15.00	1254	7.90	6950
		12997	18/01/75	73.20	82.30	5182	2840		264		286		239	298	1222	18	4.00	1823	7.85	9000
111402	?	46087	23/03/92			5798	3200		512	420	440	0.10	310	360	1400	16		2271	7.70	10000
81391	Batesford Limes	13017	29/01/64	167.30	189.60	3390	1566	12	604		303		126	181				1060	8.20	
		13018	03/03/64			3310	1384		932		262		20	184				810	8.10	
		13019	05/03/64	167.30	176.80	2900	1269	48	403										8.30	
		13020	16/03/64	167.30	189.60	2788	1263	35	455		199		77	142			36.00	778	8.30	
		13021	17/03/64	167.30	189.60	2764	1187	13			200		86	127			32.00	736	8.30	
		13022	24/04/64	167.30	189.60	2900	1212		620		200		112	120				773	8.10	
		13023	05/05/64	167.30	189.60	2627	1165		477		210		101	117				732	7.80	
		13024	08/05/64	167.30	189.60	2702	1202	40	485										8.50	
		13025	24/06/64	167.30	189.60	2711	1184	44	481										8.10	
81403	Brighton Group	35360	24/01/83			5980	2800		732	600	370	63.00	91	160	1900	18	28.00	892	7.50	10000
82982	New Volcanics	13786	19/10/71	21.30	22.90	9332							26	277				1327	7.49	13742
		13787	19/10/71	21.30	22.90	17558	8670		267				233	586				2995	7.88	24957
82989	New Volcanics	13794	02/06/80	4.00	8.50	22622	11390		545		2346	0.70	197	625	7300	210	33.00	3064	8.10	32300
82990	New Volcanics	13795	02/06/80	11.00	12.00	15513	8140		462		1233	2.90	164	484	4885	123	60.00	2401	8.10	24000
82991	Undifferentiated	13796	02/06/80		0.50	39725	21980		221		3061	3.60	639	1489	11860	440	4.00	7725	7.70	50600
104989	Undifferentiated	21778	26/04/78	9.10	13.70	61113	34950		359		3780		927	2513	18546	24	6.00	12669	7.15	84000

Table 5.3 Groundwater chemistry of the principal aquifers in the Geelong-North Shore Area.

5.4.2 Newer Volcanics

The Newer Volcanics system is exposed along the northern and western edges of Corio Bay. The formation covers the north-western portion of the Geelong-North Shore study region.

The aquifer occurs as a sheet-like cover filling the deep valleys of the Barwon and Moorabool River. Different rates of cooling in different parts of the flow has resulted in the development of three different rock types. There is a coarse-grained columnar basalt occurring at the valleys with thicknesses greater than 18 m, an intermediate layer of fine-grained black basalt of irregular columns and a contrasting, thin highly vesicular basalt, found across the plains (Bowler, 1963).

The aquifer overlies the Brighton Group and is largely unconfined. Due to the nature of the aquifer the groundwater regime of the Newer Volcanics tends to vary considerably. Aquifer parameters such as groundwater flow, storage capacity, transmissivity, water quality and yield vary depending on the fractures and joints of the aquifer. The transmissivity of the aquifer can be grouped into two categories depending on the joints and thickness of the system (Riha and Kenley, 1978), namely:

- Poorly jointed and/or thin basalt – Transmissivity = 50 m^2/day
  Hydraulic Conductivity = 1 m^2/day

- Highly jointed and/or thick basalt – Transmissivity = 300 m^2/day
  Hydraulic Conductivity = 6 m^2/day

Yields from the bores in the Newer Volcanics range from 0.7 to 5.0 L/sec, depending on the density and on the number of joints and fractures of the aquifer material.

The salinity of the aquifer ranges from 8,000 to 20,000 mg/L TDS. Salinities are generally lower in areas close to zones of natural recharge (Leonard, 1992). Groundwater is typically hard with high concentrations of chloride, sulphate, calcium, bicarbonate, magnesium and sodium. Total nitrate concentrations, for bores with available data, are less than 5 mg/L with pH average of 8.1.

Recharge activity for the Newer Volcanics unit is mainly via rainfall infiltration in areas where the aquifer outcrops, infiltration from the alluvium sediments near the shoreline of Corio Bay or via discharge from localised streams. Groundwater flows are generally in a southerly direction towards Corio Bay. Discharge occurs either into surface streams, Corio Bay or springs around the periphery of the lava flows, (Leonard, 1992).

5.4.3 Brighton Group (Moorabool Viaduct Sands)

The Moorabool Viaduct Sands represents the Geelong equivalent of the Brighton Group which is exposed around the western and south-eastern edges of Corio Bay. The occurrence of the aquifer exists throughout the Geelong North region, excluding areas around Barrabool Hill and Dog Rock. The aquifer comprises of sands, silt, clay and mottled yellow red-brown sandstone. The aquifer is generally unconfined.

Transmissivity readings of 75 m^2/day were obtained in the Point Wilson area, reflecting the higher sand content and greater thickness of the unit around the Geelong district (ICI Australia, 1978).

Yield potential of the Moorabool Viaduct Sands unit ranges from 0.5 L/sec (south edge of Corio Bay) to 15 L/sec (northern shores of Corio Bay).

From the data search undertaken around the Geelong-North Shore region only one bore provided groundwater chemistry data for the Brighton Group. This bore is located near Hermsley (southern margin of Corio Bay) and is used to depict the hydrochemical characteristics of the unit as no other water quality data is available for the Brighton Group in the Geelong-North Shore region. The water quality from this bore is poor with a salinity of 6,000 mg/L TDS. High concentrations of chloride, bicarbonate, sodium, magnesium, sulphate and calcium were observed for this unit. A total nitrate analysis of 63 mg/L (as N) was recorded from this bore.

Recharge to the aquifer is by direct infiltration of rainfall in areas where the unit is exposed or from the overlying aquifer which predominantly includes the Newer Volcanics and to a lesser extent from the Quaternary deposits.

5.5 Groundwater Contamination

The Geelong-North Shore area is an area where groundwater contamination is expected to be high, due to the number of industrial sites, waste disposal sites and chemical sites surrounding Corio Bay. Table 5.1 illustrates the potential contamination risk due to former or current industrial or other land uses for the Geelong-North Shore region.

Known and potential contaminated sites have been compiled using data from the EPA, RWC and private sources. Two sites within the Geelong-North Shore area are currently listed on the EPA State Register of Confirmed Contaminated Sites.

The first site is currently vacant land located at Douro Street, Geelong North. Former uses of part of the land included a car wrecking yard and industrial operations involving plastics and pigment manufacturing. Other parts of the site were used as a clay quarry for brickworks and as the disposal site for gasworks residues. Contamination of concern includes high levels of inorganics (cadmium, lead, zinc and mercury). Organic chemical contaminants and phenols were also found to exist on the site. This particular site is situated on the Brighton aquifer system.

The second site is located at Buckley Grove, Moolap and is presently used for chemical manufacturing use (wood preservatives). In the past the area was used for agricultural purposes.

5.6 Modelling

Analytical modelling, groundwater flow and nutrient or toxicants load calculations have been undertaken for the Werribee Delta, western suburbs and Nepean Peninsula areas. The calculations rely upon the existence of a suitable amount and coverage of groundwater data. Data on potentiometric surfaces, hydraulic gradients, aquifer morphology (to enable cross-sectional areas to be calculated) and toxicant analysis are required for the Geelong-North Shore area to enable such calculations to be made. This detailed data does not currently exist, although a summary of available data is presented in Appendix L.

Previous hydrogeological studies (e.g. Leonard, 1992) have been too broad to provide the detail required to fully understand the toxicant loads which may discharge to the Bay in this area.

As a result, reliable estimates of the toxicant load to Corio Bay from aquifers in the Geelong-North Shore area cannot be attempted with the sparse data currently available.

5.7 Discussion

Due to a lack of data further hydrochemical and hydrogeological investigations are necessary to reliably estimate the nutrient loads to the Bay from groundwater discharge in the Geelong region. It is suggested that a drilling program be undertaken along the shores of Corio Bay to obtain both hydrochemical and hydrogeological parameters of the aquifer systems in the high risk category (i.e. the Undifferentiated sediments, Newer Volcanics and Brighton Group).

The lack of groundwater data from this region is of concern. Ongoing monitoring of aquifer levels and chemistry in the relevant aquifers would provide an insight into the extent of pollution and the extent of transmission of contaminants to the Bay.

6.0 SUMMARY OF ASSESSMENT OF NUTRIENT AND TOXICANT RISK

This first phase of the groundwater component of the Port Phillip Bay Environmental Study has assessed the contaminant risk to the Bay from discharging groundwater in the following areas:

- Northern Werribee Delta - Nutrients
- Nepean Peninsula - Nutrients
- Western Suburbs - Toxicants
- Geelong-North Shore - Toxicants

Available data has allowed estimates of nutrient or toxicant loads to be calculated for three of these study zones. Insufficient data has precluded the calculation of reliable toxicant loads from the Geelong-North Shore area.

Estimates of the nutrient and toxicant loads from the three areas are presented in Table 6.1. As a comparison, results from surface water inputs are also included.

TABLE 6.1. Summary of nutrient and toxicant loads

	N (t/yr)	P (t/yr)	Cd (t/yr)	Pb (t/yr)	Fe (t/yr)	Phenols (t/yr)
North-east Werribee Delta (Groundwater)	10 - 24	1 - 3				
South-west Werribee Delta (Groundwater)	20	3				
Nepean Peninsula (Groundwater)	18.7	3				
Western Suburbs (Groundwater)	12.2 - 16.9	1.7 - 2.4	0.2 - 0.3	1.0 - 1.4	136.3 - 187.9	6.1 - 8.4
Patterson River*	516	333	0.08	7.13	1073.33	
Mordialloc Main Drain*	332	269	0.03	3.29	445.13	
Mordialloc Settlement Drain*	223	113	0.07	7.65	870.07	
Kororoit Creek*	103	167	0.02	0.49	62.91	
Werribee Treatment Complex#	4000	1200				

* Source: NSR (1993)
\# Source: CPI (1992)

The above table shows that, in terms of nutrients, groundwater appears to play only a minor role in total contributions to Port Phillip Bay. Heavy metal loads, in particular cadmium and lead, appear to be high when compared to estimates for surface water systems. The high groundwater loads may be a function of the non-representativeness of some of the groundwater analyses in the western suburbs. As mentioned earlier, the bores tended to be sited near known polluting activities, yielding a biased result. The Geelong-North Shore area would be expected to yield similar values for toxicant loads as those from the western suburbs. This assumption is reinforced by the prominence of Cowies Creek as a source of surface water toxicants (NSR, 1993). Groundwater discharge to Cowies Creek may explain the high toxicant levels encountered.

7.0 CONCLUSIONS

This study has provided the Port Phillip Bay Environmental Study with estimates of nutrient and toxicant contributions to the Bay from groundwater. From the data analysed, it is apparent that nutrients appear to play a minor role when compared with surface inputs. Toxicants are more of an issue. A number of factors render the toxicant loads important in the overall input of contaminants to the Bay. These are:

- The results of this study show that in terms of nutrients, groundwater only makes a minor contribution to the waters of Port Phillip Bay. Heavy metal loads (in particular lead and cadmium) do however appear to be high compared to those from surface streams.

- Rate of migration. From numerical and analytical modelling of hydrodynamic dispersion and solute transport (Section 3.6), it appears that the solute can migrate at rates between 500 and 1000 m per year. This means that there is the possibility of a large "contaminant front" discharging to the Bay in the future. The migration of solutes in the Newer Volcanics aquifer needs further investigation.

- Sufficiency of data. The Geelong-North Shore area suffers from a lack of hard data with which to estimate the toxicant risk to the Bay. Establishment of monitoring bores and field testing may be warranted. A further matter is the known existence of groundwater, soils and geotechnical data collected for site assessments of local industry in the western suburbs and Geelong-North Shore areas. Some of this information has been accessible for this study, but it is suspected that there are many more investigations which are not known to the EPA or the study team.

- Surface water monitoring. Monitoring of toxicant and nutrient loads to Port Phillip Bay from streams must be sited to also pick up the groundwater baseflow which contributes to the streams. This is particularly important in the western suburbs and Geelong-North Shore areas. Approximately 90% of groundwater discharge to the Bay from the Newer Volcanics aquifer of the western suburbs has been found to occur via initial discharge to streams. Existing stream monitoring sites would miss the important groundwater baseflow component.

- Representativeness of monitoring bores. The groundwater data collated from the western suburbs was originally collected for studies of known polluting areas or for specialist geotechnical studies. The issue of the extent of a large scale, combined, contaminant plume was not identified as one worthy of monitoring. As a result, given the heterogeneity of the aquifer, it is not possible to accurately estimate the extent and path of any large scale contaminant plumes.

8.0 RECOMMENDATIONS

After analysis of the data and presentation of these results, several recommendations for future directions become apparent:

- Surface water monitoring needs to be established in the streams traversing the western suburbs and Geelong-North Shore areas to pick up the important groundwater baseflow component. Sampling of the stream estuary at various depths will detect toxicant loads discharging to the Bay.

- Data needs to be collected to more accurately characterise the interaction of the Newer Volcanics aquifer with the streams of the western suburbs. A number of trial sites should be established adjacent to streams or between streams and known contaminating sites, monitoring hydraulic gradients, groundwater velocity, groundwater quality and dispersion parameters. A site may also need to be established along the coastal margin of the western suburbs (Altona - Williamstown) to trace the bulk water quality changes and the subsequent discharge to the Bay.

- The migration of solutes in the Newer Volcanics Basalt aquifer needs further investigation. More detailed solute transport modelling of the abovementioned trial sites, in conjunction with field observations, is required. This is suggested to enable more realistic estimates of toxicant migration rates, salt water interfaces and solute flow paths. Such a study would focus on the trial site(s) in the highest priority area, the western suburbs. The data produced may also be of relevance to the basalt areas of the Geelong-North Shore area.

- Groundwater monitoring sites need to be established in the Geelong-North Shore area to define the aquifer morphology, aquifer parameters and to provide a means of monitoring variations in chemistry and water level. Suggested sites would be along the coastal margin.

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