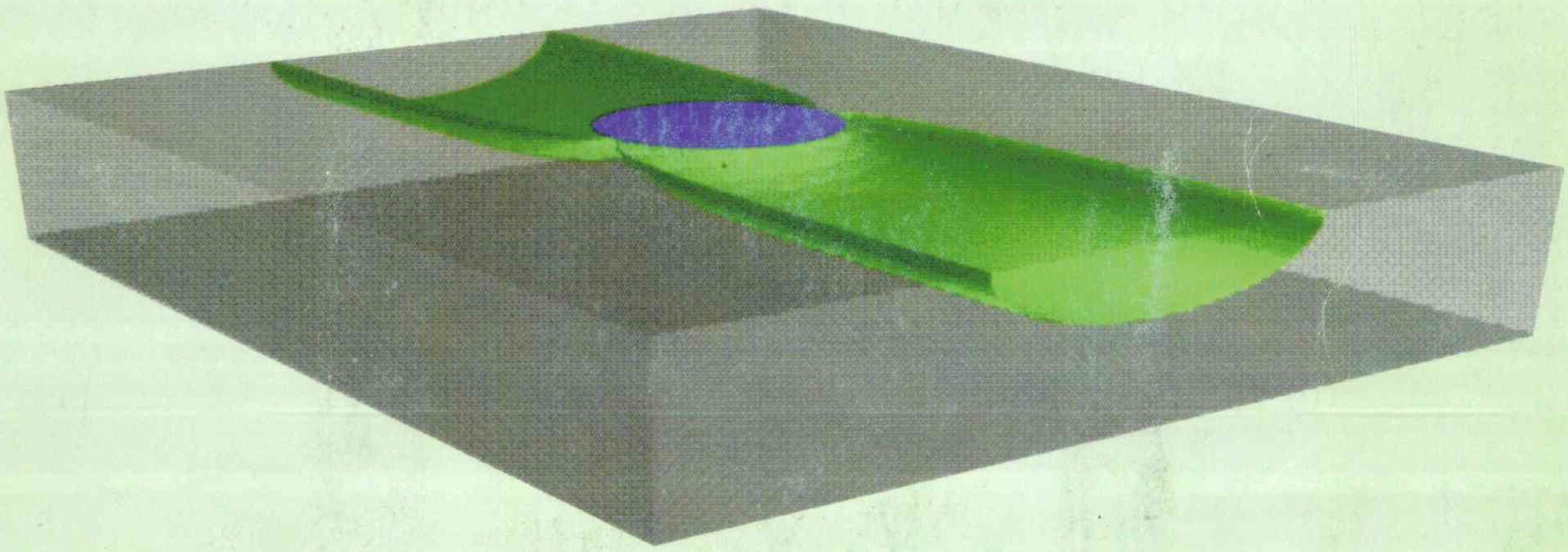
WETLANDS OF THE SWAN COASTAL PLAIN

Volume 3



Interaction Between Lakes, Wetlands and Unconfined Aquifers

V Turner, A D Barr, M G Trefry,
litis, C J Harris and C D Johnston

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Preface

The wetlands of the Swan Coastal Plain provide a habitat for many millions of animals and plants. Prior to European settlement the flora and fauna of these wetlands would have been as rich and diverse as the wetlands of Kakadu today. Many of the wetlands that once existed on the Swan Coastal Plain have been filled, drained, mined for peat or clay, or cleared of vegetation. As the urban and rural areas continue to expand and engulf more wetlands, the remaining wetlands become pressured by vegetation clearing, grazing, flooding, organic and metal contamination, nutrient enrichment and groundwater extraction. Clearly, there is a need to establish the extent of the wetland resource and develop strategies for wetland management.

Good management requires detailed information about the groundwater, the plants and the animals that live in these wetlands and the processes that bind them together. Good wetland management requires wetland boundary description, detailed information about surface and groundwater hydrology and the plants and animals that depend on them. Research provides information that enables managers to restore unhealthy wetlands and protect and conserve healthy ones.

Jeff Kite at the Water Authority of Western Australia, and the late Jenny Arnold, Bob Humphries and John Sutton at the Environmental Protection Authority and the Australian Water Resources Advisory Council (now Land and Water Resources Research and Development Corporation) have been instrumental in coordinating and funding five wetland research projects. The first of these projects commenced in 1988 and each project ran for at least 3 years and cost a total of $1.2 million in funding.

Research proposals for these projects were developed by Lloyd Townley and Jeff Turner at CSIRO Division of Water Resources, by Stuart Halse and Jim Lane at the Department of Conservation and Land Management in Woodvale in association with Rodney Vervest and Roger Jaensch of the Royal Australasian Ornithologists Union and Tony Ford at the Water Authority, by Jenny Davis and Arthur McComb at Murdoch University and Ron Rosich at the Water Authority. Other people that have been involved in coordinating these projects are Brian Kavanagh, Charlie Nicholson, Paul Lavery, Roy Stone and Karen Hillman.

Together with information on mapping and classification of wetlands by the V & C Semeniuk Research Group, initiated and coordinated by Alan Hill at the Water Authority, the results of this research form volumes 2 to 7 in this series Wetlands of the Swan Coastal Plain.

In order to encourage use of the information resulting from this research, volume 1 is a synthesis of information on wetlands of the Swan Coastal Plain and a tool for managers of wetlands.

Shirley Balla
May 1993

Wetlands of the Swan Coastal Plain

Volume 1. **Their nature and management.**
S A Balla.

Volume 2. **Wetland mapping, classification and evaluation.**
A Hill, C Semeniuk & V Semeniuk.

Volume 3. **Interaction between lakes, wetlands and unconfined aquifers.**
L Townley, J Turner, A Barr, M Trefry, K Wright, V Gailitis, C Harris & C Johnston.

Volume 4. **The effects of altered water levels on wetland plants.**
R H Froend, R C C Farrell, C F Wilkins, C C Wilson & A J McComb.

Volume 5. **Managing Perth's wetlands to conserve the aquatic fauna.**
S A Balla & J A Davis.

Volume 6. **Wetland classification on the basis of water quality and invertebrate community data.**
J A Davis, R S Rosich, J S Bradley, J E Growns, L G Schmidt & F Cheal.

Volume 7. **Waterbird usage of wetlands on the Swan Coastal Plain.**
A W Storey, R M Vervest, G B Pearson & S A Halse.

Table of Contents

Acknowledgements

This report summarises the results of a three-year project entitled "Interaction between lakes, wetlands and unconfined aquifers", which was funded as Project P88/12 under a Partnership Research Program of the Australian Water Research Advisory Council (AWRAC). After the creation of the Land and Water Resources Research and Development Corporation (LWRRDC), the Project became known as R&D Project CWW5. Partners in funding were the Water Authority of Western Australia (WAWA) and the Environmental Protection Authority (EPA) of Western Australia. The project funding provided support for one full-time and two half-time scientists/technicians for three years (Anthony Barr, Vit Gailitis and Ken Wright, respectively), as well as support for field instrumentation, drilling and computing. Additional support was provided by the Geological Survey of Western Australia (GSWA), particularly in relation to extensive drilling undertaken near Nowergup Lake at the time the Project was starting. The CSIRO Division of Water Resources contributed significantly by providing salary and other support for Lloyd Townley, Jeffrey Turner, Michael Trefry, Ken Wright, Chris Harris, Colin Johnston, Viv Baker and Peter Alt-Epping at various stages of the project. Overall support for the project was provided roughly in equal thirds by CSIRO, LWRRDC and the partner organisations in Western Australia.

The authors are grateful to many individuals who assisted in various ways throughout the project. In particular, we acknowledge the assistance of:

- Viv Baker (CSIRO) for her skill, attention to detail and patience in preparing text, equations, Figures and Tables for this and other reports throughout this project
- Simon Nield (Mackie Martin and Associates) for his ongoing cooperation and interest in our modelling efforts and for early assistance in identifying and summarising relevant literature
- Jeff Kite, Paul Lavery and Shirley Balla (WAWA), and Bob Humphries, John Sutton and Garry Middle (EPA), for their support in establishing and coordinating the project on behalf of the partner organisations
- Mike Martin (GSWA) for substantial assistance with field work at Nowergup Lake
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- staff of the CSIRO Division of Water Resources Chemistry Laboratory

While all authors of this report contributed to the final result, particular responsibilities were as follows:

- Lloyd Townley, Principal Investigator, responsible for mathematical and computer modelling
- Jeffrey Turner, Principal Investigator, responsible for chemical, isotope and other field measurements and analysis
- Anthony Barr, responsible for two-dimensional modelling in vertical section and in plan
- Michael Trefry, responsible for three-dimensional modelling and modelling of the Pinjar-Nowergup transect
- Ken Wright, responsible for drilling, water level measurements, operation of climate stations, sediment sampling and analysis, and data analysis
- Vit Gailitis, responsible for analysis of isotope concentrations, water sampling and chemical analysis
- Chris Harris, responsible for drilling and water sampling
- Colin Johnston, responsible for relating two- and three-dimensional modelling

Executive Summary

This report summarises the results of a three-year study of the interaction between lakes, wetlands and unconfined aquifers. Apart from a scientific goal of achieving greater understanding of lake-aquifer interaction, the project had three specific practical objectives relating to (i) the identification of capture zones, (ii) management of water levels and (iii) the development of effective parameters for groundwater flow models in plan.

This report is based on a much more detailed report by Townley *et al.* [1993]. The latter contains a full description of the technical aspects of the groundwater modelling carried out during this study. A series of journal papers is also being prepared, and readers with specific interests should contact the senior author of this report for appropriate references.

Identification of Capture and Release Zones

This report focuses on lakes, rather than on sumplands or damplands. We have obtained evidence by modelling and field work to support the conclusion that the majority of lakes on the Swan Coastal Plain act as flow-through lakes which capture groundwater on their upgradient side and discharge lakewater on their downgradient side.

During this project, we have developed two-dimensional models in vertical section, two-dimensional models in plan and three-dimensional models in order to study the shape of capture and release zones as a function of nearby aquifer flows and net groundwater recharge. The depth of a capture zone depends mostly on the length of a lake, in the direction of average groundwater flow, relative to the thickness of the aquifer. The depth of a capture zone also depends on aquifer anisotropy, the resistance of low conductivity bottom sediments, aquifer inflows and outflows, and recharge. The width of a capture zone in plan is roughly twice the width of the lake.

The depth of a release zone is closely related to the depth of a capture zone. Groundwater seepage into and out of a flow-through lake is concentrated near the upgradient and downgradient edges of the lake.

The shape of a lake's release zone can be identified in the field by taking water samples and analysing for isotopic and hydrogeochemical concentrations. We have studied the release zones of Nowergup Lake, Mariginiup Lake, Jandabup Lake and Thomsons Lake using isotopic and hydrogeochemical tracers. In particular, isotopic and hydrogeochemical tracers have shown that outflow from Lake Pinjar (its release zone) becomes inflow to Nowergup Lake (its capture zone), a distance of 5.75 km downgradient.

The most cost-effective way to learn about a lake's release zone and hence its groundwater flow regime is to install a nest of piezometers or a multi-level piezometer at the middle of the downgradient side of a lake. Measurements of piezometric heads upgradient and downgradient of a lake can in principle give information about the geometry of capture and release zones, but are not as conclusive as isotopic and hydrogeochemical data.

The concentration of isotopes and chloride in lakewater and a lake's release zone can assist in the determination of a lake's water balance. Capture zone geometries vary seasonally as lake levels and surface areas fluctuate. Capture zones of lakes on the Swan Coastal Plain can be determined by regional scale modelling, coupled with results of idealised modelling of isolated lakes. Nitrate, phosphate, petroleum products and pesticides can all be carried by groundwater, but some are degraded or retarded, thus reducing their rate of movement through an aquifer.

Management of Water Levels

Water levels in lakes on the Swan Coastal Plain fluctuate seasonally, and some lakes dry out at the end of summer. Lake levels can be effectively maintained by pumping relatively small volumes of groundwater into the lakes for a few months each year. Artificial water level maintenance can lead to an improvement in lake water quality. In order to minimise its impact on lakes, pumping for public or private water supply should be located as far away as possible, both in space and in time.

Average groundwater and lake levels depend on long-term average recharge, whereas seasonal fluctuations depend on the deviations between fluctuating recharge and the long-term average. Long-term fluctuations in groundwater and lake levels depend on long-term fluctuations in recharge. Lake levels can fluctuate either more or less than nearby groundwater levels, depending on whether a lake is driven by surface water inflows or by groundwater inflows.

Effective Parameters for Models in Plan

Groundwater flow patterns near shallow lakes are fundamentally three-dimensional, but we have developed approximate methods for representing lakes in two-dimensional regional models of aquifer flow. We have developed guidelines for assigning large transmissivities to represent circular lakes in a one-layered model of a regional aquifer. We have also developed guidelines for assigning leakage coefficients to represent circular lakes in a two-layered model of a regional aquifer. Field data on the hydraulic conductivities of lake linings confirm that they are often low, but we have not related measured values to effective values needed to represent lakes in two-layered plan models.

Other Findings

We have reviewed the development of isotope balance equations for evaporating water bodies, and summarised previous literature in a concise unified framework. We have summarised the correct way of determining the angle between equipotentials and directions of flow in a vertically exaggerated cross-section through an anisotropic medium. We have developed the theory for a fully coupled groundwater and surface water model, which solves simultaneously for groundwater and surface water levels in a vertical section or in three dimensions. In the process of reviewing simple methods for predicting the movement of phosphate fronts, we have discovered inconsistencies and developed a new method for predicting travel distance. Finally, we have developed a method for combining measurements of all the components of a lake water balance to obtain estimates of the same components which are constrained to satisfy an exact water balance.

Management Implications

Lakes and wetlands are intimately coupled to the unconfined aquifer of the Swan Coastal Plain. The water quality of a lake depends on the quality of groundwater and surface water entering the lake, as well as on chemical and biological processes taking place within the lake. The concept of a "groundwater capture zone" for a lake has implications for management, in that it defines the shape of a region at the land surface within which any recharge will ultimately pass through the lake. The capture zone defines perhaps the largest "buffer zone" that could be required to be protected, in order to protect the quality of a surface water body in perpetuity.

There are many activities that may not be desirable within the capture zone of a lake, or at least within some short distance of the lake. These activities depend on the risk of contamination of recharge to groundwater, and on the nature of the particular contaminants. Although groundwater on the Swan Coastal Plain flows generally towards rivers or the ocean, capture zones are not restricted to the upgradient side of the lakes. During the winter season when recharge occurs, groundwater on the downgradient side of a lake may flow in a reverse direction, counter to the average regional flow. Capture zones may therefore extend some distance downgradient of lakes.

Although water level fluctuations are natural and may even be desirable, water levels can be effectively maintained by pumping groundwater into a lake for as little as a few months per year. Artificial maintenance can in principle maintain lake levels well above regional groundwater levels. Depending on the source of the groundwater, the water quality of a lake can even be improved. The source should preferably be as far away from the lake as possible, preferably from below the unconfined aquifer.

One finding of this study is that groundwater flow regimes near lakes can be either dominated by surface water or by groundwater. Natural lakes may have been groundwater dominated, but networks of drains may have changed the balance of some lakes so that they are now surface water dominated. The role of drains in modifying and controlling the behaviour of lakes and wetlands is not fully understood and needs careful investigation.

There appears to be some tendency at present for the responsibility for wetlands management to be passed on to local councils and Shires. While implementation of management plans for controlling access, vegetation and fauna may be handled effectively at this level of government, it is not possible for individual councils and Shires to manage the water balance of lakes in isolation from others. All lakes and wetlands on the Swan Coastal Plain are interconnected by a regional scale aquifer. Lake and wetlands will not continue to exist if the water balance of the unconfined aquifer of the Swan Coastal Plain is disrupted to the extent that they are no longer "wet". In our view, the water balance of the Swan Coastal Plain and maintenance of the condition of lakes and wetlands can best be managed at a regional level, by a suitably skilled interdisciplinary team of experts.

1. Introduction

1.1 Preamble

A sequence of dry years in the late 1970's resulted in declining groundwater levels in the Perth region, restrictions on water use and an increased awareness of lakes and wetlands for their contribution to the aesthetic and environmental quality of the Perth metropolitan area. The Perth Urban Water Balance Study [Cargeeg *et al.*, 1987*a,b*] was initiated by the Metropolitan Water Authority in 1982 and completed by the Water Authority of Western Australia (WAWA) in 1987. One of the primary outcomes of that Study was the development of a regional scale water balance model for a major portion of the Swan Coastal Plain, a region containing dozens of lakes and hundreds of wetlands. A less well-known outcome was that it incorporated, for the first time in a study of its kind, two significant studies of the region's wetlands. Both were initiated by the Department of Conservation of Environment and completed under the name of the Environmental Protection Authority (EPA).

In late 1987 and early 1988, a number of groups of researchers in Western Australia independently approached the Water Authority and the EPA seeking support for applications under the Partnership Research Program of the Australian Water Research Advisory Council (AWRAC). The research described here was one of five major studies initiated in 1989 with joint support from these three organisations. The other four studies focused on water quality (particularly the use of macro invertebrates as indicators of the health of wetlands), aquatic fauna, water birds and fringing vegetation. The outcomes of all five are summarised in other volumes in this series of reports.

The Water Authority of Western Australia is responsible for managing the State's water resources. In some areas on the Swan Coastal Plain, the Water Authority pumps groundwater to provide drinking water supplies for Perth. Where legislation allows, it also allocates groundwater to private and public users. The review process whereby new groundwater pumping schemes are approved has resulted in the definition by the EPA of desirable minimum and absolute minimum water levels in a number of lakes. One of the major issues relating to lakes and wetlands is therefore that of water level management.

Another major issue is that of lake water quality and the role of "buffer zones" in protecting water quality. Both the EPA and the Water Authority are involved in the process through which permission is given for specific land use activities at particular locations. The EPA has been particularly keen to find a scientific basis for statements concerning the implications of particular land use activities upgradient of lakes and wetlands.

A third issue of interest has been the scientific basis for the methods already being used by the Water Authority to incorporate lakes and wetlands in computer simulation models of regional groundwater flow. Although the regional scale model developed during the Perth Urban Water Balance Study did not include individual lakes, other models at more local scales have done so since. The methods used have been intuitively correct but not documented in the literature, thus the issue in this case was to validate and improve the methods in use.

To summarise, there were three major areas of interest to the sponsors of this project at the time it started:

- management of lake water levels,
- buffer zones for wetland protection, and
- development of scientifically-based computer modelling methods for lakes.

All of these are reflected in the project's objectives described below.

1.2 Regional Setting

Before presenting the project's formal objectives, it is worthwhile reviewing the physical setting of the lakes and wetlands of the Swan Coastal Plain.

Figure 1.2.1 shows the Perth region and several major land use categories in the region. Major wetlands are shown in black. Figure 1.2.2 shows regional groundwater levels and in particular the Gnangara and Jandakot Mounds. With the exception of the Gwelup borefield which is located within the urban area, Water Authority borefields are generally associated with pine plantations or uncleared land under native vegetation upgradient of major wetlands. Figures 1.2.3 and 1.2.4 clearly show the association of many wetlands with boundaries between geologic units [see also Allen, 1981]. The locations of the three sections in Figure 1.2.4 are shown in Figure 1.2.3. Figure 1.2.5 provides the names of many of the wetlands on the Swan Coastal Plain, and is consistent with those approved by the Department of Land Administration (DOLA) Nomenclature Committee.

Much of our knowledge of hydrogeology of the Swan Coastal Plain is due to the efforts of the Geological Survey of Western Australia (GSWA). Numerous contributions by GSWA staff are referenced throughout this report.

1.3 Lakes, Wetlands and Aquifers

In general, throughout this report, we use the words "lakes" and "wetlands" to mean shallow surface water bodies which are either permanently or seasonally wet. This definition is less precise than the definitions of "lakes", "sumplands" and "damplands" provided by Semeniuk [1987], but is sufficient to allow systematic study of groundwater flow near water bodies on the Swan Coastal Plain.

This report assumes that the reader has some understanding about how regional aquifers behave, but in order to set the scene, a brief introduction is presented here. Aquifers are layers of geological material (such as sands, silts, clays, and in many cases, fractured rock) which are capable of transmitting significant quantities of water. Significant in this context means that the rate of flow is large enough to be non-negligible relative to other flows in the hydrological system. Aquifers transmit water downgradient from source areas towards sinks. For the Swan Coastal Plain, the sinks are the Indian Ocean, and the Swan and Canning Rivers. The source area is the whole of the Swan Coastal Plain, because recharge occurs virtually everywhere. The source areas furthest from the sinks are at the tops of regional groundwater mounds, known as the Gnangara Mound in the north and the Jandakot Mound in the south (Figure 1.2.2).

The uppermost aquifer on the Swan Coastal Plain is an unconfined aquifer, which has as its upper boundary a "water table". The water table is the top of the saturated zone, and the elevation of the water table provides the driving force for flow towards the regional sinks. The shape of the regional water table explains the use of the term "mound" in the explanation above. In recharge areas, groundwater flow has a slightly downward component of flow, from the water table towards the the bottom of the aquifer, or even towards deeper aquifers. Near discharge areas, there is a slight upward component of flow. Groundwater is always flowing "downhill", but the "hill" is defined by the variation of a quantity known as "piezometric head". Piezometric head is a combination of elevation and pressure. At the water table, piezometric head is equal to water table elevation. In recharge and discharge areas, piezometric head decreases and increases with depth, respectively. Because aquifers are typically shallow compared to their horizontal extent, and because gradients in water table elevation are typically small, groundwater flow in aquifers can be visualised as being essentially horizontal.

This report focuses on the special role of lakes in a regional aquifer system. Shallow lakes occur on the Swan Coastal Plain where the regional groundwater level intersects the undulating land surface. Within each lake, the water surface is horizontal, thus the piezometric head at the bed of the lake is everywhere equal to the elevation of the lake surface. This creates a region beneath each lake where there is effectively no horizontal gradient, and where the groundwater flow tends to stagnate. At the same time, a water body itself provides less resistance to flow than an aquifer, so groundwater tends to rise towards a lake on the upgradient side, travel through the lake and then discharge to the aquifer at the downgradient side. The water body acts as a conduit, or a short circuit in the lake-aquifer system. It causes flow to deviate from being essentially horizontal, i.e. it induces significant upward and downward components of flow. It is this fact which makes lakes particularly important in the context of a regional flow system. Lakes interrupt the essentially horizontal movement of groundwater by diverting flow through the water bodies themselves.

1.4 Objectives

The objectives of this project, as defined in our original proposal to AWRAC, were as follows:

(i) to understand fully the hydrological behaviour of shallow lakes in steady and transient flow situations by a careful sequence of numerical experiments in two and three dimensions. This will result in the definition of an upstream "capture zone", in which all groundwater flow and any groundwater pollution will eventually pass through the body of the lake; it will also define the zone downstream of a lake in which water quality is influenced by a shallow lake;

(ii) to validate predictions of lake-aquifer interaction using physical, hydrogeological, chemical and stable isotopic measurements in the field;

(iii) to calibrate two-dimensional plan models of aquifer flow against three-dimensional local models of shallow lakes, in order to determine effective transmissivities in the vicinity of lakes for use in plan models. This will allow the use of simpler two-dimensional models in further studies of lake-aquifer management strategies; and

(iv) to investigate and make recommendations on management issues such as the rates of solute and nutrient transport into shallow lakes, possible strategies to reduce the impact of pollution upgradient of a wetland, pumping strategies in the vicinity of wetlands to reduce the effects on lake water levels and strategies for artificial maintenance of lake levels.

All of these objectives have been addressed during the course of this project. But more progress has been made on some than on others. The first three objectives have been addressed very thoroughly, although many questions have been raised, and there is scope for further research. Significant progress has been made on the fourth objective, but more work is needed on pumping strategies for artificially maintaining lake water levels.

1.5 Structure of the Report

The structure of this report is intended to allow water resources managers to quickly see the current state of our knowledge about how shallow lakes and wetlands interact with unconfined aquifers.

Chapter 2 provides a brief review of literature on lake-aquifer interaction and also, to a lesser extent, on lake hydrology and hydrogeochemistry.

Chapters 3, 4 and 5 focus on the three main questions which led to this study. Chapter 6 presents other findings, and Chapter 7 identifies topics that need further research.

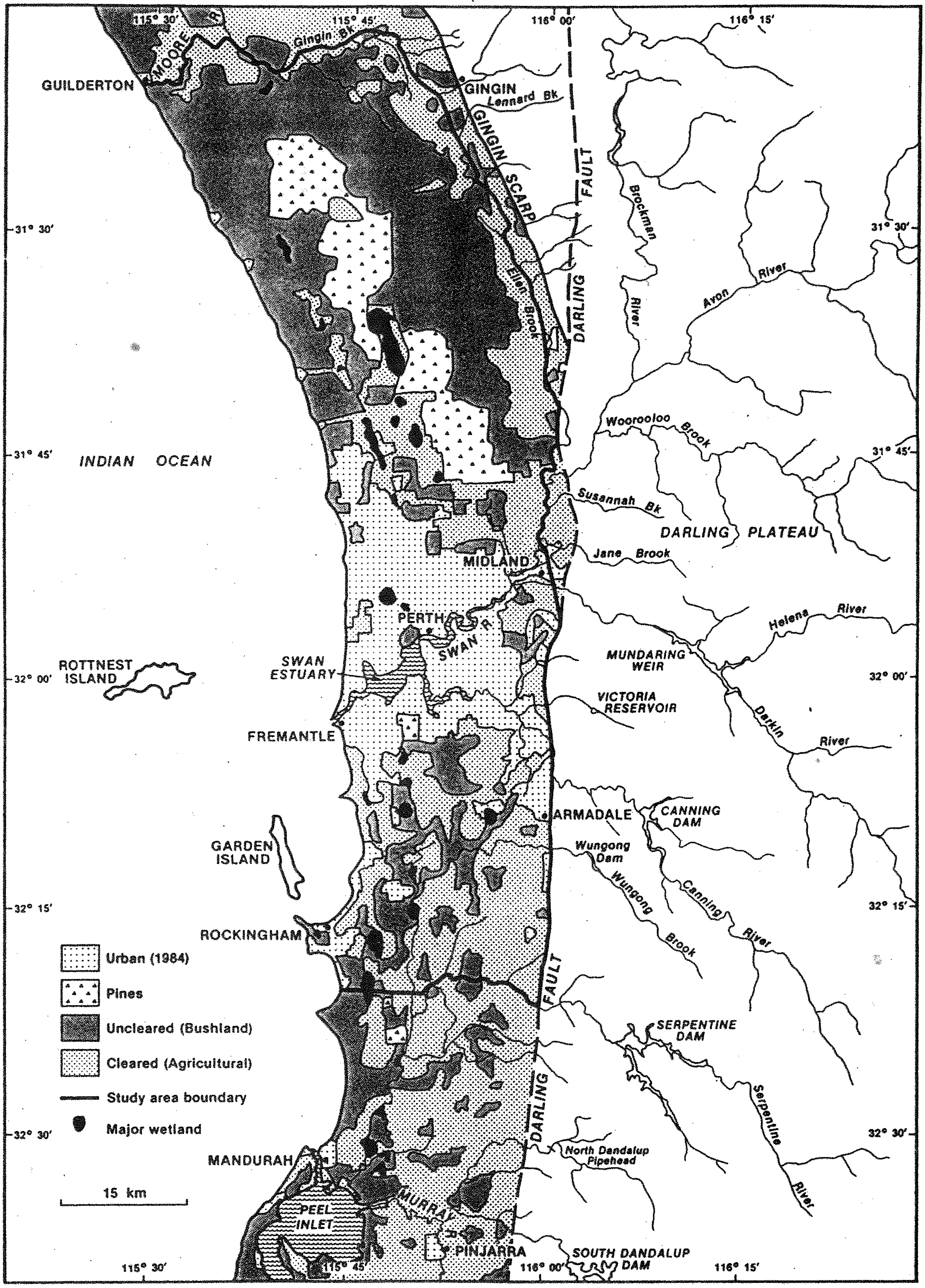


Figure 1.2.1 Map of the Swan Coastal Plain, showing wetlands in relation to major land use categories [from Smith and Allen, 1987]

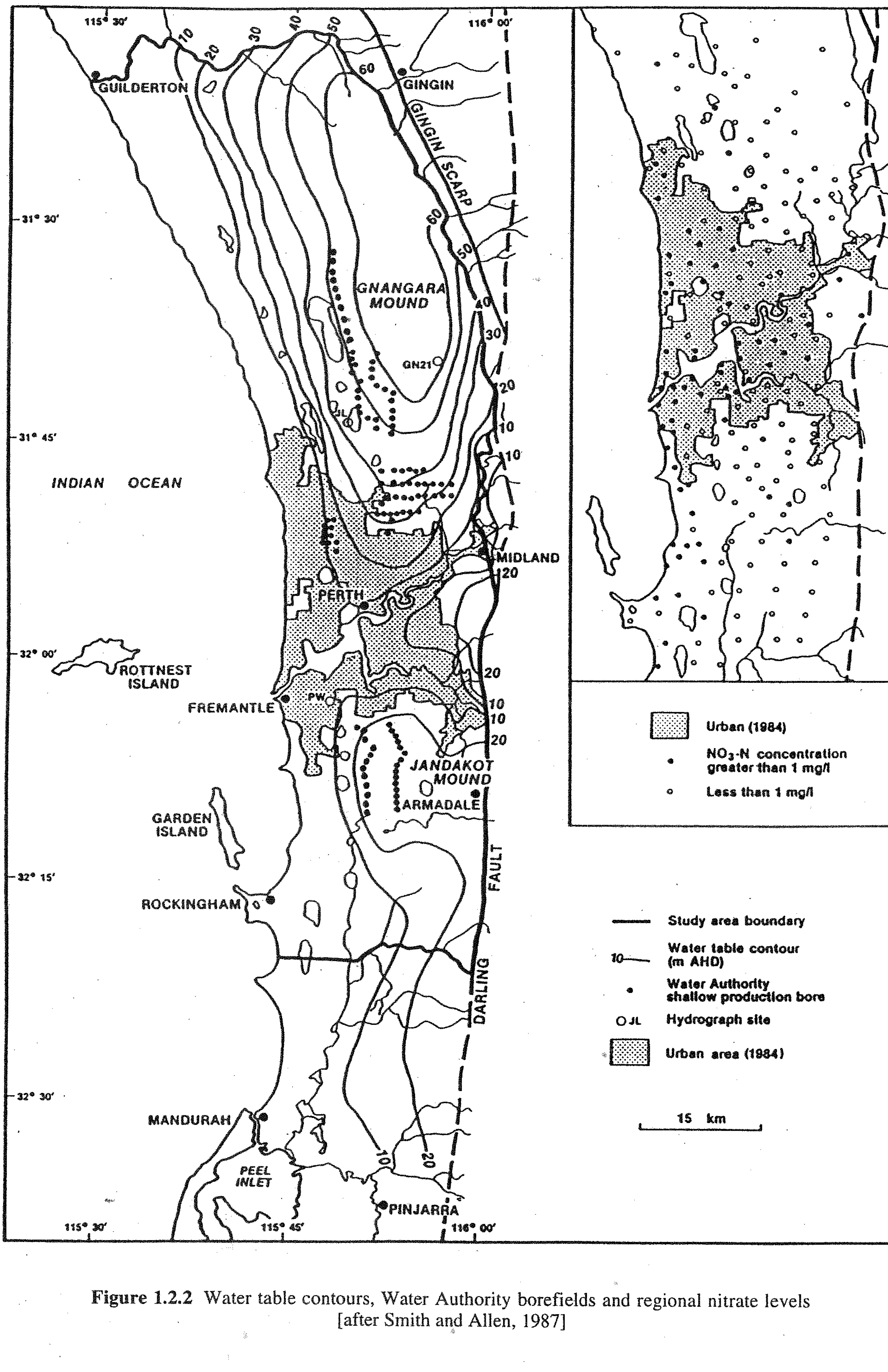


Figure 1.2.2 Water table contours, Water Authority borefields and regional nitrate levels [after Smith and Allen, 1987]

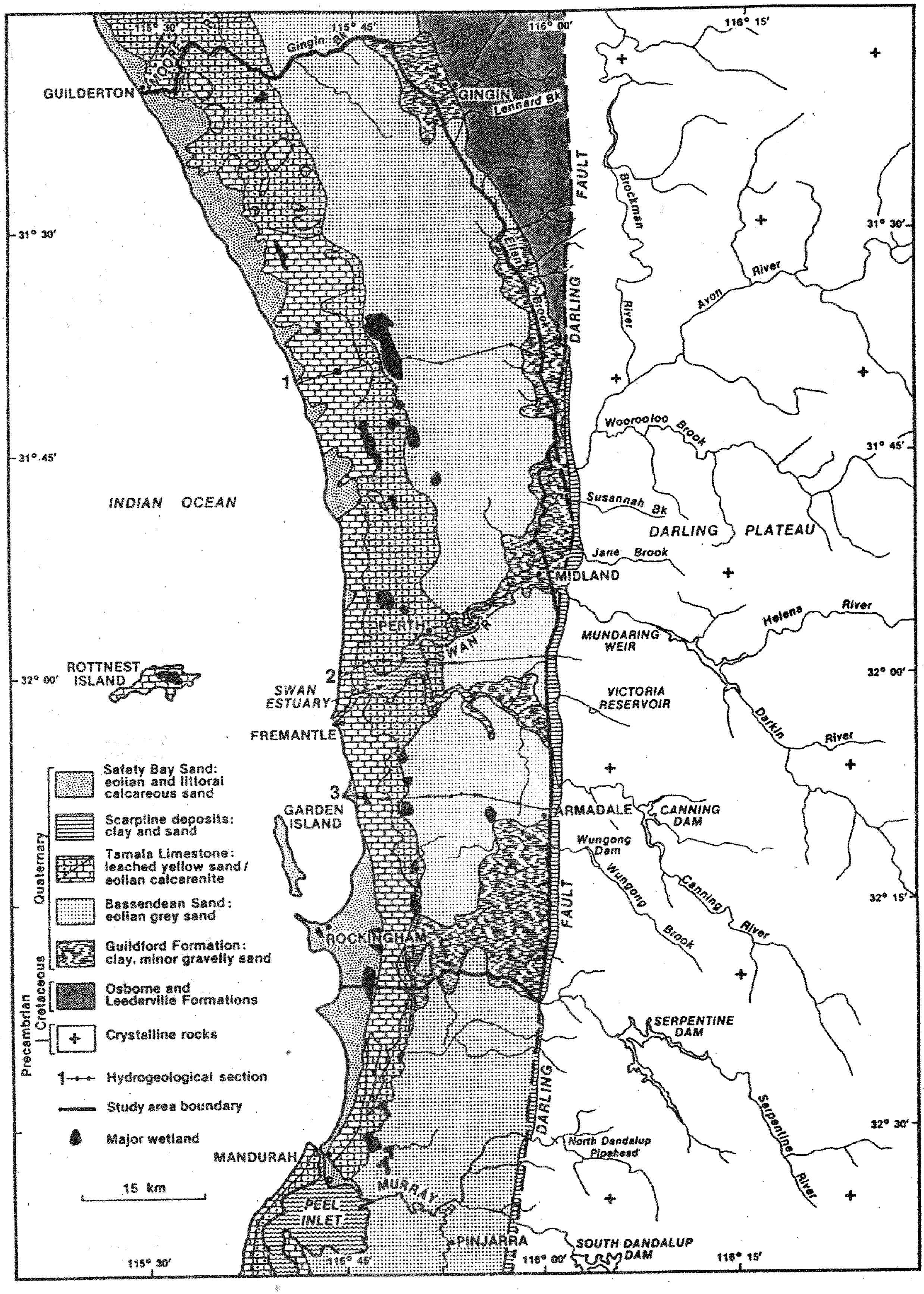


Figure 1.2.3 Wetlands in relation to generalised geology [from Smith and Allen, 1987]

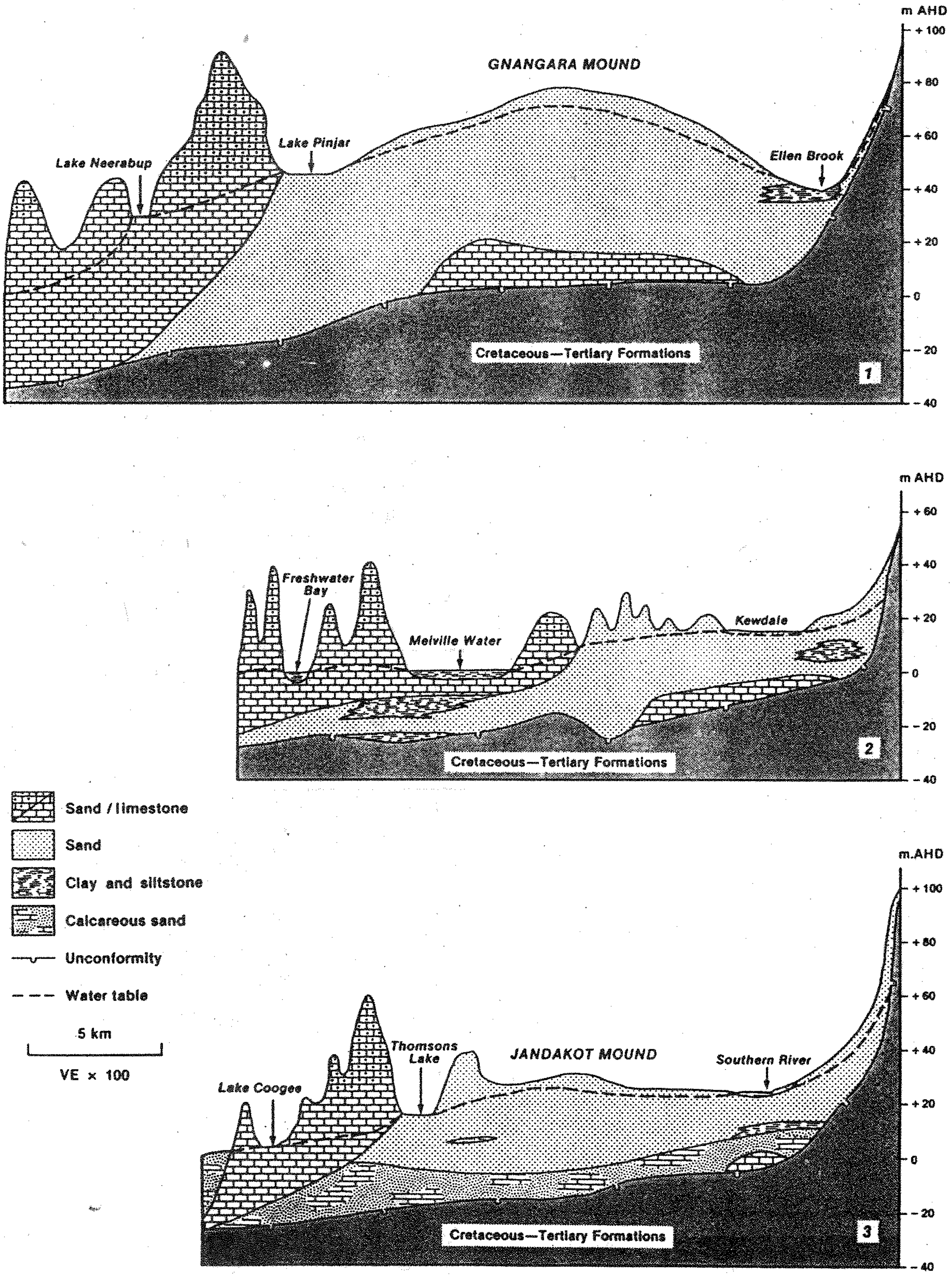


Figure 1.2.4 Generalised geological sections [from Smith and Allen, 1987]

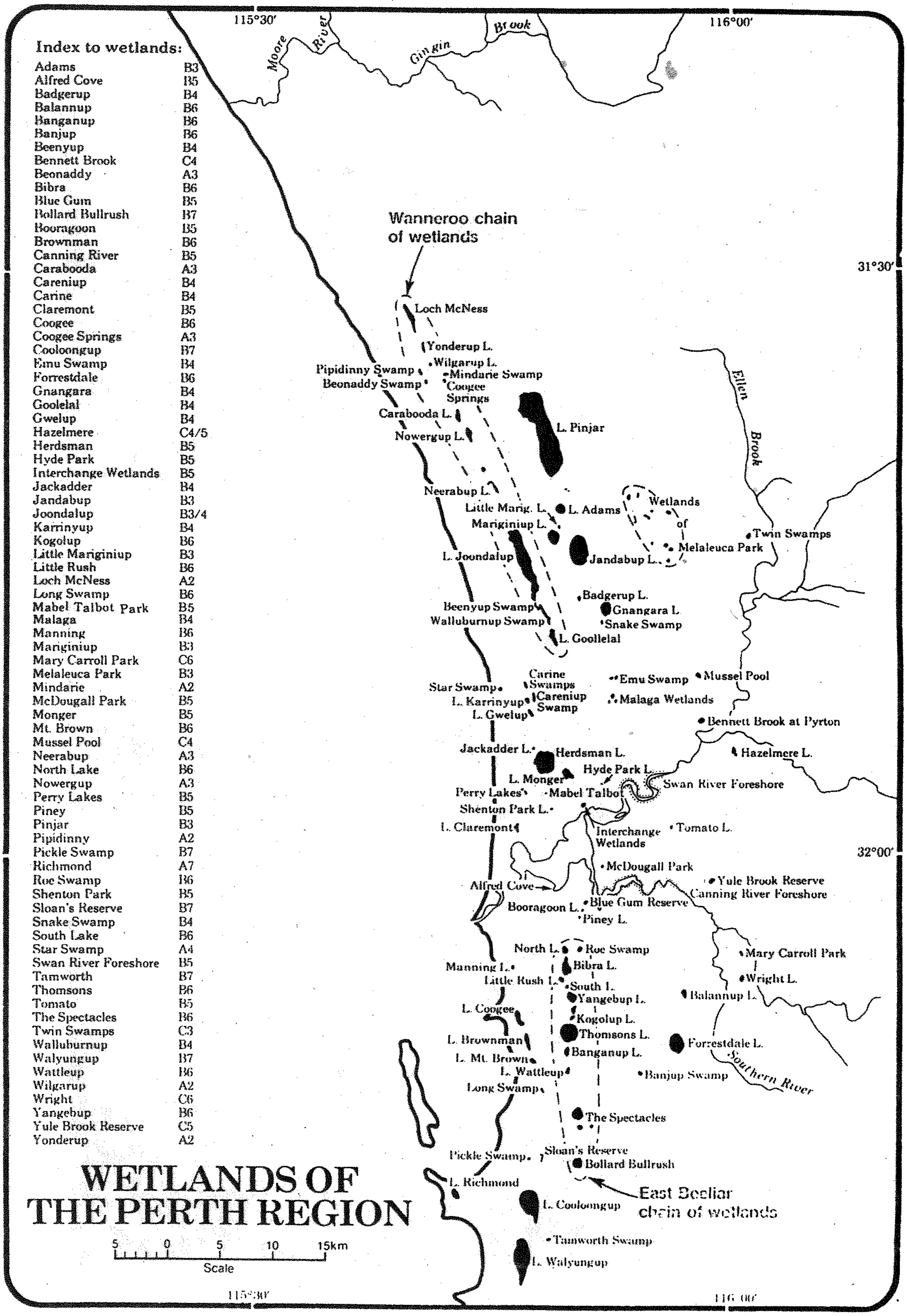


Figure 1.2.5 Wetlands of the Perth region [from Cargeeg *et al.*, 1987*a*]

2. Literature Review

2.1 Classification Systems for Lakes and Wetlands

It has long been recognised that both lakes and wetlands of the Swan Coastal Plain are affected by and affect the behaviour of the regional unconfined aquifer. Allen [1981, pp.38-39] defined six classes of lakes on the coastal plain, on the basis of their topographic location. In inferred order of decreasing age, they are: Bambun-type, Gnangara-type, Forrestdale-type, Joondalup-type, Gwelup-type and Cooloongup-type. The lakes developed in the coastal belt (Joondalup and Cooloongup types) are typically elongated and parallel to the coast, whereas the other lakes are nearly circular in plan. Classifications based on a wider range of criteria include those by Semeniuk [1987, 1989] and Arnold [1990].

In the international literature, there have been numerous attempts to develop meaningful classification systems based at least partly on hydrological setting. Stephenson [1971] discussed the desirability of developing a classification system based on the position of lakes in a groundwater flow system and interchange of water between groundwater and surface water bodies. His work led to a detailed inventory of 63 lakes in twelve states and provinces of North America, and eventually to a classification system which was based on groundwater flow patterns [Born *et al.*, 1979].

Born *et al.* suggested a primary classification based on a distinction between "recharge", "discharge" and "flow-through" lakes. A lake is classified as a recharge lake if lakewater recharges the aquifer over the entire lakebed, as a discharge lake if the aquifer discharges groundwater into the lake over the entire lakebed, or as a flow-through lake if water moves into the lake and out of the lake in different areas of the lakebed. This terminology is of critical importance to this project, because our own classification of flow patterns [Nield, 1990; Nield *et al.*, 1994] uses the same primary groupings. Gat [1979] provided a sketchy outline of a similar classification scheme, aimed at relating observations of isotopic abundances and lake salinity to groundwater flow regimes.

Another attempt to incorporate hydrological characteristics into a classification system was that by Winter [1977]. Winter carried out a principal component analysis on data from 150 lakes, but his results do not appear to have been adopted by researchers or managers. Geldner and Kaleris [1980] systematically analysed flow in a cross-section without surface recharge or discharge. The style of their approach was similar to that adopted in this project, but our own work, by including recharge, has added a significant level of complexity.

2.2 Studies of Lakes and Wetlands on the Swan Coastal Plain

The first detailed hydrogeological study of an individual lake on the Swan Coastal Plain was that of Jandabup Lake by Allen [1980]. Allen demonstrated that the lake received groundwater on its upgradient side and discharged water of increased salinity on its downgradient side, i.e. that it was a flow-through lake in the terminology of Born *et al.* [1979]. Because flow directions were inferred on the basis of equipotentials displayed on vertically exaggerated cross-sections, Allen concluded that only a relatively shallow layer within the unconfined aquifer was interacting with the lake. An observation that Allen's interpretation of flow directions was not correct [Don McFarlane, pers. comm.] led directly to investigations by Oo [1985], Townley and Davidson [1988] and eventually to the proposal for this project.

Detailed hydrological studies of other lakes on the Swan Coastal Plain include those of Lake Mariginiup [Hall, 1983], Bibra Lake [Davidson, 1983], Lake Joondalup [Congdon, 1985], North Lake [Bayley *et al.*, 1989] and the lakes within Yalgorup National Park [Commander, 1988]. McFarlane [1984] studied plumes downgradient of Mason Gardens and Shenton Park Lake. Studies that include small components of hydrology include those of Jandabup Lake [Department of Conservation and Environment, 1984] and Star Swamp [Loneragan *et al.*, 1984]. More significant reviews (preceding the studies concurrent with this project) which relate several different lakes include those by Davis and Rolls [1987], Carbon *et al.* [1988] and Burke and Knott [1989]. Moore and Turner [1989] documented the first use of stable isotopic data in interpreting lake-groundwater interaction on the Swan Coastal Plain.

Numerous other lakes have been studied elsewhere in Australia. The study of flow systems near salt lakes is of great interest because of the large number of such lakes in Australia (see Teller *et al.* [1982], Macumber [1983] and Prendergast [1989]). But because of the significant evaporative loss in salt lakes, relative to regional flows and recharge, such flow systems are unlikely to occur on the Swan Coastal Plain. Stokes and Sheridan [1985] described studies of Lake Toolibin in the wheat belt of Western Australia. The State Rivers and Water Supply Commission of Victoria [1982] described a study of Lake Charm. The flow systems beneath Lake Boemingen on Fraser Island off the central coast of Queensland have also been studied [M. Riesser-Steffens, pers. comm.]. There are a great many lakes in various hydrologic settings in Australia, and many could benefit from interpretation on the basis of hydrology, as presented in this report.

2.3 Modelling of Groundwater Flow Patterns near Surface Water Bodies

As a broad generalisation, previous modelling of the interaction between lakes and aquifers has focused either on particular lakes, with emphasis on the water balance of those lakes, or on a class of similar lakes, in which case numerical experiments have been performed to understand the features of a particular type of flow system or alternatively to evaluate particular modelling techiques. Existing results are not easily transferrable to previously unstudied lakes; their usefulness lies in the methods described rather than the general applicability of their results.

Following classical studies of regional flows by Tóth [1963] and Freeze [1969], the work by Winter [1976] was (prior to this project) the most complete analysis of regional groundwater flow systems beneath lakes. Winter's modelling demonstrates the occurrence of local and regional flow systems, but focuses on lakes which would be classified by Born *et al.* [1979] as discharge lakes. His attention at this time was focused on lakes typical of the central northern United States, and many of lakes in this region are discharge lakes. A criticism of this work, and indeed of the earlier work by Tóth and Freeze, is that by assuming the geometry of the upper boundary of the flow domain, i.e. the shape of the water table, there is also an implicit assumption about the spatial distribution of recharge or discharge at the water table – an assumption which is rarely checked for consistency with other data.

Numerous papers by Winter [1978, 1981*a*, 1983] are applicable to discharge lakes, located between groundwater mounds, or to lakes described by Born *et al.* [1979] as shallow discharge - deep recharge lakes. In a sequence of hypothetical modelling studies, results are presented for flow in a two-dimensional cross-section [Winter, 1981*a*; Pfannkuch and Winter, 1984; Winter and Pfannkuch, 1984], for three-dimensional saturated flow [Winter, 1978] and for transient two-dimensional variably-saturated flow, in which unsaturated flow above the water table is taken into account [Winter, 1983; Anderson and Munter, 1984; Winter, 1984]. An interesting feature of this class of flow system is the presence of a stagnation point, at some depth below the lake bed, whenever a lake behaves as a discharge lake. Stagnation points play a very important role in all of our analysis. Winter's purpose in including unsaturated flow was to demonstrate the sensitivity of results to spatially varying percolation to the water table. From our point of view, this reinforces our belief that assuming the location of the water table in cross-sectional models can be misleading.

McBride and Pfannkuch [1975] and Lee *et al.* [1980] presented two-dimensional results for a region near the shoreline of a large lake. Both approaches included the effects of anisotropy and focused on the spatial distribution of seepage. Anderson and Munter [1981] presented transient two-dimensional modelling results, both in cross-section and in plan, in order to show that seasonal reversals in the direction of flow can occur near flow-through lakes. They argued that the combination of two-dimensional models allows an adequate understanding of the three-dimensional flow system and similar arguments are used in this report. Their review of previous research is useful and concise, however their simulations of a particular lake in Wisconsin can not be immediately generalised to flow-through lakes of different physical dimensions.

Winter [1986] studied a system of interdunal lakes which are not unlike the wetlands near Perth, except that they are relatively more closely spaced and therefore have a higher likelihood of affecting each other's flow regimes. While we support his emphasis on temporal and spatial variations in seeking accurate simulations of wetland systems, we also recognise the need of environmental scientists for simple indicators of wetland behaviour. For this reason, we focus in this report on a class of lakes which is considerably simpler than those studied earlier by Winter and others.

Apart from the above studies which are based mainly on the use of numerical models, there are numerous studies of lakes and other surface water bodies using analytical or quasi-analytical methods. Classic texts by Muskat [1946], Polubarinova-Kochina [1962] and Aravin and Numerov [1965] all include solutions that can be applied to shallow lakes. Geldner and Kaleris [1980] and Kaleris [1986] utilise an important solution by Aravin and Numerov [1965, pp. 168-172] which allows the identification of several fundamentally different flow regimes, in a vertical section in the absence of recharge. Numerous authors, especially in international conferences, continue to provide specialised analytical solutions which are of limited value: the paper by Khublarian and Putyrskiy [1988] is an example.

Zheng *et al.* [1988] propose a model intended to apply to interceptor ditches, or to relatively small lakes, and Chambers and Bahr [1992] have attempted to apply this model. From our point of view, the approach of Zheng *et al.* has serious limitations, related to the assumption of a particular shape for the water table. A quasi-analytical approach developed by Townley and Davidson [1988] applies both in vertical section and in plan, and could easily be applied to the data utilised by Chambers and Bahr.

Several recent studies by Cherkauer and Nader [1989], Cherkauer and Zager [1989] and Cherkauer and McKereghan [1991] address specific issues such as the effects heterogeneities below the lakebed, and the effects of a tortuous shoreline on rates of seepage. Heterogeneities had previously been studied by Krabbenhoft and Anderson [1986]. Brainard and Gelhar [1991] report solutions for three-dimensional flow towards a sloping river or stream, an important contribution to a very small number of three-dimensional studies reported in the literature. Papers by Lee *et al.* [1980] and Krabbenhoft *et al.* [1990*a*, *b*] are significant because they recognise the need for solving the advection-dispersion equation for solute transport in interpreting the motion of tracers.

Major advances in the identification of flow patterns near surface water bodies have been reported by Nield [1990] and Nield *et al.* [1994]. Readable summaries of this work include those by Townley *et al.* [1988], Townley *et al.* [1991] and Townley and Turner [1990; 1992]. This report summarises and extends these results.

2.4 Distribution of Bottom Seepage

Extremely large lakes embedded in regional flow systems are characterised by the fact that no underflow passes beneath the centre of the lake. It is well known that inflow to a large lake is largest at the shoreline (in fact, seepage is often observed to cause rivulets in beach sands) and decreases approximately exponentially with distance offshore (see comments by Born *et al.* [1979, p.35], Lee *et al.* [1980, pp.57,60] and Winter [1981*b*, p.106]). This result is verified by the experimental data of John and Lock [1977] and Lee *et al.* [1980], and also by two-dimensional cross-sectional numerical experiments by McBride and Pfannkuch [1975] and Lee *et al.* [1980]. The assumption in these numerical experiments of a no-flow boundary beneath the centre of a lake, presumably on the basis of symmetry, is shown in our research to apply for large flow-through lakes as well.

McBride and Pfannkuch [1975] found that the majority of seepage occurs close to the shore. Although their model assumed a perfect hydraulic connection between lake and aquifer, with no spatial variation in the resistance of the lakebed, seepage decreased approximately exponentially with distance off-shore in the majority of cases studied. Their results were supported by seepage measurements from Lake Sallie in West-Central Minnesota, where seepage was found to decrease by approximately an order of magnitude for every 60m moved off-shore. But numerous field and modelling studies have since demonstrated that heterogeneities in the aquifer and lake sediments can cause large variations from the expected exponential decay-type of seepage distribution [Munter and Anderson, 1981; Krabbenhoft and Anderson, 1986; Bruckner *et al.*, 1989; Cornett *et al.*, 1989; Cherkauer and Nader, 1989].

Lee *et al.* [1980] observed an exponential decay in seepage with distance from the lake shore, while investigating movement of a tracer through the bed of Perch Lake, Ontario. Numerical modelling of a simplified system revealed that high values for aquifer anisotropy caused more evenly distributed seepage on the lakebed.

Using an electrical analogue method, Pfannkuch and Winter [1984] simulated seepage through lakebeds in isotropic systems, and found that seepage tended to be more evenly distributed on the beds of lakes with small ratios of lake length to aquifer thickness. They noted that since this ratio decreases when anisotropic systems are converted to their equivalent isotropic counterparts, lakes embedded in highly anisotropic aquifers should also tend to have relatively uniform seepage through their beds.

In a companion study, Winter and Pfannkuch [1984] used a finite difference model to examine flow patterns near lakes for various values of anisotropy and aquifer thickness. For a given water table configuration, they observed a tendency for flow-through rather than discharge behaviour in settings with a low ratio of lake width to aquifer depth, or high anisotropy. These observations appear to contradict those made in the first part of the study [Pfannkuch and Winter, 1984], in which seepage was found to be more evenly distributed (with therefore less possibility of flow-through) in these settings. One reason for this is that the distribution of resistivity of lake sediments plays a dominant role in controlling seepage through the lakebed in the second study. Another is that in the second study the same steady-state water table configuration was imposed as a model boundary condition with different values of anisotropy, thereby applying different forcing to the lake-aquifer system.

Nield [1990] and Townley and Nield [1994] addressed many of these issues by examining steady-state lakebed seepage distributions in the analytical solution of Aravin and Numerov [1965] and the numerical model of surface water - groundwater interaction described by Nield *et al.* [1994]. Their approach is followed in this report.

2.5 Lake Water Balance Studies

The water balance of a lake can be described by a water balance equation which states that the rate of change of stored volume within the water body is equal to some of inflows minus outflows. In a discrete form, the change in lake level during any time interval is equal to the sum of inflow volumes minus the sum of outflow volumes during that time interval, all divided by the surface area of the lake.

Inflows to a lake include direct rainfall onto the lake surface, surface inflow from a nearby surface catchment or surface capture zone, and groundwater inflow from a groundwater capture zone. Outflows include evaporation from the surface, surface outflows to rivers, streams or drains, and groundwater outflow to a groundwater release zone.

All components of the water balance are extremely difficult to measure. For this reason, unknown components are often measured by difference, i.e. by estimating all components except one, and then by calculating the remaining component from the water balance equation. Scientists studying evaporation have on occasion estimated rainfall, surface flows and net groundwater inflow and calculated evaporation by difference. Scientists studying groundwater flows have estimated evaporation and calculated the net groundwater inflow. Such circular arguments are prone to errors [Winter, 1981*b*], and do little to enhance our understanding of lake water balances.

Regional scale models, such as the Perth Urban Water Balance Model [Cargeeg *et al.*, 1987*a,b*], have the potential to properly combine all components of the regional water balance, as long as grids are sufficiently fine near lakes and wetlands to adequately describe dynamic changes in the area of open water. Recent research has emphasised the importance of measuring evaporation and coupling estimates of components of the water balance to other balances such as a thermal balance, a chloride balance and a balance of stable isotopic abundances.

2.6 The Role of Bottom Sediments

Bottom sediments affect both the physical interaction between a lake and the underlying groundwater flow system and the chemistry of the lake waters. The physical effect of bottom sediments is to add resistance along a flow path between the regional groundwater flow system and the body of the lake, thus tending to reduce the degree of inter-connection. The chemical effect is to provide surface area for sorption of phosphate and metal ions, thus acting to reduce their concentration in the water body, at least until the sorption capacity of the sediments is exceeded.

Geldner [1981] identified bottom resistance as being of particular importance in surface water – groundwater interaction, and developed a "bulk clogging parameter" to represent the net effect on the flow of lakebed conductivity variations, for several river reaches in Europe. Values thus obtained showed a large variability in time, which Geldner suggested may be due to seasonally varying biological processes in the lake sediments. Geldner's representation of bottom sediment resistance was adopted by Nield [1990] and Nield *et al.* [1994] and plays an important role in the treatment of bottom resistance during this project.

Other authors investigating stream-aquifer interaction [e.g. Rushton and Tomlinson, 1979; Mishra and Seth, 1988] have suggested that nonlinear leakage coefficients should be used to model the dependence of seepage from rivers on the difference between aquifer and river heads. Under transient conditions this coefficient depends not just on the river stage, but also on the direction of change of river stage, due to the effects of bank storage. These considerations are of less importance in lake-aquifer interaction, as lake levels vary at a much slower rate than river levels.

Later, Barwell and Lee [1981] developed a method for estimating aquifer anisotropy from observations of the advection of a tracer into a lake. They pointed out, however, that anisotropy estimates will be affected by the presence of resistive lakebed sediments.

Due to evaporation and other processes in a surface water body, chemical characteristics of surface water and groundwater are usually quite different. This can lead to observable differences in sediment characteristics depending on the direction of flow through the sediments [Moore and Turner, 1989]. In addition, sharp concentration gradients in phosphorus, dissolved oxygen, and other constituents of interstitial water are typically observed close to the sediment-water interface. Models of sediment processes usually represent the movement of these constituents by diffusion only [Straskraba and Gnauck, 1985], but the presence of even a small advective flux can significantly alter sediment concentration gradients [Cornett *et al.*, 1989]. In this way, bottom seepage may have a major influence on biological and chemical processes in the sediment. Knowledge of bottom seepage distributions and their seasonal variation may have profound effects on our understanding of other aspects of limnology.

In a study of other saline lakes close to Lake Clifton in Western Australia, Burke and Knott [1989] identified groundwater inflow as a major influence on lake processes, contributing to winter stratification and affecting the chemical composition of water in the lake and sediment. They suggested that organic carbon production by benthic microbial communities can influence the permeability of lake sediments, reducing outflow from the lake and thus contributing to its evaporative concentration. The build-up of low conductivity sediments in regions of lakewater outflow has also been suggested as a cause of increased water table gradients on the downgradient sides of small ponds in North America [Stross and Spangler, 1980].

2.7 Nutrient Loads to Surface Water Bodies

Numerous experimental studies in the laboratory and field have demonstrated a major role for phosphorus and nitrogen in determining the dynamics of algal populations in lakes and wetlands. Light penetration, trace element concentrations, temperature and silica content are some of the other frequently cited controlling factors. Descriptions of the character and processes controlling eutrophication in temperate and tropical region lakes have been given by Vollenweider [1968], OECD [1982], Thornton [1987] and Ryding and Rast [1989].

The important roles of processes at the sediment-water interface such as sediment dynamics, sediment mass balance, nutrient loadings and the release of nutrients from sediments have been presented in the proceedings of a symposium [Sly, 1982]. Classification schemes designed to assist in eutrophication risk assessment of particular water bodies based on nutrient concentrations, chlorophyll *a* concentration and light penetration characteristics have been given by Vollenweider [1968] and OECD [1982]. The more recently published OECD classification is an "open boundary" scheme that describes the trophic condition of a water body. The open boundary scheme is an attempt to the overcome limitations imposed by using fixed values to delineate the trophic status of open water bodies. In the open boundary classification scheme, a water body is considered to be correctly classified if no more than one of the designated parameters of phosphorus, nitrogen, chlorophyll *a* and Secchi disc light penetration depth deviates from its geometric mean by more than ±2 standard deviations. Table 2.7.1 is a modified version of the OECD classification, after Rast *et al.* [1989]. As pointed out by Rast, the overlap in the range of values in Table 2.7.1 indicates the still somewhat subjective nature of trophic classification schemes.

Because phosphorus and nitrogen are the two nutrients essential for algal growth, it follows that for control of eutrophication in surface water bodies over the long term, strategies should be developed to limit or even eliminate altogether the quantity of these nutrients entering a water body. Identification of which nutrient in the water body is limiting to algal growth, either nitrogen or phosphorus or sometimes both, allows focus on the nutrient input to the water body that should be controlled. This is a fundamental point, essential for the control of eutrophication. The concept of a limiting nutrient has its basis in the photosynthetic reaction [Rast *et al.*, 1989; Stumm and Morgan, 1970]. In its simplest form this reaction may be written:

$$CO_2 + NO_3^- + PO_4^{2-} + H_2O + H^+ \\ = \text{algal protoplasm} + O_2$$

Bioassay of the carbon, nitrogen and phosphorus content of algal protoplasm material shows that these elements occur in the atomic ratio 106C: 16N: 1P. This ratio has become a widely used reference ratio for determining the limiting nutrient for algal growth in water bodies. Since measurements of the concentrations of these elements in water bodies are made in mass units, it is more useful in practice to use the corresponding mass ratio of the biologically available forms of nitrogen and phosphorus. Thus the atomic mass ratio corresponds to a mass ratio of 7.2N:1P [Rast *et al.*, 1989]. Measurements of this ratio in lake water provide a useful operational guide as to which of either nitrogen or phosphorus is likely to be limiting nutrient. As pointed out by Rast *et al.* [1989], some latitude should be accepted in interpretation of the mass ratio. In addressing the question of transport of nutrients into a shallow lake or wetland by groundwater flow, the mass ratio of the nutrient input into a water body will determine which nutrient is limiting to algal growth. Because it is frequently found that nitrate is a more mobile species in groundwater, it is more likely to be transported via groundwater inflow into a lake. In such cases this will result in phosphate being a limiting species for algal growth in the water body.

Davis *et al.* [1993], in a project parallel to this one, considered wetlands of the Swan Coastal Plain to be phosphorus-limited if their minimum annual total nitrogen to total phosphorus ratio (TN:TP) was greater than 17. Conversely, wetlands with TN:TP less than 17 were considered to be potentially nitrogen-limited. Of 55 wetlands examined in this way, about 30% were nitrogen-limited. The high mobility of nitrate in groundwater therefore ensures that transport of groundwater into wetlands is an important issue for wetlands of the Swan Coastal Plain.

Many studies have been concerned with the process of estimating nutrient loads to a water body. The chapter by Loehr *et al.* [1989] provides a useful review of the role of point and diffuse sources of nutrient loads into water bodies and provides a summary of methods available for nutrient load estimation.

It is generally considered that nutrient loading into lakes via groundwater inflow is not important [Loehr *et al.*, 1989], primarily because of low phosphorus concentrations in groundwater. This view has evidently been adopted by several authors in modelling the nutrient mass balance of water bodies [Snodgrass and O'Melia, 1975; Lam *et al.*, 1982; Jørgensen *et al.*, 1982; and Golterman, 1982] where the role of groundwater flow as a component of the phosphate loading to water bodies was not considered. In a case study at Lake Mendota, Wisconsin, Brock *et al.*, [1982] showed that although groundwater seepage was a significant source of water inflow, nutrient concentrations were sufficiently low that they did not form a significant source of nutrient loading. Similar conclusions are frequently found, and riparian zones around lakes are often regarded as sinks for nutrients transported toward lakes by groundwater flow [Reddy and Rao, 1983]. Once the adsorptive or assimilative capacity of such zones is exceeded, then the retained nutrients can be remobilised and transported into the surface water body.

Studies where nutrient transport via groundwater inflow into surface water bodies is found to be significant are quite uncommon. A study reported by Vanek [1991] showed that significant concentrations of phosphorus of up to 9 mgL^{-1} PO_4–P were in the inflowing groundwater to Lake Bysjon in southern Sweden. In this case, the phosphorus was found to be released from sandy sediments of the riparian zone following its accumulation by adsorption on the Al and Fe phases of the sandy, non-calcareous soils. Release of the phosphorus and its transport toward the lake occurred following the onset of anaerobic conditions in the soils and mobilisation of the iron-bound phosphate. Vanek [1991] recognised the importance of delineating the geometric boundaries of the groundwater capture zone of Lake Bysjon in the assessment of the role of groundwater inflow in its eutrophication. However, although a schematic groundwater capture zone was proposed, no quantitative hydrogeological basis was given to account for either the direction of groundwater flow nor the shape in plan or cross-section of the groundwater capture zone for the lake. Enell (1982) noted the development of filamentous algae forming dense mats in the groundwater discharge zones of Lake Bysjon and these were ascribed to the utilisation of the groundwater-derived phosphorus. Kerfoot and Skinner [1981] found a close association between the location of groundwater plumes entering Crystal Lake, Michigan, and the growth of attached macrophytes, in particular *Cladophora sp.* The low concentrations of phosphorus measured in the inflowing groundwater (17 μgL^{-1} total P) were considered to be sufficient to support the growth of algae and plants attached to the sediments but not high enough to influence the surrounding lake water.

Frequently it is observed that the phosphorus concentration in a lake is lower than expected from the measured inputs in surface water runoff. In such cases the low concentrations of phosphorus can be accounted for by its exchange with the lake bottom sediments [Jørgensen *et al.*, 1975]. The process of phosphorus exchange between the lake water and sediments is determined by the phosphorus concentration in the interstitial water, the phophorus concentration in the water phase and the adsorption characterisitcs of the sediment for phosphorus under aerobic and anaerobic conditions. Groundwater flowing through lakebed sediments and carrying some phosphorus load will undergo phosphorus removal by adsorption on the sediments. This process will be separate from the uptake of phosphorus from the water column by sediments.

Experimental studies of the influence of groundwater movement on phosphorus release from sediments have been carried out by Liere and Mur [1981]. They simulated the flow of groundwater through lake sediment cores in the laboratory and showed the rapid movement of phosphorus out of the core materials in comparison to experiments where no simulated groundwater flow was allowed.

2.8 Nutrient and Contaminant Transport on the Swan Coastal Plain

The predominantly light textured, sandy soils of the Swan Coastal Plain and their generally poor nutrient and water retention characteristics have been recognised as important contributing factors to contamination of the shallow, unconfined groundwaters of the region [Whelan *et al.*, 1981; Yeates, 1988; Gerritse and Schofield, 1989; Pionke *et al.*, 1990; and Sharma *et al.*, 1991*a*].

The vulnerability of the shallow groundwater to contamination by the nutrient species, nitrate and phosphate, as well as pesticides is well recognised [Gerritse *et al.*, 1990]. It is clear that where groundwater contamination occurs within the capture zone of a shallow lake or wetland, there is a possibility that transport by groundwater flow will contribute to the nutrient and contaminant load of a lake. Examples of such sources of groundwater contamination on the Swan Coastal Plain are septic tank leachates [Whelan *et al.*, 1981], landfill leachate [Barber *et al.*, 1991], industrial waste discharges [Hirschberg, 1988] and application of fertilisers and pesticides in areas of intensive horticultural practice or pastures [Schofield *et al.*, 1985, Singh, 1989; Sharma *et al.*, 1991*b*]. The proximity of some horticultural areas to shallow lakes and wetlands and their location within the groundwater capture zones of lakes explain the motivation for our study of the potential for transport of nutrients into lakes via groundwater inflow.

Nitrate and phosphate

The dominant form of nitrogen found in groundwaters of the Swan Coastal Plain is nitrate. Reduced forms of nitrogen as ammonium and NO_2^- are invariably found in much smaller concentrations, as is organic nitrogen. Reconnaisance surveys on the levels of nitrate in groundwaters of the Swan Coastal Plain have been reported by Davidson and Jack [1983], Gerritse *et al.* [1988] and Pionke *et al.* [1990]. High nitrate concentrations of between 280 and 840 mgL^{-1} were reported by Pionke *et al.* [1990] in percolate beneath the root zone from two irrigated market gardens.

Denitrification has been identified as being an important process in acting as a sink for nitrate in groundwater. The highly spatially variable nitrate concentrations observed in groundwater are probably due to spatial variations in the redox conditions of groundwater that in turn depend upon the availability of dissolved organic carbon. Sharma *et al.* [1991] carried out an assessment of the transport of nitrate and chloride from a market garden on the Swan Coastal Plain. This report was the first to attempt calculation of nitrate travel times in soils and groundwaters in relation to market gardens on the Swan Coastal Plain. The authors concluded that nitrate and chloride would move at the same velocity as groundwater. Because neither nitrate nor phosphate was detected in a transect of monitoring bores on the groundwater downgradient side of the market garden, they further concluded that there was negligible export of water and dissolved solutes from the market garden. Modelling of nitrate transport in the unsaturated zone led to the conclusion that nitrate transport rates were rapid with average travel times for water and nitrate of the order of 0.05 md^{-1}.

The important conclusions to emerge from these studies are that the potential for nitrate loss from market gardens via groundwater transport is significant. Dilution and denitrification are the important processes that can act to ameliorate nitrate contamination of groundwater. But in the absence of these processes, nitrate will migrate at effectively the same velocity as the groundwater. In view of these results, and those obtained in the present study, there is a clear case for the design of irrigation groundwater pumping strategies (e.g. the siting of groundwater irrigation pumps) that capture and return nitrate contaminated groundwater that otherwise would migrate off-site from market gardens.

The leaching and transport of phosphate via saturated groundwater flow in soils and shallow groundwaters of the Swan Coastal Plain has been the subject of numerous studies. Phosphate differs from nitrate in that it is readily adsorbed to the soil surface, thus movement of phosphate is retarded relative to groundwater. Most experimental measurements of phosphate adsorption and desorption involve batch equilibration of the soil or sediment of interest with solutions of varying phosphate concentrations. The adsorption and desorption characteristics of a particular soil are then described by adsorption models such as the well-known Freundlich or Langmuir isotherms. Numerous attempts have been made to combine these isotherms with simple groundwater flow and transport models, in order to predict travel times of phosphate movement. These attempts are reviewed more fully by Townley *et al.* [1993].

Barrow [1980] studied soils from a number of sources, including Western Australia, and found that the relationship between adsorbed phosphate and the concentration in solution could be described by a modified Freundlich isotherm. Whelan *et al.* [1981] differentiated the adsorption capacity of Bassendean and Spearwood sands, demonstrating the enhanced capacity of the latter soil type to adsorb phosphate to over 200 mg P/kg soil. They demonstrated that for Spearwood sands under high phosphate input, breakthrough of phosphate to the water table at a depth of 7 m could occur in less than 5 years. For Bassendean sand, lower values for the phosphate adsorption capacity of less than 40 mg P/kg soil were determined.

In principle, travel times for phosphate relative to water in soil types of the Swan Coastal Plain can be estimated using adsorption parameters determined from previous work [Barrow, 1980; Gerritse, 1989]. The computed travel times for phosphate are usually determined for groundwater transport in the saturated zone only. In this sense, the predicted travel times are conservative estimates, because additional retardation in the unsaturated zone is not considered. In some instances retardation in the unsaturated zone may be significant, especially when the depth of the unsaturated zone is substantial. Estimates of travel times for phosphate in the unsaturated zone of several Western Australian soil types have been given by Sharma *et al.* [1991] and Gerritse [1993*a*]. For Spearwood sands, a realistic range of transport times over a 1m depth of unsaturated zone ranged was estimated to be 5 and 255 years depending on recharge rate and the phosphorus source strength at input [Sharma *et al.*, 1991]. Gerritse [1993*a*] claims that adsorption of phosphate in soils depends on both concentration and time, and estimates travel times for phosphate of between 80 and 630 years per metre of topsoil under a range of recharge rates and phosphate input conditions. Recent work by Gerritse [1993*b*] suggests that the mobilities of phosphate relative to water in soils of Rottnest Island are about 5% of the infiltration rates for water at artificially high infiltration rates of between 0.5 and 1 cmd^{-1} used for land application of wastewater. The review presented in Appendix A of Townley *et al.* [1993] casts doubt on these estimates, because of inconsistencies or errors in the equations used.

Pionke *et al.* [1990] pointed out the usefulness of non-nitrogen and non-phosphorus chemical parameters as indicators of potential negative impacts of horticultural land use on groundwater quality. The ratios of relatively conservative and mobile ions such as Na^+, K^+, Mg^{2+} and SO_4^{2+} to Cl^- that are associated with fertiliser application can be used as indicators of groundwater contamination by fertilisers. The SO_4^{2+}/Cl^- ratio in particular was confirmed as a useful indicator of the impacts of horticulture on groundwater quality. The basis for its application is that the SO_4^{2+}/Cl^- ratio in marine aerosol input to groundwater is between 0.15 and 0.3 (the seawater ratio of SO_4^{2+}/Cl^- is 0.14). Several factors that can affect the SO_4^{2+}/Cl^- ratio need to be considered regarding sulphate transformations, such as sulphate reduction, sulphide deposition and whether sulphate is a significant component of fertiliser. Pionke *et al.* [1990] reported that groundwaters affected by irrigated horticulture have ratios between 0.4 and 0.7 and found a ratio of 0.63 ± 0.3 for groundwaters considered to be affected by horticulture in the Coogee area. In quoting these results, we have converted values reported by Pionke *et al.* [1990] as SO_4^{2+}–S/Cl^- to SO_4^{2+}/Cl^-.

Petroleum products

When petroleum makes contact with water, the monoaromatic hydrocarbons that are components of it, i.e. benzene, toluene, ethylbenzene and the three xylene isomers (collectively referred to as BTEX compounds), comprise 60% of the soluble mass that goes into solution. Contamination of groundwater by BTEX compounds and napthalene from petroleum has only recently been recognised as a groundwater pollution issue on the Swan Coastal Plain [Barber *et al.*, 1991].

The transport characteristics of these compounds in groundwater under reducing conditions encountered in Bassendean sands of the Swan Coastal Plain has recently been reported by Thierren *et al.* [1992*a, b*]. These papers report the findings of field and laboratory studies into the migration of BTEX compounds in groundwater from a leaking underground petroleum storage tank The most important parameters that determine the transport of these compounds in groundwater were identified as the groundwater redox condition, retardation coefficients for the compounds, natural degradation processes due to biological activity and the groundwater flow rate. Under oxic conditions, the degradation of toluene, p-xylene and napthalene is rapid compared to the rate under anoxic conditions. However, as the available oxygen is depleted by microbial respiration, and because the replenishment rate of oxygen can be slow, anaerobic conditions can develop and persist. Under anoxic conditions, benzene in particular shows no significant biodegradation and the rates of degradation of the other BTEX compounds decrease. Retardation factors [Mackay, 1991] also indicate that benzene is the most mobile compound and napthalene the least. In the field, the combination of transport characteristics leads to benzene being the most persistent compound dissolved in groundwater. With a groundwater flow velocity of 150 my^{-1}, a plume of benzene 420 m long was detected. Toluene was completely degraded over a 200 m flow path. Transport modelling of the plume indicated that it had taken 4 years to develop from the onset of leakage.

Pesticides

There have been very few studies of pesticide transport specifically in Australian soils but in general, the topic of pesticide transport has been very extensively studied. A review of mathematical models of reactive solute transport of pesticides was presented by Rao and Jessup [1983]. Sorption and degradation of pesticides are the primary processes that lead to their attenuation in the environment. The organic matter content and clay content of soils have been widely identified as two major controls on the amount of pesticide sorption on soils.

Singh [1989] has provided the most relevant study of pesticide transport under conditions of relevance to tranport of pesticides in groundwater in relation to wetlands of the Swan Coastal Plain. Singh reported on the transport of four herbicides (diquat, paraquat, linuron and simazine) and one nematicide (fenamiphos) in Bassendean Sand. These pesticides represent classes of chemicals widely used in Australian agriculture. Diquat and paraquat are cationic herbicides and are known for their strong adsorption to soils and high solubility in solution. Linuron is non-ionic and belongs to the phenylurea group of herbicides. It has an intermediate solubility in water. Simazine belongs to the s-triazine group of herbicides and has a low solubility in water. Fenamiphos is a broad spectrum organo-phosphate nematicide and pesticide. It is non-ionic but polar and is highly soluble in water.

Jury *et al.* [1983] developed an equilibrium adsorption model incorporating degradation to describe of pesticide transport in soil. This model assumed steady water flow, equilibrium sorption, a depth dependent degradation and assumed a uniform organic carbon content in the soil profile. The model was subsequently used to screen the pollution potential of pesticides [Jury *et al.*, 1987]. Singh [1989] modified the former model to account for the frequently observed exponential decrease in organic carbon content with depth in the soil profile. Kookana *et al.* [1992] have since studied time dependent sorption of pesticides.

Table 2.7.1 OECD boundary values for open trophic classification system (annual mean values) [after Rast *et al.*, 1989]

Parameter		Oligotrophic	Mesotrophic	Eutrophic
Total phosphorus	$\bar{x}$	8.0	26.7	84.4
(µg P/L)	$\bar{x} \pm 2\sigma_x$	2.9 – 22.1	7.9 – 90.8	7.9 – 90.8
Total nitrogen	$\bar{x}$	661	753	1875
(µg N/L)	$\bar{x} \pm 2\sigma_x$	208 – 2103	313 – 1816	395 – 8913
Chlorophyll *a*	$\bar{x}$	1.7	4.7	14.3
(µg/L)	$\bar{x} \pm 2\sigma_x$	0.4 – 7.1	1.9 – 11.6	3.1 – 66
Chlorophyll *a* peak value	$\bar{x}$	4.2	16.1	42.6
(µg/L)	$\bar{x} \pm 2\sigma_x$	1.5 – 13	4.9 – 52.5	6.7 – 270
Secchi depth	$\bar{x}$	9.9	4.2	2.45
(m)	$\bar{x} \pm 2\sigma_x$	3.6 – 27.5	1.4 – 13	0.9 – 6.7

3. Identification of Capture and Release Zones

3.1 Definition of Flow Regimes

Wetlands of the Swan Coastal Plain, near Perth, Western Australia, can be classified according to geomorphological criteria [Semeniuk, 1987] as being lakes, sumplands or damplands. Lakes are wetlands in depressions in the land surface which have an exposed open water surface throughout the year. Sumplands have an open water surface for part of the year, but are dry for part of the year. Damplands never have an exposed open water surface, but are waterlogged during most of the year. This report focuses on lakes, but the results can be applied to sumplands when changing surface areas are taken into account.

In general, there are three kinds of water table lakes [Born *et al.*, 1979]. Those that receive groundwater over the whole of their bottom surface are known as "discharge" lakes. Those that release lakewater to the aquifer over the whole of their bottom surface are called "recharge" lakes. And those that receive water and release water over different parts of their bottom surface are "flow-through" lakes. All the lakes that have been studied in detail on the Swan Coastal Plain appear to act as flow-through lakes, which capture groundwater on their upgradient side and discharge lakewater on their downgradient side. Some smaller lakes may act as discharge or recharge lakes for short periods during the year.

Figure 3.1.1 shows a schematic illustration of a flow-through lake. The Figure defines a "capture zone", within which any recharge eventually flows through the lake, and a "release zone", which contains water which has passed through the lake. The shapes of capture and release zones may have important implications for land use management.

Figure 3.1.1 also defines regional (far field) and local (near field) scales. Any study of the unconfined aquifer of the Swan Coastal Plain should be carried out at a regional scale, and must take into account the spatial variation of aquifer properties and recharge that cause the water table to take its shape. On the other hand, much can be learned by studying an individual lake in isolation from its neighbours, and much of the effort during this project was focused on local scale behavior. More research is required to incorporate all our findings into regional scale studies.

Contaminants in groundwater are carried in the direction of groundwater flow, but also mix laterally within the aquifer and can be retarded by sorption or decay. Contaminants which are transported at the same velocity as groundwater are said to be carried by the process of "advection". Because groundwater follows a tortuous path through a porous matrix and through and around regions of varying hydraulic conductivity, contaminants are spread both longitudinally and laterally by the process of "dispersion". Dispersion is an important mechanism for mixing in porous media, but there is increasing evidence that advective flow patterns can often explain the dominant behaviour of non-reactive contaminants. Most of the modelling in this report is based on advection without dispersion.

Some contaminants become attached to the porous medium by the process of "adsorption", and others decay naturally, sometimes with assistance from bacteria. These processes affect travel times and intensities of contamination within a capture zone, rather than the shape of the capture zone itself.

In order to identify flow patterns near shallow lakes, we have developed two-dimensional models in vertical section, two-dimensional models in plan and three-dimensional models. These models have allowed a systematic study of the shape of capture and release zones as a function of nearby aquifer flows and net groundwater recharge.

Although the physical region near a lake is three-dimensional, it is not possible to use three-dimensional computer models for all studies of lake-aquifer interaction. As a result, we have used two-dimensional models in vertical section and in plan and identified the circumstances under which these models give good approximations to true three-dimensional behaviour.

Figures 3.1.2 to 3.1.4 define a two-dimensional vertical section, on which many of our results are based. A lake is represented by a layer at the surface of the aquifer of negligible (infinitesimal) thickness. The justification for this is that lakes on the Swan Coastal Plain are rarely more than 1 or 2 m deep, and this depth is negligible compared to the saturated thickness of the unconfined aquifer, which is typically 50 to 100 m.

The equation for steady flow of groundwater in a homogeneous isotropic vertical section is Laplace's equation. By solving Laplace's equation, we can compute both piezometric heads within the vertical section and a mathematical quantity known as "streamfunction". Contours of equal value of streamfunction show the paths along which water will travel.

A complete description of our model in vertical section is given by Nield *et al.* [1994]. We use a linear triangular finite element model known as AQUIFEM-N [Townley, 1993*a*] to solve three problems with unit values of U_+, U_- and R. These three solutions are superimposed to give the correct solution for arbitrary values of these boundary fluxes, and sophisticated post-processing produces flow nets and identifies the locations of "dividing streamlines". The latter define the shapes of the capture and release zones under any combination of boundary conditions.

To facilitate the use of our results by others, we have developed an interactive computer package called FlowThru [Townley *et al.*, 1992]. This package allows users to specify simple lake geometries, aquifer properties and boundary fluxes, and then predicts groundwater flow patterns, including the shapes of capture and release zones, in a vertical section. FlowThru applies to the near field of a lake, as defined in Figure 3.1.1. Although FlowThru predicts a total of 17 flow-through, 11 recharge and 11 discharge regimes, we have identified a subset of these, i.e. 9 flow-through, 5 recharge and 5 discharge regimes, which can occur in a vertical section when inflow and outflow at the lateral boundaries are in the same direction (see Figure 3.1.5).

Most lakes on the Swan Coastal Plain probably act as flow-through lakes with regimes FT1, FT2 or FT3. Smaller lakes, or lakes affected by high anisotropy in the aquifer (as discussed below), may act as recharge or discharge lakes, with regimes R1, R2, R4, D1, D2 or D4. The other regimes are less likely to occur, though we do not yet have enough field evidence to be sure. Notice that regime D4 suggests that the capture zone of the lake extends beyond the lake, and that recharge downgradient of the lake can flow against the average regional gradient towards the lake.

Figures 3.1.6 to 3.1.8 define a two-dimensional model in plan (see also Townley and Davidson, 1988). Solutions in plan are obtained using AQUIFEM-N. But we have also implemented a so-called "particle-tracking" method to predict travel paths in two-dimensional plan models and to calculate the shapes of capture and release zones. We argue that there are only three distinctly different flow regimes in plan when inflow and outflow at the lateral boundaries are in the same direction (Figure 3.1.9).

Figures 3.1.10, 3.1.11 and 3.1.12 define a three-dimensional model of the near field of a shallow lake. We obtain solutions for steady flow in three dimensions using a finite difference model which was especially developed for use with large numbers of nodes [Ababou *et al.*, 1988]. Some of our solutions use up to a million nodes. We have developed three-dimensional particle-tracking techniques, as well as three-dimensional visualisation of dividing surfaces to display three-dimensional groundwater flow patterns near lakes. We have shown that every flow regime which occurs in a vertical section also occurs in three dimensions, and that the flow regimes seen in plan also describe the apparent flow regime seen at the upper boundary of a three-dimensional region. Plate 3.1.1 shows three "dividing surfaces" for a lake with $2a/B = 1$ and flow-regimes corresponding to regimes FT1, FT2 and D4. This sequence corresponds to steadily increasing recharge with aquifer flows U_- and U_+ held constant and equal.

Townley *et al.* [1993] provide complete details of the computer modelling techniques developed and applied during this project.

3.2 Geometry of Capture Zones

The hierarchy of models developed during this project allows us to make quantitative predictions of the geometries of capture and release zones of shallow flow-through lakes.

Through systematic modelling, we now know that the depth of a capture zone on the centreline through a lake in the direction of regional aquifer flow depends mainly on the ratio of "lake length", $2a$, to the thickness of the aquifer, B. Length is defined to be in the direction of average flow, with the other horizontal dimension of a lake being called its "width", even if the width is larger than the length. In a vertical section through an isotropic aquifer (i.e. with equal hydraulic conductivities in the horizontal and vertical directions), we predict the depth of the capture zone at a distance of $2B$ upgradient of the lake.

As a rule of thumb, a water body with length equal to the aquifer thickness in an isotropic aquifer draws water from roughly the top half of the aquifer. A water body five or ten times longer than the aquifer thickness draws water from virtually the whole thickness of the aquifer, and discharges water to the same depth.

The simplest way to understand these results is to think of each droplet of water following the path of least resistance from its source (in a recharge area) towards its discharge point (a distant river or the ocean). It is easier for water at the base of an aquifer to rise 50 m into a wetland, to travel hundreds of metres horizontally in the water body itself and then to flow 50 m downwards again, than to travel hundreds of metres along the bottom of the aquifer, where there is negligible driving force because the water surface above is flat.

There is a good electrical analogue to this phenomenon. Consider a rectangular sheet of metal representing the vertical section in Figure 3.1.3, with a strip of copper at the upper surface representing the lake. If a potential difference (voltage drop) is applied between the two ends of the sheet, and if the copper has lower electrical resistance than the metal sheet, the current will "short-circuit" through the copper. This is completely analogous to the path of water through a lake at the surface of an aquifer.

When $2a/B \approx 1$, we say that a lake is "short", whereas when $2a/B \geq 4$, a lake is "long". Figure 3.2.1 shows likely flow regimes in a vertical section through short and long lakes. Figure 3.2.2 defines the depth of the capture zone, b_+, for a flow-through lake. Figure 3.2.3 shows the fundamental relationship between the depth of a capture zone relative to the aquifer thickness, b_+/B, and lake length for a flow-through lake, as obtained using FlowThru. Figure 3.2.4 confirms that fully three-dimensional modelling predicts a result which is very close to that predicted by FlowThru. This gives us confidence that FlowThru can be used as a tool to predict real three-dimensional behaviour, even though it is a two-dimensional model.

The ratio of horizontal to vertical hydraulic conductivity is known as the anisotropy ratio. We can show that a lake in an anisotropic aquifer behaves like a lake in an equivalent isotropic aquifer which has an effective $2a/B$ which is smaller by the square root of the anisotropy ratio. A lake with a physical $2a/B = 10$ and an anisotropy ratio of 100 therefore behaves like a lake in an isotropic aquifer with $2a/B = 1$. The depth of the capture zone predicted by FlowThru applies at a distance of $2B$ multiplied by the square root of the anisotropy ratio. Anisotropy therefore has a very significant effect on capture zone geometry.

A lake with a continuous lining of low conductivity sediments also behaves like a lake with a smaller $2a/B$. This is because the low conductivity lining tends to isolate the lake from the aquifer and therefore there is less tendency for water to be attracted towards the lake. This effect is shown in Figures 3.2.5 to 3.2.7, where increasing values of the parameter D (see Townley *et al.*, 1993; Nield *et al.*, 1994] represent increasing sediment resistance. Although we have not carried out numerical experiments, we believe that a non-uniform lake lining, especially one in which the sediments are located near the middle of the lake, will have little effect on the capture zone geometry (Figure 3.2.8*d*). It follows that FlowThru probably applies reasonably well to any lakes with sandy beaches, regardless of the hydraulic conductivity of sediments in the middle of the lakes.

The ratio of aquifer flows upgradient and downgradient of a lake is a key variable in describing flow regimes and capture zone geometries. We use U_-/U_+ to denote the ratio of downgradient to upgradient flow, or equivalently, the ratio of water table gradients in a homogenous aquifer. Capture zone depth increases as U_-/U_+ decreases, and vice versa (Figure 3.2.9), but the effect is almost negligible for long lakes. Capture zone depths are also affected by recharge (Figure 3.2.10), though it is more instructive to consider the joint effects of U_-/U_+ and recharge, with the net flux Q from a lake into the underlying aquifer held constant (Figure 3.2.11).

Two-dimensional modelling in plan shows that the width of the capture zone for a circular lake depends on the degree of isolation of the lake, measured by a/W where $2W$ is the distance between the centres of adjacent lakes (Figure 3.1.7). When a/W is small, a lake is effectively isolated and the width of the capture zone approaches twice the lake diameter (Figures 3.2.12 and 3.2.13). As $a/W \rightarrow 1$, a circular lake is almost touching its neighbours and the width of its capture zone approaches the lake diameter.

Two-dimensional modelling assumes that flow is essentially horizontal everywhere and that a lake is effectively a cylinder of water extending to the bottom of the aquifer (Figure 3.1.6). Based on two-dimensional modelling in vertical section, we argue that a long lake has capture and release zones that extend to the bottom of the aquifer, so that flow is almost horizontal over most of the region. Furthermore, flow is negligible beneath the middle of the lake, so that the aquifer below the lake is almost hydrostatic, as if it were part of the lake. There is thus good justification for accepting the results of two-dimensional modelling in plan for long lakes.

Three-dimensional modelling of flow-through lakes with no recharge allows us to predict the width of capture zones at the land surface. Figures 3.2.14 and 3.2.15 show that capture zone width approaches twice the diameter of a circular lake as the lake becomes longer in the direction of regional flow and when a lake is isolated from its neighbours.

There are many symmetries in our modelling results, and as a result it is possible to infer the depth of a release zone from knowledge of the depth of a lake's capture zone. From a modelling point of view, we usually predict capture zone geometry and infer the shape of the release zone. In the field, we attempt to identify the release zone and then infer the capture zone.

3.3 Bottom Seepage

Groundwater seepage into and out of a flow-through lake is concentrated near the upgradient and downgradient edges of the lake. Model simulations allow us to predict the spatial distribution of seepage between a lake and its underlying aquifer. FlowThru predicts bottom seepage in a two-dimensional vertical section. We can prove that seepage rates do not decay exponentially, as has been suggested by some researchers, but that there is certainly a rapid decline in seepage with distance from the shore (Figures 3.3.1 and 3.3.2). Seepage on the bottom of a circular lake can be calculated by three-dimensional modelling. Although the results are not as accurate as we would like, the results confirm that seepage is greatest near the leading and trailing edges of the lake (Figure 3.3.3).

Seepage rates can also be measured in the field. Figure 3.3.4 shows measurements from three seepage meters at Nowergup Lake in 1989. Townley et al. [1993] provide more details of field experience with seepage measurements. Consistent measurements are difficult to obtain, and there are many potential sources of error. As a result, seepage measurements are not as useful as is sometimes believed.

3.4 Identifying Release Zones Using Isotopes and Hydrogeochemistry

The shape of a lake's release zone can be identified in the field by taking water samples and analysing for isotopic and hydrogeochemical concentrations.

Evaporation from the open water surface of a lake causes concentrations of natural isotopes, chloride and other ions to increase in the lakewater. The naturally occurring heavy isotopes of water are known as oxygen-18 and deuterium (^{18}O and ^{2}H), and water molecules containing these isotopes do not evaporate as easily as the more abundant light isotopic water molecules. Chloride is also concentrated by evaporation, but can also be concentrated by plant water uptake. Only the isotope signatures provide unequivocal evidence that water has been subjected to evaporation.

Groundwater on the downgradient side of any flow-through lake has the same isotopic and hydrogeochemical signature as lakewater. As a result, it is possible to identify the shape of a lake's release zone by analysing groundwater samples for these constituents.

We have studied the release zones of Nowergup Lake, Mariginiup Lake, Jandabup Lake and Thomsons Lake using isotopic and hydrogeochemical tracers. Figures 3.4.1, 3.4.2 and 3.4.3 demonstrate the clear differences between concentrations of oxygen-18 and deuterium in upgradient groundwater, lakewater and downgradient groundwater at Nowergup, Jandabup and Thomsons Lakes. Figures 3.4.4, 3.4.5 and 3.4.6 show generalised cross-sections through these three lakes and demonstrate that the depth of a lake's release zone can easily be identified using isotopic data. In all cases, flow is from right to left, i.e. from east to west. Figures 3.4.7, 3.4.8 and 3.4.9 show that chloride data lead to the same conclusions.

The release zone at Nowergup Lake extends to a depth of about 0.6*B*, where *B* is the thickness of the aquifer in superficial formations near the lake. A similar depth is inferred at Mariginiup Lake, although fewer data are available there, and the flow system near the latter seems to be more complex. Jandabup and Thomsons Lakes have release zones that extend to the bottom the aquifer, consistent with these lakes being very long in the direction of regional flow.

Table 3.4.1 summarises the findings of our field investigations on release zones using isotopic and hydrogeochemical tracers. It is possible to use observations of release zone depth to infer likely values of aquifer anisotropy at a large scale. Anisotropy ratios of 50 or 100 are not inconsistent with our observations.

3.5 The Pinjar-Nowergup Transect

Isotopic and hydrogeochemical tracers have shown that outflow from Lake Pinjar becomes inflow to Nowergup Lake, a distance of 5.75 km downgradient.

During our study at Nowergup Lake, we recognised isotopic and hydrogeochemical anomalies near the upgradient edge of the lake that seemed to indicate that groundwater at mid-depths in the superficial formations had been previously evaporated. It was suspected that these anomalies could be due to the influence of groundwater within the release zone of Lake Pinjar. Consequently, further drilling and sampling confirmed that Lake Pinjar's release zone extends almost to the bottom of the aquifer and that Nowergup Lake receives water from the upper part of the release zone. Figures 3.5.1 and 3.5.2 show generalised cross-sections along the Pinjar-Nowergup transect, with deuterium and choride concentrations which confirm the connection between the two lakes. The regional aquifer flow is again from east to west.

Plate 3.5.1 shows the results of computer simulations of groundwater flow and transport along a cross-section through these lakes. Plumes of isotopes or chloride are shown to emanate from the lakes, with streamlines superimposed. These results are the only examples of multi-dimensional transport modelling in this report. They confirm that the observed pattern of concentrations can be explained by reasonable combinations of recharge, hydraulic conductivities, and dispersivities. The best results are obtained with an anisotropy ratio of 100 and a ratio of longitudinal to transverse dispersivity of 1000. The latter is very high, and supports an increasing body of evidence that lateral dispersion may be less important than advection in controlling the movement of tracers or contaminants in groundwater.

3.6 Monitoring Techniques

The most cost-effective way to learn about a lake's release zone and hence its groundwater flow regime is to install a nest of piezometers or a multi-level piezometer at the middle of the downgradient side of a lake.

Isotopic and hydrogeochemical data from a nest of bores, with screens positioned at appropriate intervals between the water table and the bottom of the aquifer, should provide clear evidence for the location of a dividing surface which separates a lake's release zone from groundwater which has passed below a lake. A knowledge of the shape of the release zone gives support to computer models which can then predict the shape of the capture zone.

3.7 Piezometric Head Measurements

Measurements of piezometric heads upgradient and downgradient of a lake can in principle give information about the geometry of capture and release zones, but are not as conclusive as isotopic and hydrogeochemical data.

The first step in understanding a lake's role in a lake-aquifer system is to measure regional water table elevations and infer average directions of aquifer flow. For flow-through lakes of the type seen on the Swan Coastal Plain, this allows the identification of the upgradient and downgradient sides of a lake. Since groundwater rises towards a lake at its upgradient shore, a nest of piezometers at that location should show increasing heads with depth. Conversely, at the downgradient shore, heads in a nest of piezometers should decrease with depth.

Investigations of this type were not particularly successful during this project, perhaps because boreholes used for water level monitoring were located and constructed many years ago for other purposes. Figures 3.7.1 and 3.7.2 show piezometric heads near Nowergup Lake, Figures 3.7.3 and 3.7.4 show heads near Mariginiup Lake, Figures 3.7.5 and 3.7.6 show heads near Jandabup Lake, and Figures 3.7.7 and 3.7.8 show heads near Thomsons Lake. Maps showing the locations of monitoring sites are given by Townley *et al.* [1993].

Data on the upgradient side of Mariginiup Lake show the most consistent upward head gradient, of the type expected. Prediction of capture zone depth based on the difference in depth-averaged heads, as proposed by Nield [1990], could not be successfully demonstrated using our field data.

Measurements of water table elevations near the stagnation or dividing points at the northern and southern ends of Nowergup Lake (Figures 3.7.9 and 3.7.10) did not give conclusive results. The purpose of these measurements was to attempt to detect subtle changes in the water table configuration near these critical points. But because gradients are so flat in these regions, it is perhaps not surprising that clear trends were not seen.

3.8 Water Balance Determinations

The concentration of isotopes and chloride in lakewater and a lake's release zone can assist in the determination of a lake's water balance.

Although we have not carried out detailed water balance calculations for any individual lakes during this project, all of our modelling is implicitly based on water balances. Estimating components of the water balance by measuring them alone is not as useful as simultaneous calculation of water, solute and thermal balances. Isotope and chloride balances are two forms of solute balances, which provide complementary information about the various components of the water balance. Chloride and bromide balances would probably be redundant, i.e. the measurements of bromide would not add any new information since bromide migrates in exactly the same way as chloride. Natural isotopes, however, provide additional information, because of the fractionation during evaporation that imparts an unequivocal signature to the residual lakewater.

Most solute and thermal balances of lakes suffer from incomplete knowledge about groundwater inflows and outflows. We have developed a new method for calculating the solute and isotope balances of flow-through lakes by integrating the results of groundwater flow calculations in a two-dimensional vertical section near shallow lakes. Figures 3.8.1 and 3.8.2 show the conceptual models on which the balances are based. Numerous Figures have been prepared for two lake lengths, $2a/B = 1$ and 4, to demonstrate that the ratios of chloride and isotope concentrations in lakewater relative to those in upgradient groundwater depend on the relative magnitudes of various components of the water balance near a lake.

We have applied this methodology to Nowergup Lake, with some success (Figures 3.8.3 to 3.8.7). For a given value of $2a/B$, and for particular magnitudes of rainfall P or evaporation E at the lake surface, Figures 3.8.3 to 3.8.6 show contours within a zone in $(U_-/U_+, RL/U_+B)$ space in which flow regimes are known to be of types FT1, FT2 or FT3. In each Figure there are two sets of contours. One set of contours has constant values of b_+/B and the other set has constant values of either ratios of chloride concentrations or ratios of isotope abundances. The basic technique is based on the fact that a nest of bores on the downgradient side of a lake allows us to estimate all three of these ratios. By preparing Figures of this kind and plotting points at the intersections of particular contours, we can obtain estimates of U_-/U_+ and RL/U_+B, and hence estimates of the components of the water balance of a lake.

Figure 3.8.7 shows predictions of dividing streamlines in a vertical section through Nowergup Lake, based on interpretation of Figures 3.8.3 to 3.8.6. Further details of this technique are given by Townley *et al.* [1993]. The technique may be useful in future for interpretation of flow regimes near relatively small lakes.

3.9 Seasonal Fluctuations in Capture Zones

Capture zone geometries vary seasonally as lake levels and surface areas fluctuate. Although we have not carried out systematic modelling of the seasonal dynamics of capture and release zones, we have argued that flow regimes near lakes must change in response to seasonal variations in rainfall minus evaporation over the lake surface, net recharge to the water table and regional aquifer flows. Figures 3.9.1 to 3.9.3 show examples of interesting systems containing more than one lake. Figures 3.9.4 and 3.9.5 illustrate the ways in which capture zones probably change.

Figure 3.9.4 is particularly interesting, because it suggests that lakes are either groundwater or surface water dominated. Groundwater dominated lakes have lake level fluctuations that are smaller than water table fluctuations, and lake levels probably lag behind groundwater levels. Surface water dominated lakes are driven by changes in lake volumes, for example due to large inputs from surface drains, thus lake level fluctuations are larger than water table fluctuations and lake levels rise and fall before the surrounding groundwater levels.

These findings are supported by modelling (e.g. Figure 3.9.6), as described by Townley *et al.* [1993]. This Figure shows dynamic fluctuations in water levels, relative to the long-term average levels, for a section of aquifer between a coast (at the left) and a groundwater divide (at the right) during a period of one year. The uppermost Figure shows a system driven by fluctuations in the surface water level, while the lowest shows a groundwater dominated system. There is an infinite number of possible responses, depending on a number of non-dimensional ratios. In spite of these results, capture zones do not need to vary as drastically as Figure 3.9.4 indicates. Some lakes on the Swan Coastal Plain may fluctuate between recharge and flow-through regimes or between flow-through and discharge regimes, without necessarily fluctuating all the way between recharge and discharge regimes.

The depth and width of a capture zone may vary significantly in lakes which have large seasonal changes in surface area. The depth and width can be estimated using the length and width of the lake at any instant in time, i.e. assuming quasi-steady behaviour of the system.

3.10 Capture Zones of Multiple Lakes

Capture zones of lakes on the Swan Coastal Plain can be determined by regional scale modelling, coupled with results of idealised modelling of isolated lakes.

The many lakes on the Swan Coastal Plain are an integral part of a regional groundwater flow system, and their behaviour therefore depends on regional hydrogeology, regional land use patterns, regional climatic variations and the behaviour of other lakes. The Perth Urban Water Balance Model [Cargeeg *et al.*, 1987*a*] provides the logical framework for assessing the behaviour of lakes on the Swan Coastal Plain.

The Perth Urban Water Balance Model combines a two-dimensional aquifer flow model, which assumes that flow in the superficial formations is essentially horizontal, and a Vertical Flux Model capable of predicting net recharge to the aquifer as a function of space and time. The model also allows lakes to be represented using a "dummy" second layer, such that groundwater flow can leak upwards into and downwards out of high conductivity zones representing the lakes. As summarised in Chapter 5, we have developed new methods for predicting effective model parameters to be used in this kind of approach. This will allow the Water Authority in the future to simulate lakes embedded in the regional aquifer more accurately, especially if model grids are redesigned with higher resolution.

We have used the Perth Urban Water Balance Model and a data set provided by the Water Authority to predict an annual average aquifer flow pattern for a large region to the south of Perth containing the Jandakot Mound. We have then used particle tracking to define the capture zones of seven lakes (Figure 3.10.1). These capture zones are predictions only, based on the Water Authority's calibration of the regional model.

All capture zones in Figure 3.10.1 extend the entire distance to the top of the Jandakot Mound. But actual capture zones may be shorter because of three-dimensional effects which can not be simulated directly at regional scale. We have developed a methodology for accounting for three-dimensional effects, based on anisotropy and the resistance of lake bottom sediments. The lengths of predicted capture zones can be shortened based on water balance arguments in a vertical section, and using the depths of capture zones predicted with FlowThru (Figures 3.10.2 and 3.10.3). A complete explanation of the method is given by Townley *et al.* [1993]. In principle it would be possible to calculate capture zones based on travel times and decay or retardation of particular contaminants, but we have not had the resources to do so.

3.11 Transport of Nutrients Within Capture Zones

Nitrate, phosphate, petroleum products and pesticides can all be carried by groundwater, but some are degraded or retarded, thus reducing their rate of movement through an aquifer.

It is well known that some potential contaminants break down naturally in the groundwater environment, while others become adsorbed to the porous matrix and even react chemically with the matrix. We have not carried out fundamental research on these issues during this project, but have reviewed available literature, with particular emphasis on local research on the Swan Coastal Plain. We have also carried out investigations at a market garden near Lake Wattleup, specifically to study the occurrence and movement of nitrate and phosphate in the unsaturated zone and just below the water table.

Nitrate is generally highly mobile, although under reducing conditions, denitrification causes nitrate to degrade into nitrogen. Figures 3.11.1 and 3.11.2 show high and variable nitrate concentrations within the area of the Lake Wattleup market garden, both above and below the water table. Table 3.11.1 shows that nitrate has travelled at least 50 m beyond the boundaries of the irrigated market garden area, although the breakthrough is unevenly distributed.

Phosphate is far less mobile, because it is readily adsorbed onto the porous matrix. Figure 3.11.3 shows that most phosphate is retained in the upper 2 m of the unsaturated zone beneath the Lake Wattleup market garden. Elevated phosphate concentrations are observed at one site beyond the market garden boundary, at a site where nitrate levels are also high. It is possible that these observations may be due to the discharge of nutrients from washing sheds located near the downgradient end of the market garden.

Batch experiments were carried out to evaluate the adsorption properties of phosphate on soil from the Lake Wattleup market garden. We have applied an approximate analytical model for the movement of a sharp front of phosphate in the unsaturated zone, using parameters from the batch experiments. Table 3.11.2 shows that travel distances in 100 years are likely to be less than 16 m in the unsaturated zone, at typical rates of recharge and fertiliser application. Lateral travel distances at or near the water table have not been calculated, but may be further, because of faster velocities of water.

Tables 3.11.3 and 3.11.4 show published results on the movement of petroleum products and pesticides in Western Australian conditions. Of the BTEX compounds in petroleum, benzene is the most mobile in groundwater. Of the pesticides, simazine is the most likely to migrate.

3.12 Implications for Land Use Planning

The capture zones of lakes have implications for land use planning, in that it may be desirable to control potentially polluting activities within the capture zones.

The groundwater capture zone of a lake allows us to define a region on the land surface within which recharge will eventually pass through the lake. If there is a source of contamination at or below the land surface within capture zone, there is ultimately a potential for contaminated water to reach the lake. The groundwater capture zone therefore allows us to define an upper bound on the size of a buffer zone needed to protect water quality in the lake for all time.

Taking into account the relative mobilities of different contaminants, it is possible in principle to calculate travel times for each contaminant within the capture zone, and therefore to define 10-year or 50-year buffer zones, based on the time each contaminant would take to reach the lake. It may be undesirable to site horticultural activities or stormwater drainage sumps, for example, within the groundwater capture zone of a lake, or at least within zones defined by 50-year or 100-year travel times for likely contaminants.

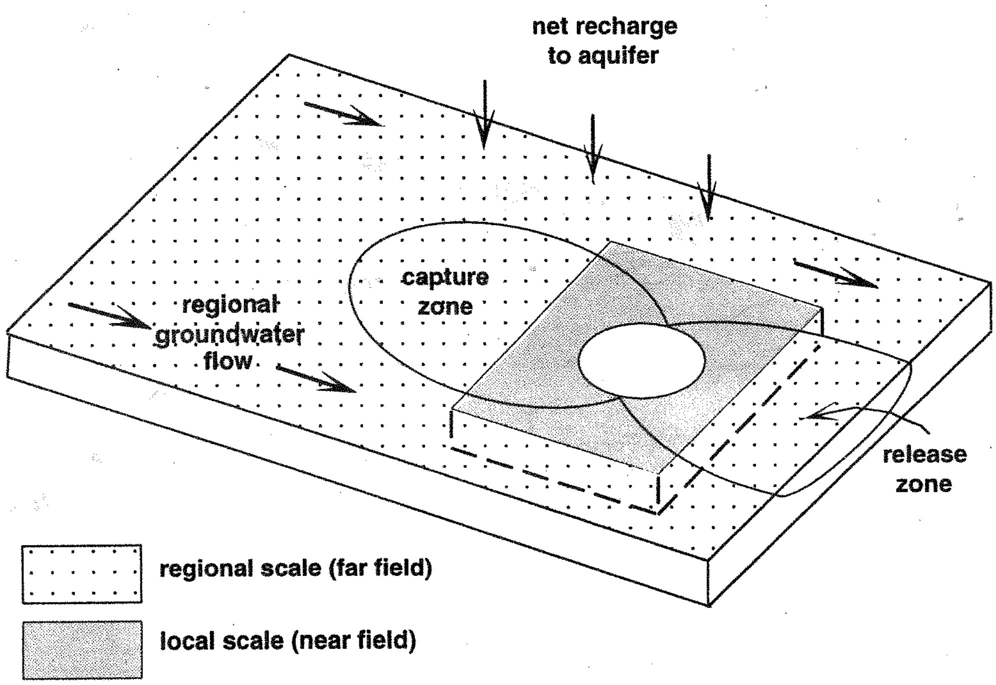


Figure 3.1.1 Relationship between regional scale and local scale

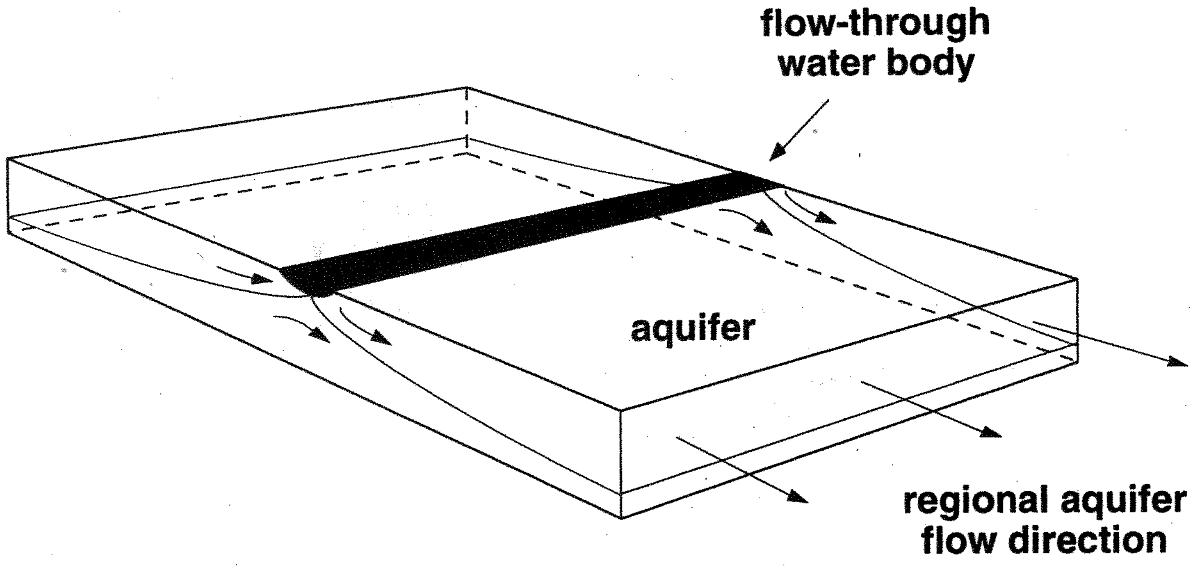


Figure 3.1.2 Groundwater flow near a long flow-through lake

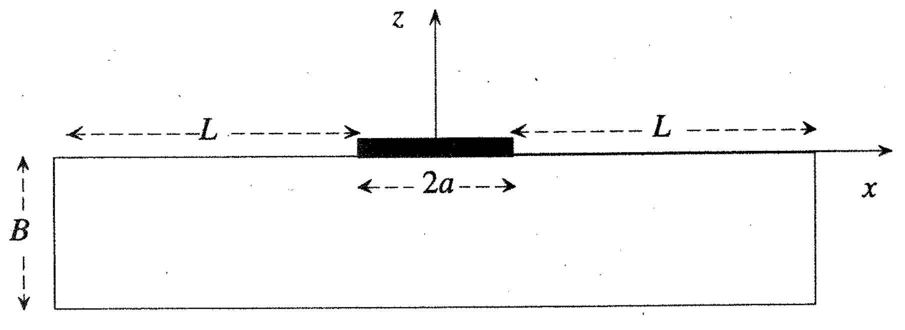


Figure 3.1.3 Idealised geometry for model of a vertical section through a shallow lake

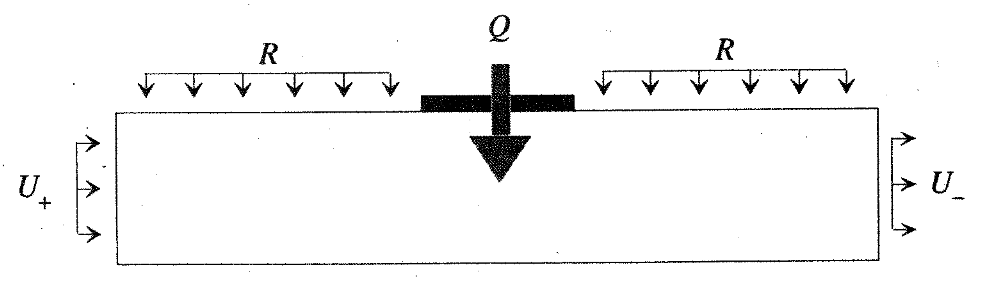


Figure 3.1.4 Boundary conditions for model of a vertical section through a shallow lake

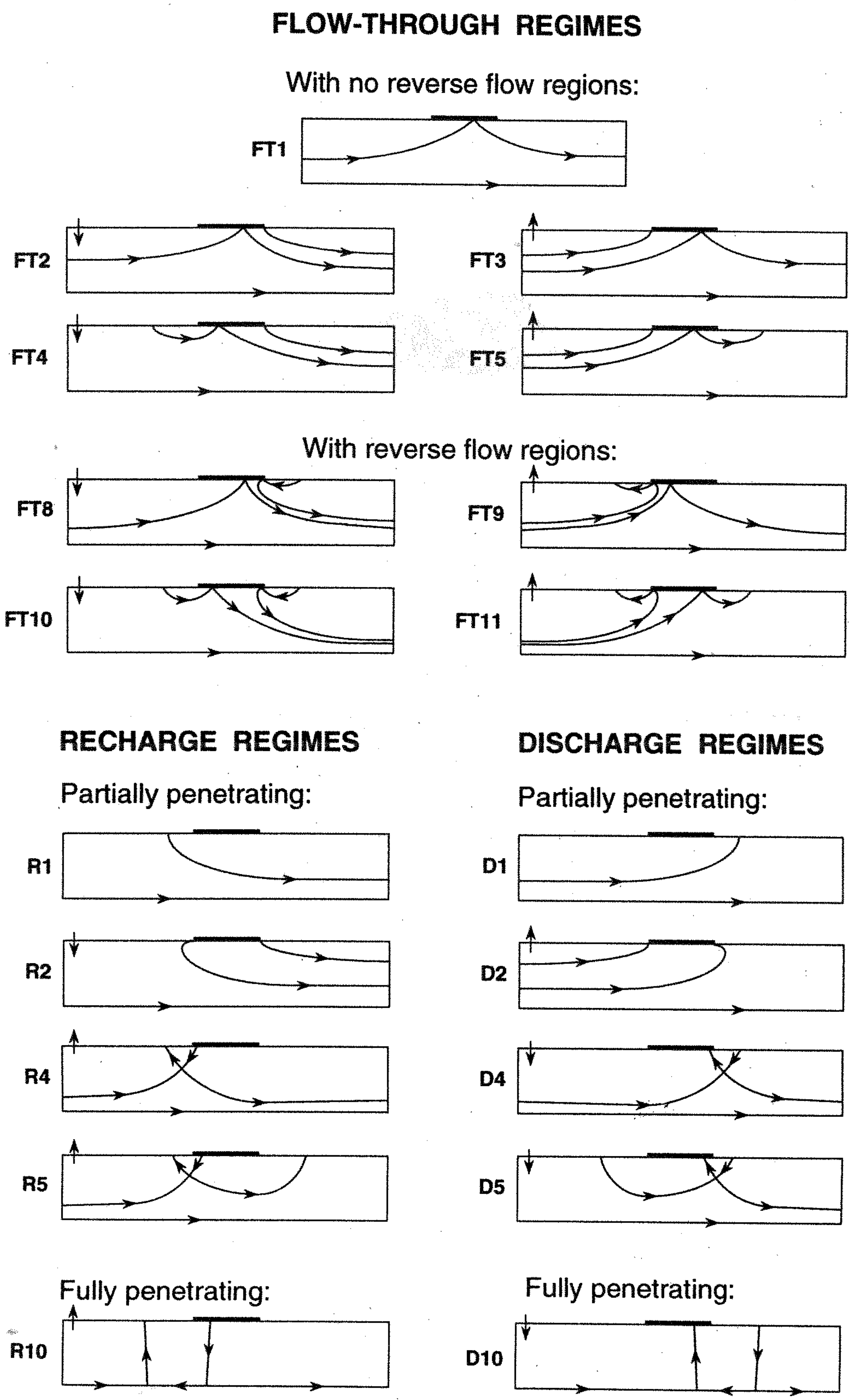


Figure 3.1.5 Schematic diagrams showing dividing streamlines for recharge, discharge and flow-through regimes which may occur in regional flow situations [after Nield *et al.*, 1994]

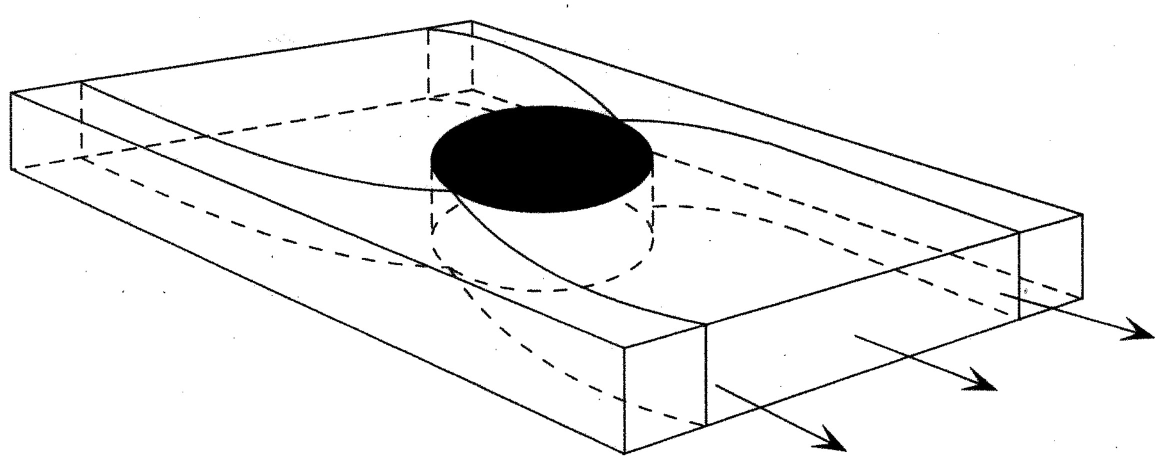


Figure 3.1.6 Groundwater flow near a circular fully-penetrating flow-through lake

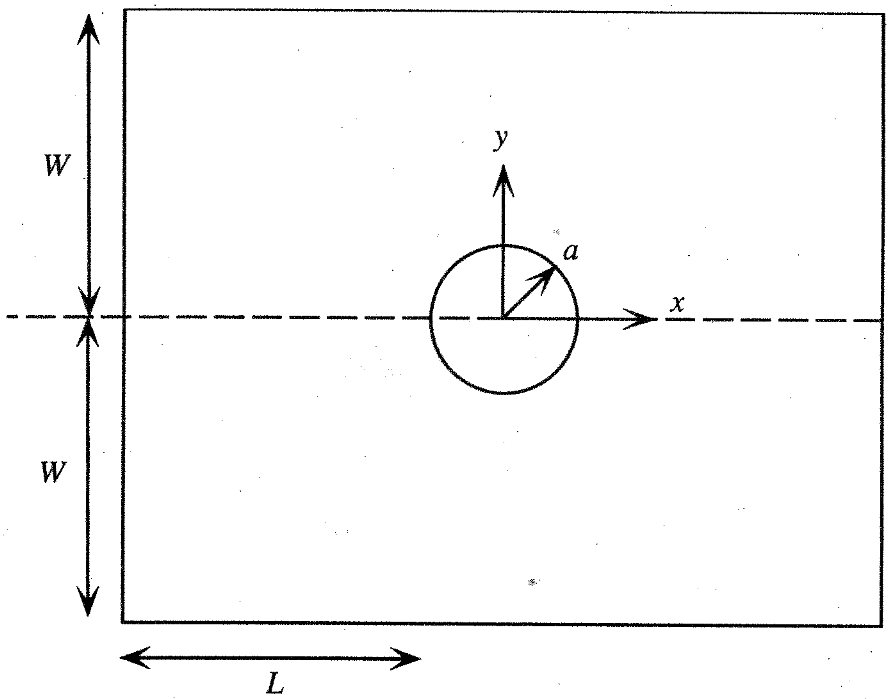


Figure 3.1.7 Idealised geometry for plan model of a fully-penetrating circular lake

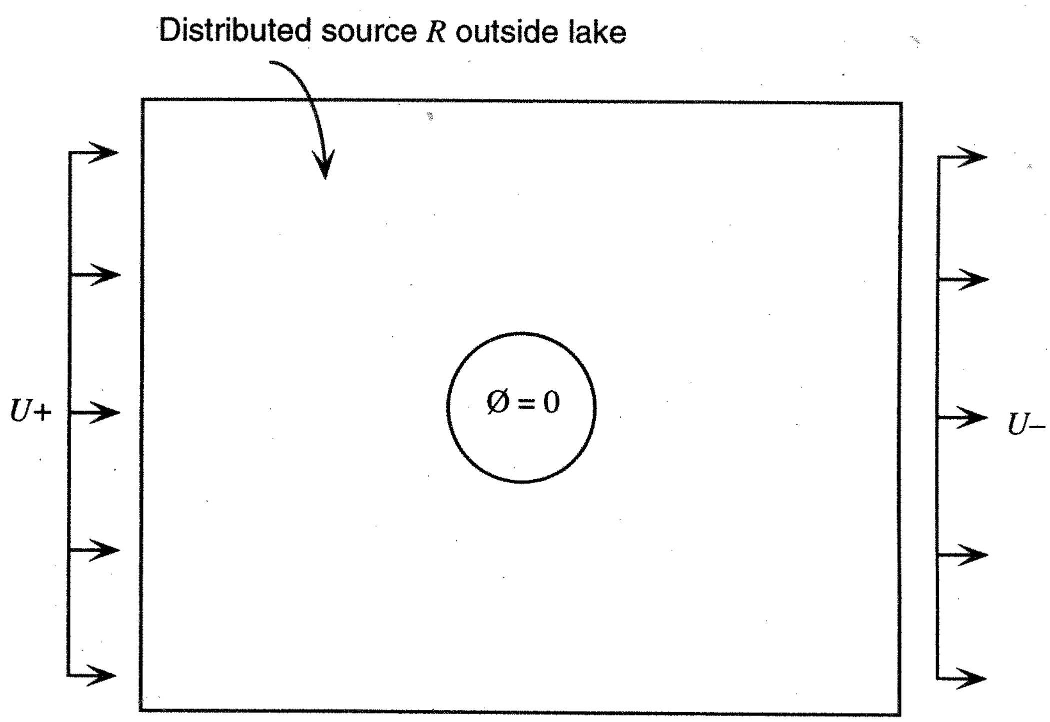


Figure 3.1.8 Boundary conditions and recharge for plan model of a fully-penetrating circular lake

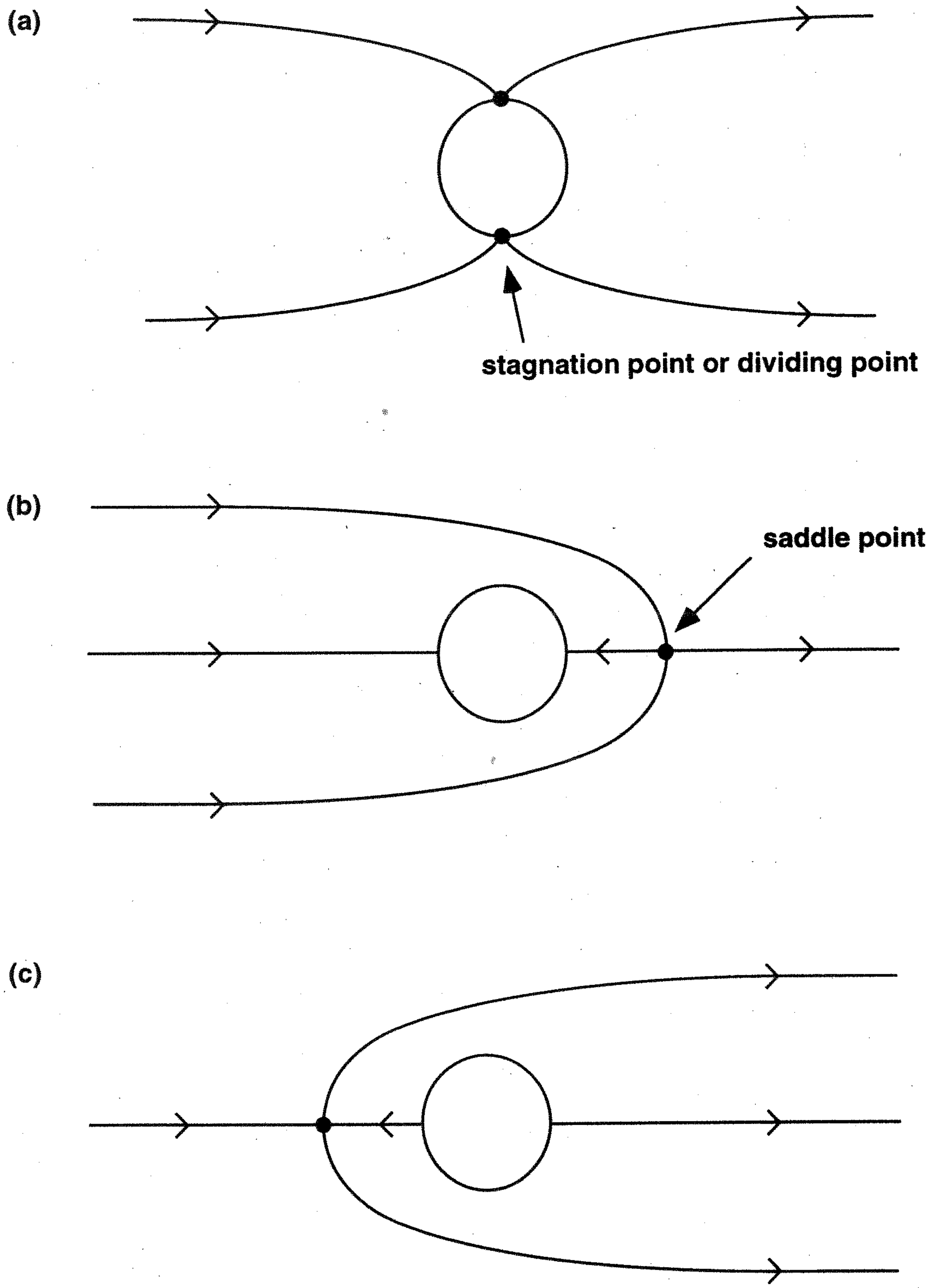


Figure 3.1.9 Possible flow regimes in plan

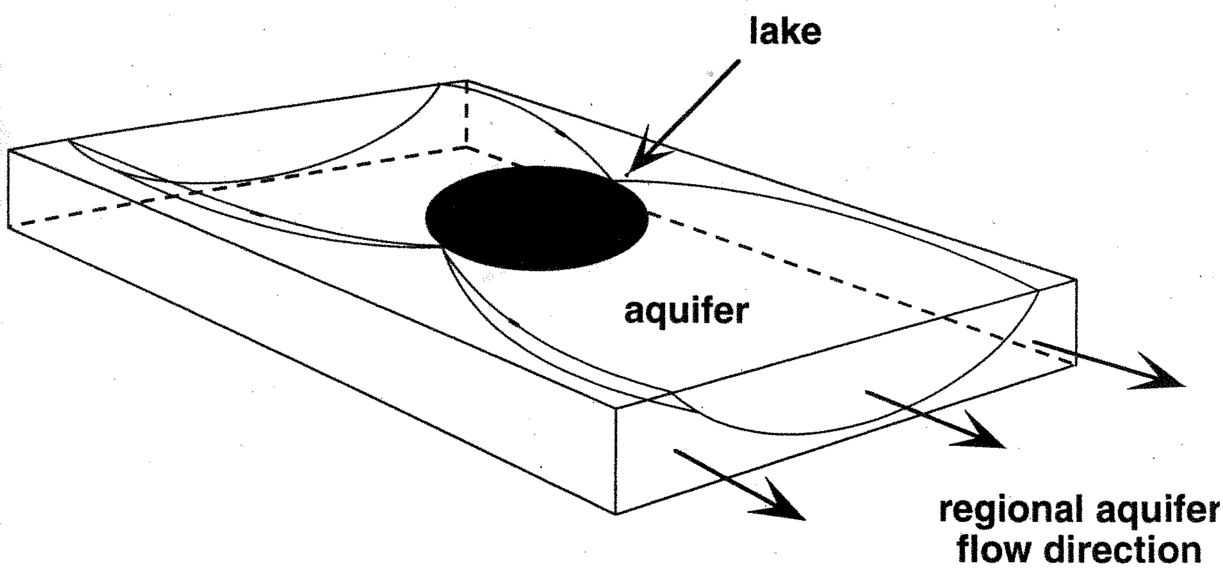


Figure 3.1.10 Three-dimensional groundwater flow near a circular flow-through lake

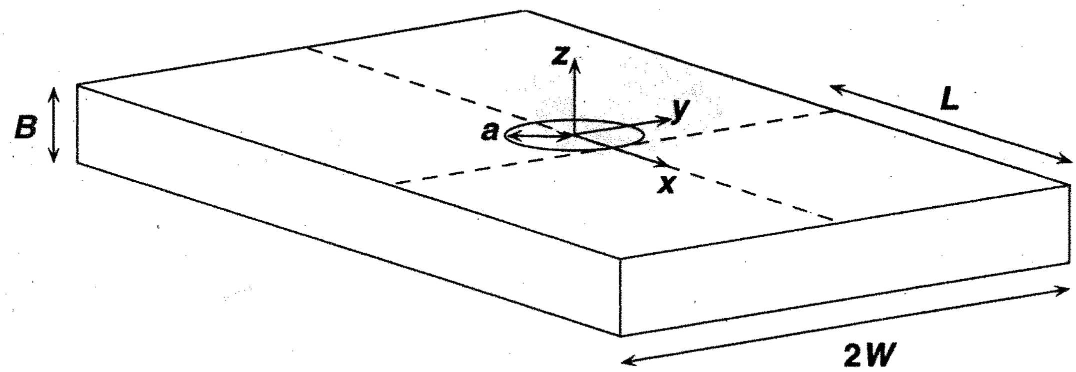


Figure 3.1.11 Idealised geometry for model of three-dimensional flow near a shallow circular lake

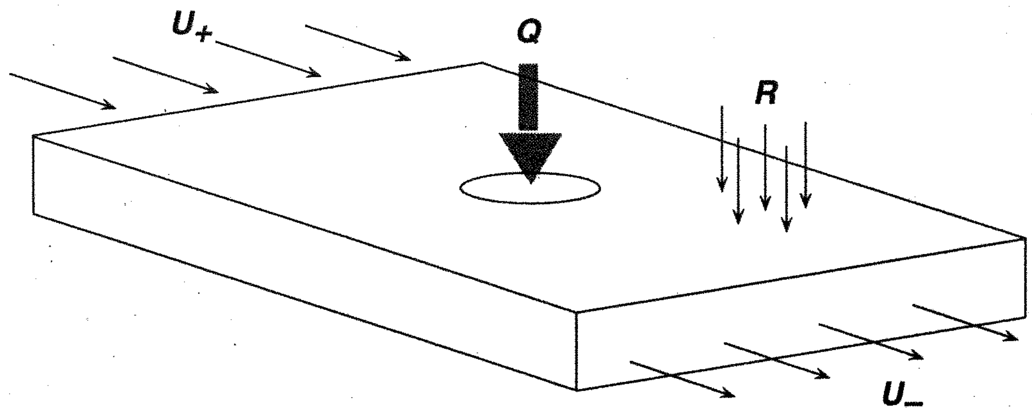


Figure 3.1.12 Boundary conditions for three-dimensional model near a shallow circular lake

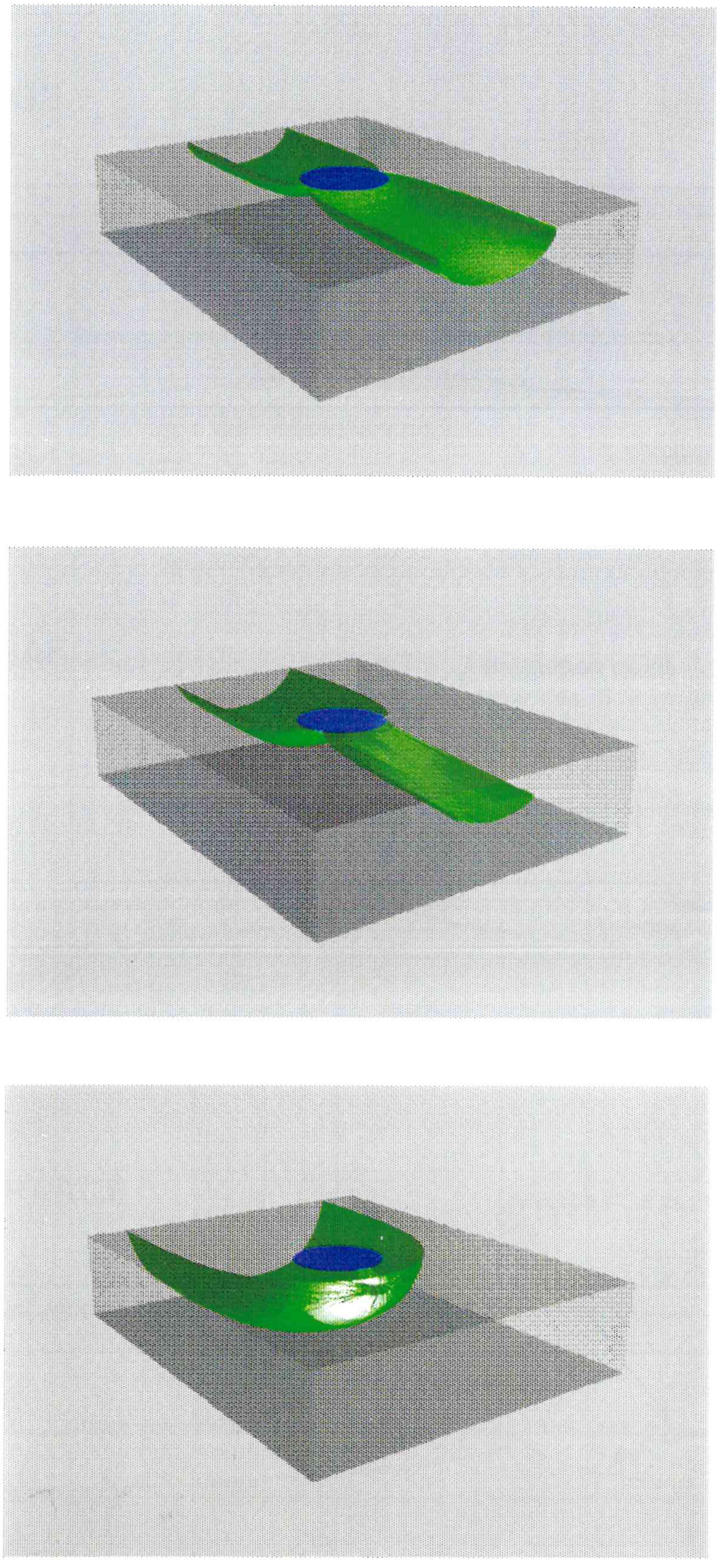


Plate 3.1.1 Dividing surfaces for steady three-dimensional flow near a shallow circular lake, corresponding to zero recharge and two successively larger values of recharge

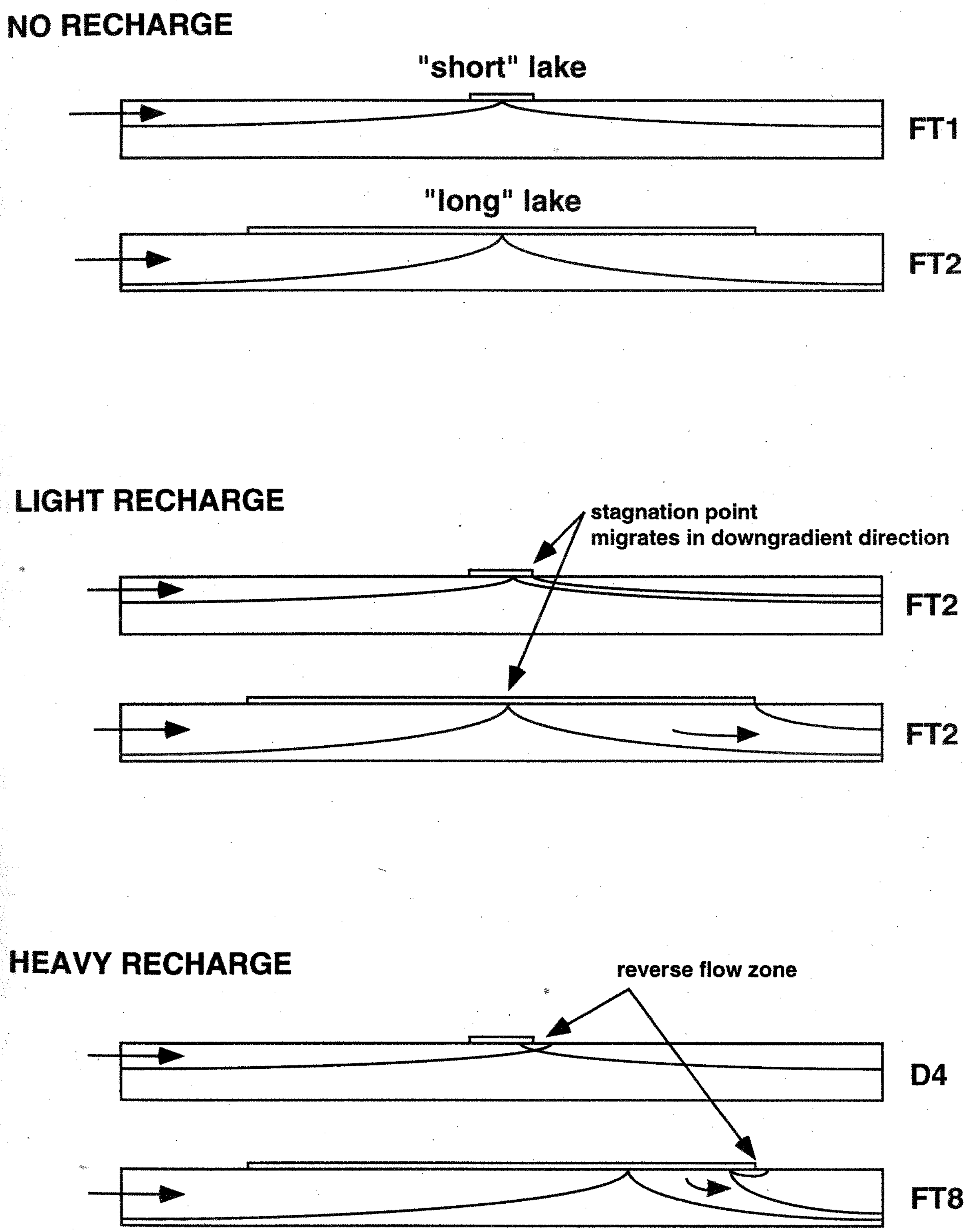


Figure 3.2.1 Likely flow patterns near "short" and "long" lake

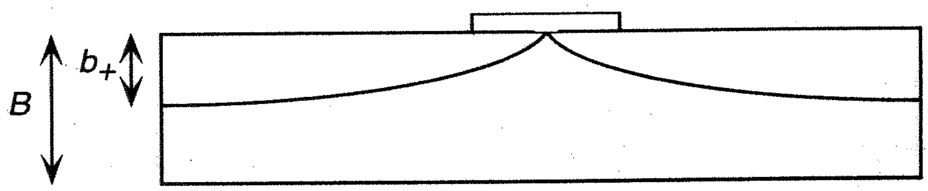


Figure 3.2.2 Definition of depth of capture zone b_+

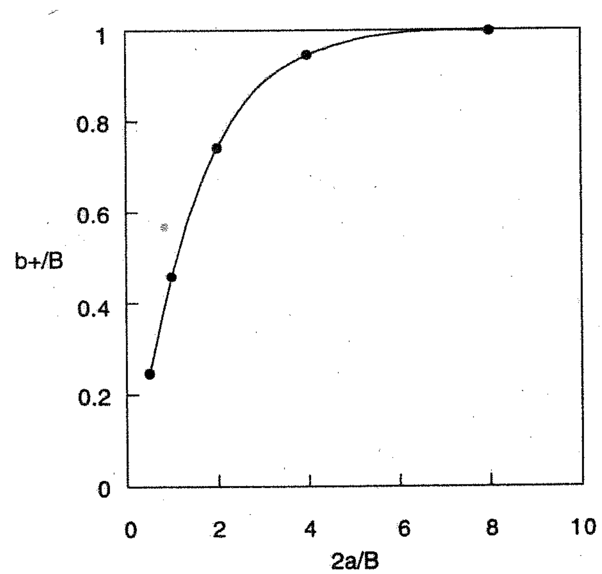


Figure 3.2.3 Depth of capture zone for flow-through lakes with $R = 0$

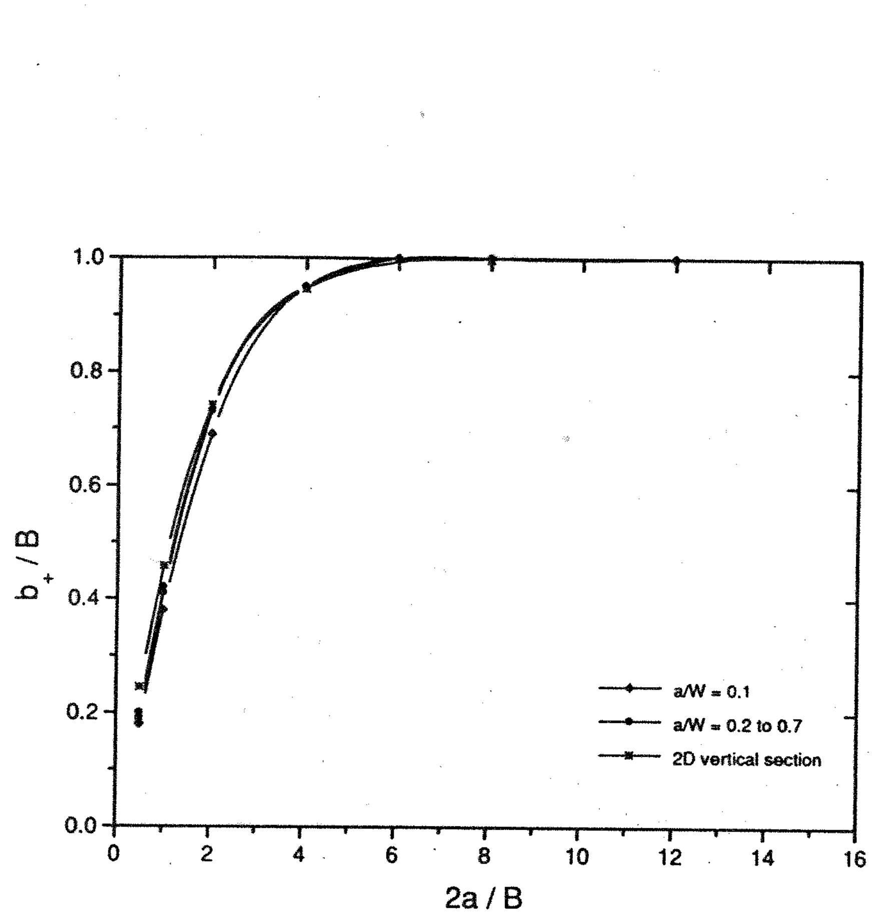


Figure 3.2.4 Depth of three-dimensional capture zone as a function of *2a/B*, for different values of *a/W*

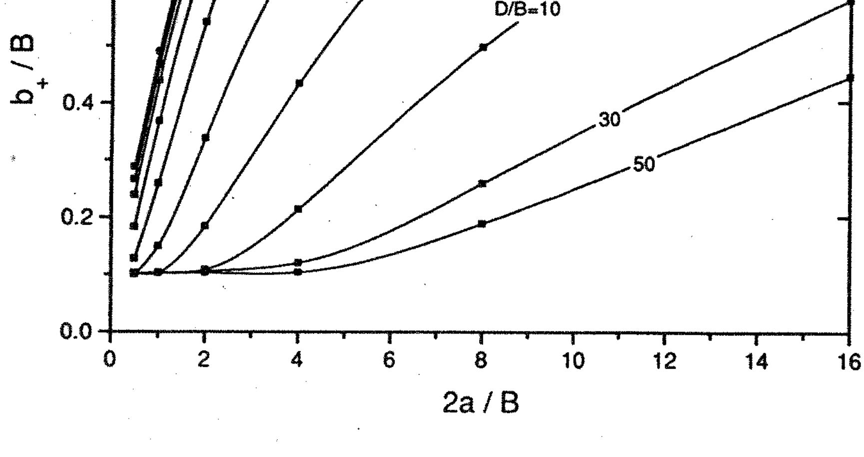


Figure 3.2.5 Depth of capture zone in a vertical section as a function of $2a/B$, with $U_-/U_+=0.9$, for different values of *D/B*

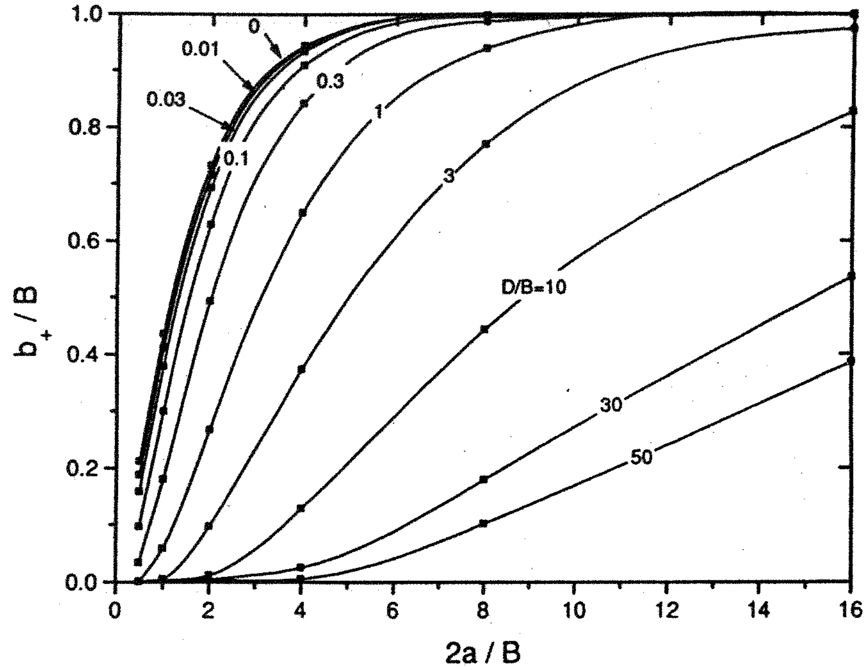


Figure 3.2.6 Depth of capture zone in a vertical section as a function of $2a/B$, with $U_-/U_+=1.1$, for different values of D/B

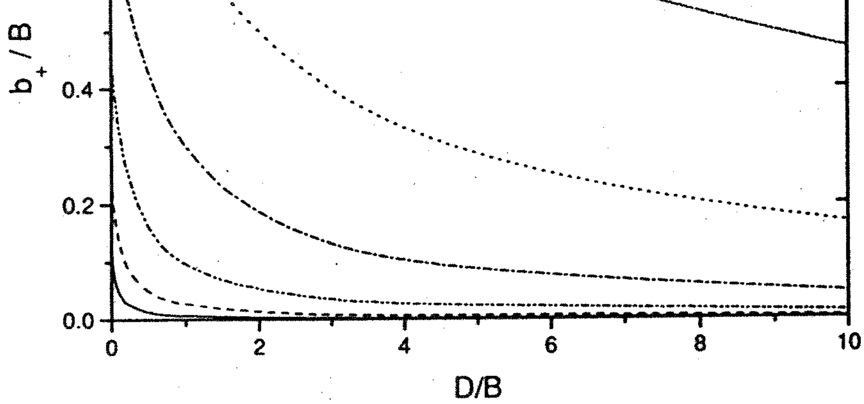


Figure 3.2.7 Depth of capture zone as a function of bottom resistance, for different $2a/B$ and K_x/K_z

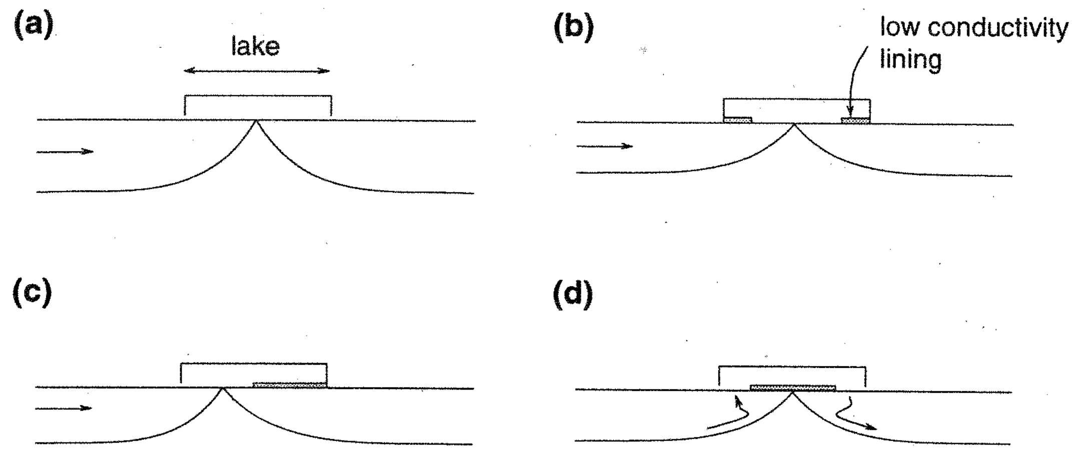


Figure 3.2.8 Possible flow patterns with non-uniform low conductivity linings

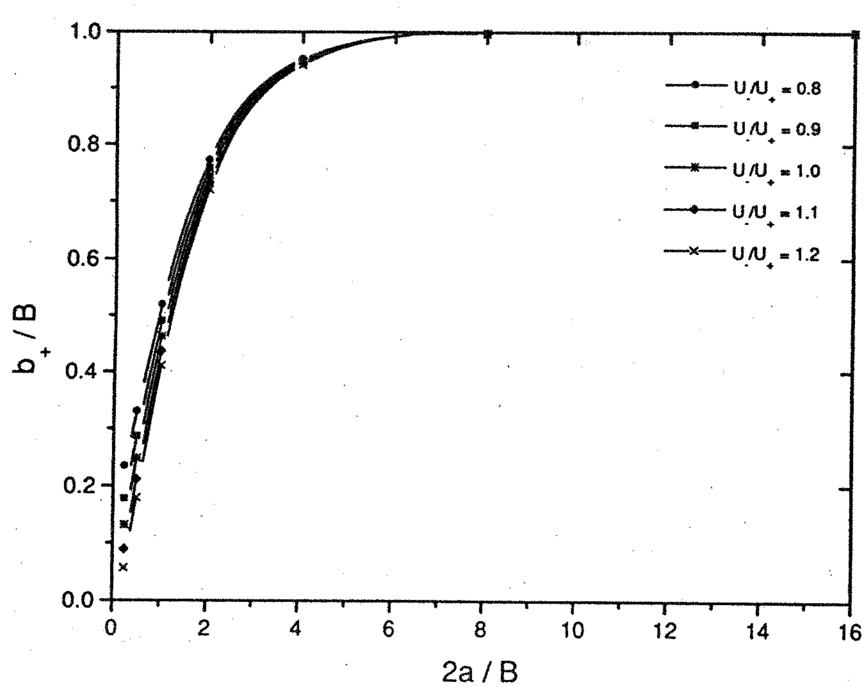


Figure 3.2.9 Depth of capture zone in a vertical section as a function of $2a/B$, for different values of U_-/U_+

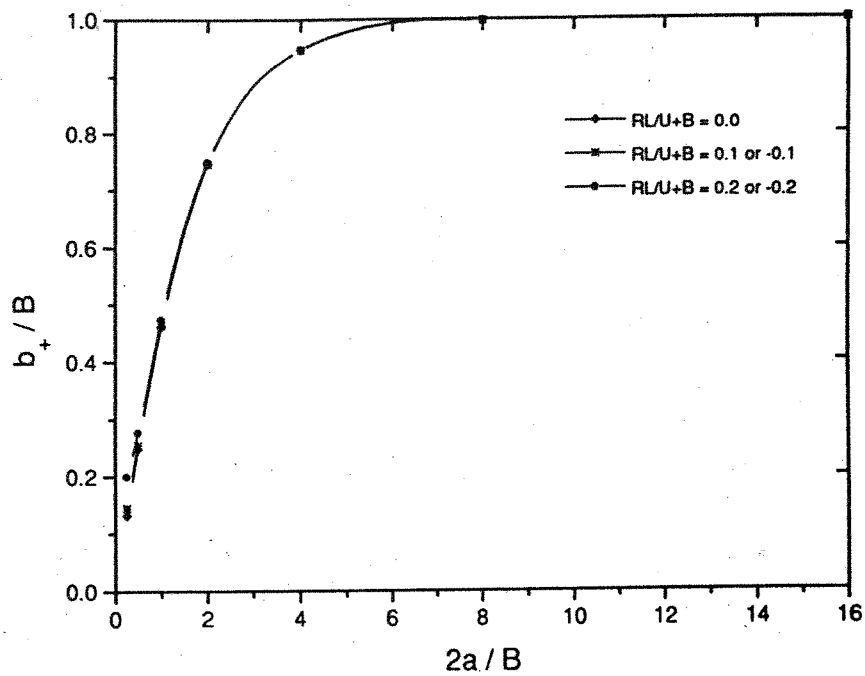


Figure 3.2.10 Depth of capture zone in a vertical section as a function of $2a/B$, for different values of RL/U_+B

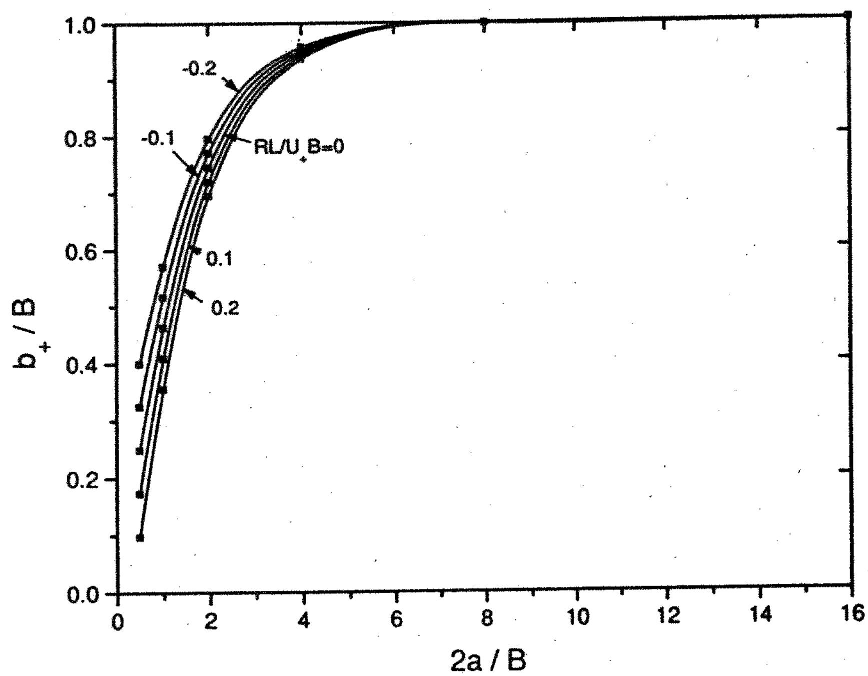


Figure 3.2.11 Depth of capture zone in a vertical section as a function of $2a/B$, with U_-/U_+ and RL/U_+B varying such that Q=0

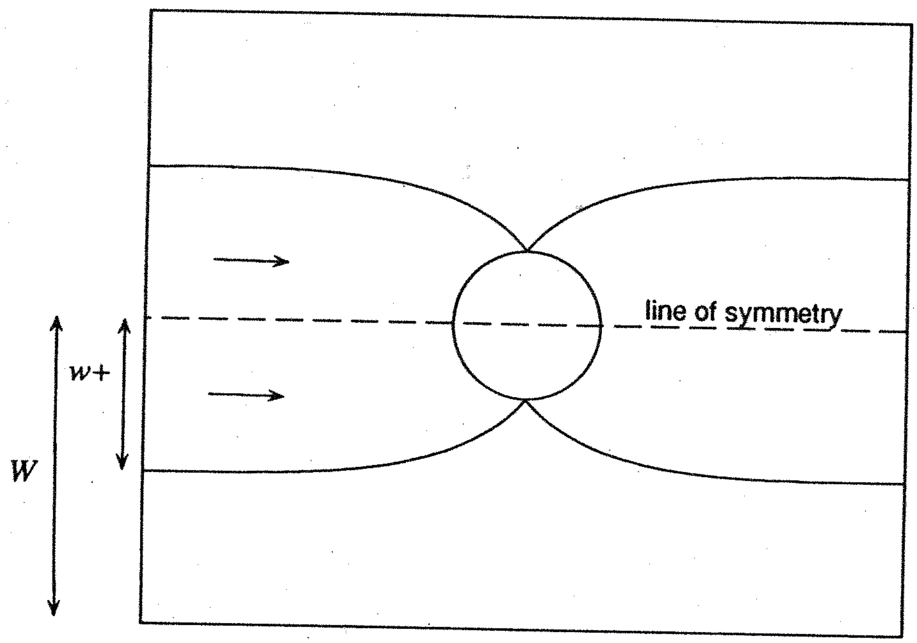


Figure 3.2.12 Definition of width of capture zone w^+

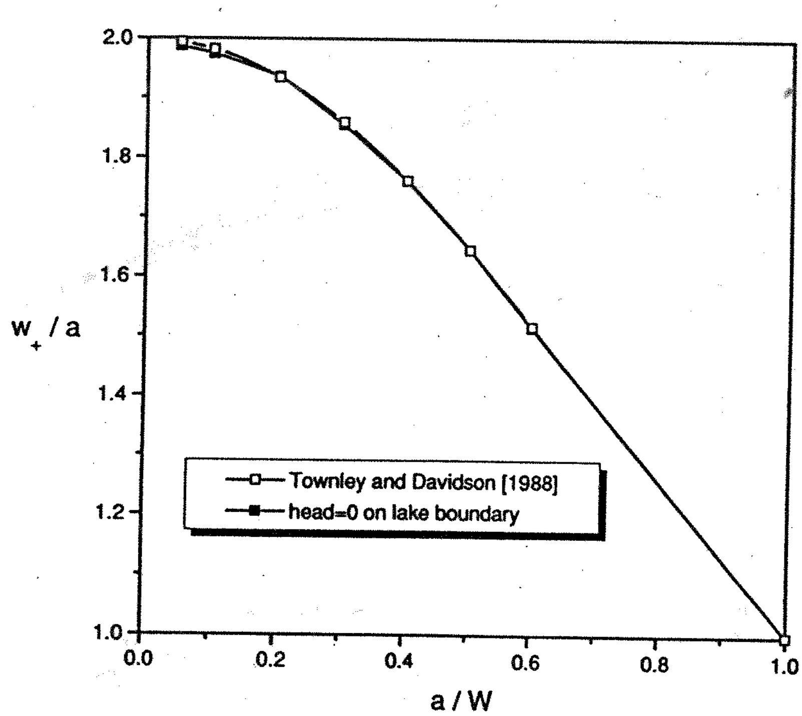


Figure 3.2.13 Width of capture zone for flow-through lakes with $R = 0$

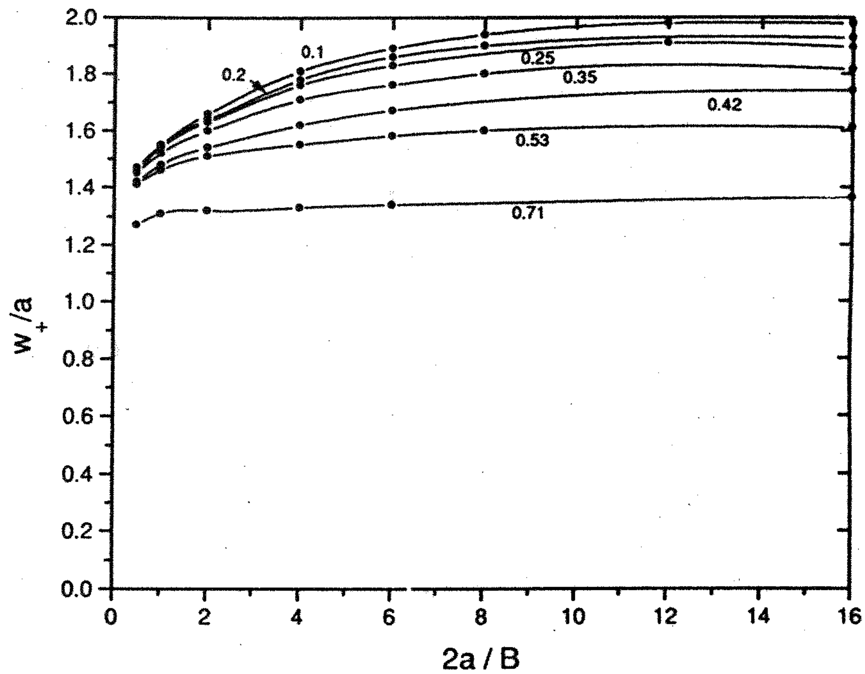


Figure 3.2.14 Width of three-dimensional capture zone as a function of $2a/B$ for different values of a/W

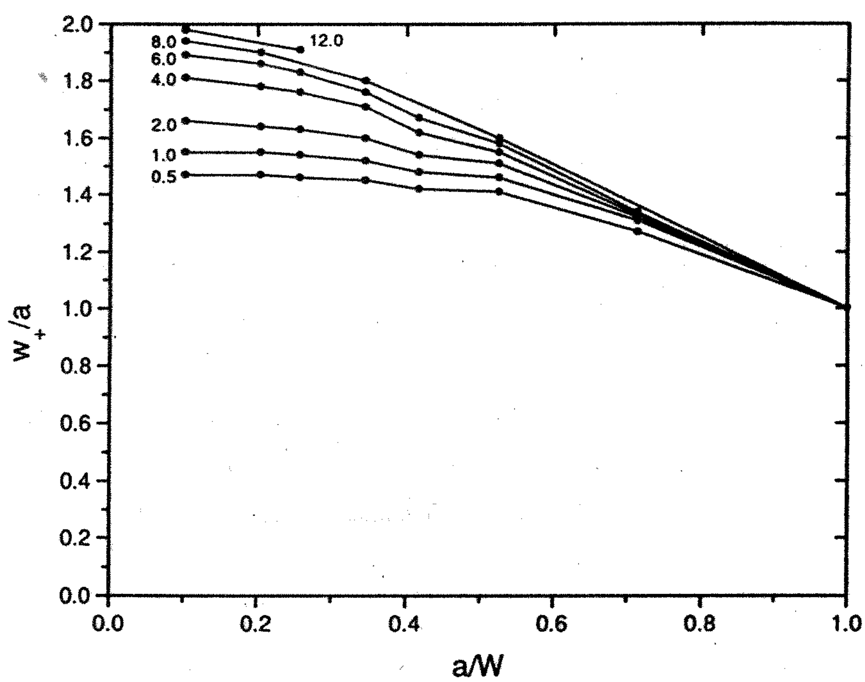


Figure 3.2.15 Width of three-dimensional capture zone as a function of a/W, for different $2a/B$

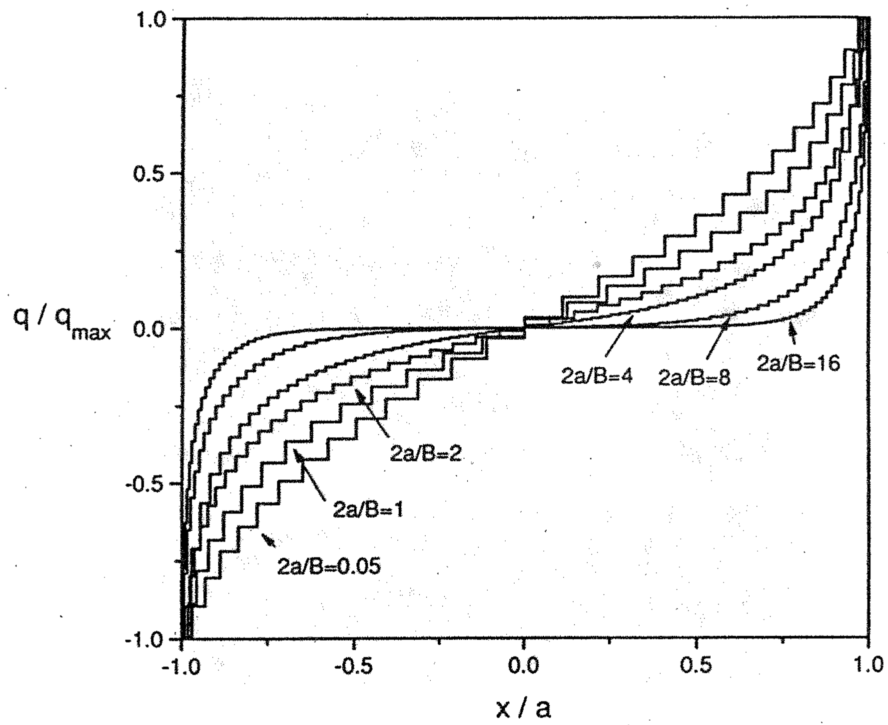


Figure 3.3.1 Effect of lake length $2a/B$ on the spatial distribution of seepage. Curves show seepage, normalised relative to maximum seepage at the edge of the water body, with $U_-/U_+ = 1$, $RL/U_+B = 0$, $D/B = 0$, and a range of values of $2a/B$

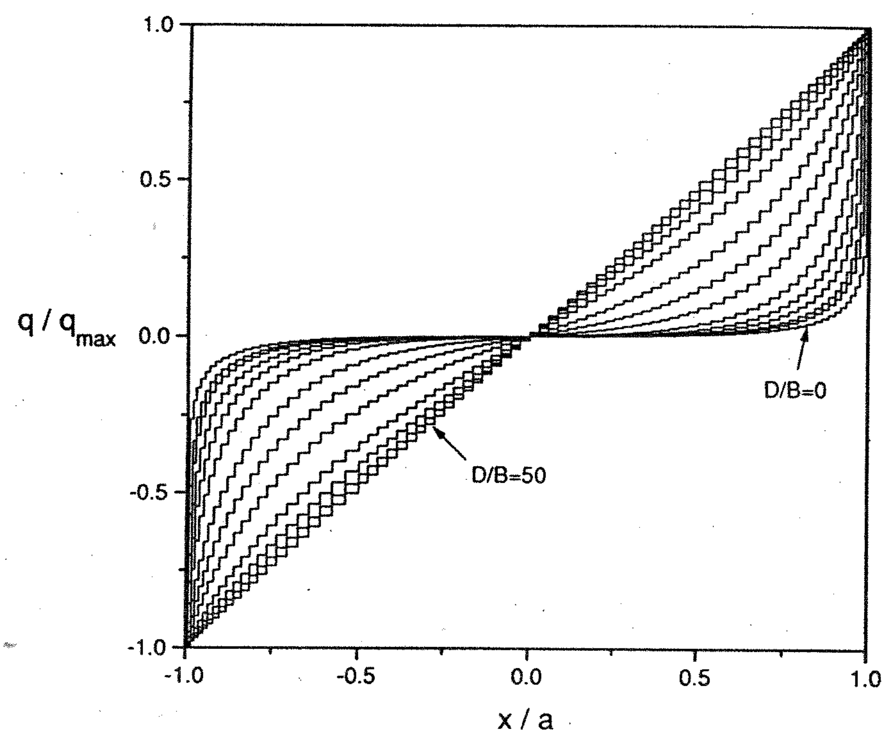


Figure 3.3.2 Effect of resistance of lake lining on the spatial distribution of seepage. Curves show seepage, normalised relative to maximum seepage at the edge of the water body, with $2a/B = 8$, $U_-/U_+ = 1$, $RL/U_+B = 0$ and a range of values of D/B

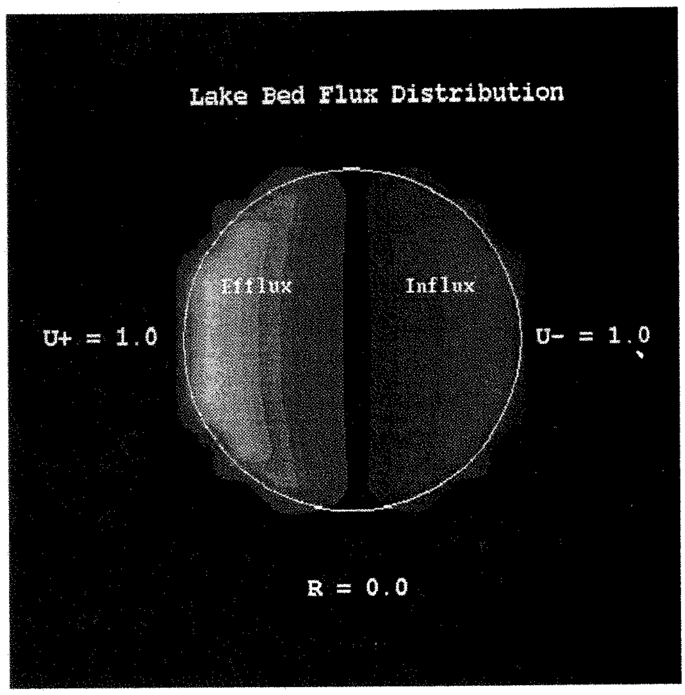


Figure 3.3.3 Spatial distribution of bottom seepage for a circular flow-through lake

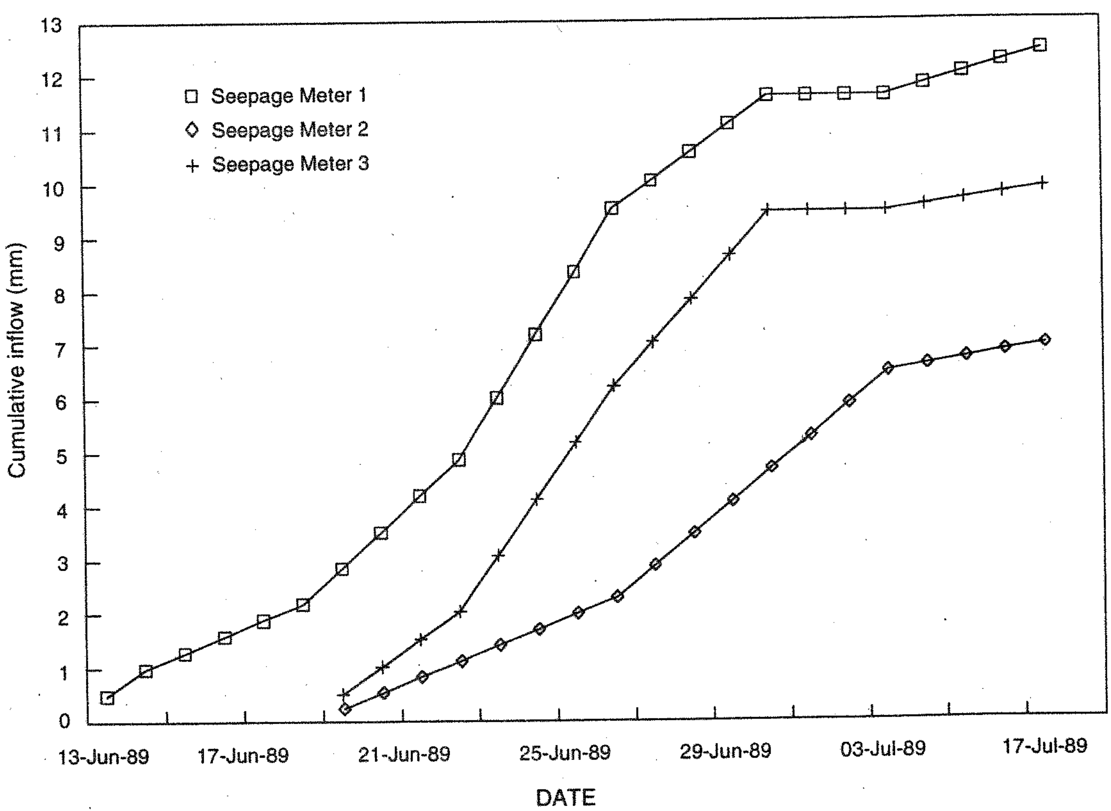


Figure 3.3.4 Cumulative groundwater seepage at Nowergup Lake

Table 3.4.1 Comparison of predicted and observed release zone depths

Lake	Length $2a$ (m)	Aquifer thickness B (m)	Actual $2a/B$	Predicted $\frac{b_-}{B}$	Observed $\frac{b_-}{B}$	Implied $2a/B^*$	Implied K_x/K_z
Nowergup Lake	500	50	10	1	0.6	1.5	45
Mariginiup Lake	800	55	14.5	1	< 0.5	< 1.2	>140
Jandabup Lake	1600	55	29	1	> 0.95	> 4	< 50
Thomsons Lake	1600	40	40	1	> 0.95	> 4	<100

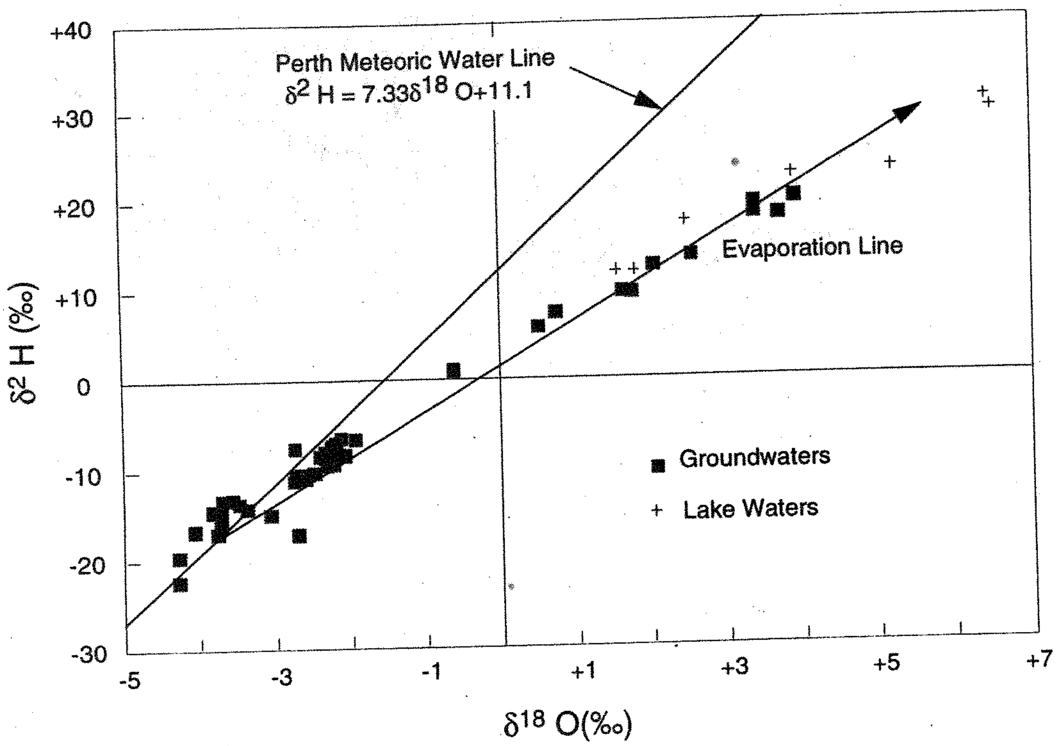


Figure 3.4.1 Stable isotope results at Nowergup Lake in 1989/90

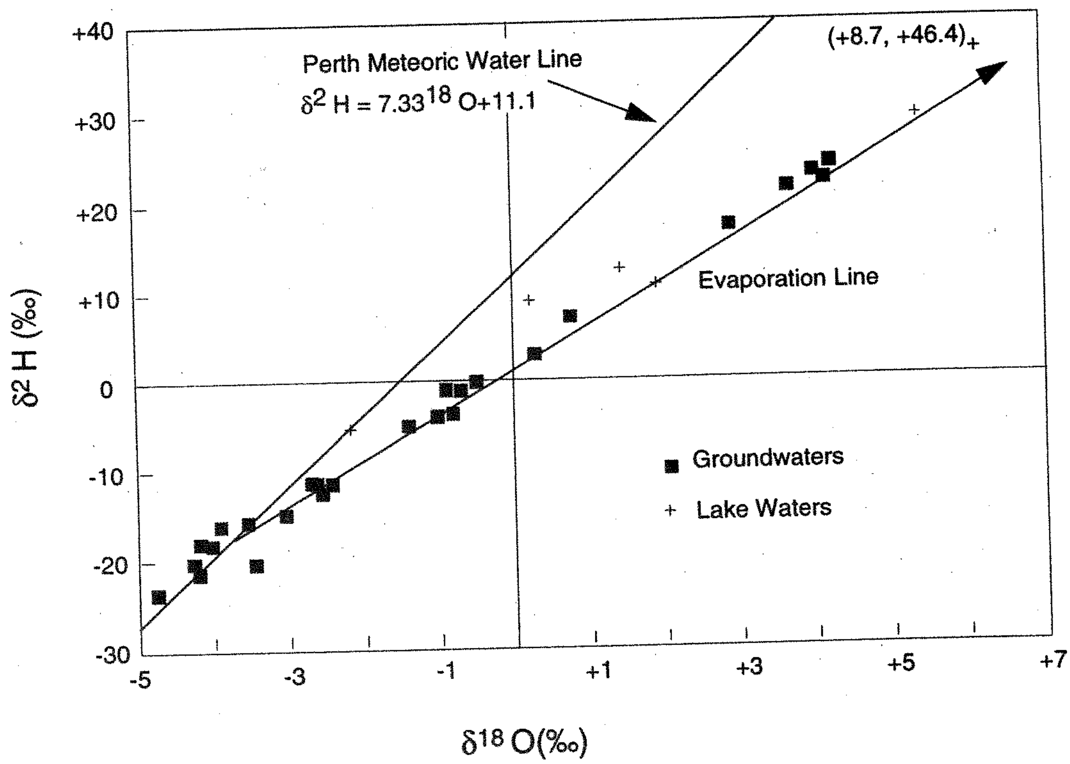


Figure 3.4.2 Stable isotope results at Jandabup Lake in 1989/90

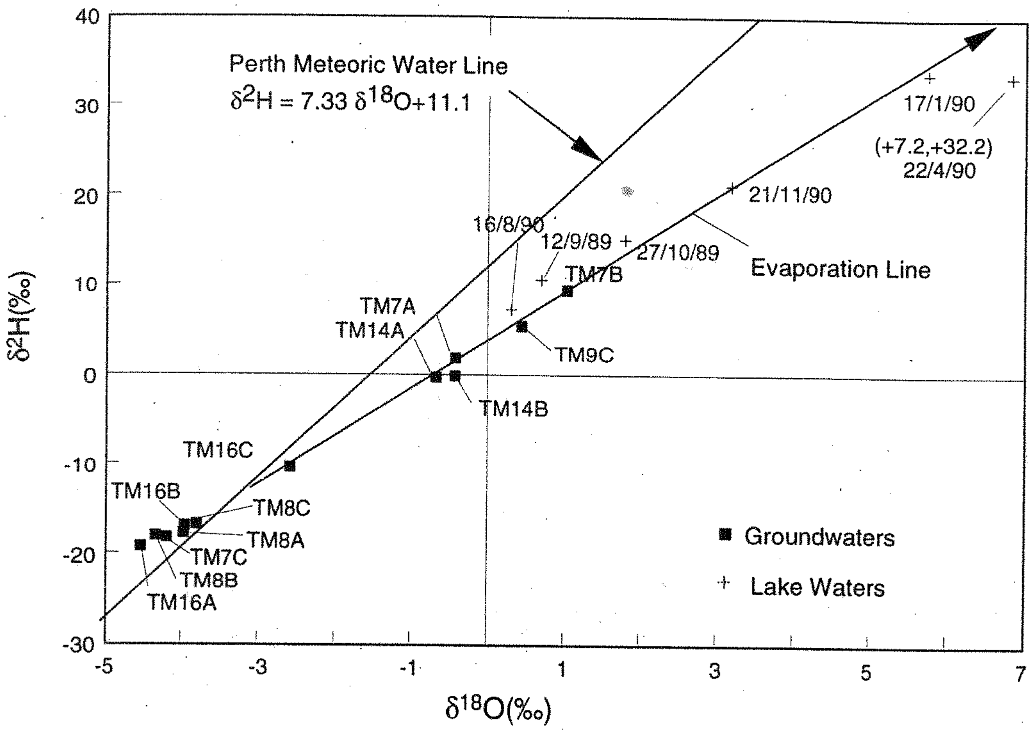


Figure 3.4.3 Stable isotope results at Thomsons Lake in 1989/90

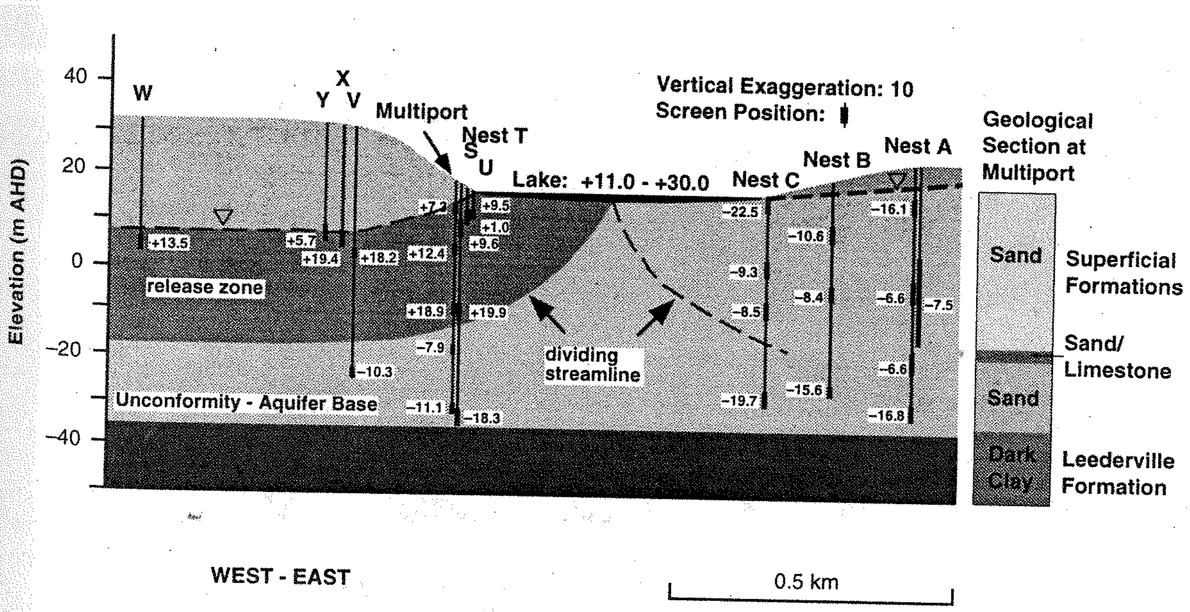


Figure 3.4.4 Deuterium concentrations (‰) in a generalised cross-section through Nowergup Lake

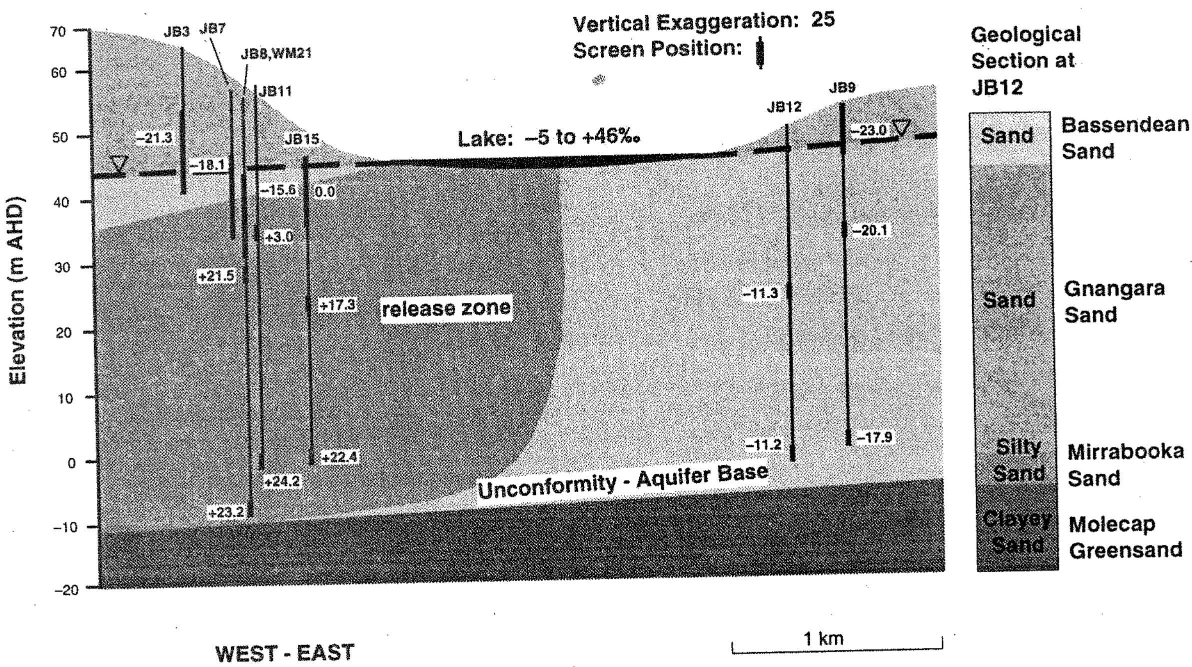


Figure 3.4.5 Deuterium concentrations (‰) in a generalised cross-section through Jandabup Lake

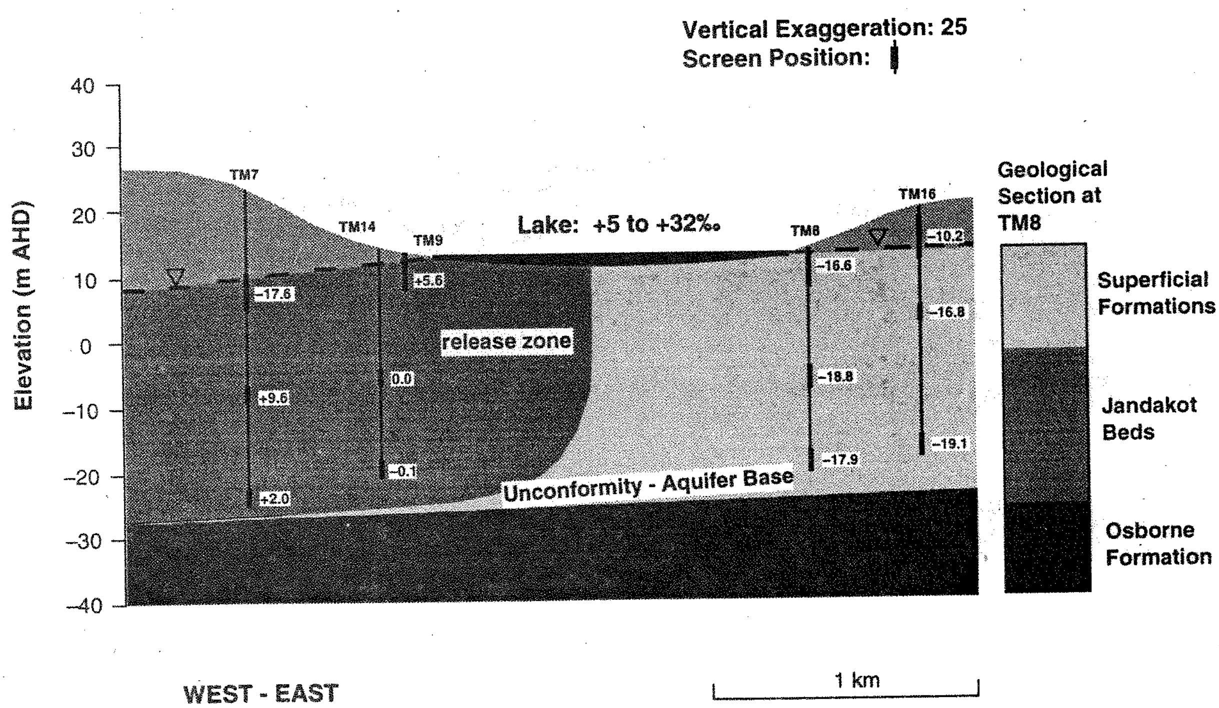


Figure 3.4.6 Deuterium concentrations (‰) in a generalised cross-section through Thomsons Lake

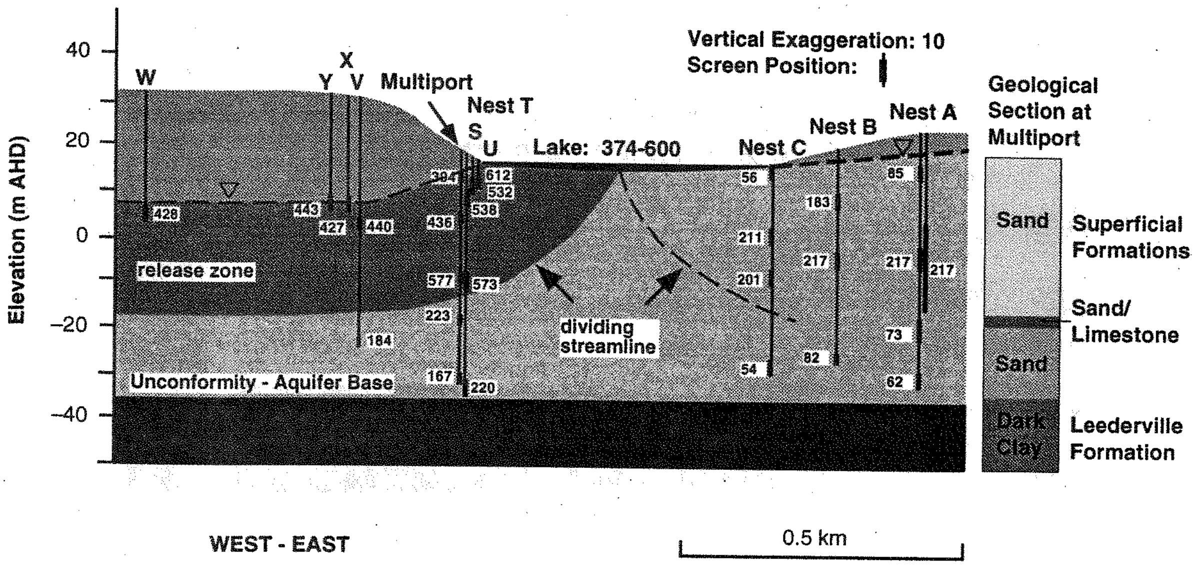


Figure 3.4.7 Chloride concentrations (mgL^{-1}) in a generalised cross-section through Nowergup Lake

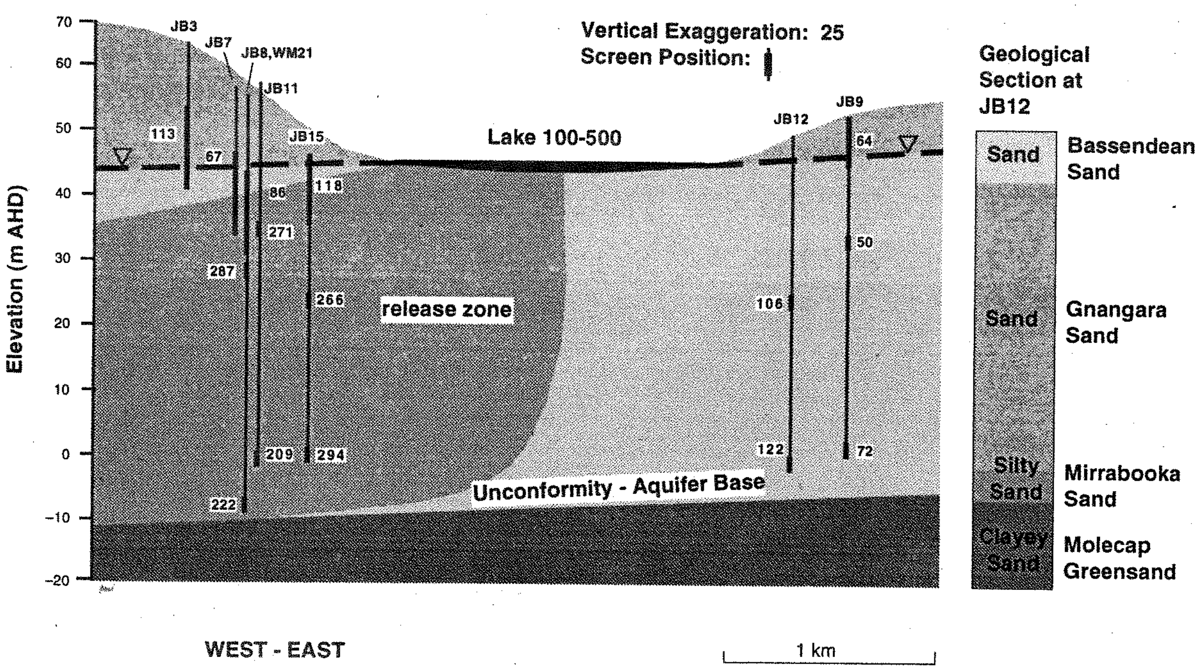


Figure 3.4.8 Chloride concentrations (mgL^{-1}) in a generalised cross-section through Jandabup Lake

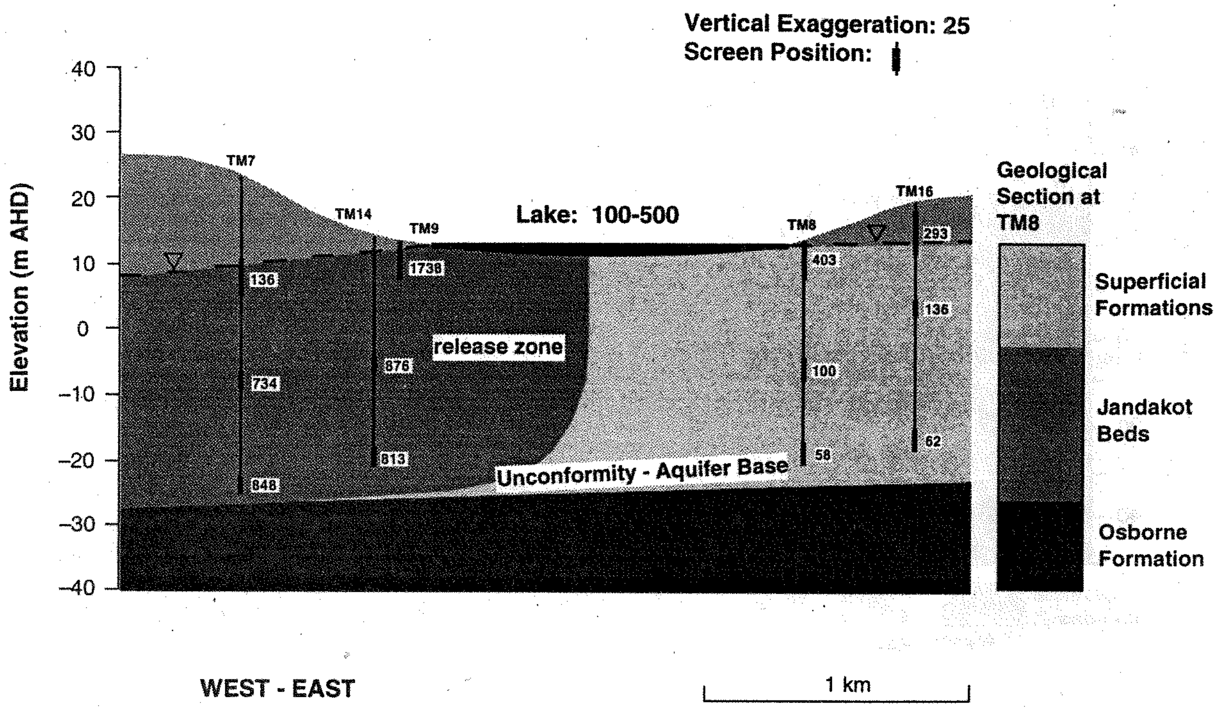


Figure 3.4.9 Chloride concentrations (mgL^{-1}) in a generalised cross-section through Thomsons Lake

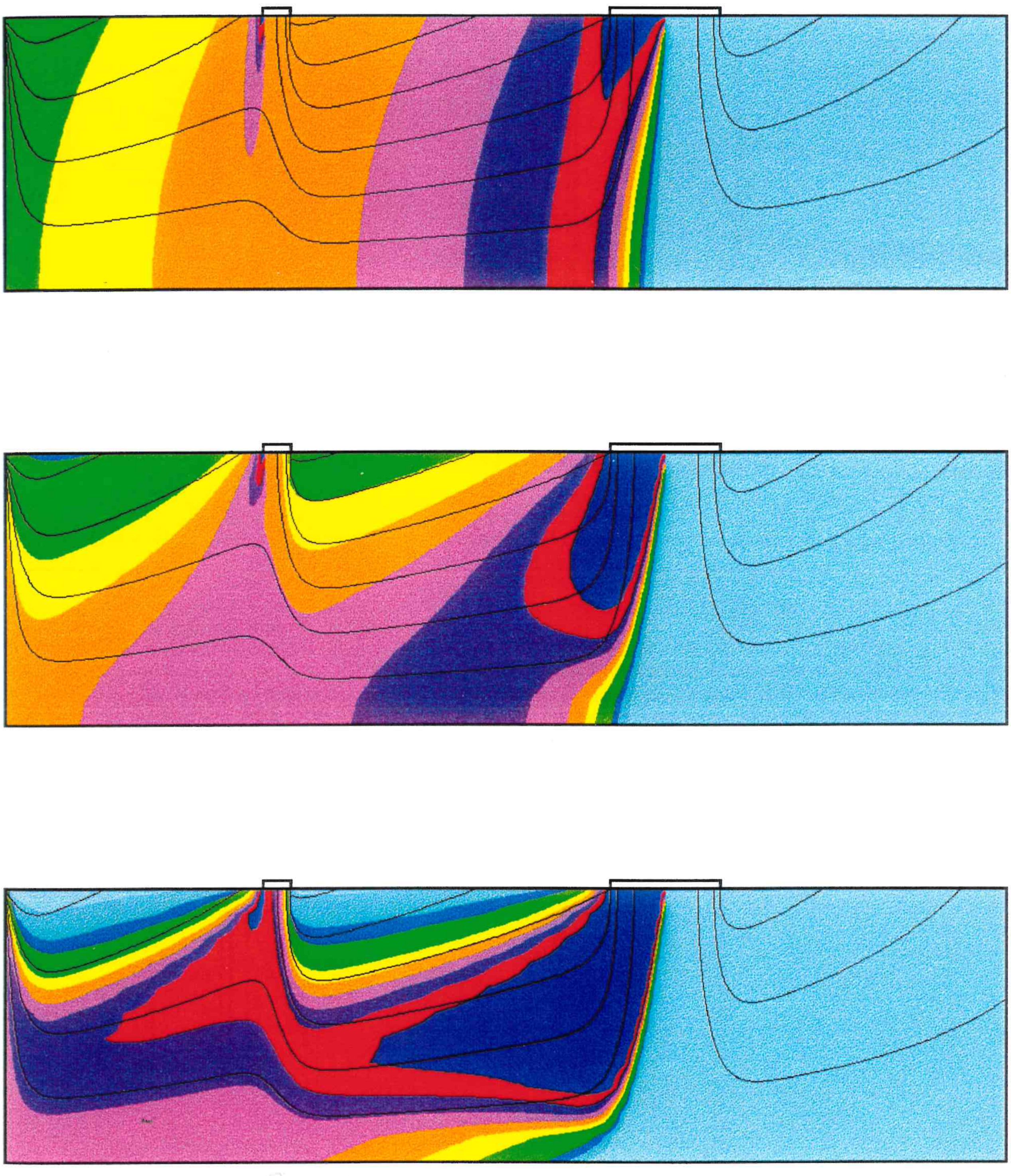


Plate 3.5.1 Streamfunction (black) with R = 0.0004md^{-1}, K_x = 20md^{-1} and K_z = 0.2md^{-1}, and concentration (in colour) with a_L = 20m and (a) a_T = 2m, (b) a_T = 0.2m and (c) a_T = 0.02m (flow from right to left past Lake Pinjar, then Nowergup Lake; length 18 km, depth 70 m)

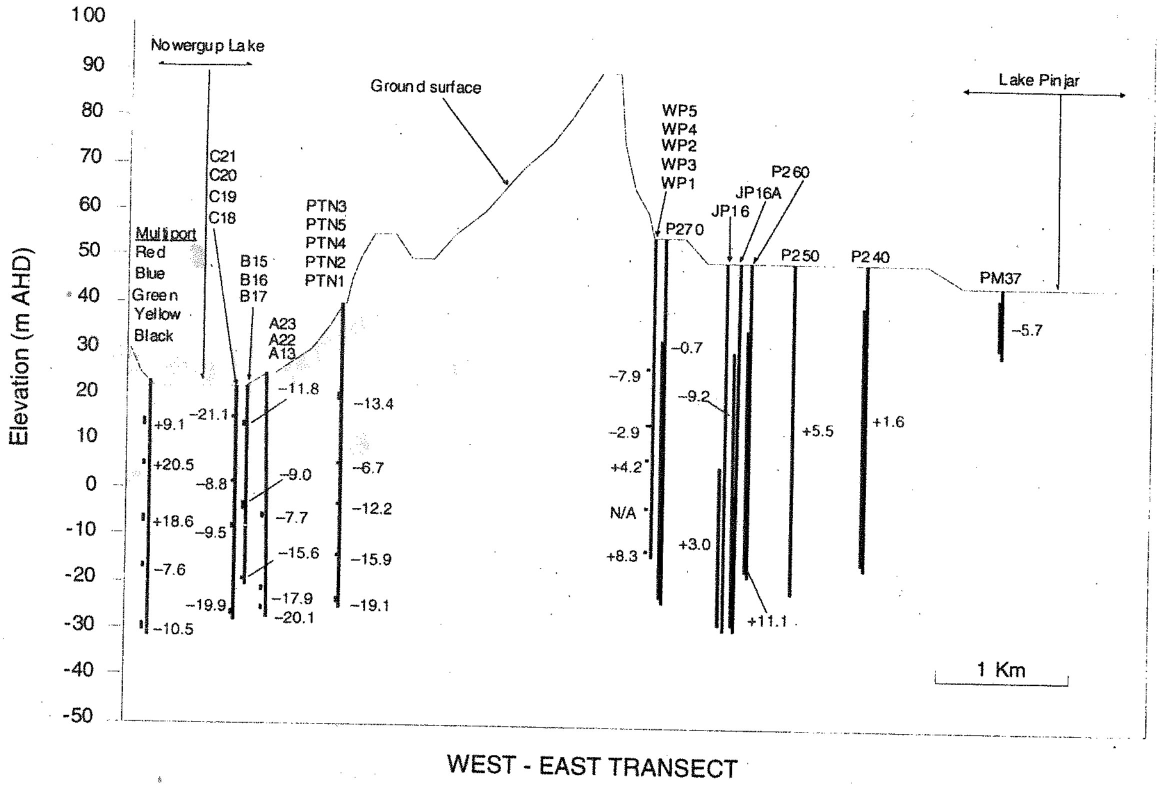


Figure 3.5.1 Deuterium compositions (‰) in a generalised cross-section (B-B') along the Pinjar-Nowergup transect

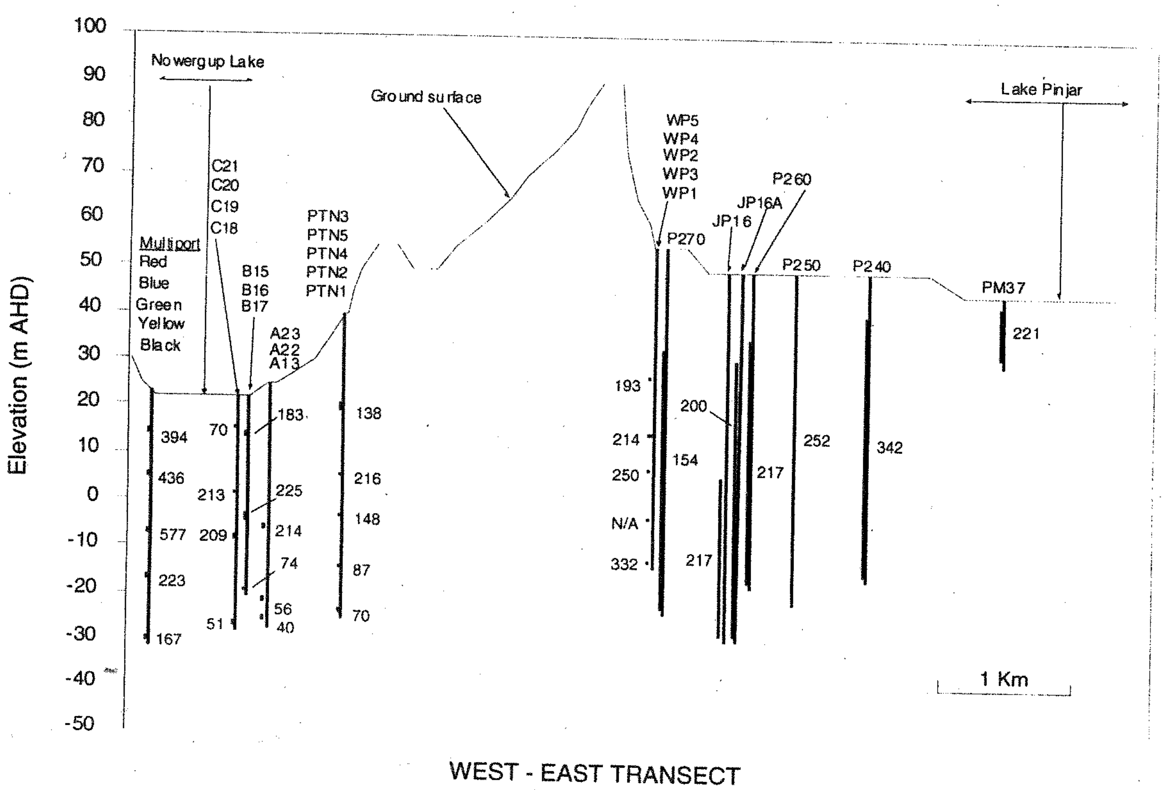


Figure 3.5.2 Chloride concentration (mgL^{-1}) in a generalised cross-section (B-B') along the Pinjar-Nowergup transect

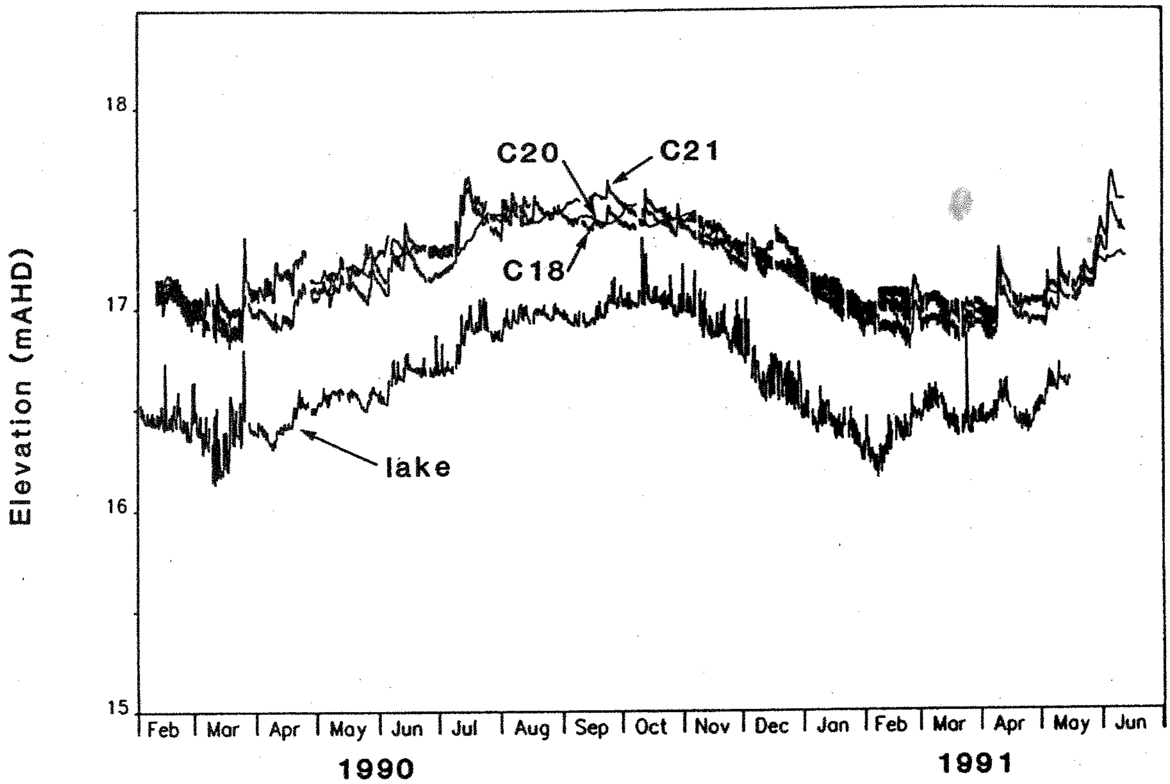


Figure 3.7.1 Lake level and piezometric heads in a nest of three bores on the eastern side of Nowergup Lake

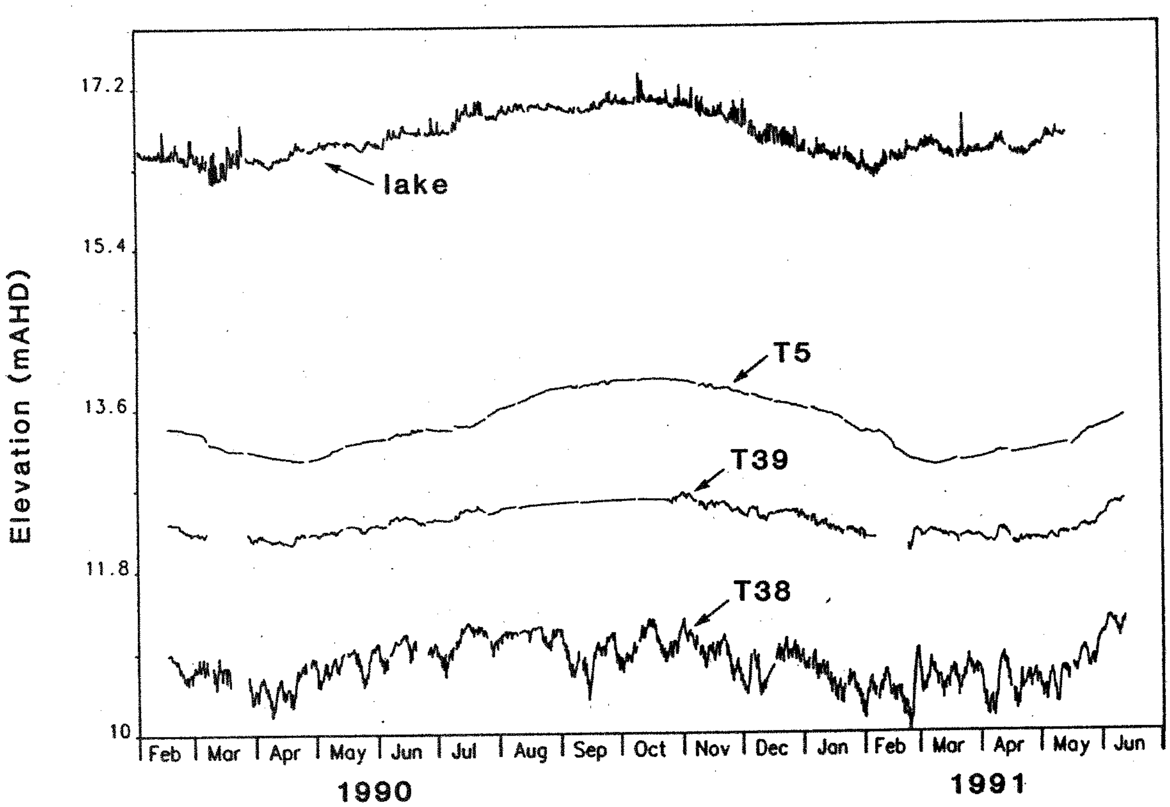


Figure 3.7.2 Lake level and piezometric heads in a nest of three bores on the western side of Nowergup Lake

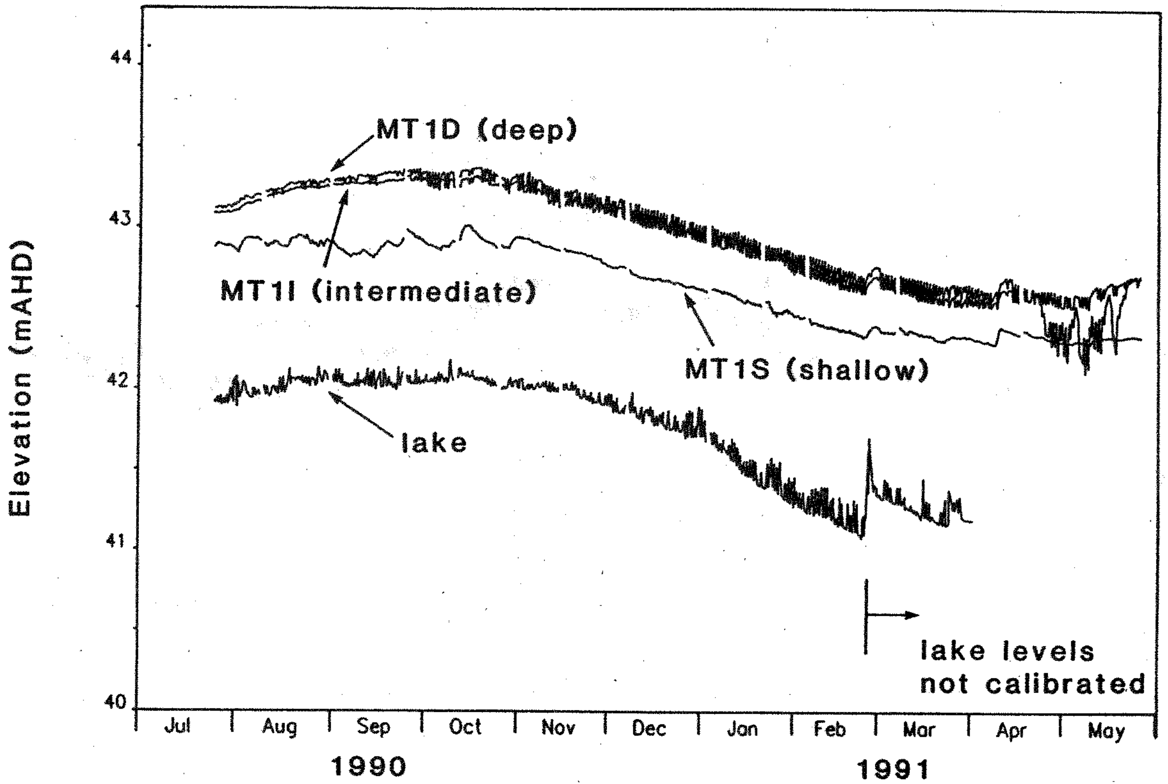


Figure 3.7.3 Lake level and piezometric heads in a nest of three piezometers on the eastern side of Mariginiup Lake

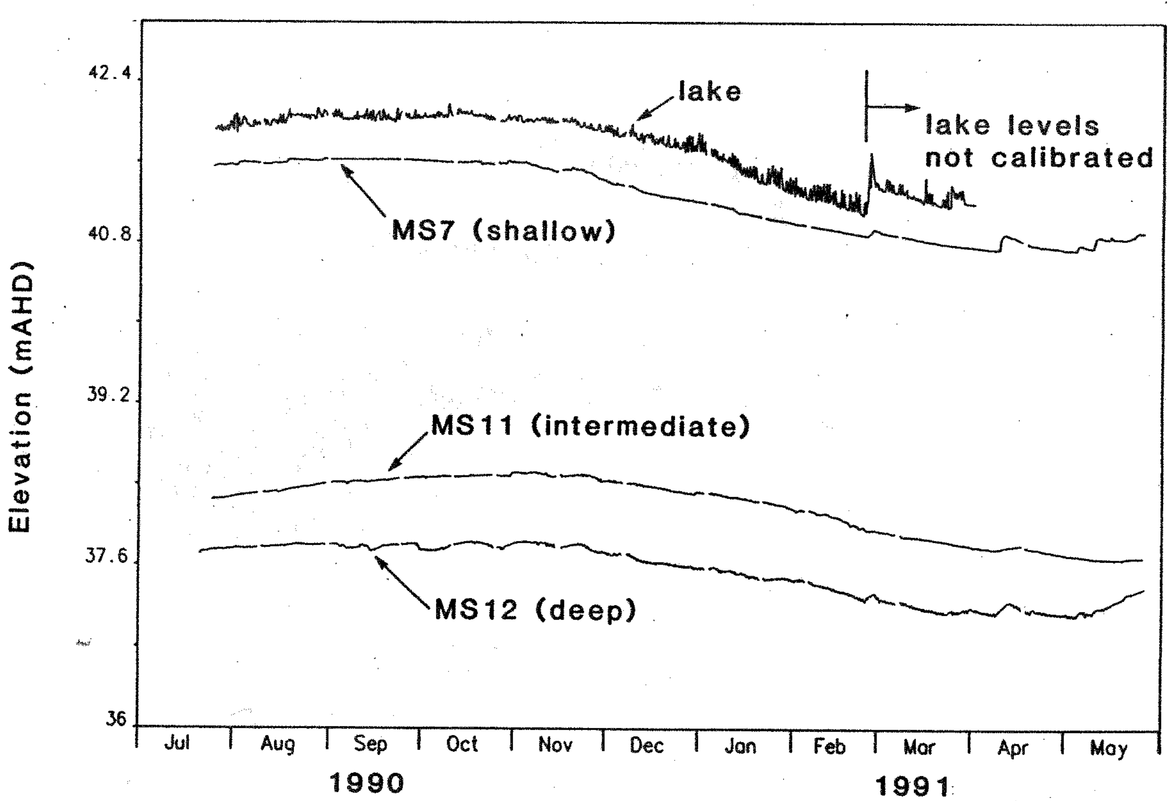


Figure 3.7.4 Lake level and piezometric heads in three boreholes on the western side of Mariginiup Lake

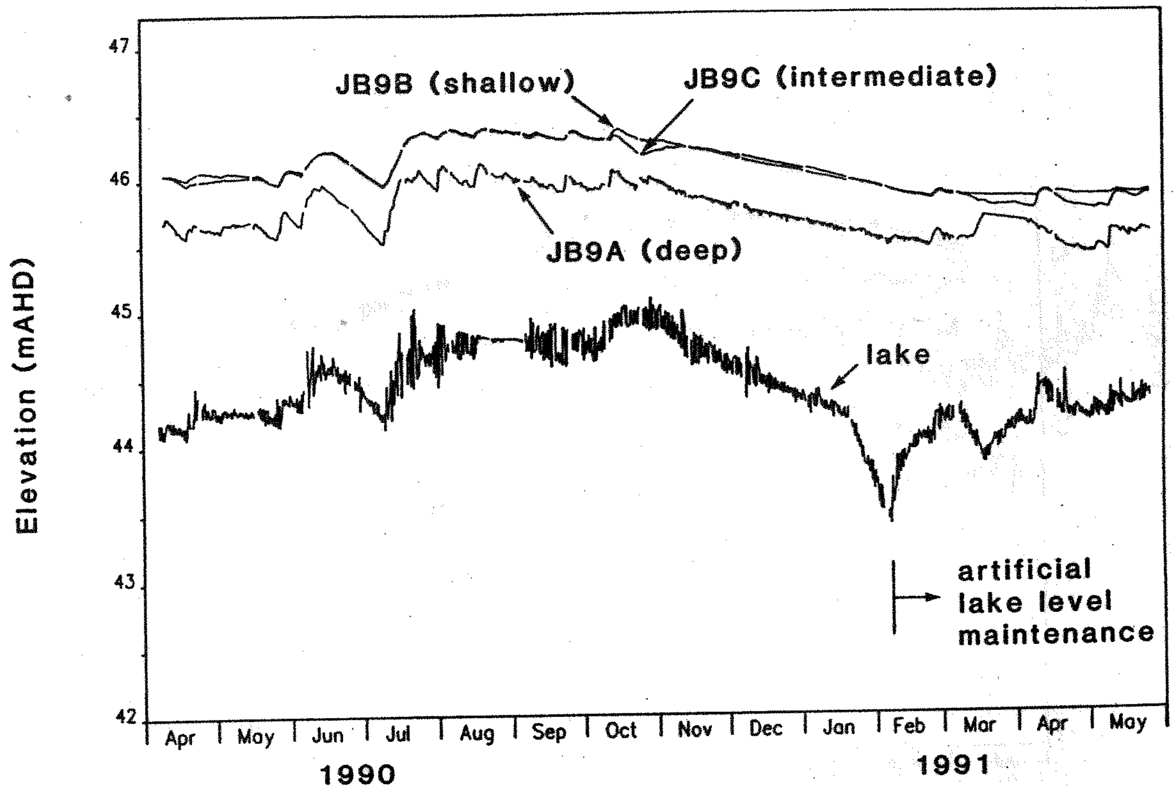


Figure 3.7.5 Lake level and piezometric heads in a nest of three piezometers on the eastern side of Jandabup Lake

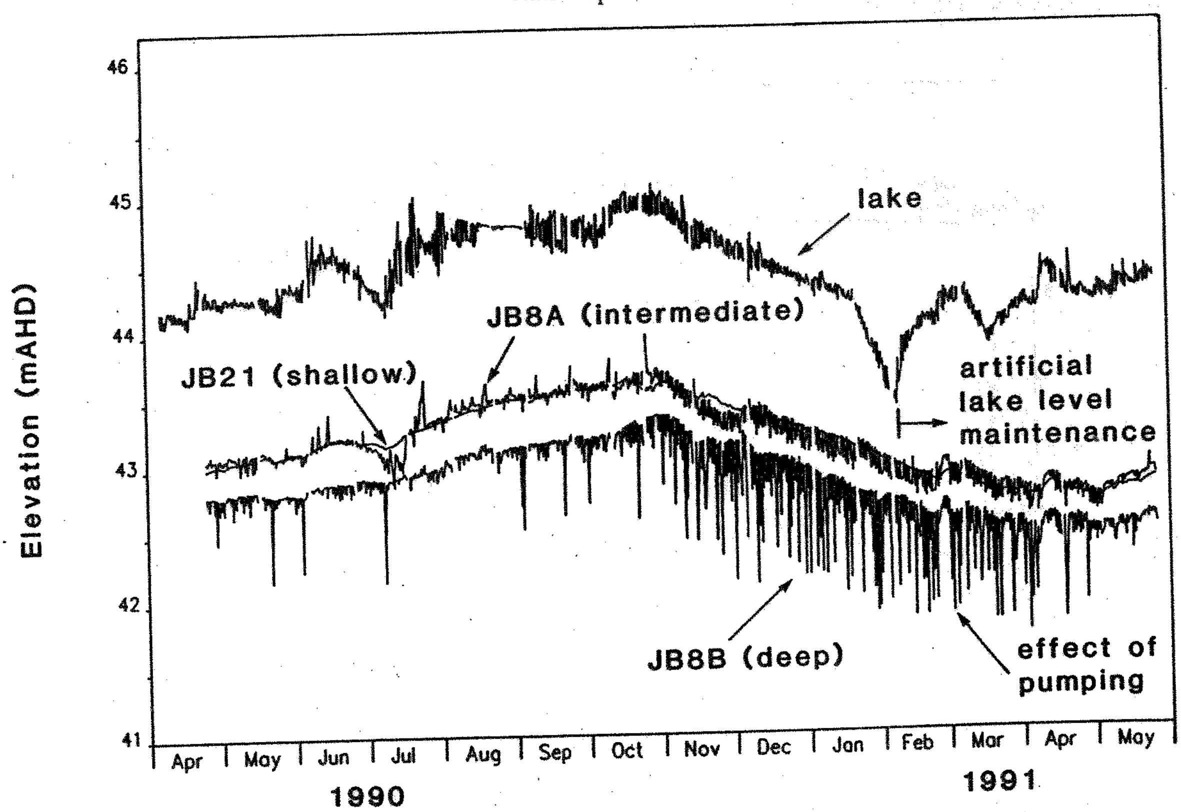


Figure 3.7.6 Lake level and piezometric head in a nest of three piezometers on the western side of Jandabup Lake

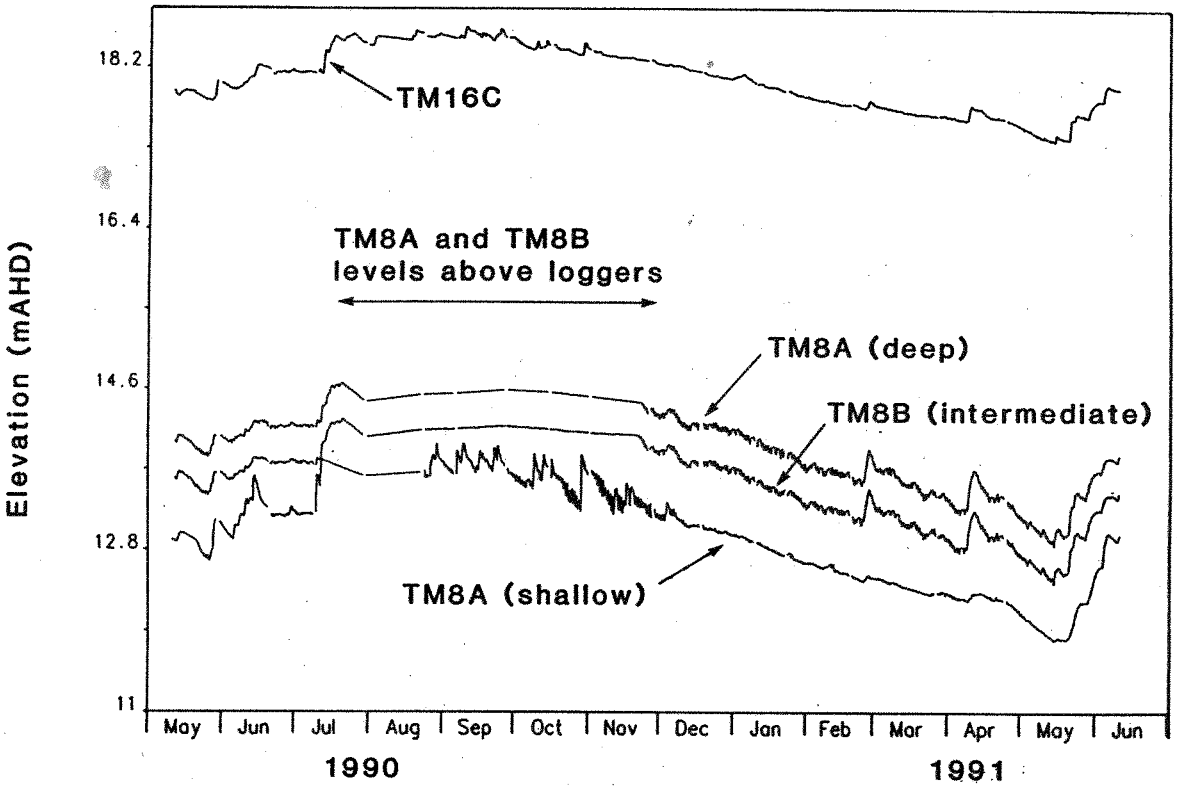


Figure 3.7.7 Piezometric heads on the eastern side of Thomsons Lake

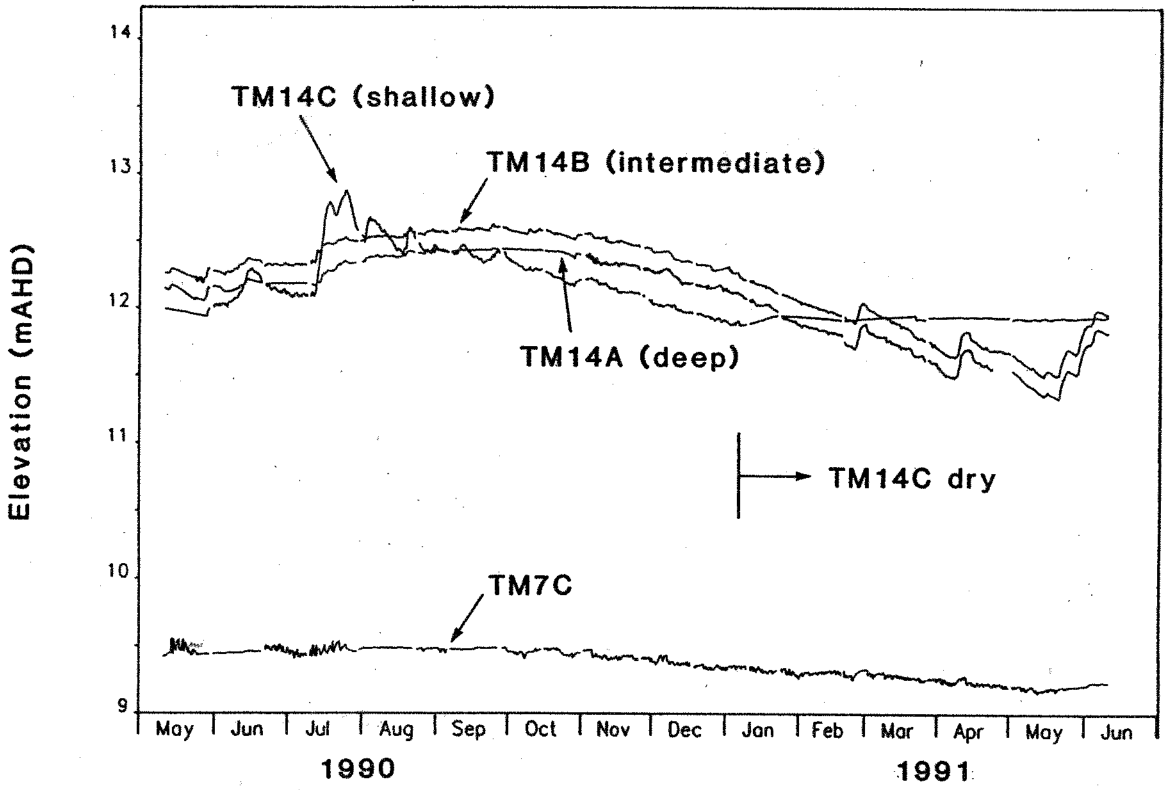


Figure 3.7.8 Piezometric heads on the western side of Thomsons Lake

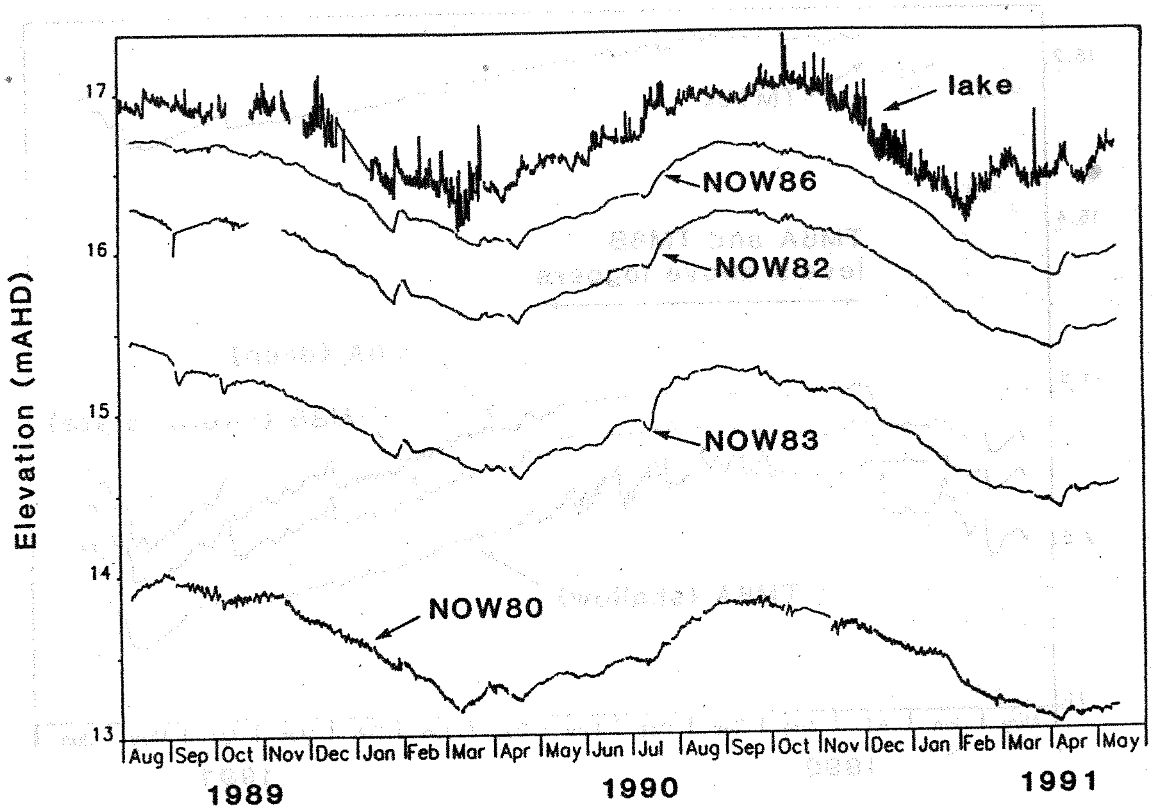


Figure 3.7.9 Lake level and groundwater levels at four shallow piezometers at northern end of Nowergup Lake

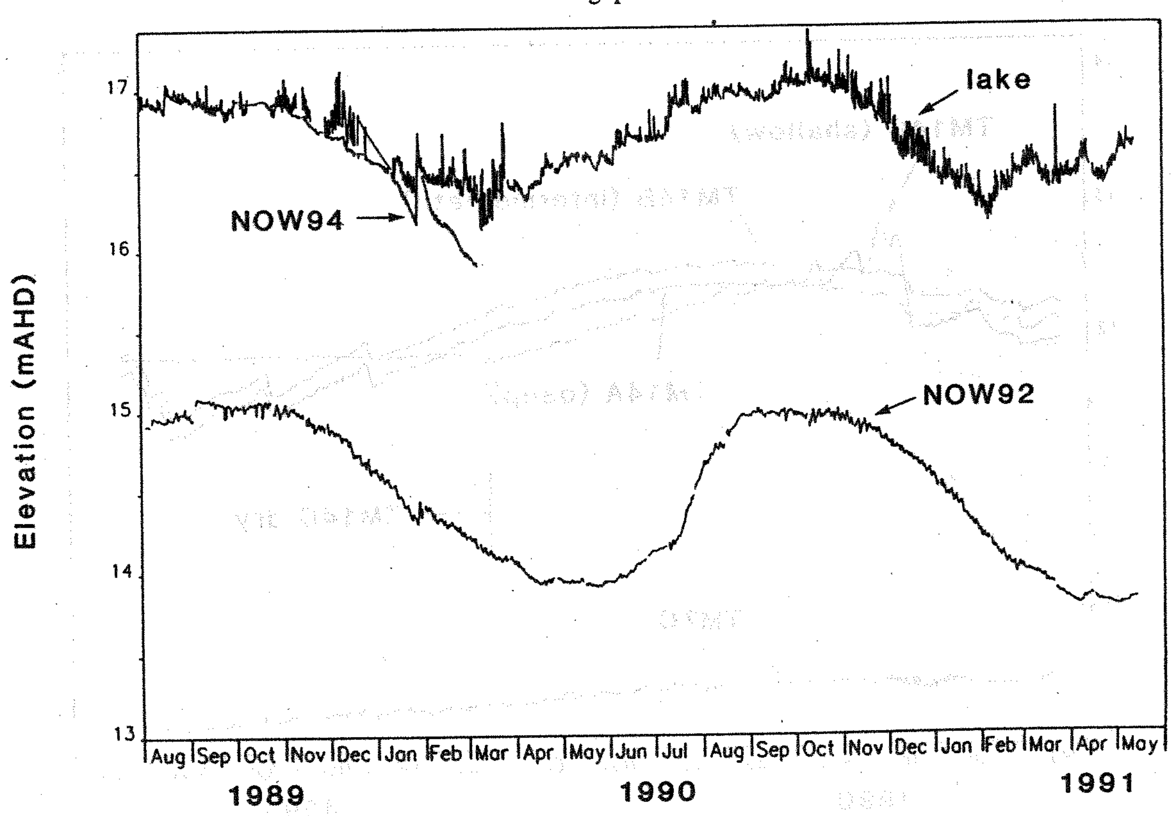


Figure 3.7.10 Lake level and groundwater levels at two shallow piezometers at southern end of Nowergup Lake

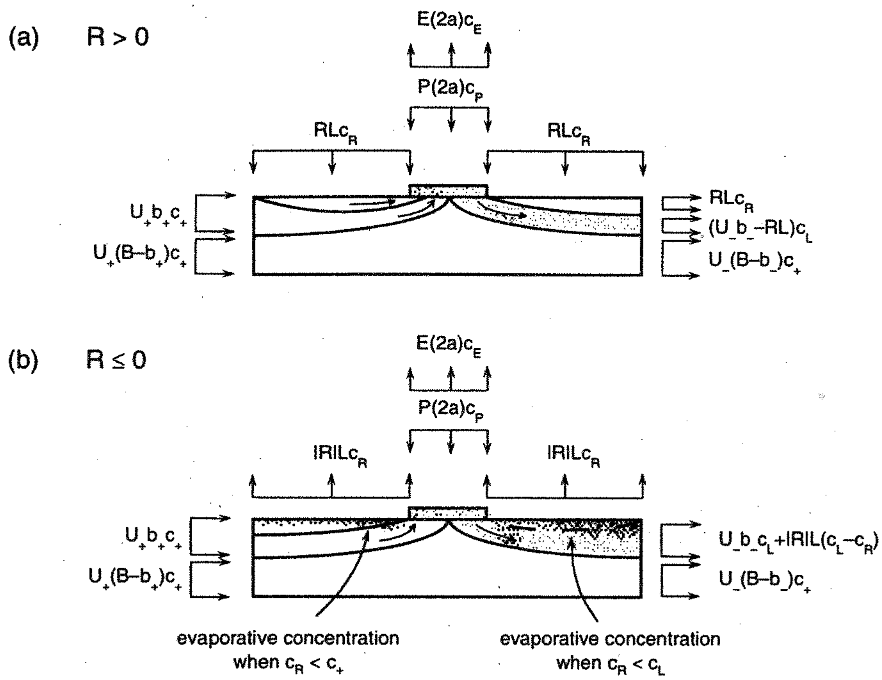


Figure 3.8.1 Solute balances for flow-through lakes

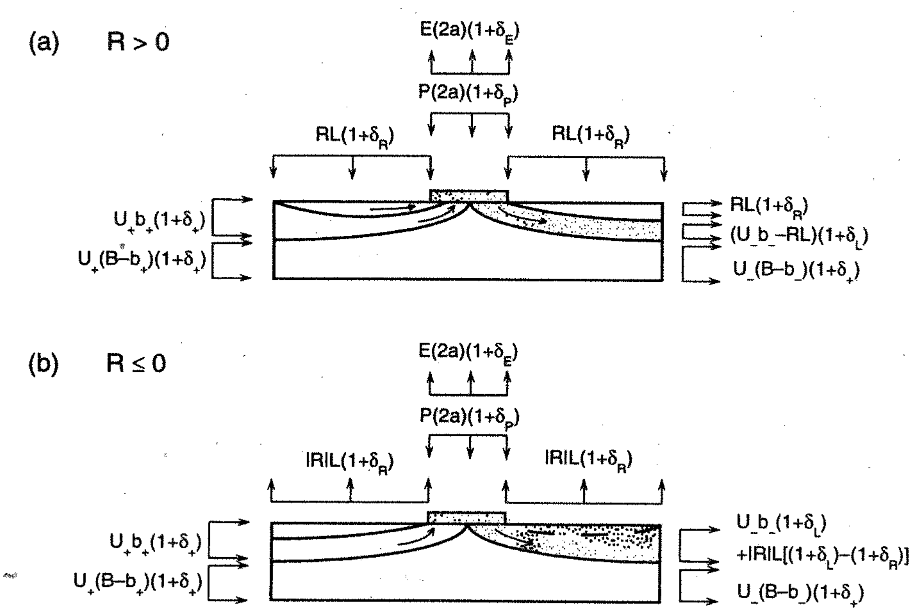


Figure 3.8.2 Isotope balances for flow-through lakes

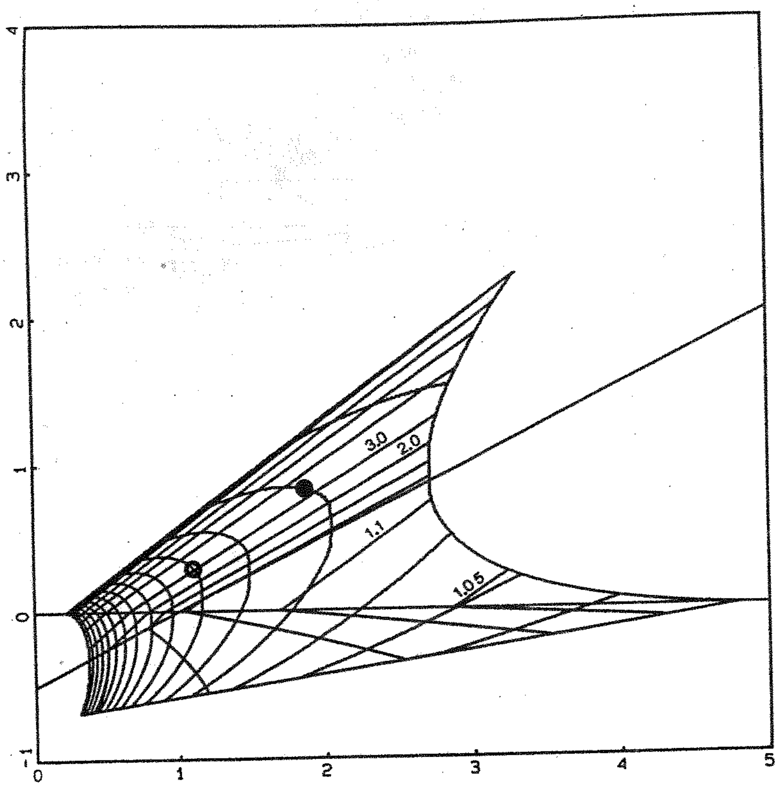


Figure 3.8.3 Flow-through domain for $2a/B$=1, showing contours of b_-/B and c_L/c_+ when $P(2a)/U_+B$=0.1 above and $E(2a)/U_+B$=0.1 below the Q=0 line, with data from Nowergup Lake superimposed

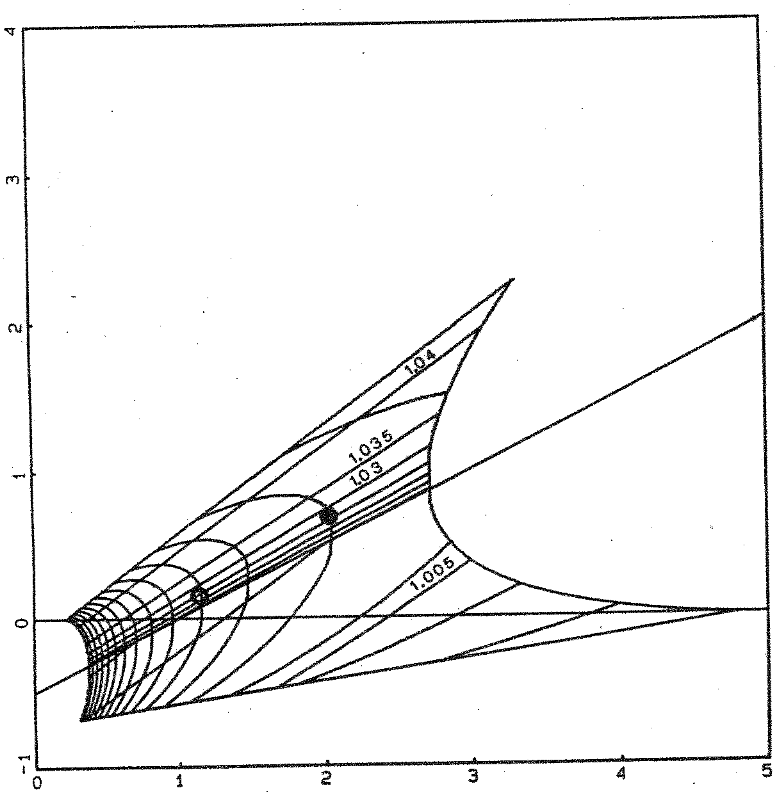


Figure 3.8.4 Flow-through domain for $2a/B$=1, showing contours of b_-/B and $(1+\delta_L)/(1+\delta_+)$ when $P(2a)/U_+B$=0.1 above and $E(2a)/U_+B$=0.1 below the Q=0 line, with data from Nowergup Lake superimposed

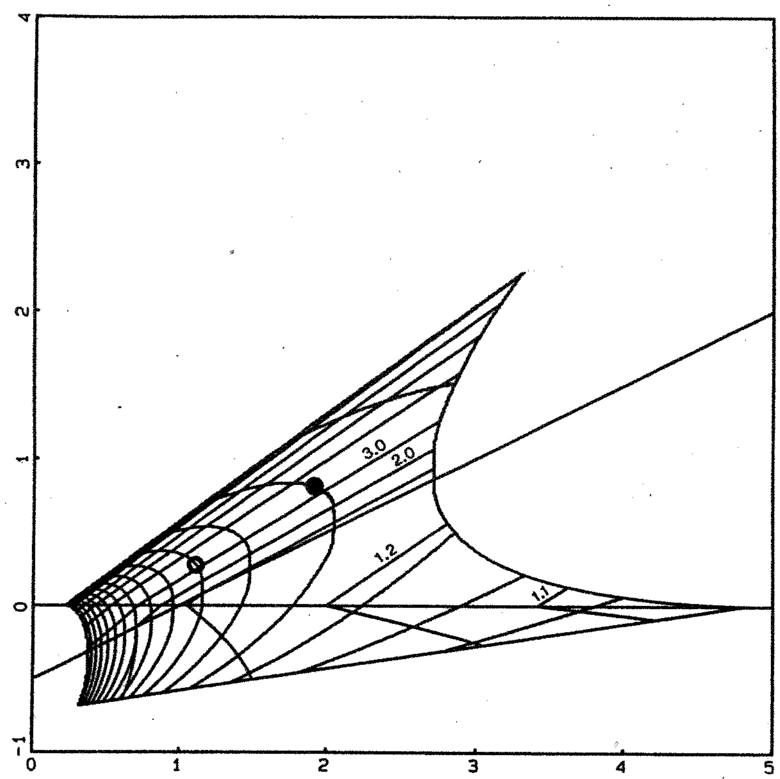


Figure 3.8.5 Flow-through domain for $2a/B$=1, showing contours of b_-/B and c_L/c_+ when $P(2a)/U_+B$=0.25 above and $E(2a)/U_+B$=0.25 below the Q=0 line, with data from Nowergup Lake superimposed

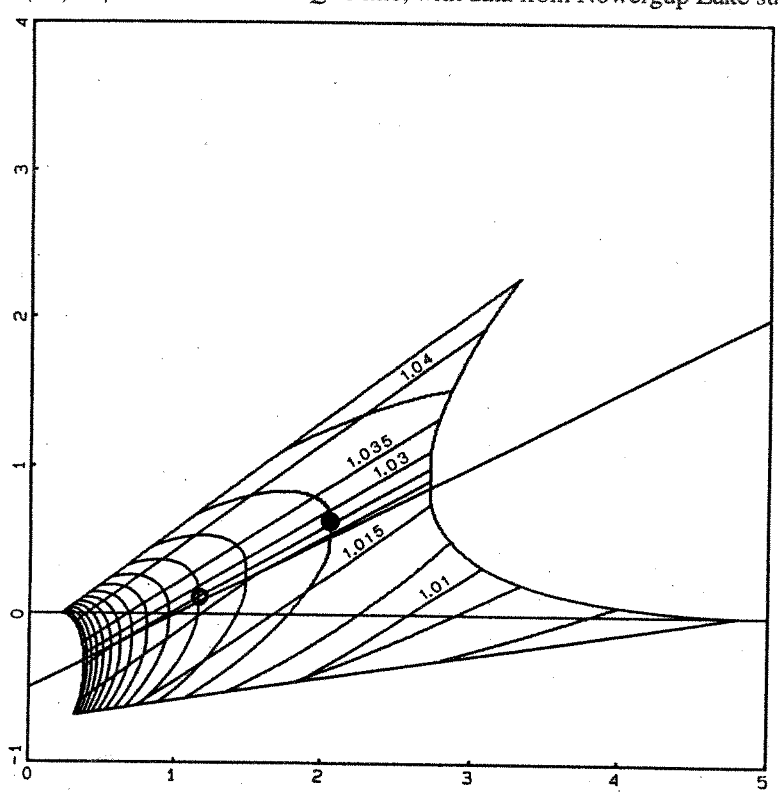


Figure 3.8.6 Flow-through domain for $2a/B$=1, showing contours of b_-/B and $(1+\delta_L)/(1+\delta_+)$ when $P(2a)/U_+B$=0.25 above and $E(2a)/U_+B$=0.25 below the Q=0 line, with data from Nowergup Lake superimposed

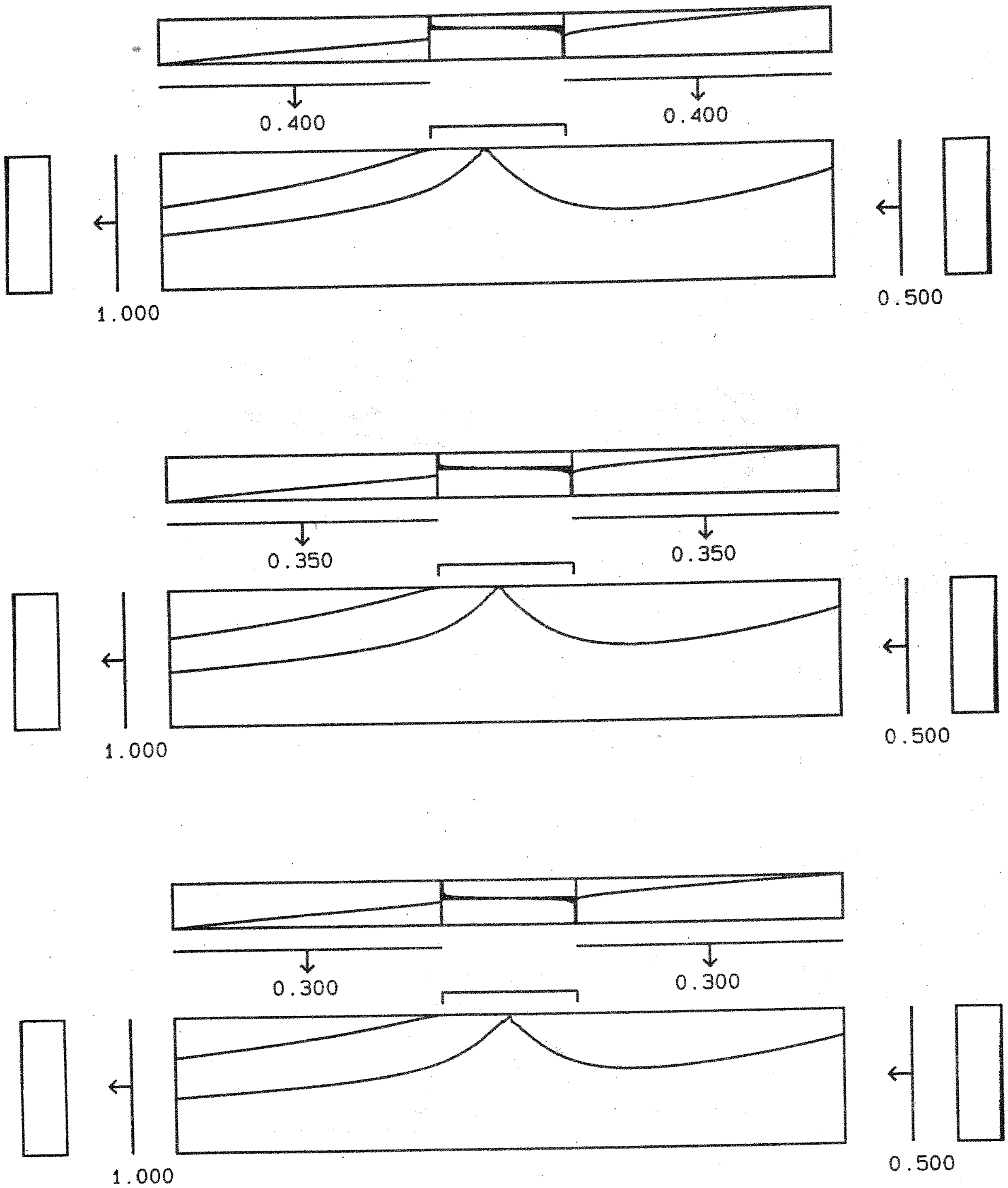


Figure 3.8.7 Dividing streamlines for three sets of parameters within the range indicated by analysis of $b\_/B$, c_L/c_+ and $(1+\delta_L)/(1+\delta_+)$ with $b\_/B$=0.6

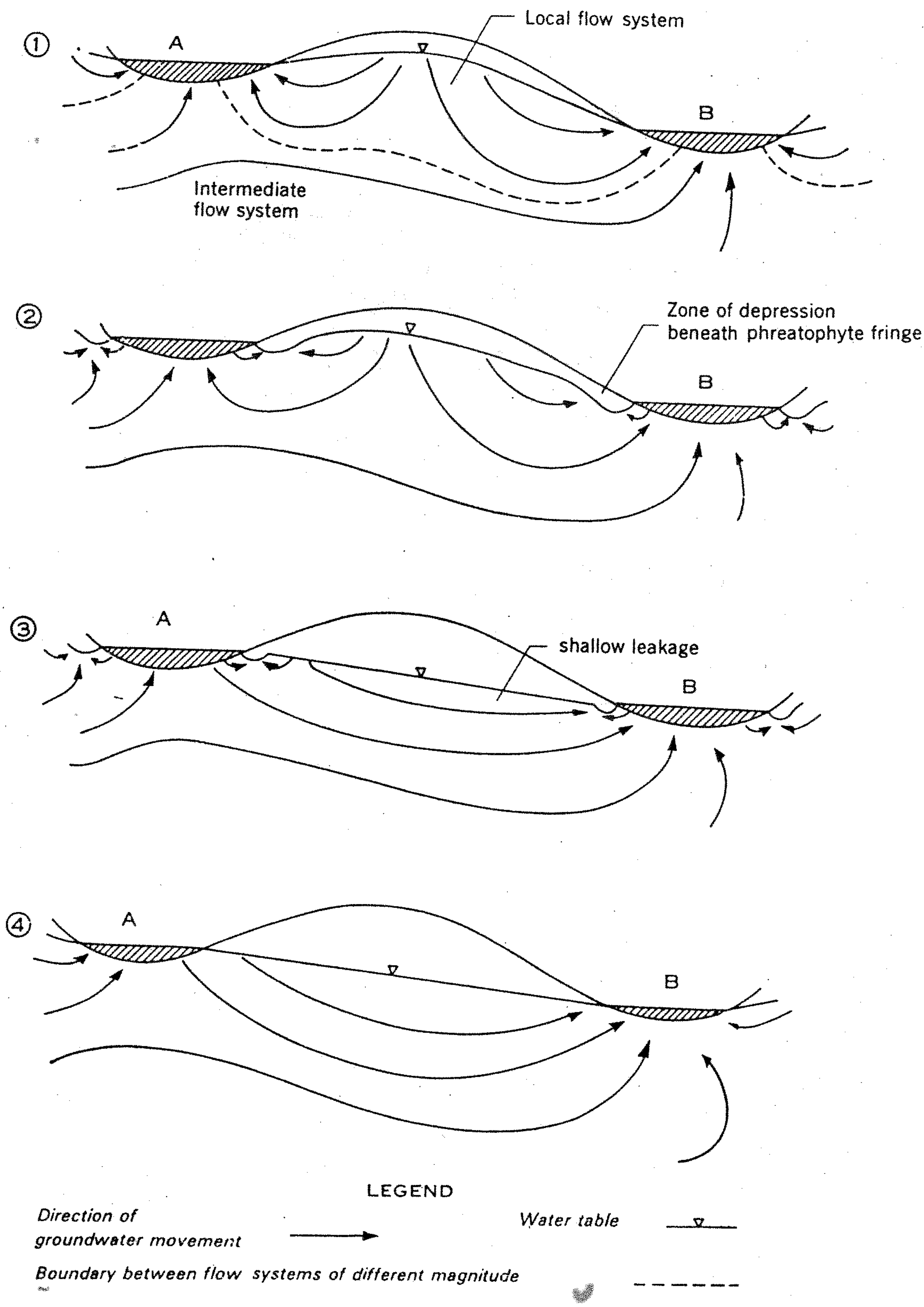
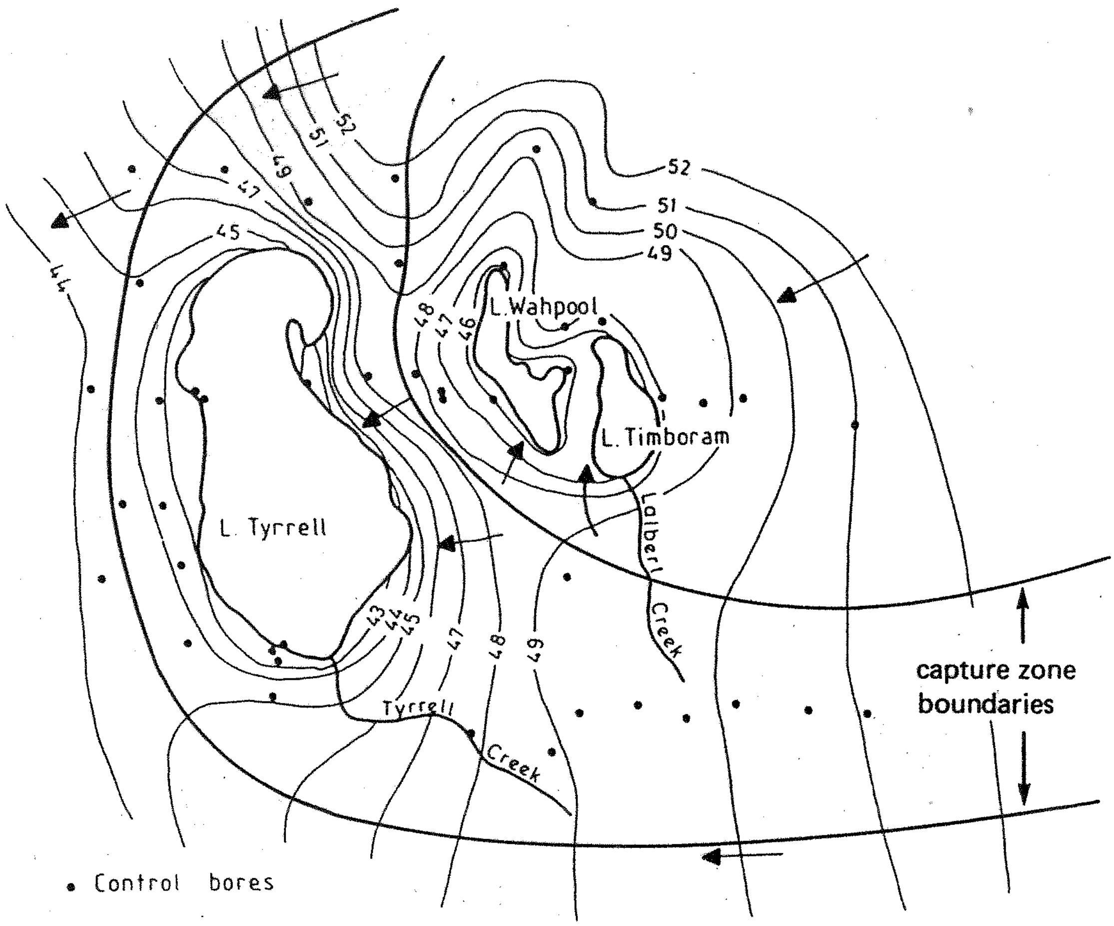


Figure 3.9.1 Diagramatic presentation of four typical flow conditions near permanent lakes in hummocky moraine: (1) Spring (2) Early summer (3) Late summer (4) Autumn and winter [from Meyboom, 1967]

(a)



(b)

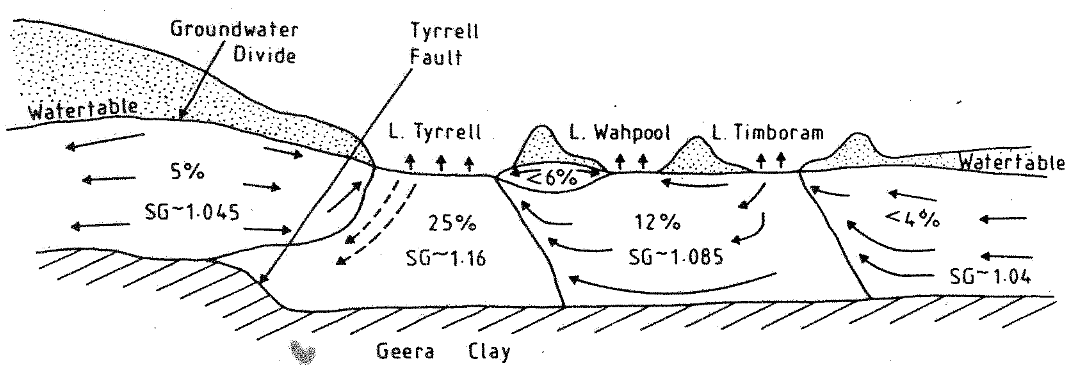


Figure 3.9.2 Groundwater levels and capture zones near Lake Tyrrell (a) in plan and (b) in vertical section [after Prendergast, 1989]

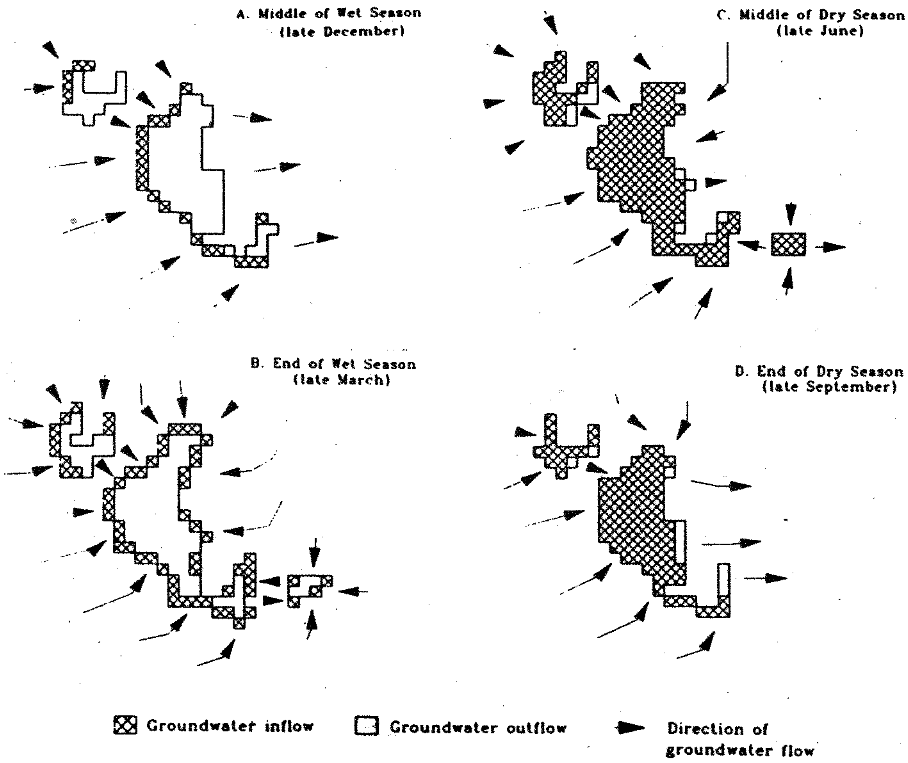


Figure 3.9.3 Transition between flow regimes in Doñana National Park, Spain (after Sacks *et al.*, 1992]

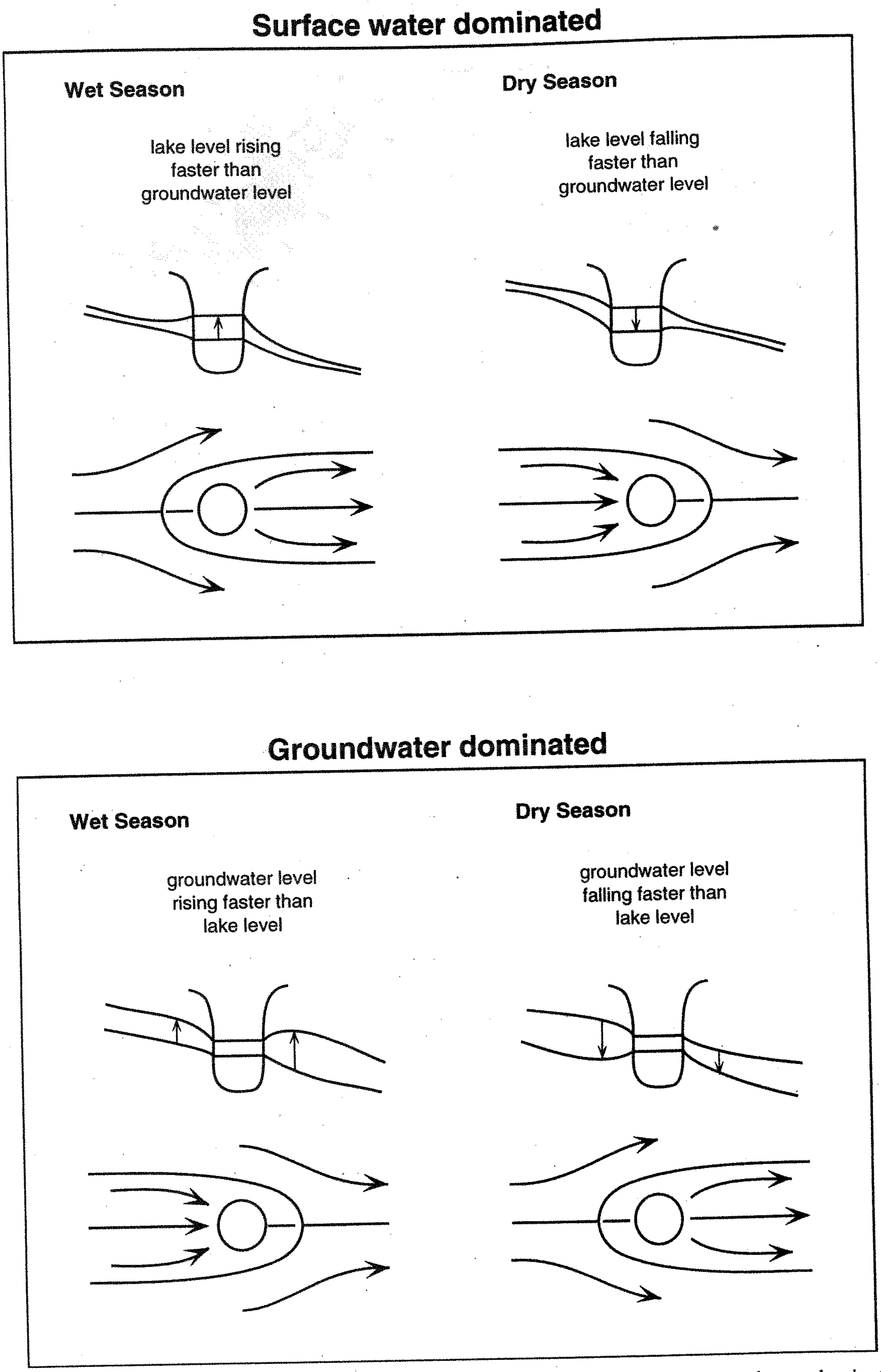


Figure 3.9.4 Fundamentally different behaviours in surface water dominated and groundwater dominated lake-aquifer systems

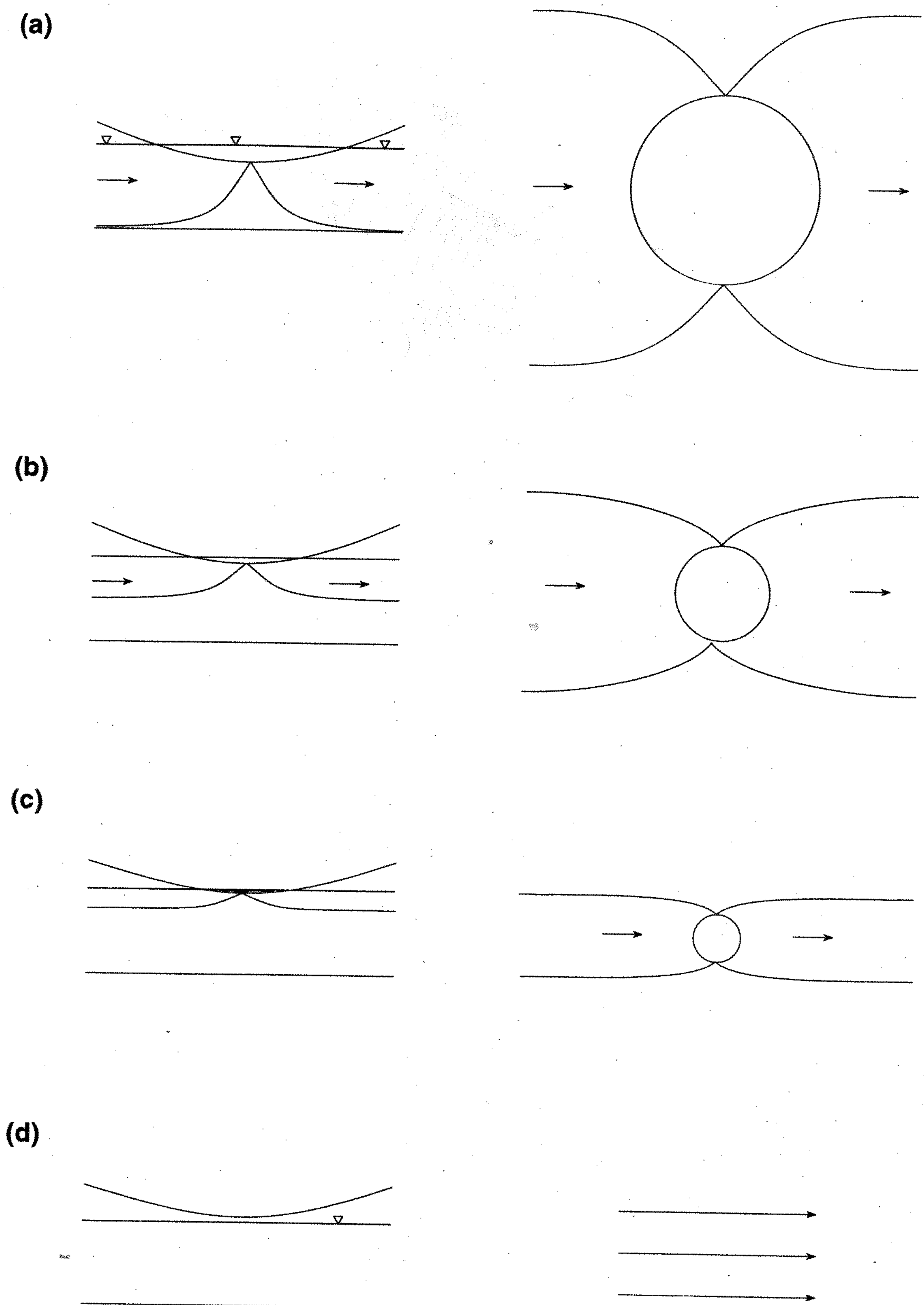


Figure 3.9.5 Transition between flow patterns as a large lake dries out

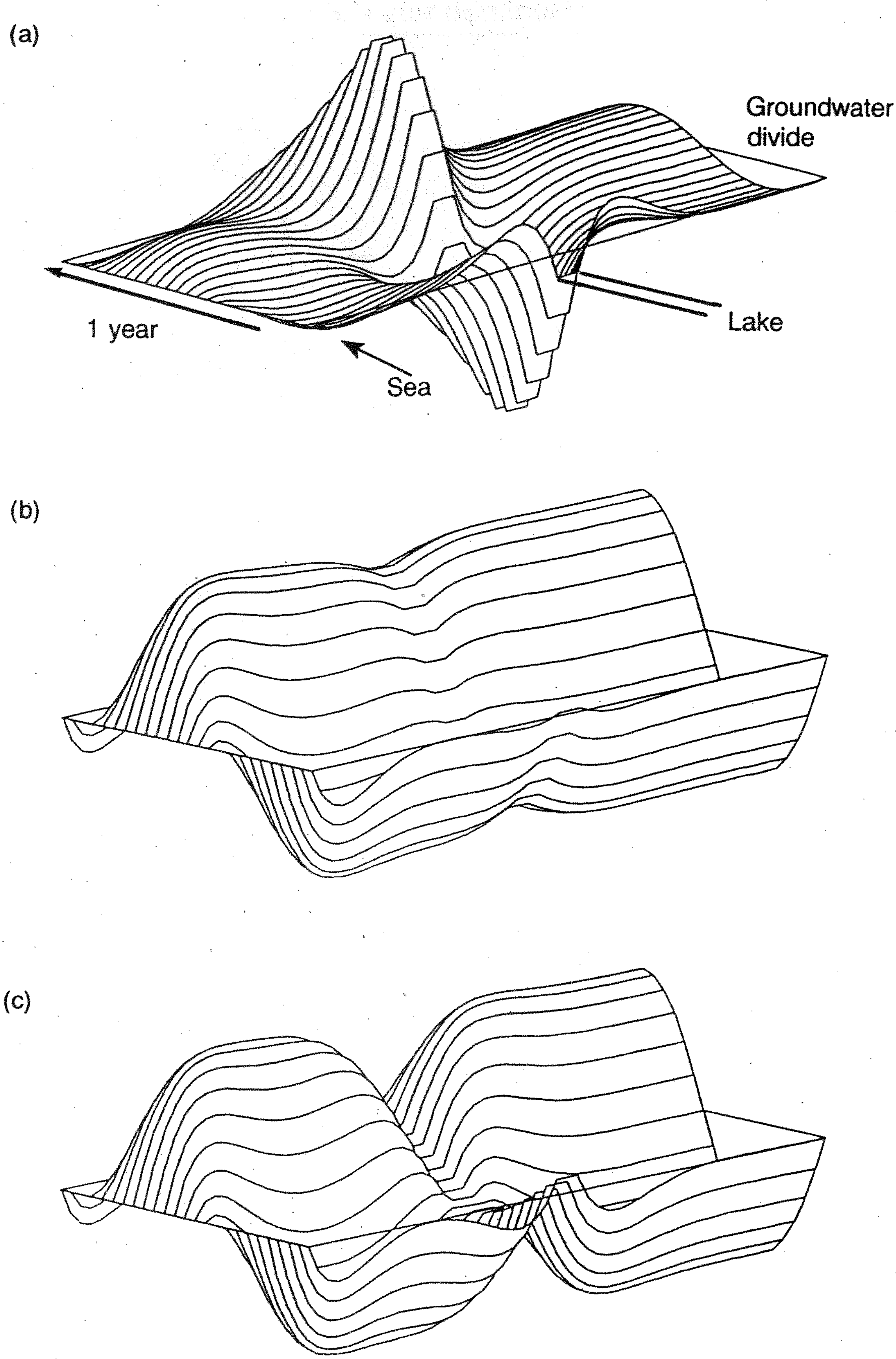


Figure 3.9.6 Fluctuations in water table elevation: (a) with $R_{Lp}S/R_p = 8$, (b) with $R_{Lp}S/R_p = 0.8$, and (c) with $R_{Lp}S/R_p = 0.08$

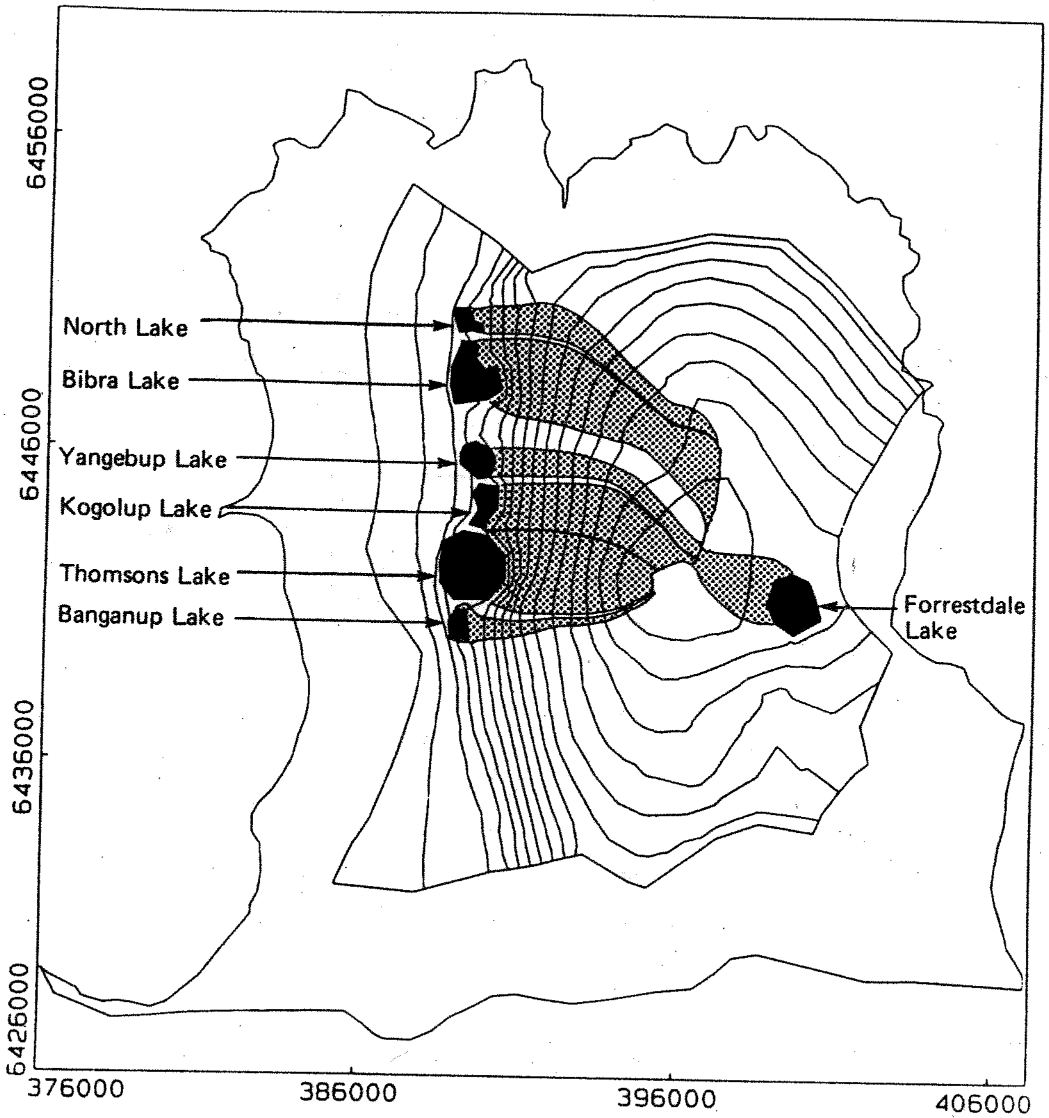


Figure 3.10.1 Predicted capture zones for seven lakes on the Jandakot Mound, without taking into account the depth of capture zones

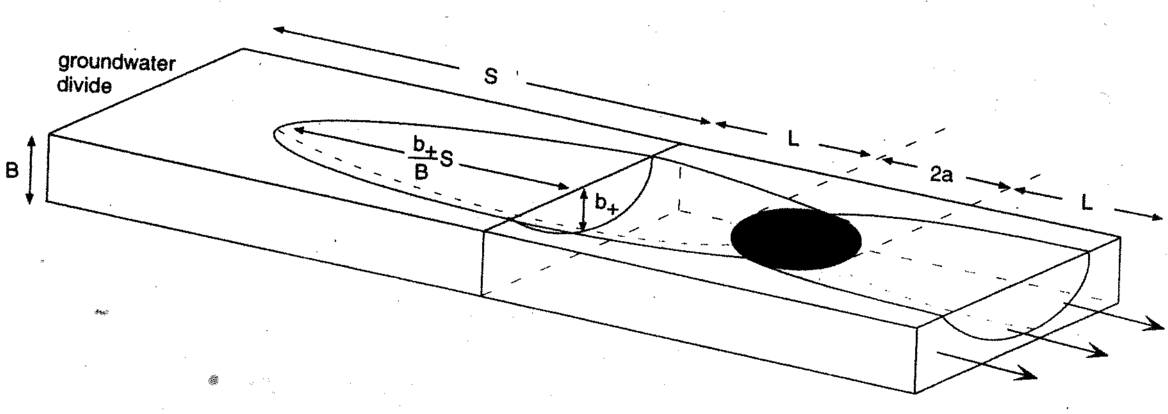


Figure 3.10.2 Schematic diagram showing extension of a capture zone towards a groundwater divide

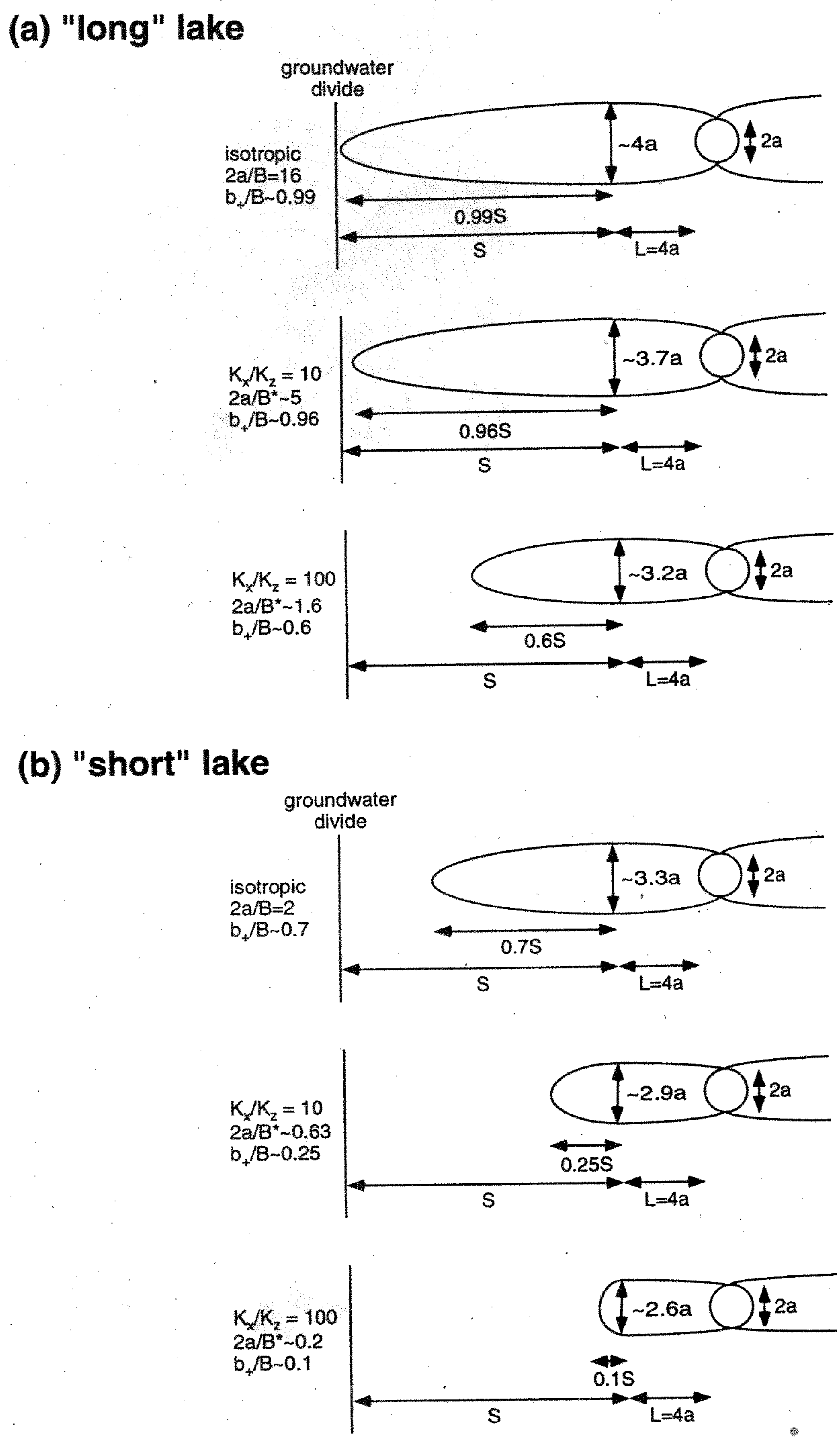


Figure 3.10.3 Capture zones for isolated (a) long and (b) short lakes, taking into account an effective $2a/B^*$ equal to $(2a/B)/(K_x/K_z)^{0.5}$

Table 3.11.1 Off-site groundwater data at Wattleup market garden

Bore Identity & Sample Date			Screen Depth (m)	Depth to Groundwater (m)	NO3 mg/L	PO4 mg/L (0=<0.03)
Site 8	Shallow	081190	10.8 to 13.8	10.22	50.5	3.83
	Shallow	081190			47.0	5.51
	Shallow	030491			124.9	4.6
	Shallow	040292			27.2	2.85
	Deep	081190	16.63 to 18.96	11.05	0.1	0
	Deep	081190			0.1	0
	Deep	030491			5.6	0
	Deep	040292			<0.04	<0.006
Site 9	Shallow	250990	9.1 to 12.1	7.06	0.0	
	Shallow	081190		7.34	0.1	0.01
	Shallow	081190			0.1	0.01
	Shallow	030491			3.8	0
	Shallow	040292			<0.04	<0.006
	Intermediate	250990	13.8 to 15.8	7.10	0.0	
	Intermediate	250990			0.0	
	Intermediate	081190		8.50	0.1	0.01
	Intermediate	081190			0.1	0.02
	Intermediate	030491			0.04	0.02
	Intermediate	040292			0.05	0.006
	Deep	130990	16.5 to 18.5	7.45	0.2	
	Deep	130990			0.0	
	Deep	130990			0.0	
	Deep	081190		8.87	0.1	0.01
	Deep	081190			0.1	0.02
	Deep	030491			0.05	0.02
	Deep	040292			0.05	<0.006
Site 10	Shallow	241090	12.7 to 15.7	12.81		
	Shallow	121190		13.04	37.0	0.01
	Shallow	040292			27.5	<0.006
	Deep	241090	17.7 to 19.7	13.65		
	Deep	121190		13.33	0.1	0.01
	Deep	121190			0.1	0.02
	Deep	030491			0.05	0.02
	Deep	040292			<0.04	0.009
Site 11	Shallow	250990	4.92 to 7.92	–	10.9	
	Shallow	121190		6.78	21.5	0.01
	Intermediate	250990	10.03 to 12.03	5.80	0.0	
	Intermediate	121190		6.62	0.1	0.01
	Intermediate	121190			0.1	0.01
	Intermediate	030491			0.1	0.01
	Intermediate	040292			<0.04	<0.006
	Deep	250990	16.35 to 18.35	5.90	0.0	
	Deep	121190		6.51	0.1	0
	Deep	121190			0.1	0
	Deep	030491			0.3	0.01
	Deep	040292			0.05	<0.006
Site 12	Shallow	121190	6.00 to 9.00	7.47	0.1	0
	Shallow	121190			0.1	0.02
	Shallow	040292			0.06	<0.006
	Intermediate	250990	10.39 to 12.39	6.75	15.2	
	Intermediate	081190		7.26	4.1	0
	Intermediate	081190			5.1	0
	Intermediate	030491			0.6	0
	Intermediate	040292			9.4	<0.006
	Deep	250990	14.86 to 16.86	6.68	170.1	
	Deep	081190		7.22	130.2	0
	Deep	081190			155.1	0
	Deep	030491			257.2	0
	Deep	040292			183.3	<0.006

(Shaded sections denote evidence of nitrate breakthrough)

Table 3.11.2 Phosphate adsorption and travel distances using soil samples from Wattleup market garden

Coefficients in modified Freundlich isotherm			
Coefficients	Soil (a) 0-25 cm	Soil (b) 50-75 cm	Soil (c) 2.75 - 3.0 m
K	12.6	17.8	41.7
m	0.3	0.3	0.2
n	0.22	0.22	0.22

	Phosphate Mobilities - Soil (b)			
Applied phosphate (kg/ha/yr)	10		100	
Recharge rate q				
cm/y	20	100	20	100
cm/d	0.055	0.274	0.055	0.274
Input phosphate concentration (mg/L)	5	1	50	10
Lower bound 100-yr travel distance (m)	0.20	0.32	1.02	1.64
Upper bound 100-yr travel distance (m)	2.01	3.29	9.76	16.3

Table 3.11.3 Transport characteristics of BTEX compounds in petroleum in relation to wetland buffer zones

	Benzene	Toluene	P-Xylene	Napthalene
$v_{groundwater}/v_{organic}$	1.02	1.04	1.12	1.32
$t_{1/2}$ oxic degradation (days)	80–160	10–20	20–280	20
Travel distance (metres)	140–300	10–30	30–480	30
$t_{1/2}$ anoxic degradation (days)	400	70–140	150–300	20–40
Travel distance (metres)	750	140–260	260–510	30–60

Source of basic data: Thierren *et al.* [1992 *a,b*]
Soil type: Bassendean Sand

Table 3.11.4 Residual concentrations of pesticides reaching a depth of 3 metres in Bassendean Sand with a recharge rate of 1 m yr^{-1} [from Singh, 1989]

K_{oc} (kg/m^3)	Half life (days)	Fraction of applied mass	Time T_H (years)	Applied Mass (kg/ha (a.i.))	Conc.$^{(ii)}$ (mg/L) at 3m depth
Simazine					
0.037	32	4.9×10^{-5}	4.5	1.0	2.5×10^{-3}
0.037	75	1.4×10^{-2}	4.5	1.0	7.0×10^{-1}
0.037	75$^{(i)}$	4.3×10^{-1}	1.2	1.0	2.1×10^{1}
Fenamiphos					
0.137	10	8.2×10^{-47}	15.1	10.0	4.1×10^{-45}
0.137	38	7.4×10^{-13}	15.1	10.0	3.7×10^{-11}
0.137	67	1.3×10^{-7}	15.1	10.0	6.5×10^{-5}
0.137	67$^{(i)}$	3.4×10^{-1}	3.5	10.0	1.7×10^{1}
Linuron					
0.210	75	5.5×10^{-10}	22.7	2.25	2.7×10^{-8}
0.210	75$^{(i)}$	1.0×10^{-1}	3.3	2.25	5.0×10^{1}
Diquat					
4.25	7300	1.3×10^{-2}	450	1.0	0.65
4.25	7300$^{(i)}$	6.3×10^{-1}	83	1.0	3.15×10^{1}

(i) calculated with decreasing organic carbon content
(ii) in 1 cm soil layer with $\theta = 0.2\ m^3\ m^{-3}$

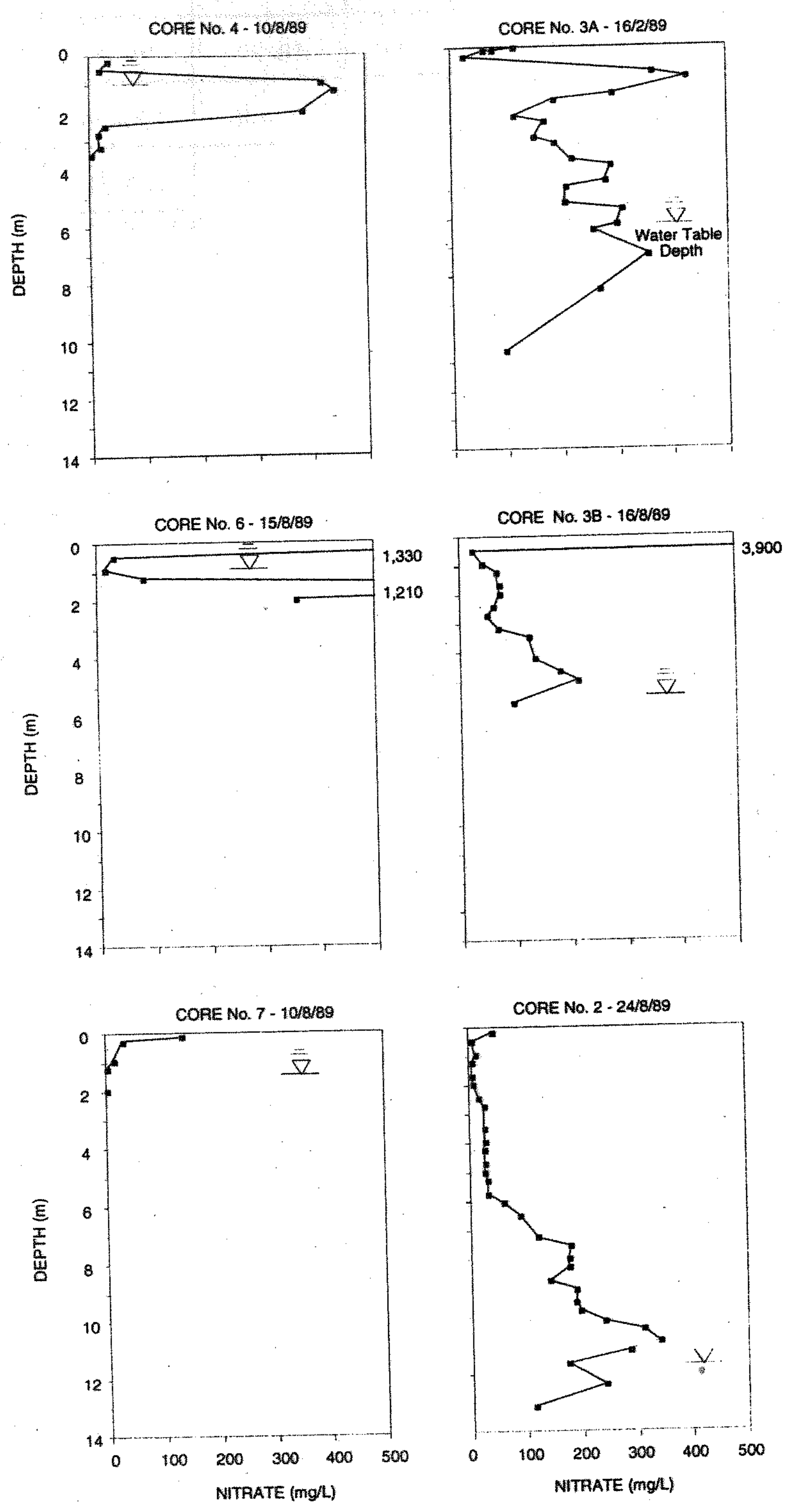


Figure 3.11.1 Distribution of nitrate in soil water in the unsaturated zone near Lake Wattleup

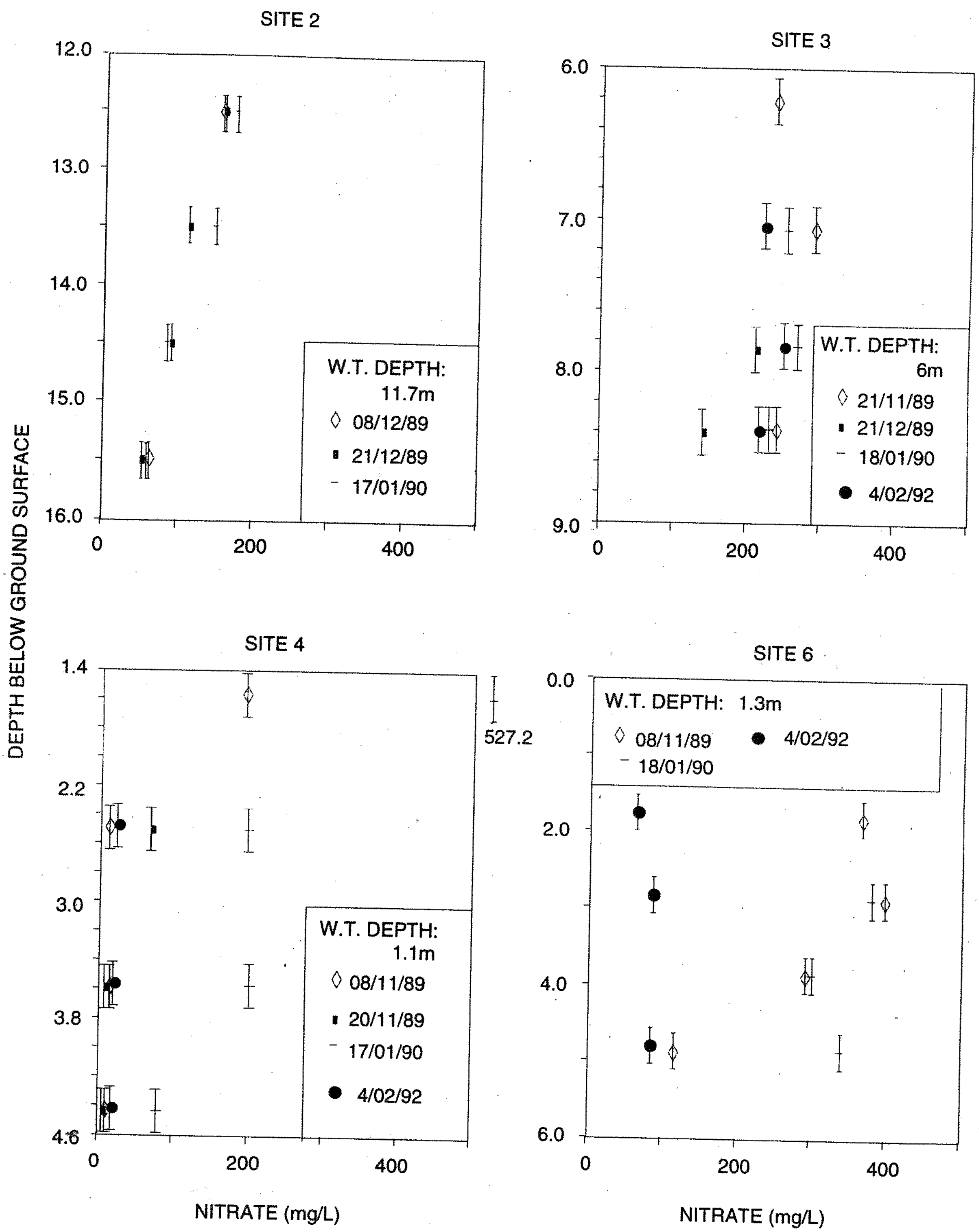


Figure 3.11.2 Distribution of nitrate below the water table near Lake Wattleup

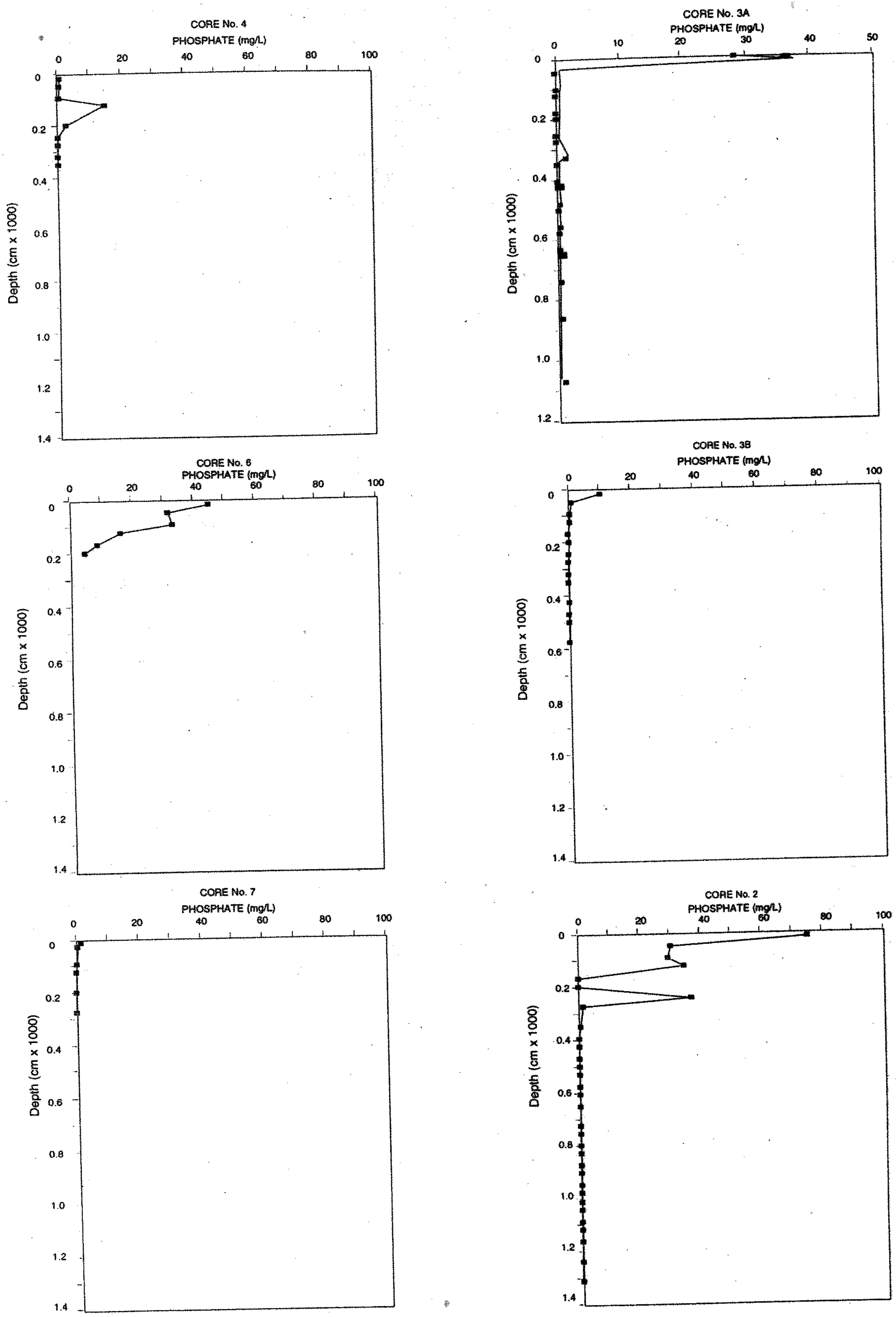


Figure 3.11.3 Distribution of phosphate in soil water in the unsaturated zone near Lake Wattleup

4. Management of Water Levels

4.1 Observed Seasonal Fluctuations

Water levels in lakes on the Swan Coastal Plain fluctuate seasonally, and some lakes dry out at the end of summer.

Figures 4.1.1 to 4.1.9 show lake levels at a number of lakes in the Perth region during the twenty-year period from 1 January 1972 to 31 December 1991. We have not carried out systematic analysis of these levels or related them to climate or land use changes. Such studies need to be done using a regional scale hydrogical model, such as the Perth Urban Water Balance Model, in order to take into account the role of the lakes in the regional scale system.

In general, lake levels vary by 0.5 to 1.5 m during the annual cycle. Some lakes have a range as low as 0.1 m. Others have a range as large as 3 m. Some lakes have steep banks and are wet all year. Others have bowl-shaped bottoms, so that they dry out at the end of summer. Water level fluctuations are natural in our mediterranean climate, and are even necessary for the health of some ecological systems. Suggestions have been made by other researchers that the rate of change of water levels is just as important as the total magnitude of the change [e.g., Froend *et al.*, 1993]. Water levels fluctuations can be increased or decreased by manmade influences, such as surface drainage or weirs to control levels.

In recent years, the Environmental Protection Authority of Western Australia has set lower limits for lake levels in some lakes on the Swan Coastal Plain. As a result, it has been proposed that a number of lakes should have their levels artificially maintained if lake levels approach the recommended or absolute minimum levels.

4.2 Artificial Water Level Maintenance

Lake levels can be effectively maintained by pumping relatively small volumes of groundwater into the lakes for a few months each year.

Figures 4.2.1 to 4.2.4 show lake level changes in Nowergup Lake during artificial water level maintenance in 1990. Lake levels respond immediately to the addition of pumped groundwater. The impact of pumping about 3000 KL d^{-1} for two weeks is to cause lake levels in Nowergup Lake to rise about 10 cm. For a lake of this size (i.e. surface area) in this hydrogeological environment, this rate of pumping is sufficient to exceed losses by evaporation and groundwater outflow. In 1990, the total period of pumping was less than six weeks, with a total pumped volume of 0.13 GL.

Artificial water level maintenance can lead to an improvement in lake water quality. Our measurements have shown that groundwater pumped into Nowergup Lake during a water level maintenance trial in 1989 contained lower concentrations of nitrate and phosphate than the lake itself [Water Authority of Western Australia, 1992]. A permanent facility established by the Water Authority for artificial water level maintenance pumps water from the deeper Leederville Formation, which is even better in quality than the superficial formations. Water level maintenance in 1990 may have improved water quality in the lake and limited the growth of a bloom of *Microcystis sp* that occurred prior to pumping [Townley *et al.*, 1993].

4.3 Minimising the Impact of Pumping

In order to minimise its impact on lakes, pumping for public or private water supply should be located as far away as possible, both in space and in time.

We have not carried out specific modelling studies to determine desirable or optimal pumping strategies for pumping for water supply, mainly because we do not yet have a model which properly handles lakes which experience significant seasonal changes in area. We have argued on an intuitive basis, however, and on the basis of other theoretical results, that pumping at a distance from a lake, either horizontally or vertically, will reduce the immediate impact of the pumping on lake water levels.

Pumping from deeper aquifers, such as the Leederville Formation, is an effective way of reducing impacts at the water table. However if additional pumping from the Leederville Formation is to be encouraged in the future, it is critically important that the Water Authority should develop and maintain a capability for predicting water balances in all major regional aquifers, i.e. not only in the superficial formations as currently modelled by the Perth Urban Water Balance Model.

A shift towards preferentially using groundwater in winter months and surface water in summer months may have advantages for the management of both groundwater and surface water supplies on the Swan Coastal Plain. Pumping groundwater in winter months will have less impact on lake and wetland levels.

4.4 The Physics of Water Level Fluctuations

Average groundwater and lake levels depend on long-term average recharge, whereas seasonal fluctuations depend on the deviations between fluctuating recharge and the long-term average.

We have argued intuitively and developed a mathematical model [Townley *et al.*, 1993] which demonstrates that average groundwater and lake levels are controlled by different quantities than fluctuations in groundwater and lake levels. Average levels are controlled by the spatial distributions of aquifer transmissivity and long-term average net recharge, i.e. the net effect of percolation to the water table, pumping, groundwater withdrawals by phreatophytic vegetation and leakage to or from lower aquifers. The amplitude or range of fluctuations, however, depends on the spatial distributions of specific yield and transmissivity, and on the amplitude or range of the deviation between instantaneous net recharge and its long-term average.

Specific yield and transmissivity can be combined to define a characteristic response time for a regional aquifer of length L, or equivalently a non-dimensional ratio of the form L^2S/TP, which is the ratio of the aquifer response time to the period of fluctations, P. The unconfined aquifer of the Swan Coastal Plain has a reasonably long response time, such that groundwater fluctuations are closely related to fluctuations in recharge and are not greatly diminished by the effects of lateral aquifer flow.

Long-term fluctuations in groundwater and lake levels depend on long-term fluctuations in recharge. A generalisation of the above is that seasonal fluctuations are superimposed on longer term changes induced by climatic variations or changes in land use. It is possible to separate the short and long term fluctuations and to analyse them separately.

Lake levels can fluctuate either more or less than nearby groundwater levels, depending on whether a lake is driven by surface water inflows or by groundwater inflows. Townley *et al.* [1993] describe a mathematical model for a lake in the middle of a one-dimensional regional aquifer and have shown that lake level fluctuations can be greater than or less than nearby water table fluctuations. Apart from the response time of the regional aquifer, the determining non-dimensional ratio is the product of the amplitude of net recharge over the lake surface and specific yield in the aquifer, divided by the amplitude of net recharge to the aquifer, $R_{Lp}S/R_p$.

Lakes which have very large fluctuations in rainfall minus evaporation, or which have large seasonal inflows of surface drainage, are likely to have level fluctuations which are larger than those in the aquifer nearby. In such cases, the lake levels drive the response of the aquifer, and peak groundwater levels lag behind peak lake levels. If the regional aquifer has larger fluctuations in net recharge to groundwater, perhaps because of significant pumping in summer months, fluctuations in groundwater levels are larger than those in the lake level, and peak lake levels lag behind peak groundwater levels.

Figure 3.9.6 illustrates the fluctuations in lake and aquifer levels for three specific situations. Figures 4.4.1 to 4.4.4 show general solutions over a range of the controlling non-dimensional ratios, $R_{Lp}S/R_p$ and L^2S/TP. The three uppermost plots are of (a) the amplitude of lake level fluctuations, (b) the amplitude of water table fluctuations at a location 0.9 of the way between the ocean and the lake, and (c) the ratio of the lake amplitude to the water table amplitude. Lake and water table amplitudes are normalised with respect to $R_pP/2\pi S$, the water table fluctuation which would occur in the absence of lateral flow. The lower plots are of (d) the phase lag between recharge and lake levels, (e) the phase lag between recharge and water table elevation at the chosen location, and (f) the phase lag between lake level and that water table fluctuation. Phase lags are expressed as fractions of a period. Some insight into the behaviour of the coupled lake-aquifer system is given by Townley [1994].

Figures 4.4.1 to 4.4.4 show the dependence of the behaviour on the size of a lake within a 10 km aquifer, and also on the depth of capture, b_+/B. In the first three Figures, the capture zone extends to the bottom of the aquifer, but in the last, it is assumed that only half of the aquifer flow is captured. A complete explanation for these Figures is provided by Townley *et al.* [1993].

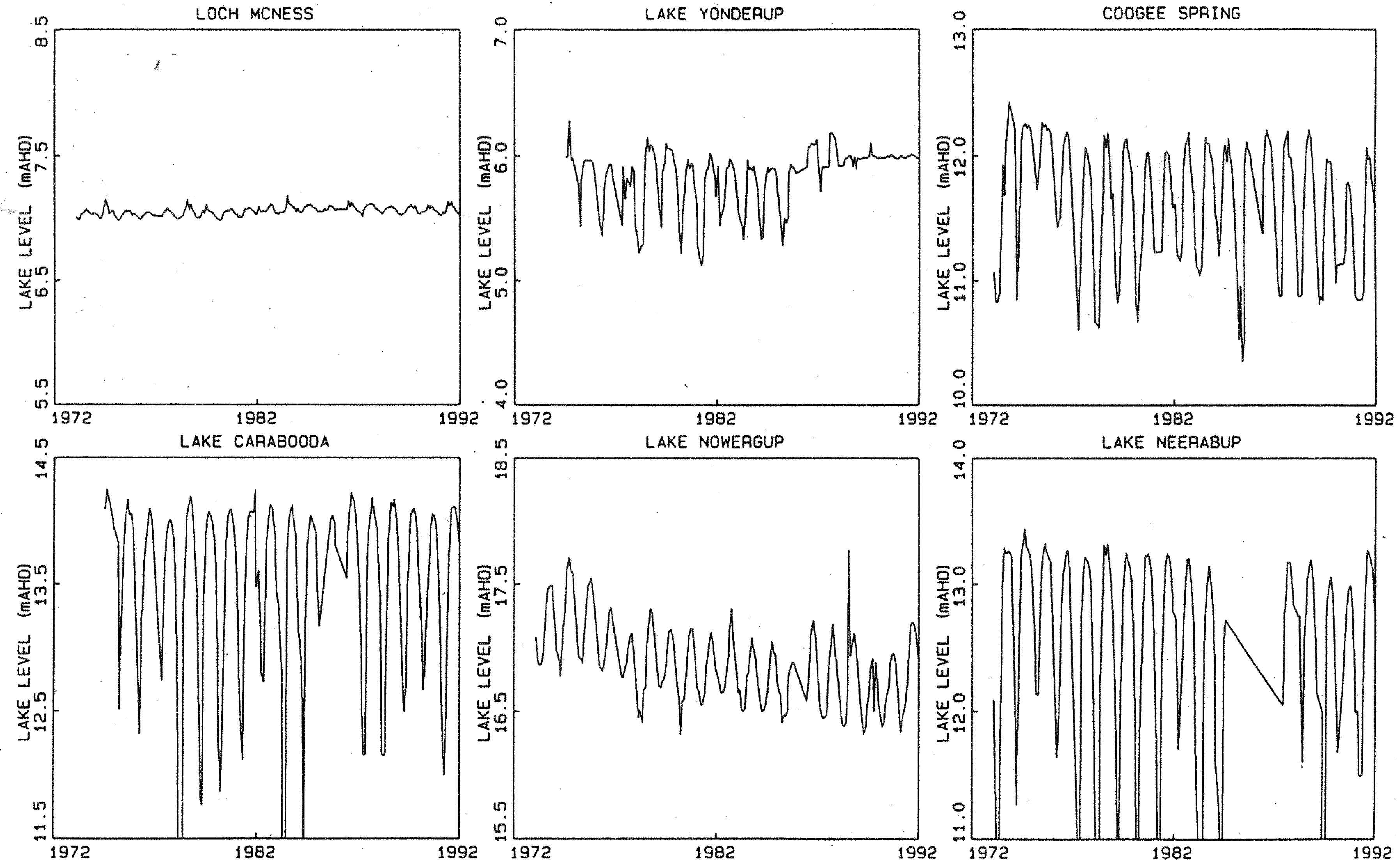


Figure 4.1.1 Lake water levels from 1 January 1972 to 31 December 1991

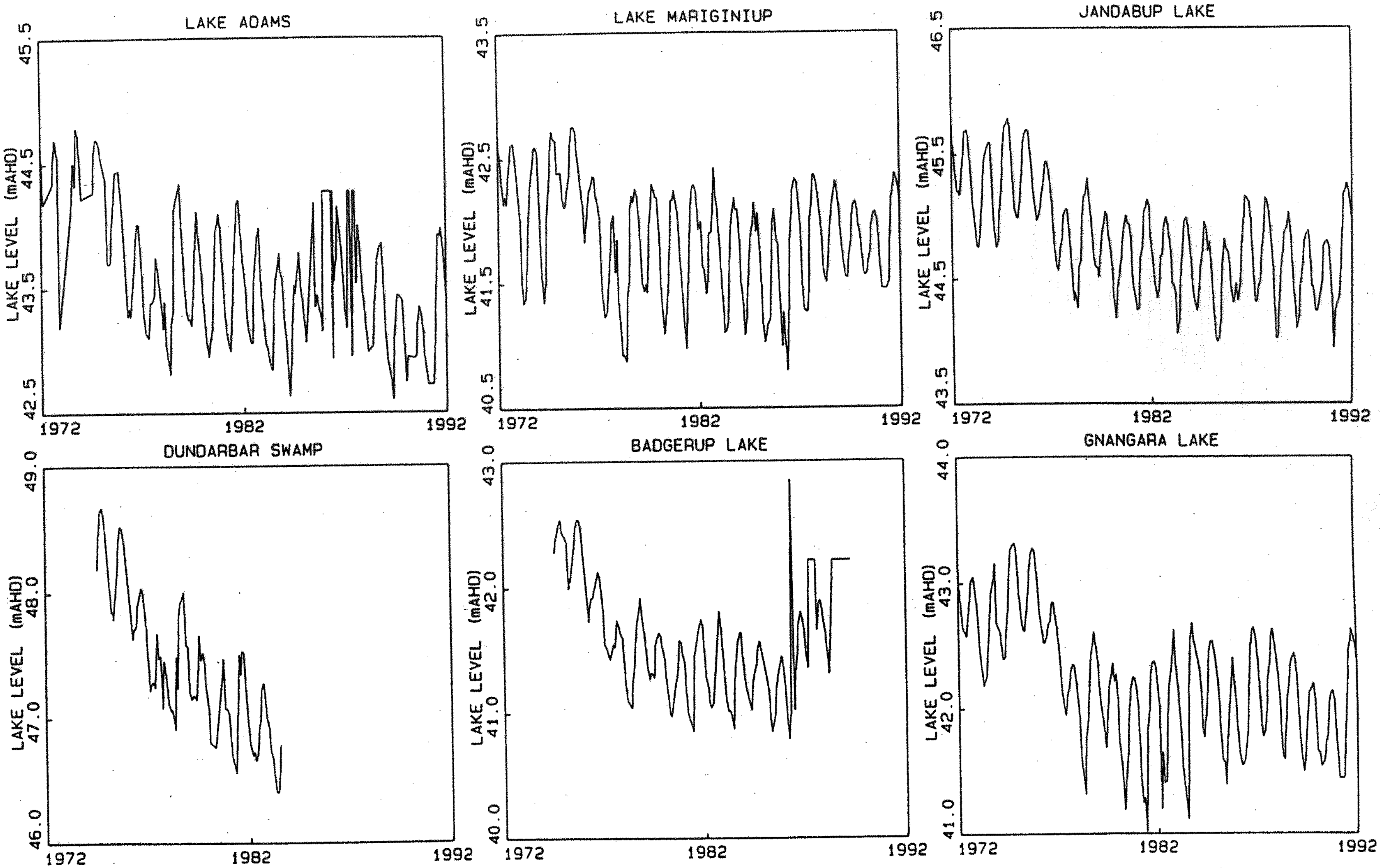


Figure 4.1.2 Lake water levels from 1 January 1972 to 31 December 1991

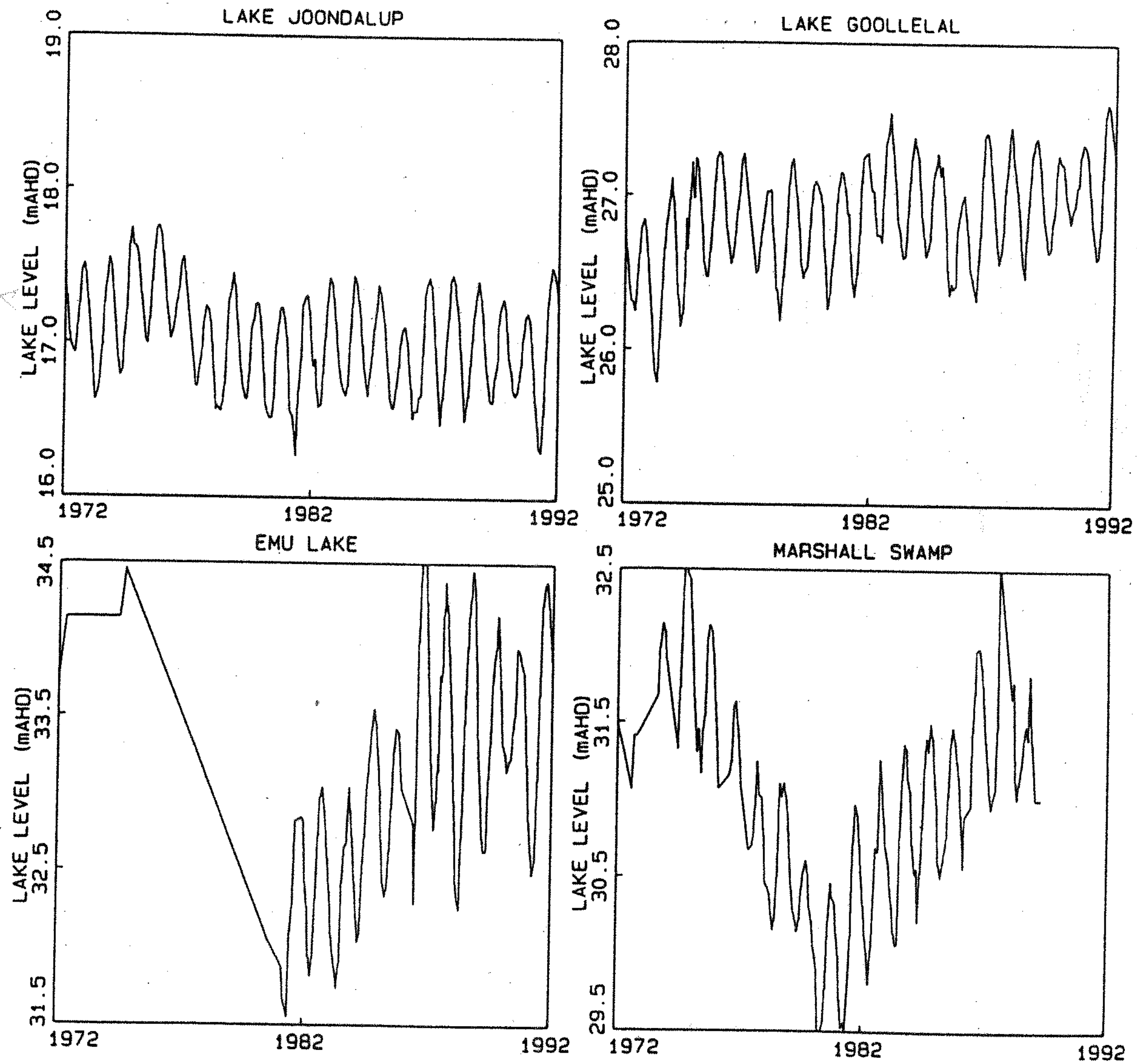


Figure 4.1.3 Lake water levels from 1 January 1972 to 31 December 1991

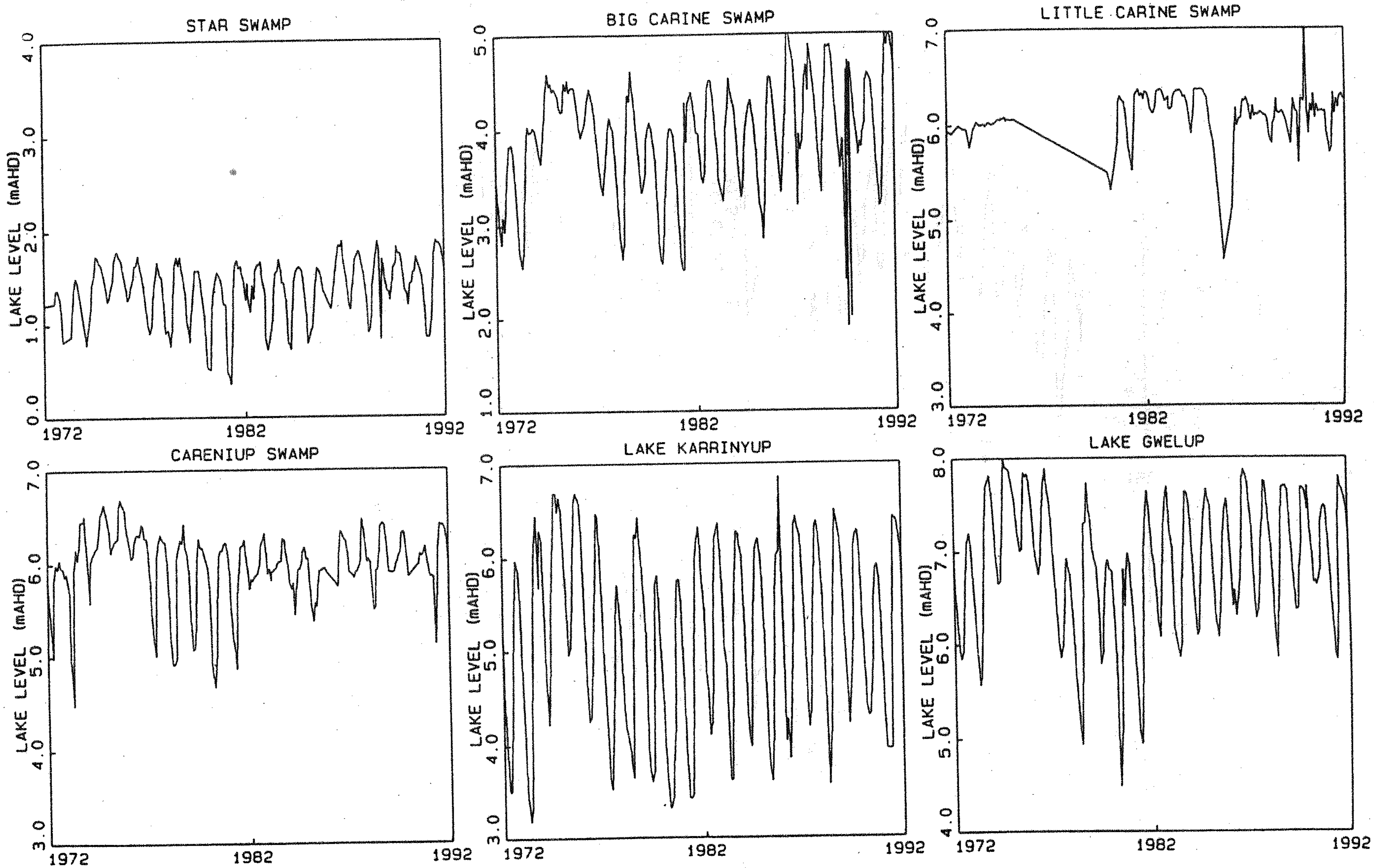


Figure 4.1.4 Lake water levels from 1 January 1972 to 31 December 1991

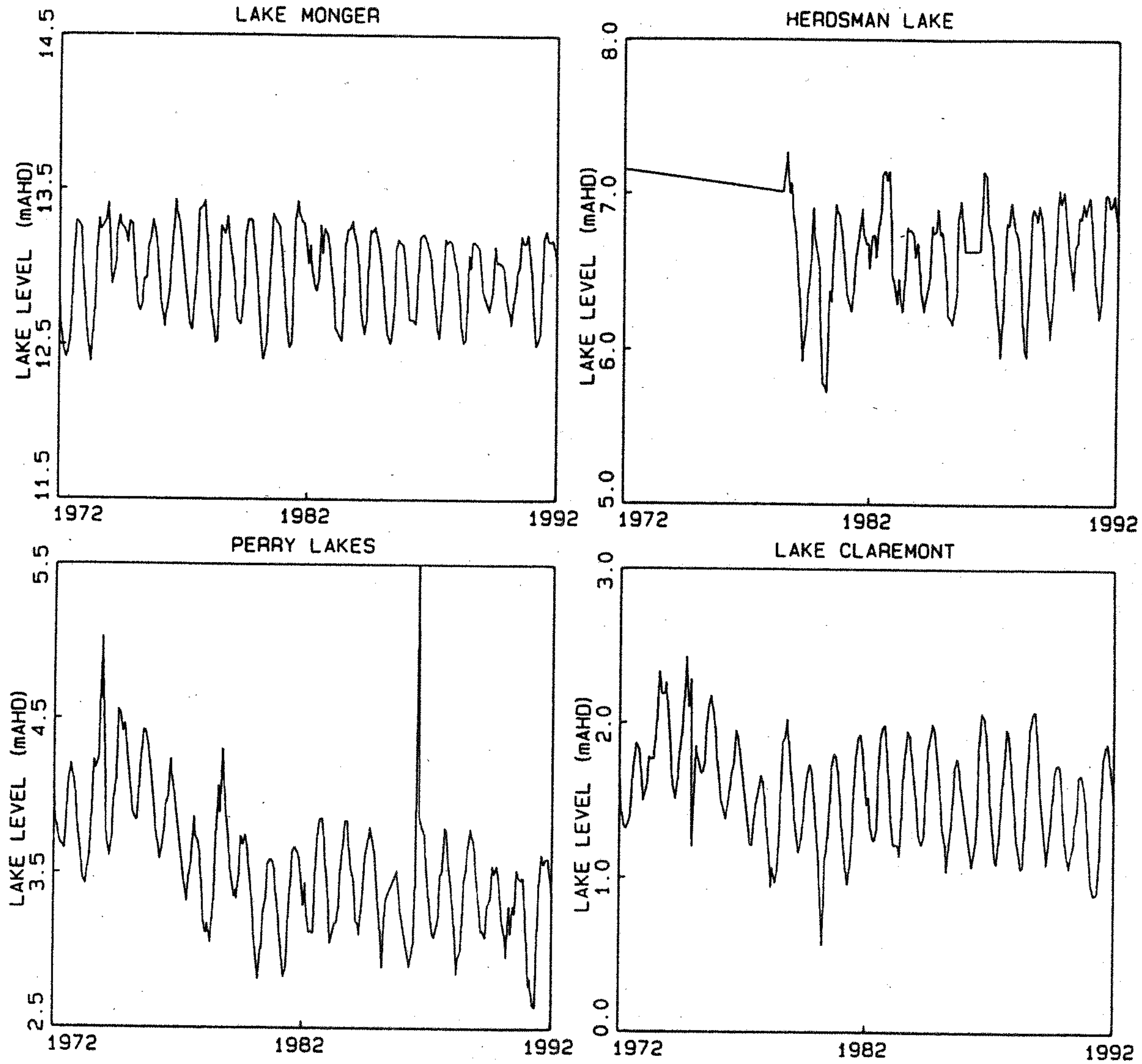


Figure 4.1.5 Lake water levels from 1 January 1972 to 31 December 1991

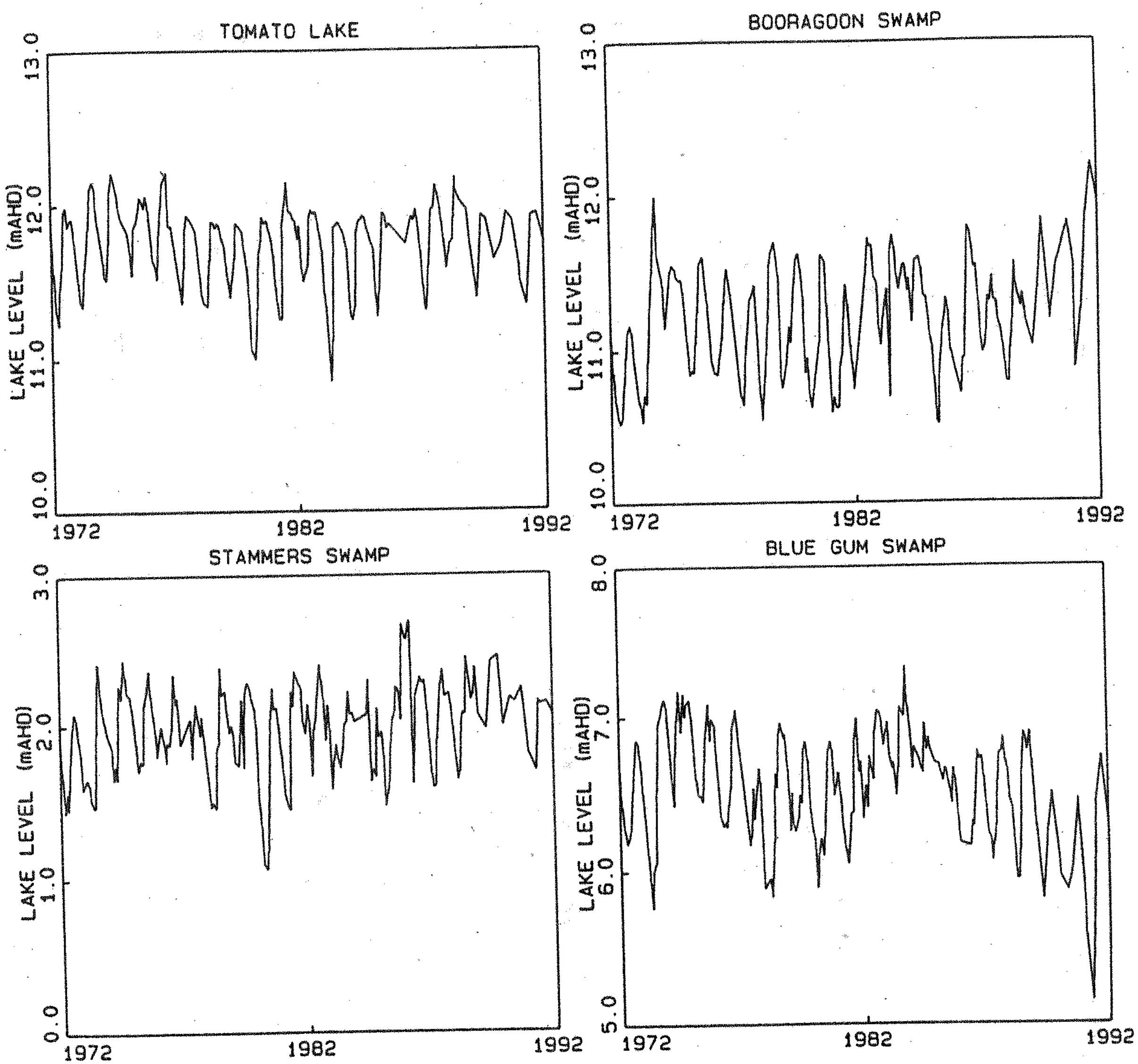


Figure 4.1.6 Lake water levels from 1 January 1972 to 31 December 1991

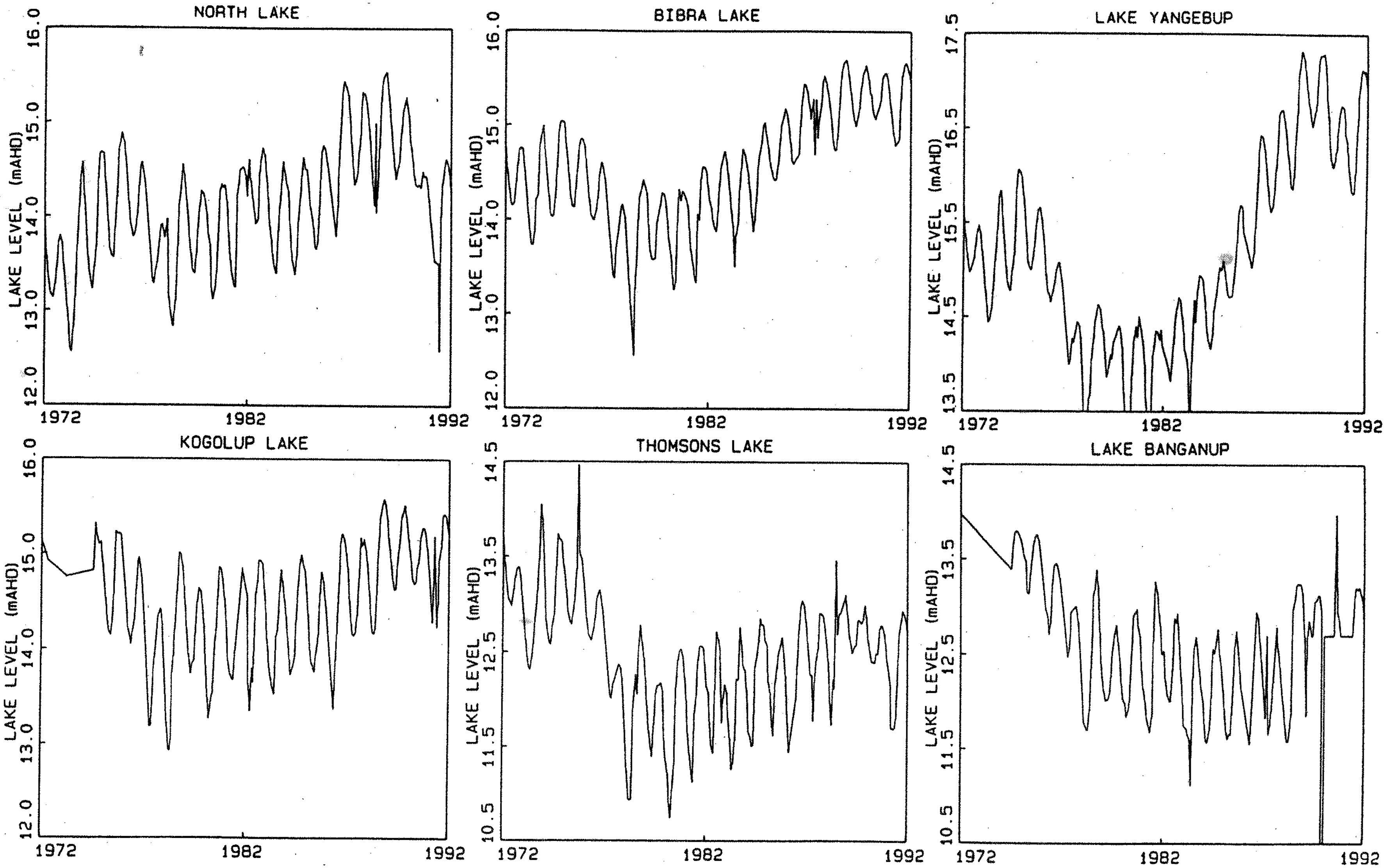


Figure 4.1.7 Lake water levels from 1 January 1972 to 31 December 1991

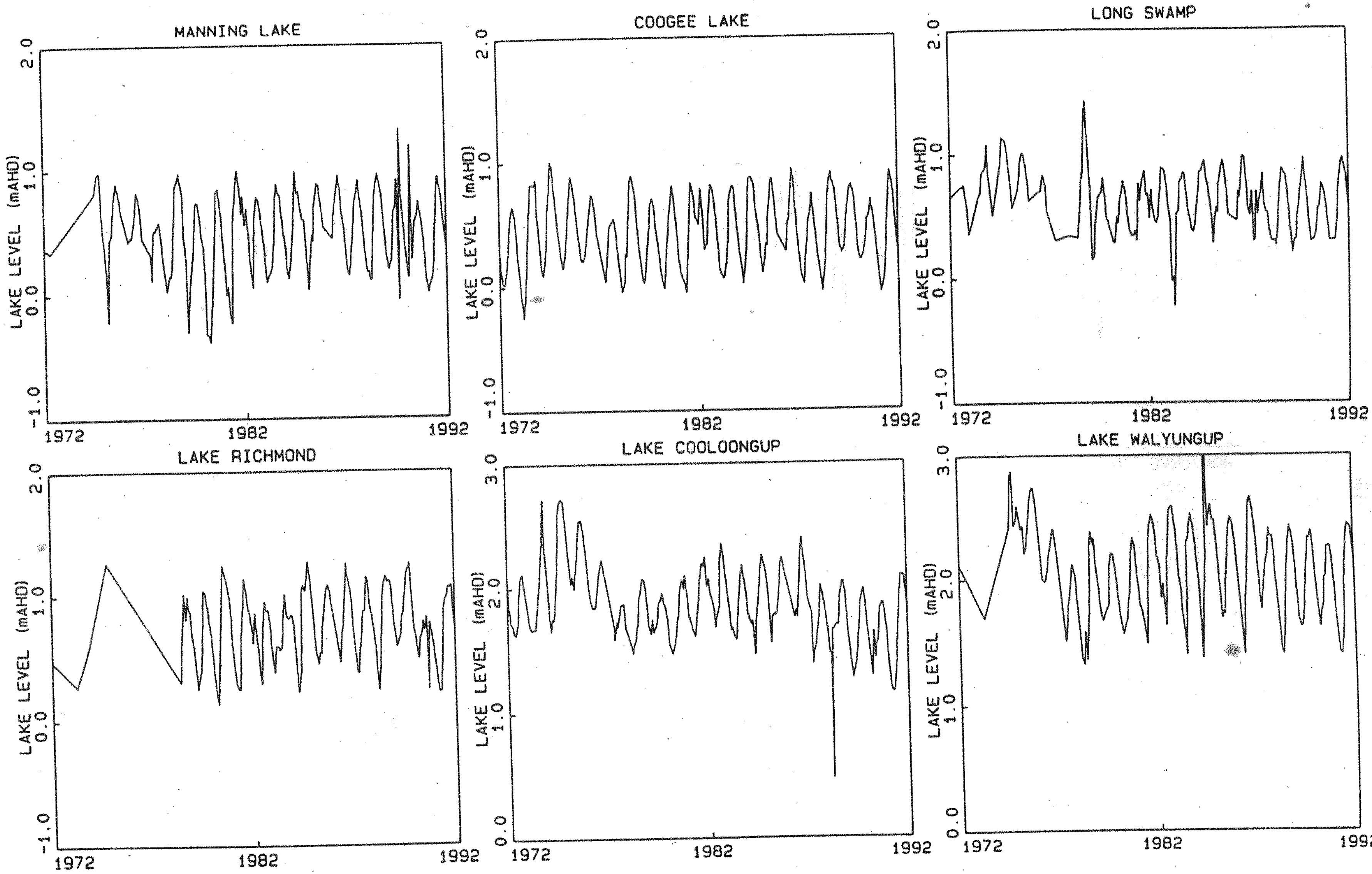


Figure 4.1.8 Lake water levels from 1 January 1972 to 31 December 1991

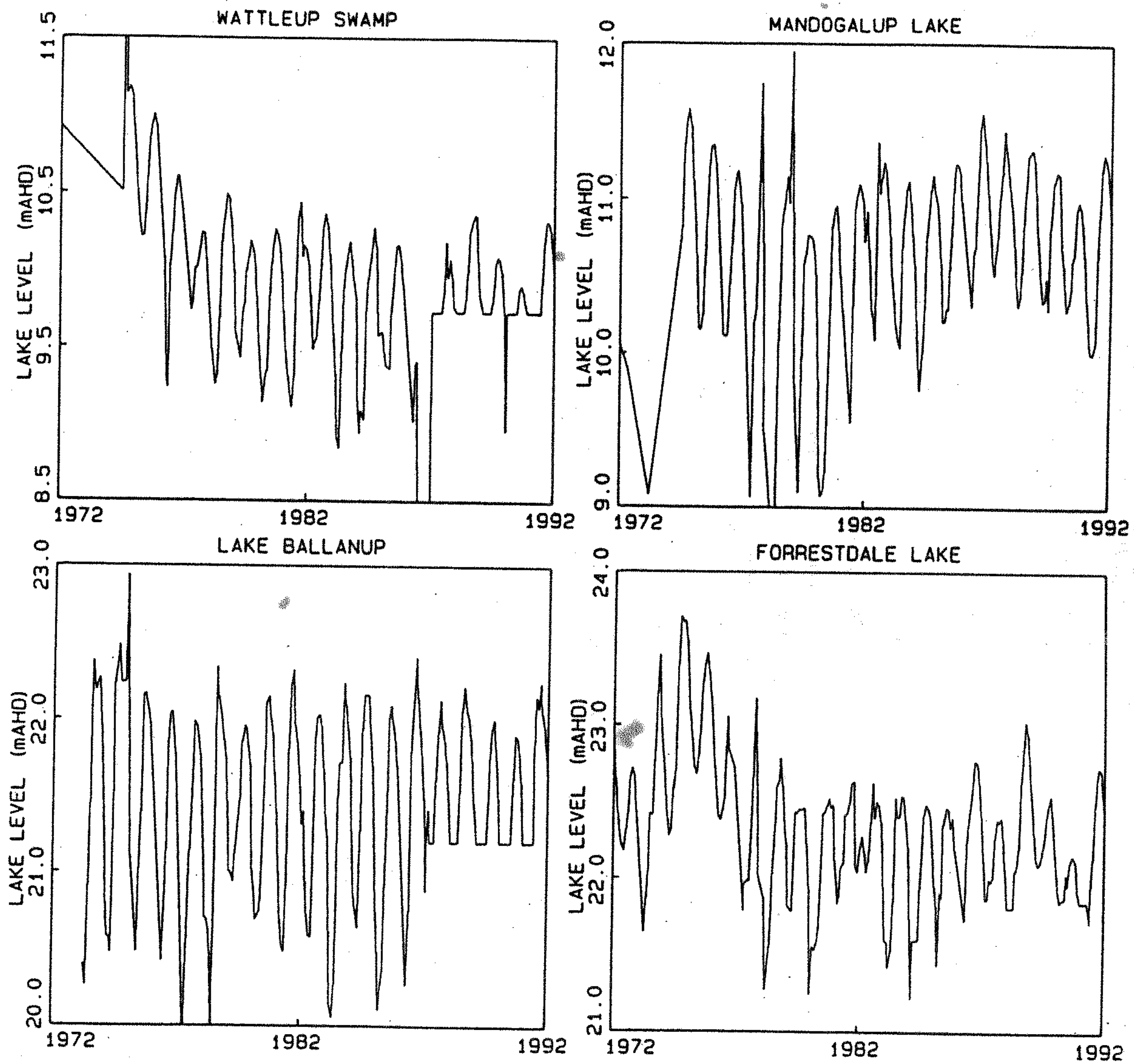


Figure 4.1.9 Lake water levels from 1 January 1972 to 31 December 1991

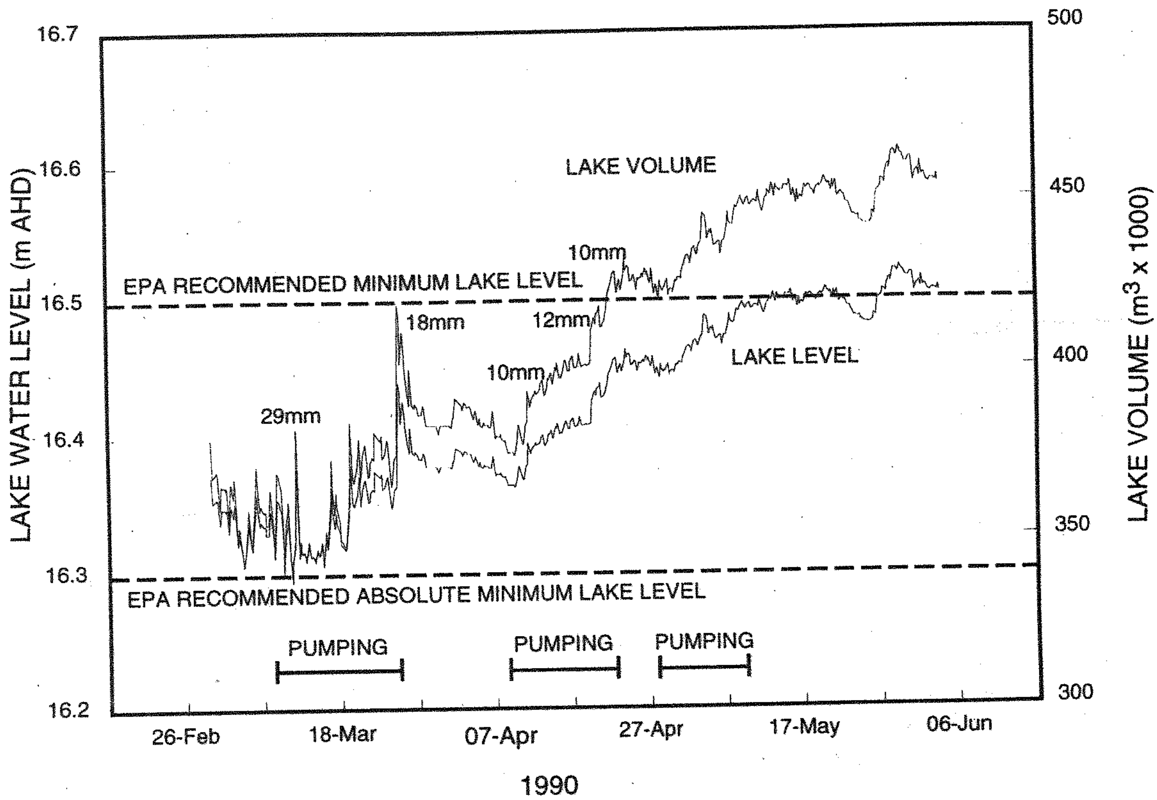


Figure 4.2.1 Lake water levels and volumes in Nowergup Lake during artificial water level maintenance in 1990

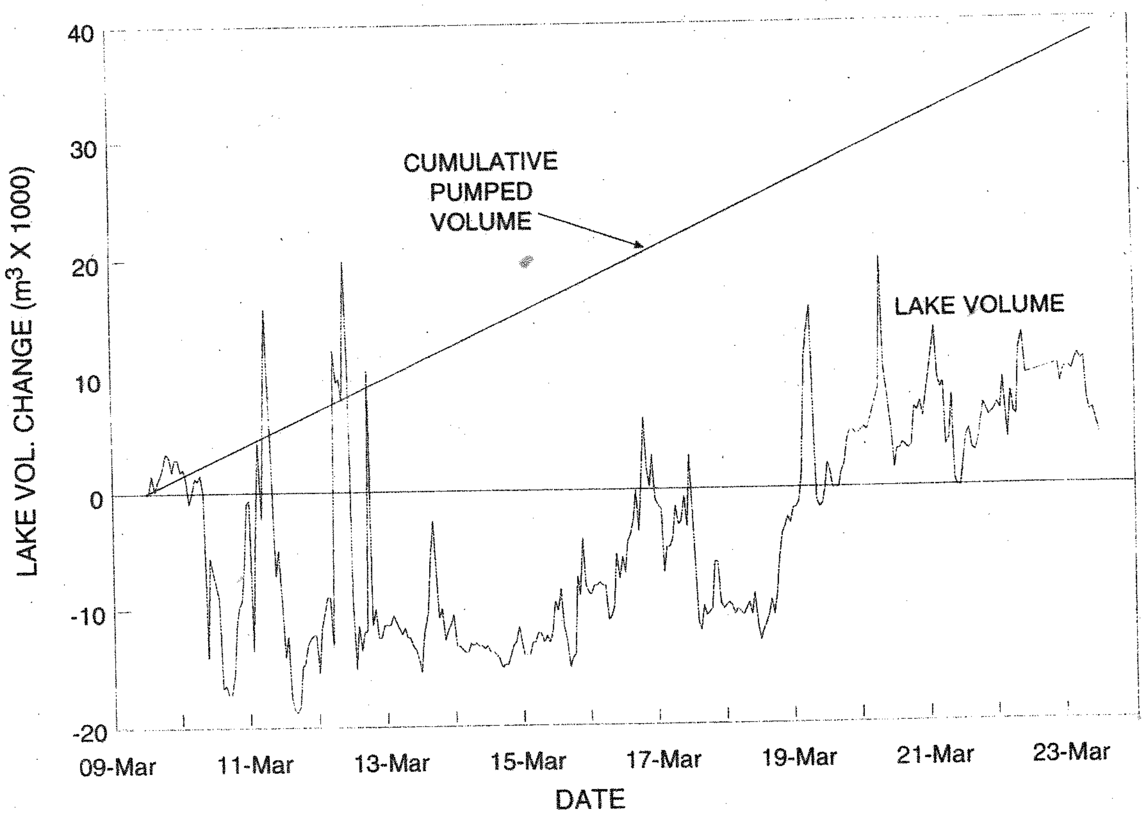


Figure 4.2.2 Hourly changes in lake volume in Nowergup Lake, 9-23 March 1990

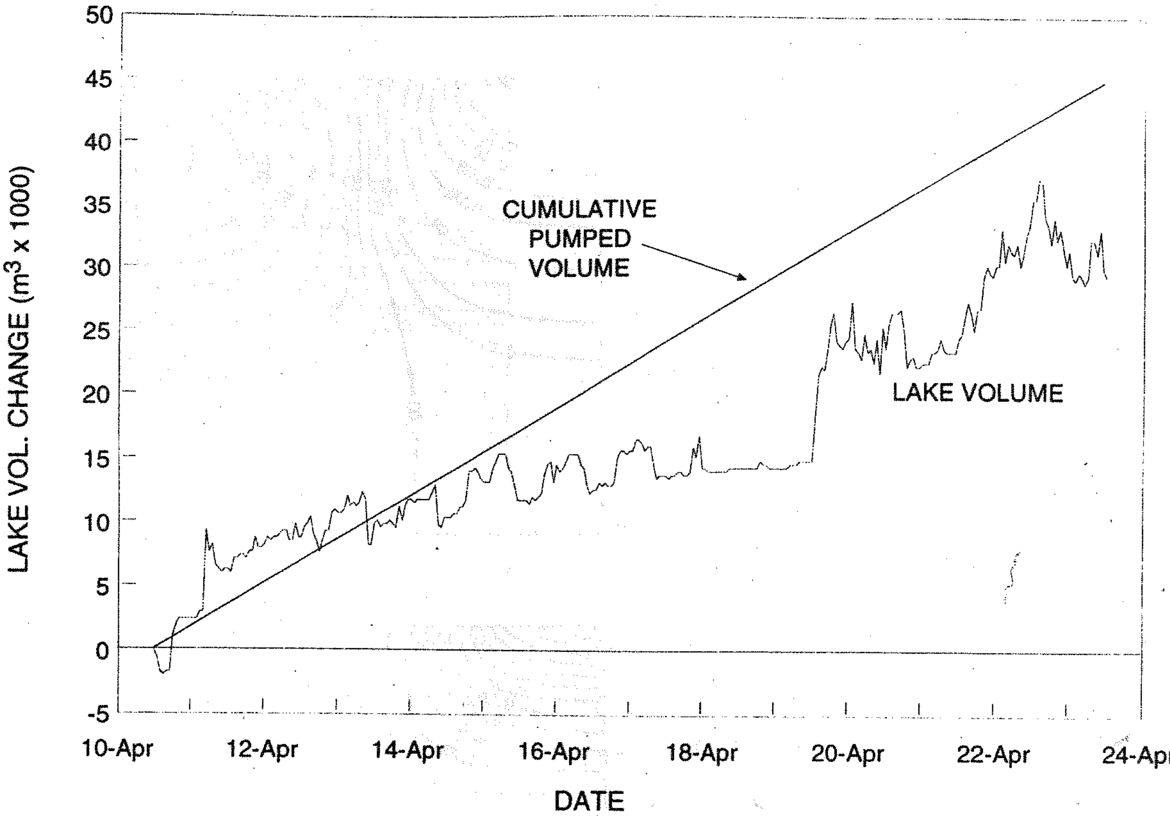


Figure 4.2.3 Hourly changes in lake volume in Nowergup Lake, 10-23 April 1990

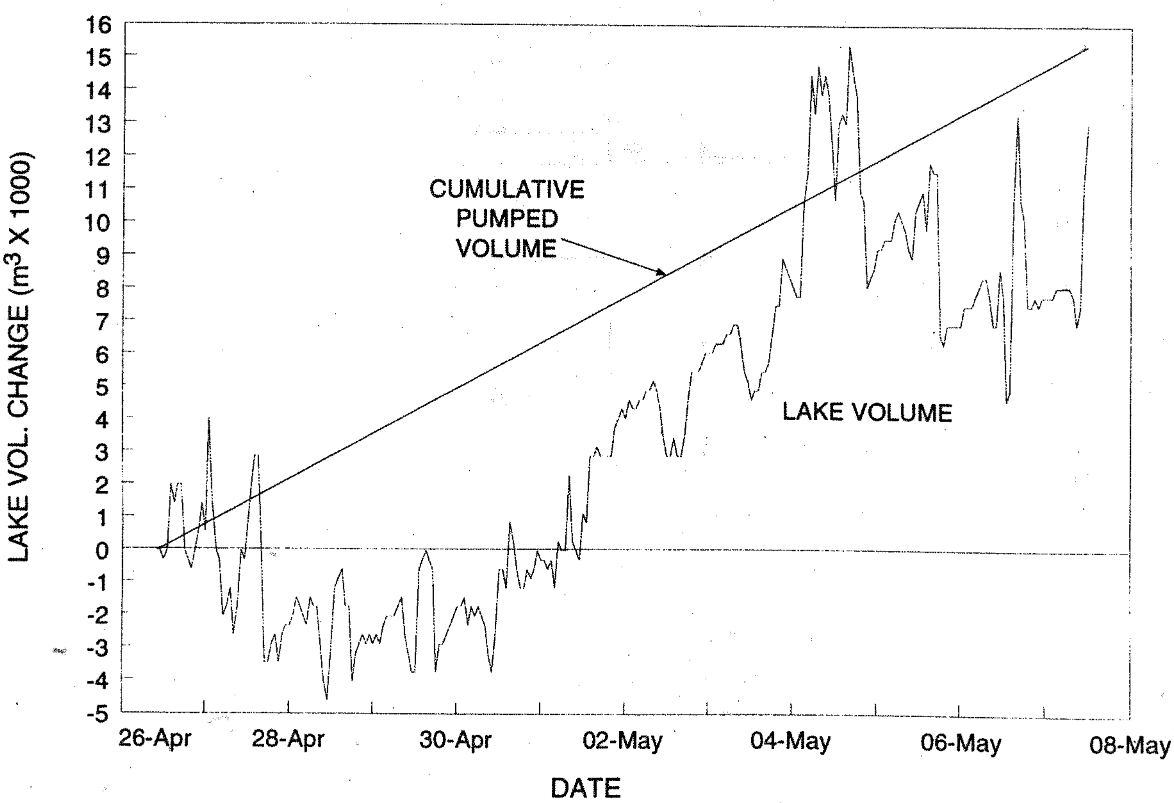


Figure 4.2.4 Hourly changes in lake volume in Nowergup Lake, 28 April - 7 May 1990

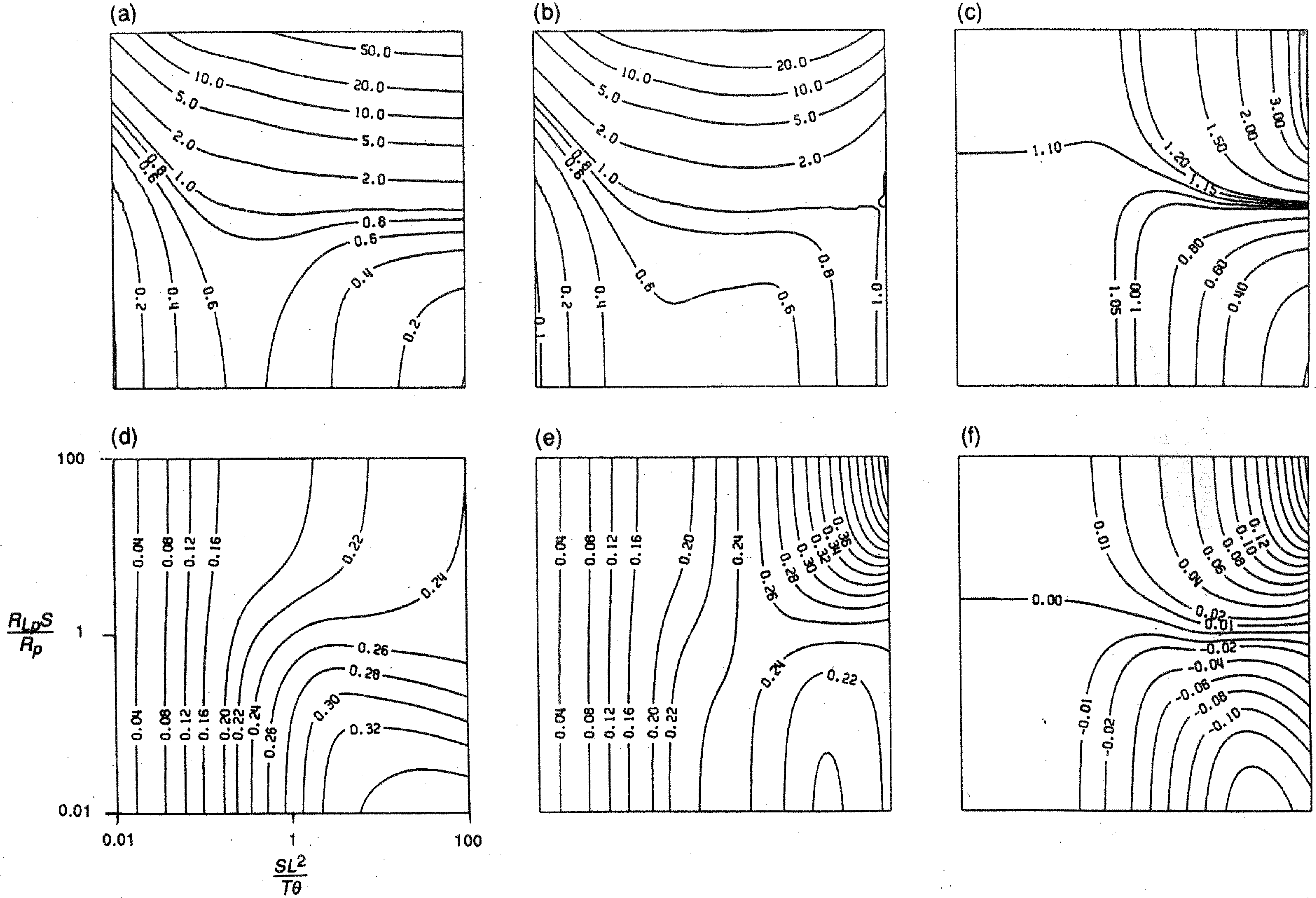


Figure 4.4.1 Non-dimensional results for 400m lake in middle of 10km aquifer

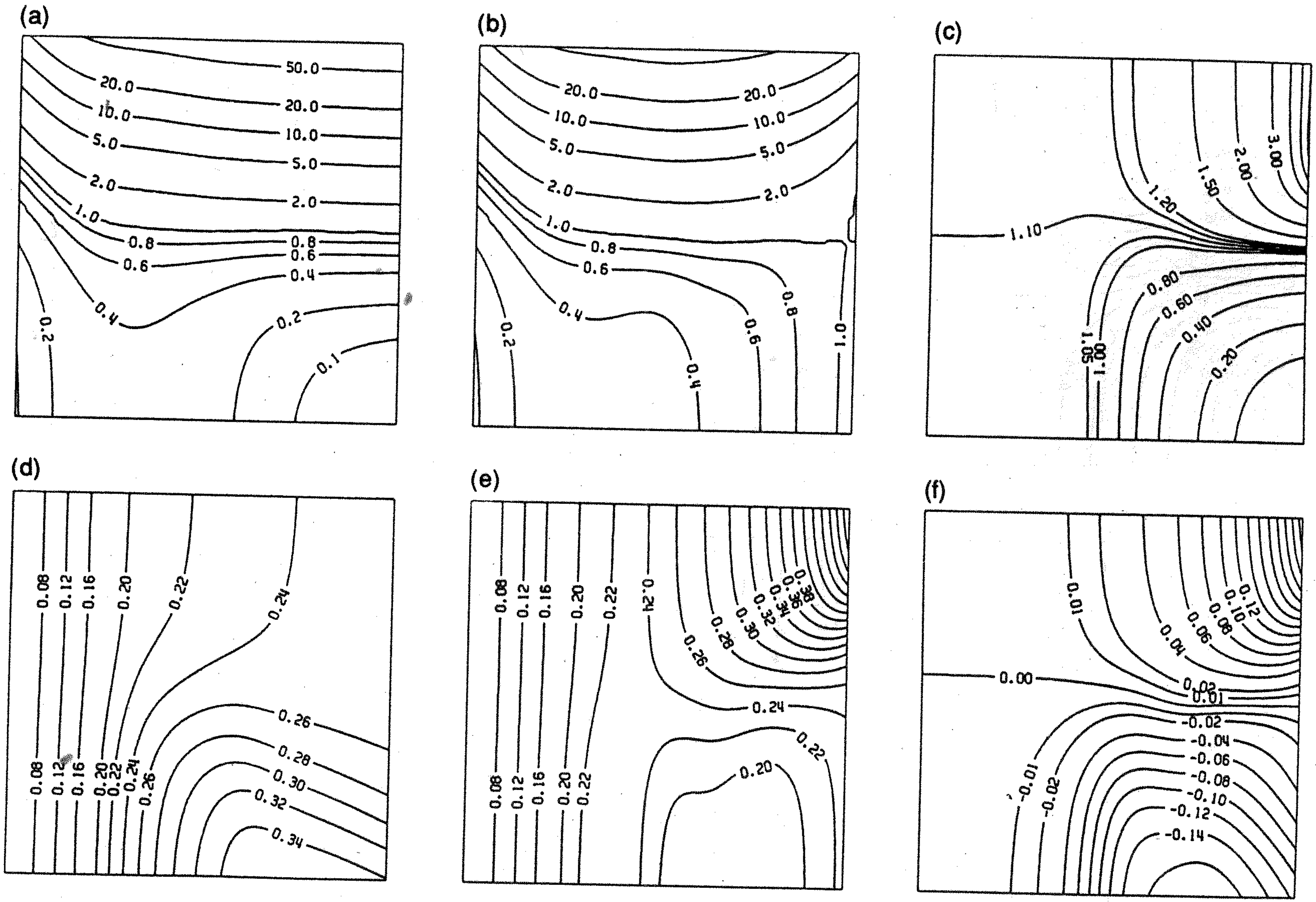


Figure 4.4.2 Non-dimensional results for 1.2km lake in middle of 10km aquifer

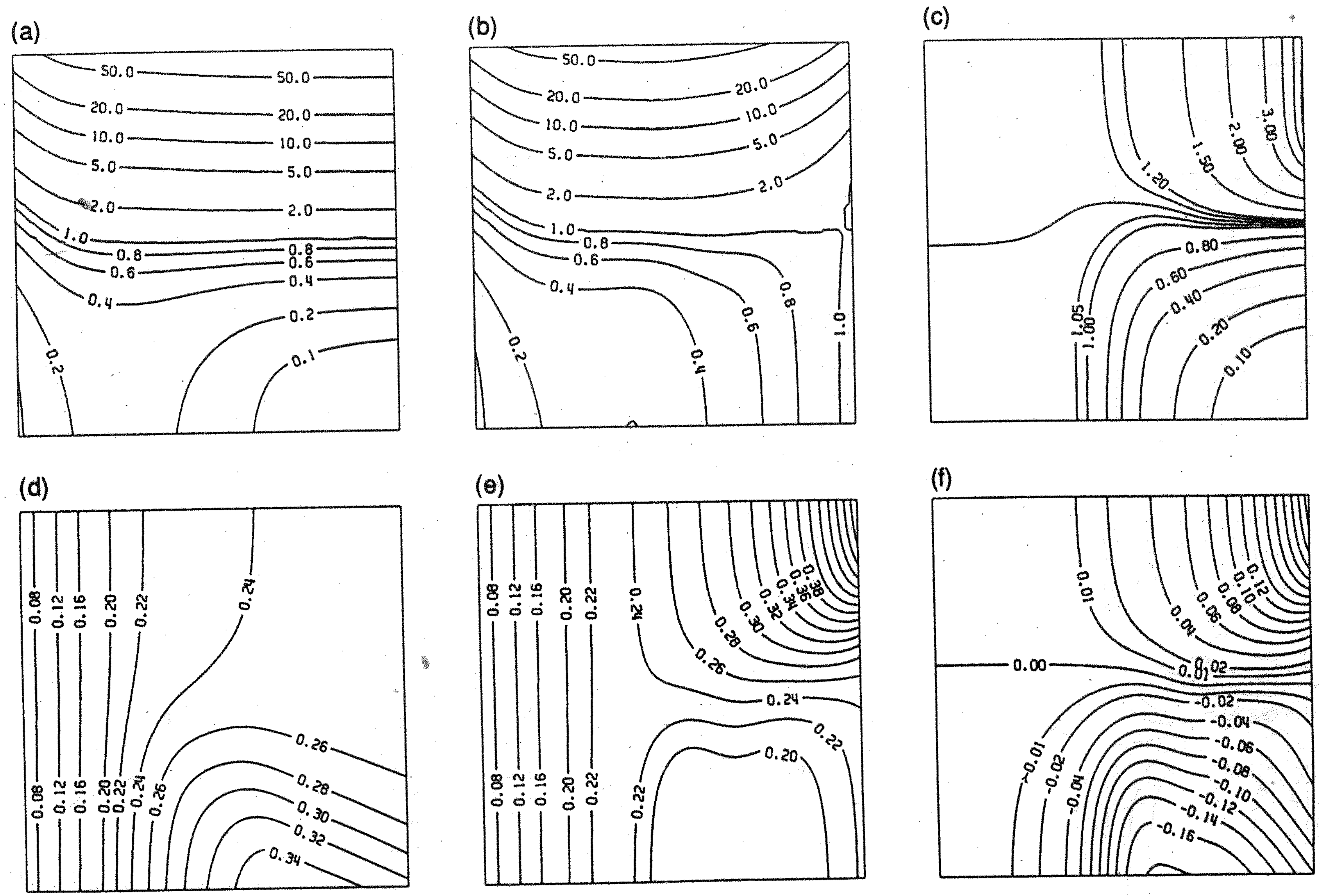


Figure 4.4.3 Non-dimensional results for 2km lake in middle of 10km aquifer

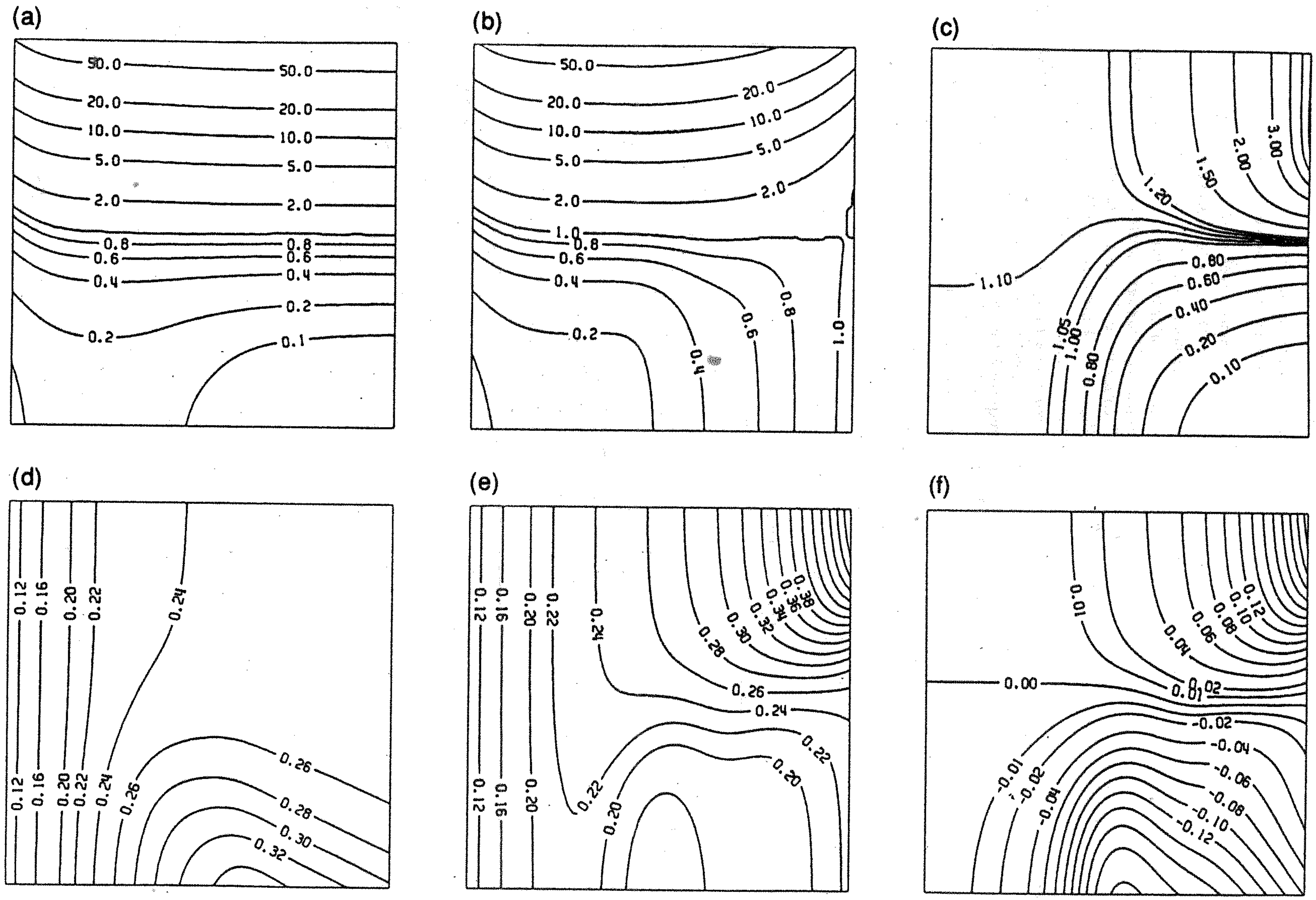


Figure 4.4.4 Non-dimensional results for 2km lake in middle of 10km aquifer, with $b_{+}/B = 0.5$

5. Effective Parameters for Models in Plan

5.1 Effective Transmissivities and Leakage Coefficients

Groundwater flow patterns near shallow lakes are fundamentally three-dimensional, but we have developed approximate methods for representing lakes in two-dimensional regional models of aquifer flow.

Managers of regional groundwater systems, such as the Water Authority of Western Australia, typically use two-dimensional models in plan to simulate the behaviour of the water table in response to climate and groundwater extraction. Such models are incapable of representing the vertical flows which occur near the upgradient and downgradient boundaries of shallow lakes. Shallow lakes in a regional aquifer attract groundwater towards them, thus one possible way of simulating the effect of a lake is to represent it as a region with a large transmissivity. Another possibility, as used by the Water Authority for many years, is to represent a lake using a "dummy" second aquifer, connected to the "real" aquifer by a leaky layer. These possibilities are illustrated in Figures 5.1.1 and 5.1.2. But until now, there has been no theoretical basis for assigning model parameters in either of these approximate methods.

During this project, we have developed guidelines for assigning large transmissivities to represent circular lakes in a one-layered model of a regional aquifer. By matching the widths of predicted capture zones using a two-dimensional model in plan and a three-dimensional model with a simple flow-through regime, we have calculated effective transmissivities which allow a two-dimensional model to reproduce the "attractiveness" of a lake in three dimensions. Figure 5.1.3 provides guidance for an effective transmissivity, T^*, relative to the regional aquifer transmissivity, T, as a function of lake length in an equivalent isotropic aquifer. To simulate a lake with $2a/B = 4$, for example, the transmissivity inside the area of the lake should be enhanced by a factor of about 8. To apply this Figure to an anisotropic aquifer, an equivalent $2a/B$ must first be determined by dividing the physical $2a/B$ by the square root of the anisotropy ratio.

We have also developed guidelines for assigning leakage coefficients to represent circular lakes in a two-layered model of a regional aquifer. By a similar matching process, we have calculated effective leakage coefficients to reproduce the "attractiveness" of a lake in three dimensions. Figure 5.1.4 provides guidance for values of D^*/B for use in a two-layered model, also as a function of lake length in an equivalent isotropic aquifer. This ratio can be converted to a leakage coefficient required by the model. Because of computing limitations, we only have confidence in these results for large lakes with $2a/B$ greater than about 2.

Complete details of how these effective parameters were obtained are provided by Townley *et al.* [1993]. It should be emphasised that Figures 5.1.3 and 5.1.4 do not remove the need for calibrating a model against available field observations. Their purpose is to provide guidance for first guesses for model parameters, or in another sense to help to explain why "best fit" model parameters converge to particular values.

5.2 Observed Hydraulic Conductivities of Lake Linings

Figures 5.2.1 to 5.2.4 present hydraulic conductivity data obtained by laboratory measurements on samples taken from the beds of six lakes. Some conductivities are very low, but we have not determined a relationship between these measured values and effective values for use in two-layered models. The effective leakage coefficients determined by matching three-dimensional capture zone widths with those in a two-layered model are based on having no low conductivity lining in the three-dimensional model. Intuitively, the effective values should be smaller if there is significant bottom resistance in the field. But counter to this is the suggestion that low conductivity linings in the middle of a lake do not affect the overall shape of a capture zone (Figure 3.2.7*d*).

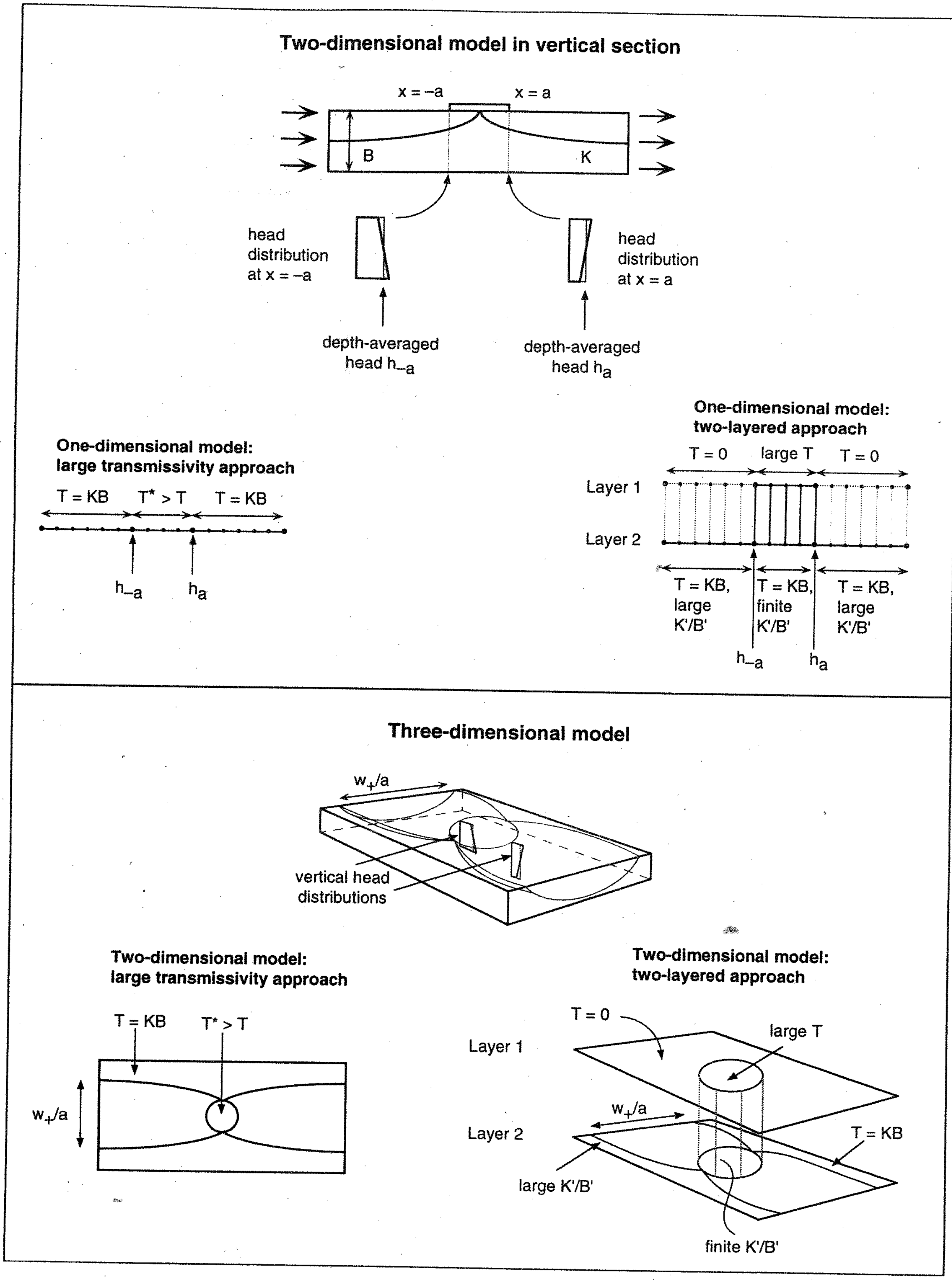


Figure 5.1.1 Overview of approximate lower-dimensional models

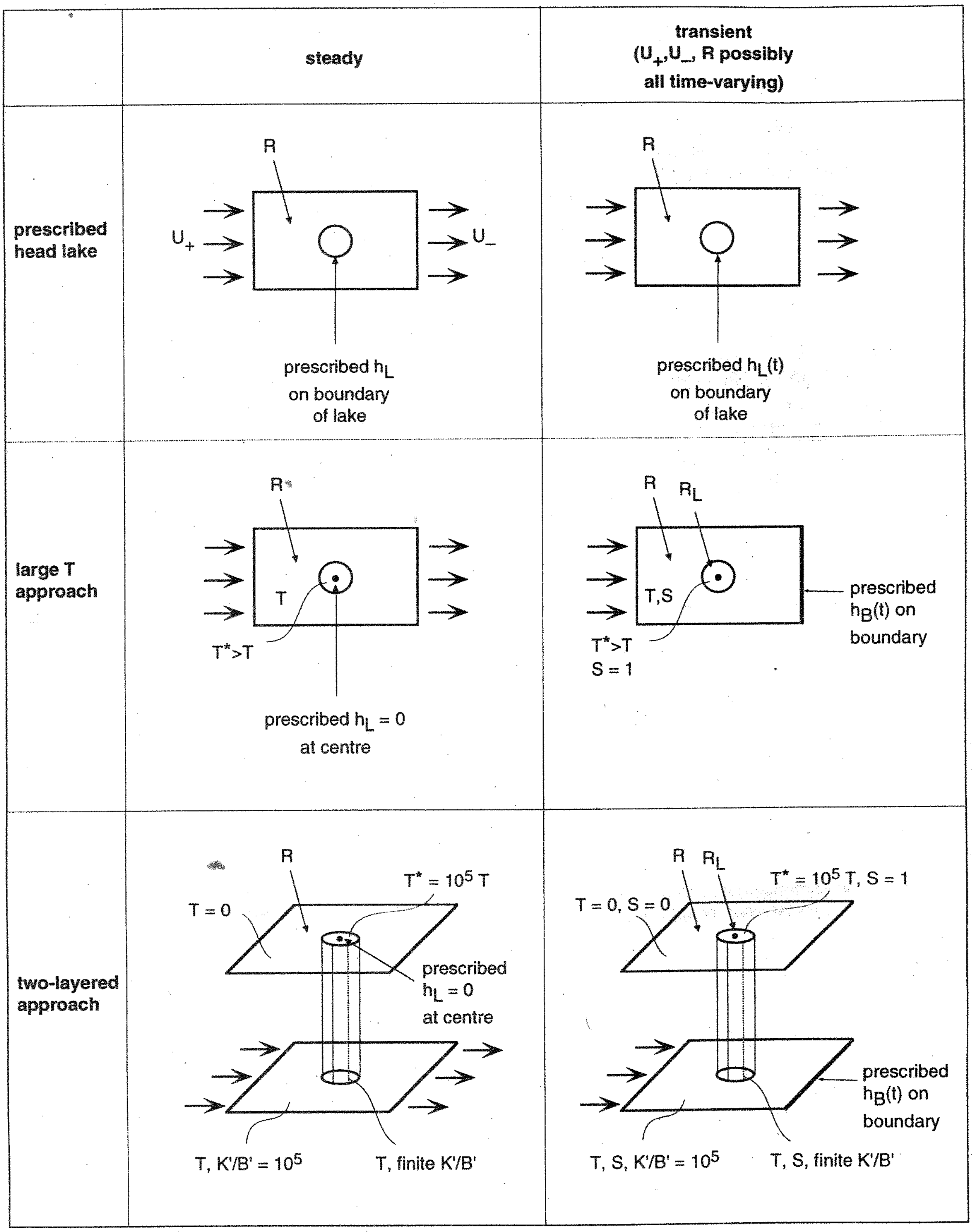


Figure 5.1.2 Summary of model parameters for different representations of flow in plan

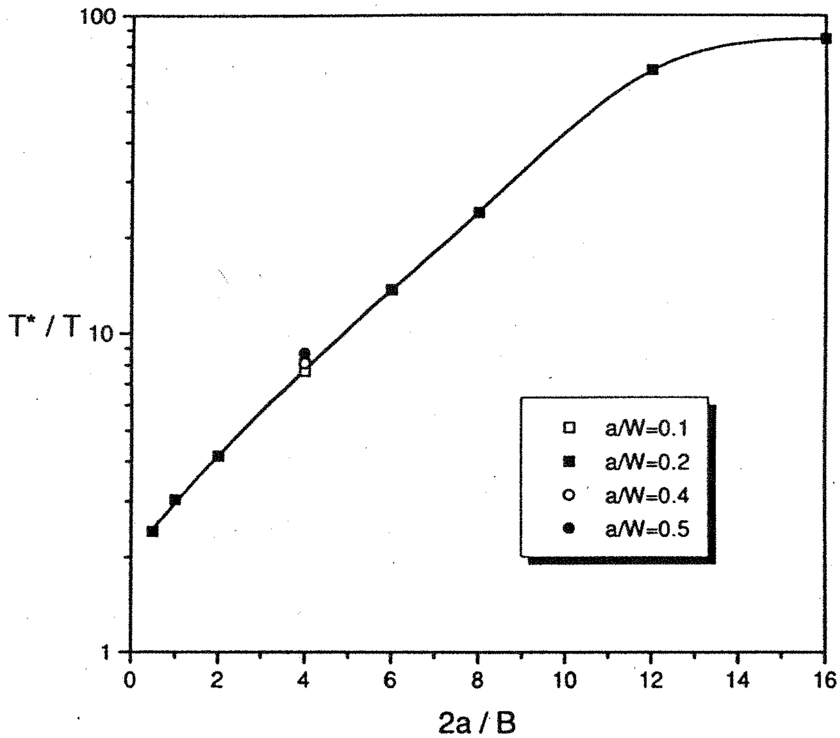


Figure 5.1.3 Effective transmissivity for use in a two-dimensional model

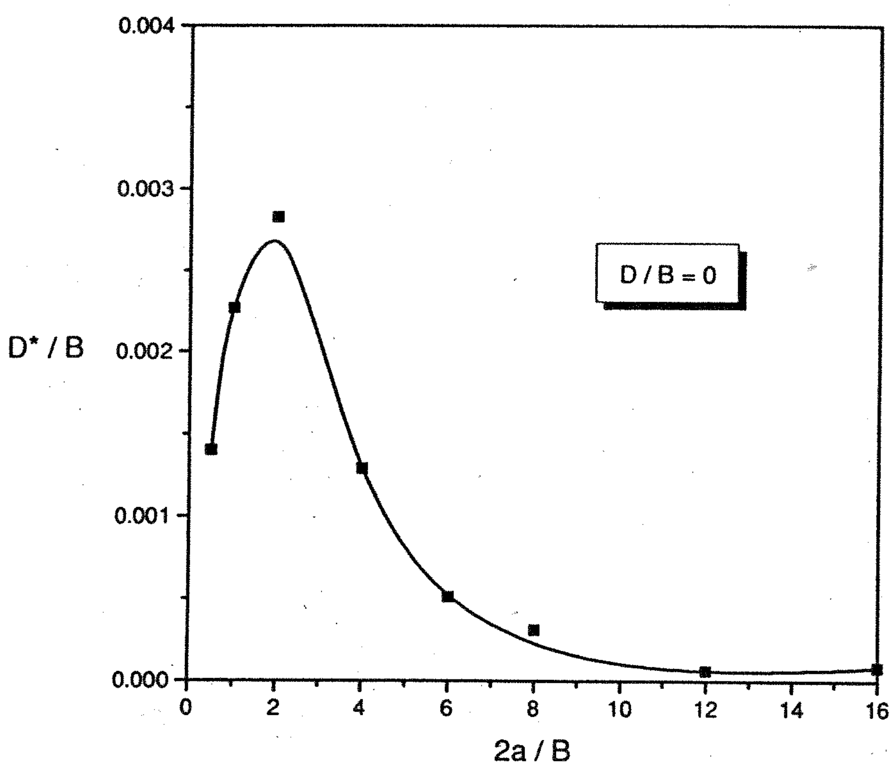


Figure 5.1.4 Effective values of D^*/B for use in a two-layered two-dimensional model

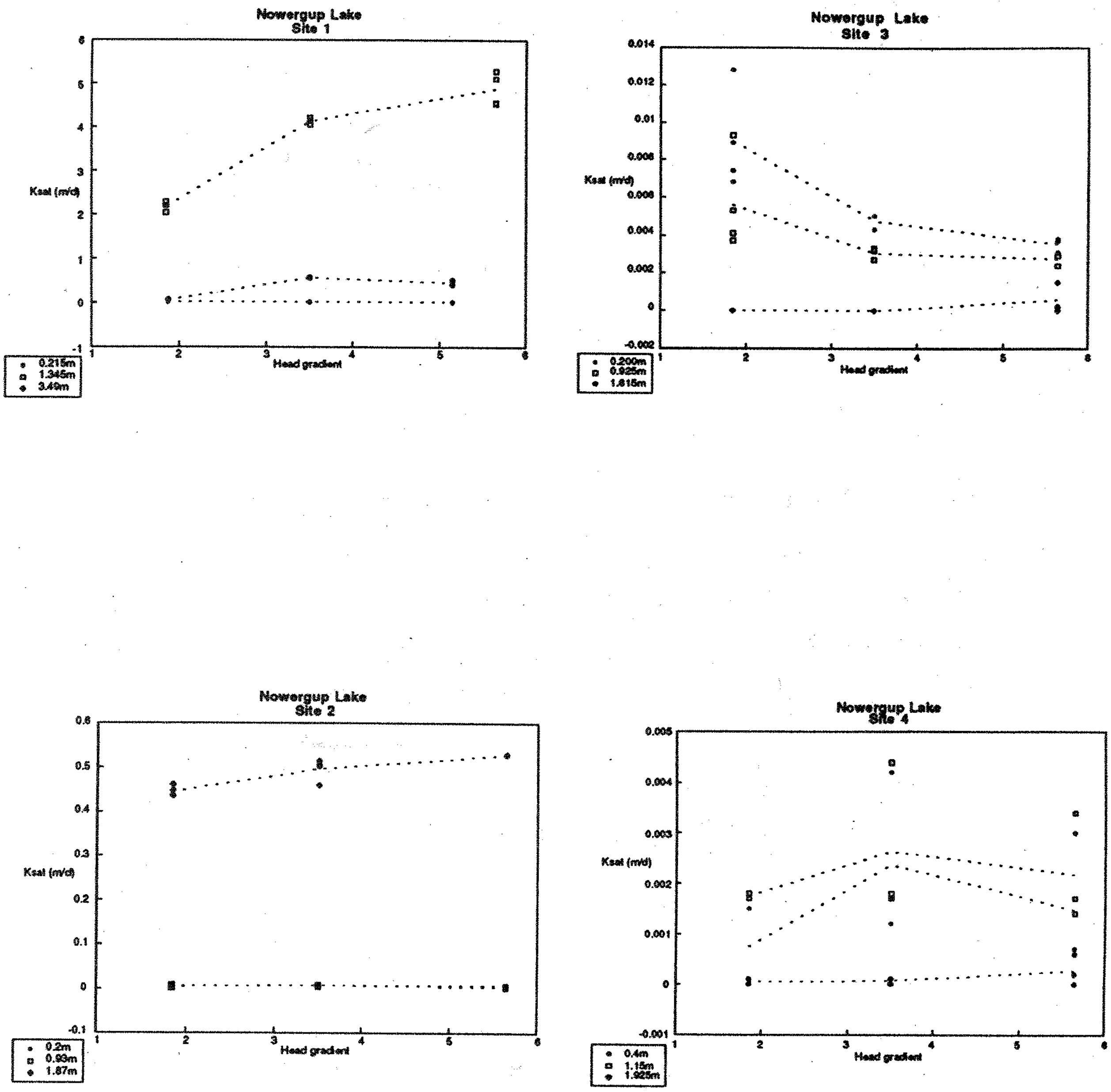


Figure 5.2.1 Hydraulic conductivities of core samples from Nowergup Lake

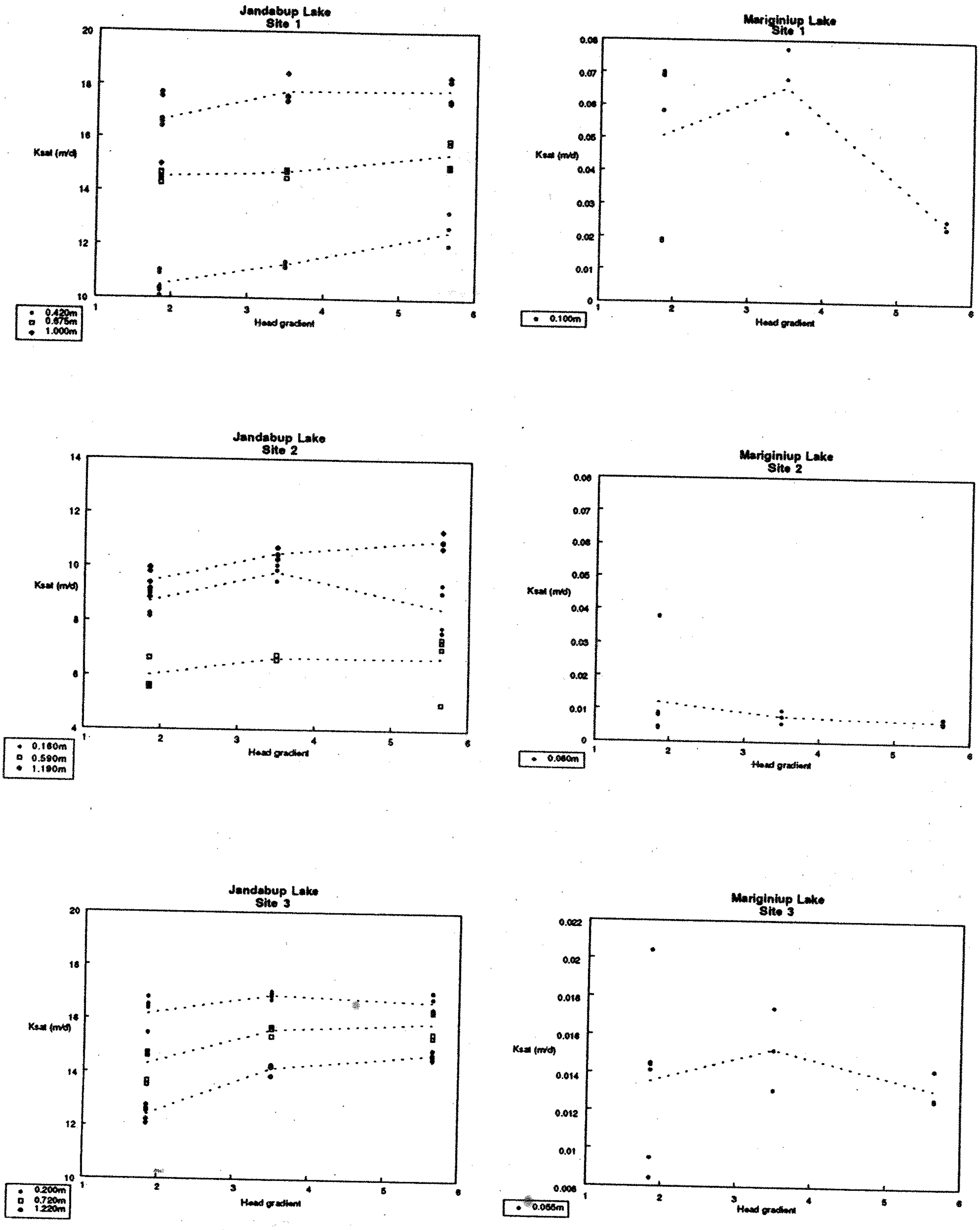


Figure 5.2.2 Hydraulic conductivities of core samples from Jandabup Lake and Mariginiup Lake

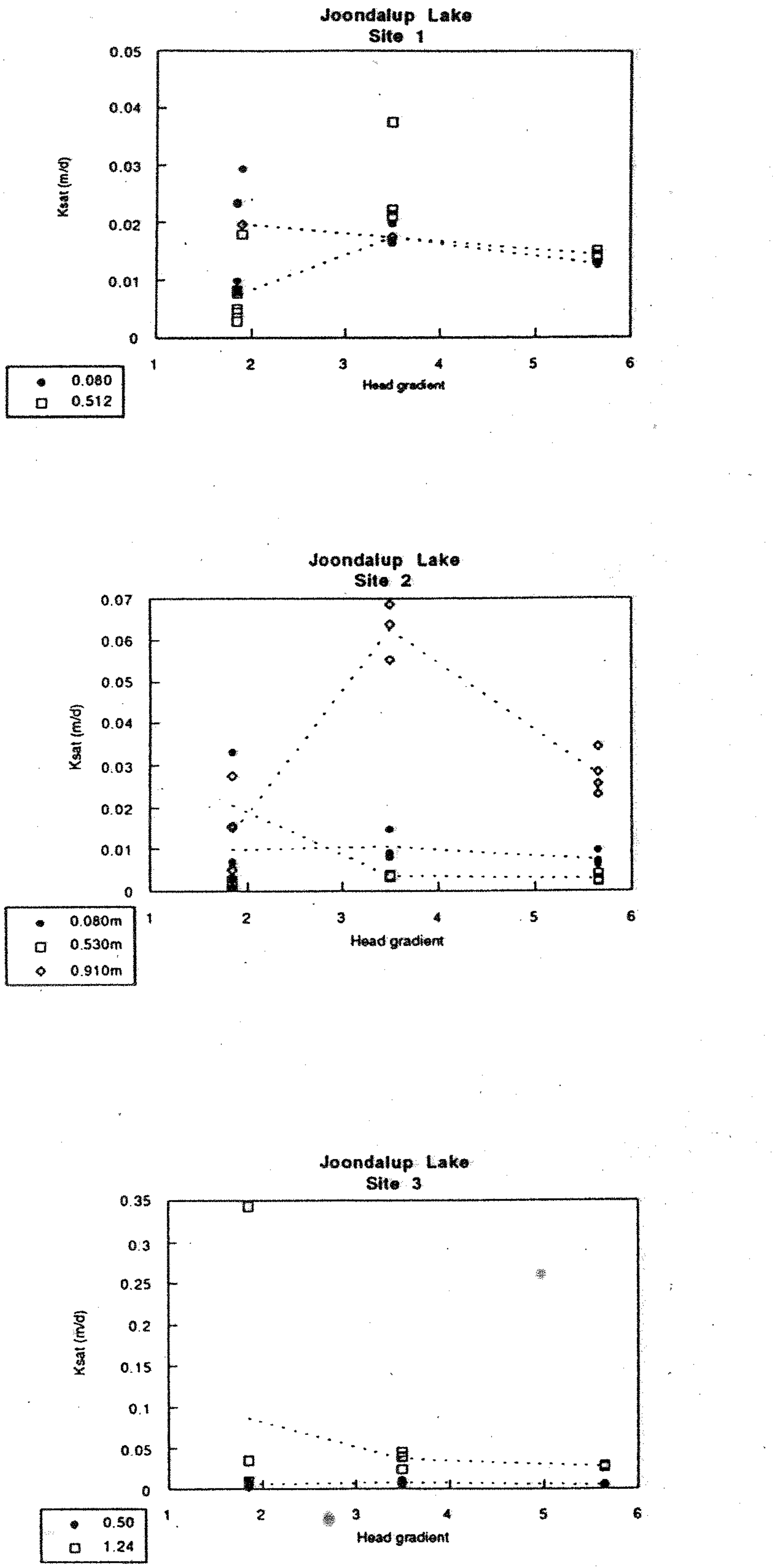


Figure 5.2.3 Hydraulic conductivities of core samples from Joondalup Lake

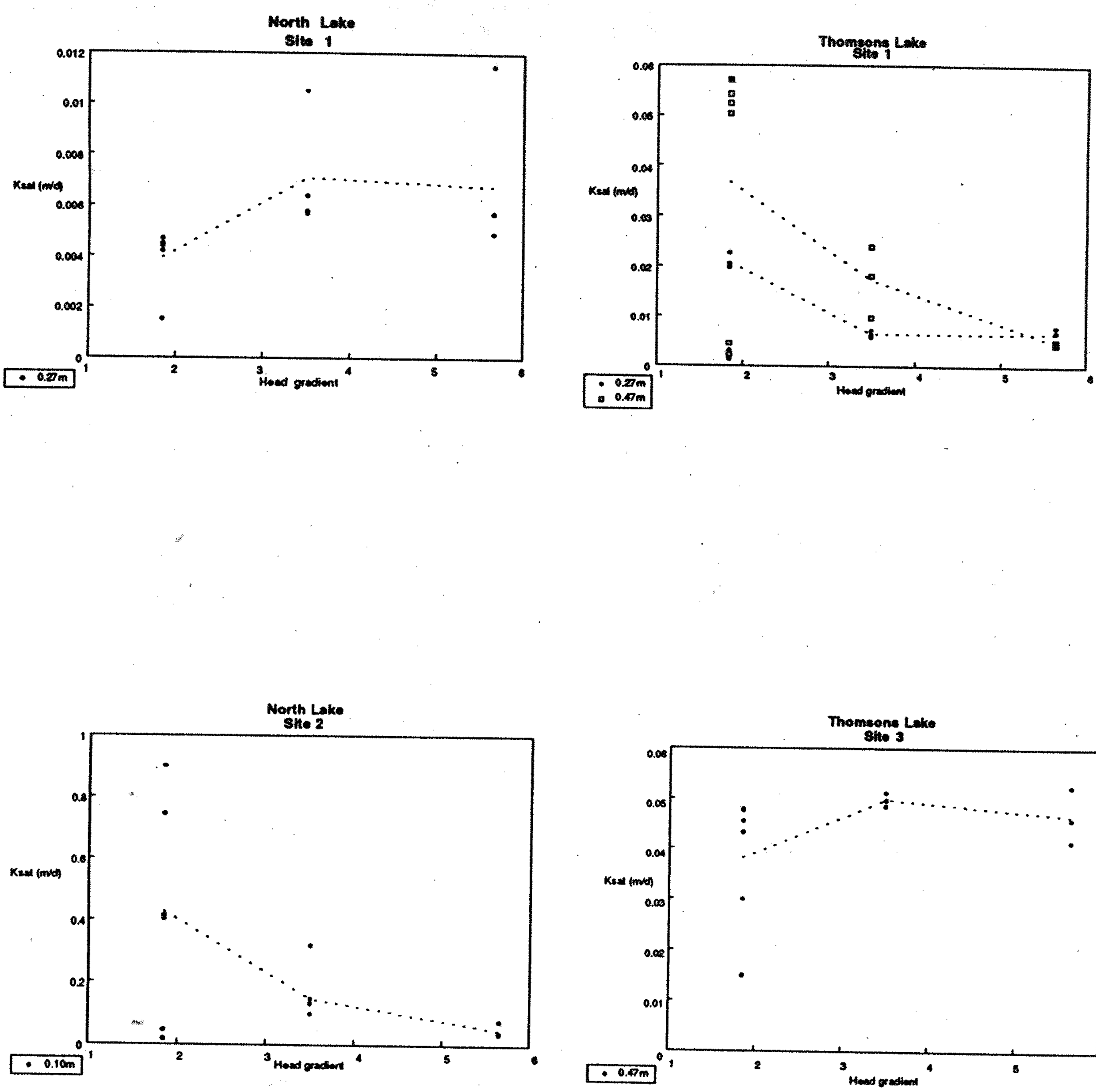


Figure 5.2.4 Hydraulic conductivities of core samples from North Lake and Thomsons Lake

6. Other Findings

During our research, we have conducted reviews of three topics and developed two mathematical approaches which we have not yet been able to implement. These activities are summarised here.

6.1 Phosphate Adsorption

In the process of reviewing simple methods for predicting the movement of phosphate fronts, we have discovered inconsistencies and developed a new method for predicting travel distance.

Appendix A of Townley *et al.* [1993] contains a review of models for phosphate adsorption, with emphasis on approximate methods for calculating travel distances and travel times of an assumed sharp front. We have discovered that several methods vary by about a factor of 10 in their prediction of travel distance after 100 years, for typical Western Australian soils. This difference may or may not be significant, depending on whether the difference is between 1 and 10 m or between 10 and 100 m. We have also developed a new method which has not yet been tested, but which takes account of continuing sorption behind a moving front. The fact that sorption continues with time means that a consistent model of a moving front must predict that the front slows down with time. Earlier models predict a constant front velocity, but our new method predicts a changing velocity.

The review suggests that it may be better in future to obtain phosphate adsorption characteristics using a phosphatostat, rather than using batch experiments. Conditions in a phosphatostat, with the concentration of dissolved phosphate held constant, agree more closely with conditions behind a travelling front.

6.2 Isotope Balance Equations

We have reviewed the development of isotope balance equations for evaporating water bodies, and summarised previous literature in a concise unified framework.

In the process of developing isotope balance equations for a flow-through lake in a vertical section (Figure 3.8.2), we reviewed a large number of papers and discovered a great diversity in notation, especially in the earlier years. The key result we needed was an expression for the isotopic concentrations in evaporating water, but this result was initially difficult to find in the literature. Appendix B in Townley *et al.* [1993] provides a comprehensive review of the development of this field, and shows how previous results are a special case of a more general result.

6.3 Directions of Flow in Anisotropic Media

We have summarised the correct way of determining the angle between equipotentials and directions of flow in a vertically exaggerated cross-section through an anisotropic medium.

There is a common misconception that the direction of groundwater flow is always at right angles to equipotentials in a vertical section. In fact the angle depends on anisotropy and the degree of vertical exaggeration of the section. A technique for determining the correct directions is presented in Appendix C of Townley *et al.* [1993].

6.4 Fully Coupled Surface Water - Groundwater Interaction

We have developed the theory for a fully coupled groundwater and surface water model, which solves simultaneously for groundwater and surface water levels in a vertical section or in three dimensions. Although we have been unable to implement it during this project, we have developed equations for a coupled model which simultaneously solves for lake elevation and the distribution of piezometric heads. The equations are written in matrix form, ready for implementation by either finite difference or finite element methods.

6.5 Uncertainty in Water Balances

We have also developed a method for combining measurements of all the components of a lake water balance to obtain estimates of the same components which are constrained to satisfy an exact water balance.

Water balance methods often involve measuring all but one of the components of a water balance and calculating the last by difference. From a philosophical point of view, it is better to measure all components, recognise that all measurements are uncertain, and then to calculate improved estimates of all components which satisfy the required water balance equation. We have developed a simple methodology for doing this, but have not yet applied it to practical lake water balance calculations.

7. Research Opportunities

Good research often asks more questions than it answers, and this project has been no exception to that rule. In this final Chapter, we present a list of possible future research activities to build on our findings:

- There is an opportunity to carry out further steady state simulations of lake-aquifer interaction, building on the results described in Chapter 3. A conscious decision was taken during this project to assume that aquifers are homogeneous in the near field, i.e. that hydraulic conductivities are spatially uniform. Clearly this is not the case, and many lakes in the Perth region occur at boundaries between different hydrogeological or geomorphological units. Site-specific models of Nowergup Lake or Thomsons Lake, for example, should probably include distinct regions with different hydraulic conductivities upgradient and downgradient of the lakes. Tóth [1963] and Freeze [1969] carried out simulations in cross-sections with regions of different conductivities, but such simulations could be performed systematically for regions containing single or multiple surface water bodies.
- Three-dimensional modelling can be extended in several ways. First, the results reported here do not include a low conductivity lining, but this is simply a special case of the heterogeneity referred to above. Leaky linings need not be continuous over the whole of a lake bottom, and could have different conductivities in inflow and outflow regions, depending on the direction of seepage. Some lakes may have virtually impermeable bottoms, except for a narrow ring around the shoreline. Second, we need to increase our ability to model very large systems with millions of nodes, especially in order to simulate interactions between several nearby surface water bodies. It may be advantageous to use special equation solvers designed for multiple processors, or to use multi-grid modelling techniques. There may be an advantage using finite element models in three dimensions, possibly with special elements which take into account the nature of known singularities near the edge of a lake. Third, there is much to be done in improving three-dimensional visualisation of flow patterns. The applicability of a three-dimensional streamfunction needs to be investigated.
- There is a need to develop modelling software which is capable of simulating lake-aquifer interaction in complex situations, in order to overcome limitations imposed by simplified models with approximate geometries and homogeneous aquifer properties. Apart from the need to operate within a user-friendly graphical interface, modelling tools are required which properly couple the surface and the subsurface in a vertical section and in three dimensions, using the theory developed during this project. Such models need to be able to handle the phreatic surface boundary condition, and the problem of changing lake surface area as lake levels change. Future models need to be much more applicable to transient problems.
- The methods described in Chapter 5 for representing lakes in two-dimensional plan models need to be properly tested in practice. These methods have been developed so that the widths of capture and release zones are matched in steady flow-through situations. We need to investigate how well the parameterisations work when flows are unsteady, and when aquifer flow towards a lake is not equal to flow away from the lake. Such work could easily be carried out by the Water Authority as part of future regional modelling studies.
- In general, our understanding of transient behaviour of lake-aquifer systems is in its infancy. We believe that groundwater flow regimes can cycle seasonally between several fundamentally different regimes, as defined in Chapter 3. There is an opportunity to carry out specific studies on individual lakes, especially small lakes or larger lakes with low conductivity linings, in order to prove that our hypotheses are correct. It may be possible to carry out periodic modelling in a vertical section using AQUIFEM-P [Townley, 1993*b*], especially to investigate the effects of anisotropy on fluctuations in flow paths near lakes. There is also an opportunity to work with the Water Authority on regional modelling using the the Perth Urban Water Balance Model. The latter includes a dynamic Vertical Flux Model which predicts the spatial and temporal variation of the source term to the aquifer flow model. Dynamic variations in this "recharge" term may influence local behaviour near surface water bodies.
- As another example of heterogeneity in large systems, we need to extend our understanding of lake-aquifer interaction to include deeper aquifers. All of the modelling reported here assumes a single homogeneous aquifer, even though the unconfined aquifer in Perth occurs in the superficial formations, which in many places rest unconformably on the Leederville Formation. Just as a lake induces upward flow from the bottom of an aquifer, it also induces upward leakage from deeper aquifers. Conversely, depending on relative magnitudes of hydraulic conductivities and other parameters, a flow-through lake may discharge water to lower

aquifers. Lakes on the Swan Coastal Plain near Perth should not be seen only as a windows into the unconfined aquifer, but also as windows into deeper systems as well. There is an opportunity to carry out systematic modelling of layered systems, on a range of scales.

- Modelling has shown that the shape of a lake's capture and release zones depends on the large-scale anisotropy of the aquifer material. Whereas aquifer tests by pumping can be used to estimate aquifer transmissivities and storage coefficients, it is very difficult in practice to determine vertical hydraulic conductivities and hence anisotropy ratios. Simply by their presence, surface water bodies induce vertical components of flow and therefore provide an opportunity for observing the effects of anisotropy. It would be useful to use field observations of release zones to infer the value of average anisotropy near a surface water body.
- There are opportunities for two types of regional hydrological studies on a range of lakes on the Swan Coastal Plain. This project has predicted many types of lake-aquifer interactions and has verified a few. In a sense it has provided a framework and tools that will allow others to make more progress in the future.

  The first type of study includes detailed studies of specific lakes. It would be timely to carry out an intense investigation of a single lake, aiming to determine its water balance, but using solute, isotope and thermal balances as well. Such an investigation would need at least two years of field monitoring, with much emphasis placed on measuring evaporation from the open water surface, and on estimating recharge and evapotranspiration nearby. Careful experimental design is needed to allow measurement of lake and water table elevations which will confirm the seasonal cycle between different flow regimes. It would be possible to use chemical and isotopic measurements to identify the widths of lake release zones in plan, since no field experiments of this kind have been carried out.

  The second type of study is more regional in nature. Now that we know some of the possible differences between different lakes, it would be timely to characterise lakes in the region based on our framework. It would be possible to build up a database of key parameters describing geometry, aquifer properties, lake linings, lake level and nearby aquifer response, chloride and isotope measurements etc., and to make and/or check predictions based on this report. A regional study of this nature could identify even more clearly the key role of surface drainage in influencing lake level fluctuations. It could also lead to an estimate of the spatial distribution of anisotropy in the unconfined aquifer.
- Although this project started with a clear emphasis on lakes on the Swan Coastal Plain, it has become clear that there are many other types of surface water bodies to which our findings could be applied. There are direct applications to long water bodies such as canals, drains, rivers and streams, and it would be interesting to carry out such work. In many parts of the world, lakes occur in river basins, as part of a surface drainage network. Such lakes can also also receive and discharge groundwater, i.e. they may act as recharge, discharge or flow-through lakes in terms of our classification, regardless of the magnitude of surface water throughflows. It would be interesting to extend the analysis by Brainard and Gelhar [1991] to include a lake in a drainage channel. Such an analysis would be relevant to many lakes in Western Australia, such as Lake Toolibin and various lakes in the Kent River basin (which is now under study by CSIRO and other Western Australian government departments). Groundwater flow patterns near junctions in a stream network also provide an interesting topic, as patterns are likely to change seasonally or during runoff events as levels rise and fall in different parts of the network.
- We have suggested that shallow lakes on the Swan Coastal Plain may be important in controlling the stability of the regional hydrologic system. As lake levels fall, the lakes become smaller in area, attract less aquifer flow and lose less water by evaporation, thus reducing the tendency for levels to fall. Conversely, as lake levels rise, the lakes become larger, attract more aquifer flow and lose more water by evaporation, thus reducing the tendency for levels to rise. It would be interesting to study historical fluctuations in lake levels, and to relate these changes to climate change. Dynamic feedback of the type suggested here can play an important stabilising role.
- All modelling in this report assumes that the concentrations of dissolved constituents in groundwater and lakewater are so small that density variations can be safely neglected. This is not the case in salt lakes in inland Australia, nor in ponds constructed for solar salt production, as at Shark Bay in Western Australia. In such cases, there are numerous effects of density differences, including the possibility of "fingering" as salty water migrates downwards [Barnes et al., 1990]. Salt lakes often form in topographically closed basins [e.g. Duffy and Al-Hussan, 1988], but can also occur in flow-through situations. Wood and Sanford [1990] and Sanford and Wood [1991], for example, have shown that the rate of outwards seepage can determine not only the concentration of salts in lakewater but also the composition of evaporite deposits. There are many opportunities

to extend our new knowledge of groundwater flow patterns near lakes and other shallow surface water bodies to more complex situations where density-driven flow and chemistry become important.

We look forward to working with government agencies, universities and other organisations in the future.

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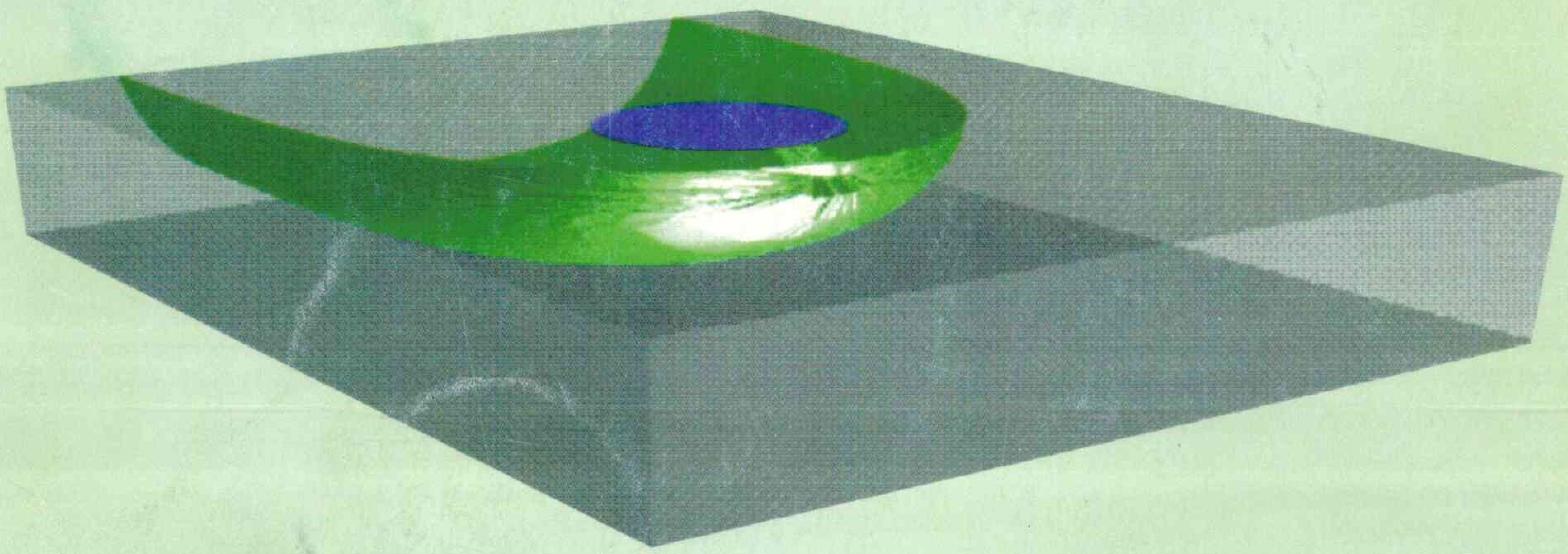

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