THE IMPACT OF STORMWATER RUNOFF ON THE HYDROLOGY AND CHEMISTRY OF AN URBAN LAKE

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ABSTRACT

The flow dynamics and water balance at an urban lake were studied in order to determine the modifications to the lake hydrology and chemistry resultant from large inputs of stormwater runoff. The investigation was aimed at providing answers to a series of environmental management problems at the wetland and to assess the potential for pollution of the groundwater through the recharge of stormwater. The boundary of the catchment contributing runoff water to the lake was established and, within this, a single sub-catchment was selected from which direct measurements of discharge in one stormwater drain were taken using a doppler instrument. From these records a rainfall/runoff relationship for the entire catchment was obtained. Variations in groundwater levels were measured continuously in five piezometers in a transect across the lake. The piezometers were installed so as to just intersect the water table. These records were then used to calibrate the modelled responses of the lake/aquifer system to stormwater recharge, calculated via the AQUIFEM-N model (Townley, unpublished).

The lake water balance was shown to be dominated by the stormwater runoff component so that the lake's 'natural flow pattern' was transformed during winter to a situation whereby it functioned as a continuous recharging source to the aquifer. Stormwater was recognised as being the major source of phosphorus to the lake, while low concentrations of nitrogen in the runoff waters transformed the water body from being hyper-eutrophic in nitrogen over summer to an oligotrophic status in winter. Concentration of nutrients via evaporation partly explained high levels of nitrogen in the lake over summer, but an analysis of flow regimes also suggested that leaching of nitrogen from a former landfill site was also restricted to this season. It was concluded that stormwater recharge at the site had no detrimental effects on the quality of groundwater in the area, due principally to the low concentration of contaminants in the runoff waters and the function of the wetland as a large settling pond for the assimilation of heavy metals and nutrients.

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1. INTRODUCTION AND LITERATURE REVIEW

1.0 Introduction

The basis of this study was the recognition of a environmental management problem at Shenton Park Lake, where low water levels in summer were resulting in concentration of nutrients via evaporation and eutrophic conditions. Associated with the eutrophic conditions were algal blooms and subsequent outbreaks of botulism among the lake's bird population. Although the sculpted surrounds of the lake, which have been fashioned into a Victorian style parkland, give the distinct impression of the wetland being a large "goldfish" bowl; the lake shares similar disturbance effects to those seen in other wetlands in the Perth metropolitan region.

Potential sources of nutrients entering the lake were identified as fertiliser runoff from surrounding parkland, excretia from the local bird population, seepage from a former landfill site on the northern boundary of the lake and stormwater runoff inputs through a series of drains.

Typically water quality related studies on wetlands are conducted so as to describe the chemistry of the wetland and emphasis is often placed on prescribing nutrient fluxes as part of an overall nutrient budget. Many of these studies are compromised, however, in the calculation of nutrient loadings, by the limited attention given to accurately determining the components of the water balance (Winter 1981). Rather than attempting an all embracing study of the water and nutrient balances, which was beyond the scope of the project, the dominant component of the lake water balance (as identified in McFarlane 1984), stormwater runoff, was investigated in detail.

Historically wetlands in Perth have been considered as convenient sites for the recharging of stormwater runoff to the groundwater. This practice is economically very efficient, but the impacts of large peak discharges of stormwater into these basins has not been extensively studied; particularly in terms of its effects on wetland hydrology. The potential of stormwater as a pollutant is of particular concern, and recent studies (Appleyard 1994) have aimed at determining the possibilty of contamination of groundwater through stormwater recharge at compensating basins.

This investigation had three basic aims.

Firstly to determine the size of the stormwater component of the water balance and the modification to the natural flow dynamics of the lake brought about by this large recharge component. This study would involve the modelling of the lake system from the end of summer and through the winter period using principally the AQUIFEM-N (Townley, unpublished) package with reference to the FLOWTHRU (Townley et al. 1992) model. Secondly, to detail the chemistry in the lake over the summer and winter seasons to determine changes in chemistry and nutrient levels with the influx of stormwater. Related to this work is a discussion on the likely impact of recharge water from the lake on the chemistry and water quality of the surrounding groundwater. The final objective was to integrate these studies in order to provide answers to the environmental management problems at the lake.

1.1 Geomorphology and hydrology of wetlands on the Swan Coastal Plain

Generally the lakes and swamps of the Perth wetlands (and the majority of lake systems on the Swan Coastal Plain) are connected to large, shallow, regional unconfined aquifers with water levels varying accordingly with the height of the water Table (Allen 1976). In consequence it can be stated that both lakes and wetlands of the Swan Coastal Plain are affected by and affect the behaviour of the regional unconfined aquifer (Townley et al. 1993). The exception to this are perched waterbodies where connectivity to the groundwater is precluded by an impervious or low permeability bed. These wetlands are common to the Pinjarra Plain and the foothills of the Darling Scarp, where duplex soils and duricrust layers prevent infiltration.

The Seminiuk (1987) classification of wetlands is based on the permanency of a waterbody in terms of its position relative to the groundwater. This classification system most accurately accommodates the process of seasonal drying which makes the hydrology and biology of these wetland types distinct. Lakes are wetlands in depressions in the land surface which have an exposed open water surface throughout the year. Sumplands have an exposed open watersurface for part of the year but are seasonally dry. Damplands never have an exposed open water surface, but are water logged during most of the year (Townley et al. 1993).

The range of wetlands on the Swan Coastal Plain is diverse owing to differences in the soil type, drainage patterns, vegetation and topography associated with the four major geomorphic units that form the coastal strip. In a progression from the Darling Scarp to the coast the major geomorphic units are the Pinjarra Plain and the Bassendean, Spearwood and Quindalup Dune systems (McArthur & Bettenay 1974), (see Figure 1.1a). These units are all sedimentary in origin and have been formed by marine, eolian and alluvial deposition over the last 2.5 million years (Balla et al. 1994). The units run parallel to the coastline and all have associated wetland systems.

The Shenton Park catchment is located on the Spearwood dune system, a formation composed of both marine and eolian deposits (McArthur & Bettenay 1974). This system is characterised as having a core of calcareous eolianite (Tamala Limestone) capped by a hard layer of secondary calcite, overlain by yellow-brown silicious sands (initially McArthur & Bettenay

1960; English 1988). It was first suggested by McArthur and Bettenay (1960) that the dunes were carbonate rich to the surface and that leaching had removed the carbonate from the overlying layers and deposited it at depth. This argument was questioned by Killigrew and Glassford (1976) who cited a relative enrichment in the overlying sands of fines, microcline, heavy minerals and kaolin spherites, as evidence that decalcification of the eolianite could not have been the mechanism for the formation of the sands. The Tamala Limestone extends along the coastal strip and is composed of various proportions of quartz sand, shell fragments and minor clay lenses (Davidson 1995).

Due to the higher proportion of fines and the presence of small amounts of iron oxides which coat the sandgrains, the soils of the Spearwood system have a greater ability to fix added nutrients than the Bassendean sands (English 1988). Both have a low inherent fertility. McFarlane (1984) identified the soil of the Shenton Park catchment as being of the Karrakatta series described by Biggs and Wilde (1980) as a podsol consisting of moderately sorted, fine to coarse grained sands, that are varicoloured dependent on the degree of leaching, iron staining and humic content. The depth of the overlying sands increases away from the coast, where the limestone may be exposed at the surface. From the drillers logs supplied by McFarlane (1984) the depth of sand across the Shenton Park catchment varies between 50-60m with limestone intrusions common (Figure 1.1b).

The Spearwood dune system is more variable in relief than either the Quindalup or Bassendean systems. The lake itself is formed in an interdunal depression, typical of many of the Swan Coastal Plain wetlands. Before urbanisation the wetland would have been classified as a sumpland experiencing seasonal drying in summer (Spillman 1985). Subsequent to clearing, the wetland became a permanent lake, experiencing large scale flooding during the early decades of this century (Spillman 1985). A restructuring of the drainage network and possible increases in extraction rates by private bores have resulted once again in seasonal drying.

1.2 Impact of urbanisation on Perth's wetlands

Disturbance effects on wetlands are largely determined by the nature of the surrounding land uses so that there are clear distinctions between wetlands in rural and urbanised catchments, Harper (1992). This is particularly evident in terms of the sources and nature of pollutants and nutrients entering the waterbody, but also incorporates less obvious impacts such as changes to the hydrology and the management of wetlands in these two land use types. To put Shenton Park in a regional perspective, the land surrounding Perth's urban wetlands was commonly initially developed for agriculture, then farmed more intensively as market gardens and finally built around or over as the suburbs spread (Balla 1994). Many areas were used as dump sites for builders' rubble in an effort to reduce the area of swampland and provide more room for housing (Balla 1994, Hunt 1980).

The response to urbanisation of the water table and associated wetlands is summarised in McFarlane (1984) from the paper by Savini and Kammerer (1961) (see Table 1.2). The transition from pre-urban uncleared vegetation to late urban development, results in fluctuations in water table elevation caused by removal of vegetation, increased shedding areas and increases in water use.

McFarlane (1984) stated that, in general, urbanisation has had a favourable effect on the quantity of groundwater in Perth, as a result of an increase in indirect recharge. It was noted, however, that large scale extraction by bores has the potential to reverse this. Davis and Rolls (1987) recognised that lake levels were in fact dropping at many sites around Perth due to an increase in demand and extraction of groundwater. Conversely, at other sites Davis & Rolls (1987) reported an increase in water levels due to increased run-off, but that the flow regime at many of these lakes had been altered. It is apparent that the influences of urbanisation cannot be suitably generalised and that localised depressions in the groundwater can, in some areas, cause lake levels to drop while increased indirect recharge in other areas has resulted in higher lake levels.

The effects on Perth's wetland water quality with the spread of urbanisation has been well documented. Balla et al. (1994) suggested that excess water, nutrients and pesticides have caused such serious degradation to our urban wetlands that they should now be considered a financial liability to the community. Lake Monger, Jackadder Lake and Bibra Lake are cited as examples and Shenton Park Lake is undoubtedly also in this category having received the lowest average score based on its natural wildlife status and recreational value of all the wetlands studied in Arnold (1990). Davis and Rolls (1987) also cite the leaching of nutrients in fertilisers and the leaching of herbicides and pesticides from gardens, market gardens and intensive horticulture as being major sources of pollution. Also associated with market gardens are rising sulphate levels from superphosphate application (McFarlane, 1984). Industrial pollution is of concern in the southern Perth region while surface water from roads is now recognised as contributing significant levels of nutrients, materials high in biological oxygen demand (BOD), heavy metals and oils to our wetlands (O'Loughlin et al. 1992). For the purposes of this study the two major problems facing Perth's wetlands as defined by Townley and Davidson (1988) also apply to the disturbance effects facing the Shenton Park catchment; these are:-

1. The drying of lakes in summer months is an annual event due to the region's Mediterranean climate in which 80% of annual rain falls in the winter months of May - September, but a large number of private bores have exacerbated the problem and led to increased frequency of drying.

2. There is a tendency for deterioration of water quality, leading to algal blooms and a reduction in species diversity (Davis & Rolls 1987).

In addition to these disturbances is the loss of fringing vegetation and the impacts of a highly modified flow regime associated with large peak discharges of stormwater runoff in winter. The potential may also exist for the leaching of pollutants from the former landfill site on the northern boundary of the lake.

1.3 Stormwater recharge and the impacts on lake and groundwater chemistry

It was suggested by O'Loughlin et al. (Governmental consultancy report 1992), that stormwater run-off has become the major source of pollutants in many urban waterways and that in general the public are ill-informed on the fate of stormwater and its potential to pollute not only wetlands but the coastal environment and groundwater.

Nature of Stormwater Contaminants (common contaminants from O'Loughlin et al. 1992)

- suspended solids
- nutrients NP
- BOD and COD (chemical oxygen demanding materials)
- micro-organisms
- toxic organics
- toxic trace metals
- oils and surfactants
- litter

The potential for recharged stormwater to reduce groundwater quality, was the basis of a study by Appleyard (1994, Perth), in which three compensating basins were selected to investigate the impact of runoff from a light industrial area, a medium-density residential area, and a major road. The results indicated that, apart from a marked reduction in the TDS downgradient of the basins and an increase in the dissolved oxygen concentration and the bromide/chloride ratio in the upper part of the aquifer, stormwater recharge had no significant impact on the local groundwater, and no deterimental effects on water quality. This was in line with the findings of Malmquist and Hard (1981, Sweden), Seaburn and Aronson (1974, New York) and McFarlane (1984).

Based on the available literature, these results are perhaps not surprising with stormwater from Perth's urban catchments apparently of good quality. Davies (1992) concluded on the basis of studies by McFarlane (1984) and Tan (1991) that the stormwater quality in Perth appears to be significantly better than that reported in many urban areas. This comparison was based largely on the results of Cordery (1977) and Gutteridge et al. (1981) on catchments in Sydney and Melbourne. The better quality of the water was attributed to lower levels of soil erosion and perhaps to lower fall-out of atmospheric pollutants (McFarlane 1984). The Tan (1991) study was, however, limited to studying nutrients, nitrogen and phosphorus whereas SS, TDS, N, P and a range of heavy metals were analysed by McFarlane (1984). Appleyard (1994) recognised that although only trace concentrations of lead were found in the groundwater, in one basin, lead concentrations in the sediments were 3,500 ppm. This indicates a strong adsorption of this element by the iron oxides and clay minerals in the bottom sediments.

From Tan (1991) the flow weighted mean concentrations of total P in runoff from sandy urban catchments typically averaged below 0.1mg/L, during the winter season (there was a noticeable first flush effect in the early rains). In the two rainfall events in which stormwater chemistry was analysed by McFarlane (1984) the figures were comparable at, 0.102 and 0.257mg/L. Tan (1991) noted that in the early rains the proportion of soluble P (orthophosphate) in total P was greater than at the end of winter. This was attributed to an increase in phosphorus washed into the stormwater system attached to soil particles, which had been eroded at higher levels in the more intense storms measured in winter. The majority of catchments studied by Tan (1991) had first flush loads of total N in the range of 1 mg/L, with the highest value of 5 mg/L. These values commonly dropped to ranges between 0.2 and 1mg/L with the average around 0.5mg/L by mid winter. These levels are not particularly high when compared to average nitrogen concentrations in groundwater of 0.5-1mg/L (Gerritse et al.1990, Appleyard 1995) and PO_4-P of 0.01 mg/L (Gerritse et al. 1990).

Based on an assumption of 50% of applied nitrogen leaching to the groundwater in a sewered, urban catchment, Gerritse et al. (1990) suggested that maximum concentrations of nitrate-N would approach 40mg/L in the groundwater. The true levels are greatly lower than this, indicating intensive denitrification. Naturally high concentrations of organic matter in the groundwater, pH values in the range of 5-7 and redox potentials between 100-300mV provide ideal conditions for the development of microbial denitrifiers in Bassendean soils (Gerritse et al. 1990). Denitrification is restricted to anaerobic conditions which are characterised as having low redox potentials, assuming that the concentration of dissolved oxygen is the main determinant on the redox potential. It is recognised that when dissolved oxygen concentrations are low, Fe^{3+}/Fe^{2+} and SO_4^{2-}/S^{2-} couples are important in buffering redox potentials (Appleyard 1995).

The Spearwood sands of the Shenton Park catchment have groundwaters lower in dissolved organic carbon and redox potentials above 300mV so that denitrification is not as extensive and higher nitrate levels in the groundwater result. Indeed, averages of nitrate concentrations taken from the Applecross Peninsula 1983-1985, by the Geological Survey of WA (Gerritse 1990) range from 5-18 mg/L. Recharge of stormwater at very localised point sources, such as Shenton Park lake and indeed at any compensating basin, allows water to move rapidly into the aquifer and it is suggested by Appleyard (1995) that this may lead to an increase in the dissolved oxygen concentrations around these basins and perhaps an increase in the redox potential over time. An increase in the redox potential could therefore lower the rates of denitrification such that increases in the level of nitrate in groundwater may result.

The concentrations of sulphate and the sulphate/chloride ratio were the most distinct differences between the groundwater beneath established and new suburbs, with both parameters increasing progressively with the age of the residential area (Appleyard 1995). This is most likely caused by the oxidation of soil held sulfides on land clearing and fertiliser inputs. A continued recharge of stormwater with high concentrations of dissolved oxygen from compensating basins may also enhance this effect.

1.4 Eutrophication of wetland ecosystems

1.4.1 Principles of eutrophication.

Harper (1992) defines eutrophication as the biological effect of an increase in concentration of plant nutrients to a waterbody, principally nitrogen and phosphorus, but on occasion silicon, potassium or calcium. The process of eutrophication is required to be described for two reasons, firstly as a potential health risk and nuisance factor and secondly in order to understand its potential in modifying the redox conditions in the lake and subsequently lake chemistry.

The health and nuisance implications associated with eutrophic waters commonly arise from the mass death of large algal blooms, that proliferate under high nutrient conditions. The death of algae and subsequent low oxygen conditions result in foul smells, an increase in the midge population (eating dead algae) and hatching of botulism spores (Balla et al. 1994). The lake biota can also suffer direct poisoning from ingestion of toxic algae, in particular cyanobacteria, but also death through suffocation.

Under low oxygen conditions bacteria in the sediments use iron and other metals as respiratory electron acceptors in place of oxygen. This process can be accelerated when the lake waters become deoxygenated following the mass death of an algal bloom. This in turn can lead to the release of phosphorus from the sediments. An increase in summer phosphorus levels may therefore be attributable both to release from the sediments and also evapoconcentration effects. This is described by Mortimer (1941, as cited in Harper 1992) in that the degree of oxidation and reduction at the sediment surface as a consequence of deoxygenation of the overlying water, determines the movement of solutes between sediment and water. Typically Shenton Park Lake should remain oxygen rich throughout most of the year given its shallow depth and the use of aerators to continually cycle water.

1.4.2 The concept of nutrient limitation

When investigating eutrophic waterbodies the N:P ratio and absolute concentrations are seen as critical in determining which nutrient controls the growth of algae but is also a determinant on the species of algae most suited to the lake conditions. It is now accepted that N:P ratios of above 30:1 do not favour the growth of cyanobacteria and at these levels green algae predominates (V.H.Smith 1983, Balla et al. 1994). At Shenton Park Lake an interesting aspect of the study was seen in the potential for the lake to develop cyanobacterial algal blooms similar to those that have plagued the Canning River and the Peel-Harvey Estuary. The lake, in summer, appears well suited to the growth of cyanobacteria which are favoured by temperatures of 20-30°C, high pH (8-10) and calm waters of low light intensity (Balla et al. 1994).

The concept of limiting nutrients on algal growth is related to the stoichiometric atomic ratio between C:N:P of 106:16:1 in plankton cells which corresponds to a mass ratio of approximately 40:7:1. An N:P ratio below 7:1 therefore suggests that nitrogen is the primary growth limiting nutrient while a ratio above this value shows potential P limitation (Ryding & Rast 1989). Variation in the cellular content of nitrogen and phosphorus in algae means that the 7:1 ratio should not be strictly applied and in the range of 5-10 either (or neither) nitrogen or phosphorus could be limiting (Chiandani and Vighi 1974). It is noted in Rast et al. (1983) that more important, in terms of management of eutrophication, is whether or not the available N or available P in the water body is reduced to growth limiting levels during the period of water quality concern. If the biologically available N and P concentrations decrease below approximately 20 μg N/L or 5-10 μg of P/L respectively, during an algal bloom peak that nutrient is likely to be the limiting one. If both nutrients decrease below this value both may be limiting.

The modelling approaches used by Rast et al. (1983) on water bodies 'close' to P limitation, provided results consistent with Schindler's (1977) remarks concerning the ultimate control on algal biomass being phosphorus, even when nitrogen appeared to be the limiting nutrient. Similar statements are recorded by Chiandani and Vighi (1974). In terms of eutrophication management, phosphorus is more easily studied, as, unlike nitrogen, it has no atmospheric component and so its fluxes can be described as either internal cycling of the nutrient, as inputs from surrounding land uses or as leaching to the groundwater. Harper (1992) noted that, commonly in rural catchments, phosphorus is the nutrient most manageable in terms of reductions, as the sources tended to be point rather than diffuse. In an urbanised catchment this is probably not the case with both sources being potentially diffuse. Phosphorus is still seen, however, as the only essential element that can be easily made to limit algal growth (Golterman 1975).

1.4.3 Modelling of phosphorus effects on eutrophication.

The archetypal phosphorus budget was conducted by Vollenweider (1968) who showed that if catchment input into a lake was calculated as a loading per unit area, lakes of different sizes could be compared. In this work the loadings of N and P were plotted against mean depth, and boundaries between oligotrophic and eutrophic lakes were defined. The Vollenweider standards for the trophic classification of lakes are listed in Figure (1.4.3a).

Many models of the eutrophication process have been developed on glacial lakes of Europe and North America which are generally deeper and show pronounced stratification in summer. The formation of pronounced epilimnion and hypolimnion layers complicates the processess of nutrient assimilation and release of nutrients from the sediments as discussed in Lam (1982). Thermal stratification is, however, precluded in the shallow lakes of the Swan Coastal Plain where there is continuous convection cycles and mixing.

Goltermann (1982) provides loading concentration models for phosphate in shallow lakes such as those found in Perth. It is stressed that phosphorus release in shallow lakes behaves differently due to resuspension of the sediments. The diagram taken from Golterman (1982) (Figure 1.4.3b) shows the solubility of phosphate in different phases bound to Ca and Fe. Golterman suggests that the assumption that the process of formation of Ca bound phosphate is slow, is only true for high concentrations of P, and that it may occur within a few days if suspended $CaCO_3$ is present at lower P concentrations.

His conclusions were that in hard waters Ca^{2+} is as important as Fe as a phosphate control mechanism. In the interstitial waters of shallow lake sediments, lower pH values may occur than in the overlying water and iron may control the phosphate concentrations there. (Wind effects in the very shallow lakes may prevent the occurrence of interstitial waters). "However, as algal growth takes place in the water, in most eutrophic shallow lakes the concentration and the pH will control the solubility of the phosphate surrounding these algae. It is therefore essential that, for hard-water eutrophic shallow lakes, the solubility product of apatite is set as a threshold for the permitted phosphate concentration" (Golterman 1982). This may have significant implications to the study of phosphate mobilisation in Shenton Park Lake if the lake is heavily concentrated in $CaCO_3$.

1.5 Water balance modelling

1.5.1 Development of a water balance equation.

Winter (1981) stated "Although water budgets are basic to understanding nutrient budgets, little effort has gone into understanding the processes and controls on water sources and sinks as they relate to lakes". The principles of water balance modelling are quite simple requiring firstly identification of the water balance components, eg. rainfall, groundwater outflow etc., and then a measurement, or best estimate, of the various parameters, to some degree of error. The components of the water balance are complex in an urban catchment due to a large range of human modifications to the catchment hydrology including interbasinal pumping of stormwater, reticulation and the presence of flood mitigation drains in wetlands.

The water balance equation developed by McFarlane (1984) for Shenton Park Lake is outlined below:-

$$R + P + G_i + Pu + F = E + G_o + D + \Delta S$$

where R = road runoff (divided into intra- and interbasinal)

P = direct precipitation inputs

Pu = pumped waters from an adjoining basin

F = fountain additions

G_i = groundwater inflow

E = evaporation

G_o = groundwater outflow

D = drained water (to an ocean outfall)

ΔS = change in surface water storage

The components of the water balance for this study are as listed by McFarlane (1984) with the exception of the bore supplying the fountain which is no longer in operation.

Techniques and experimental procedures are well developed for the estimation of the majority of the water balance components, but the interaction of lakes and groundwater is the most difficult parameter to measure and requires further discussion.

1.5.2 Groundwater flow as a component of the water balance.

As a component of the water balance, groundwater is now recognised in the majority of studies, as significant in terms of the water budget of a lake. Prior to articles by Winter (1978b), Born et al. (1979) and Anderson and Munter (1981), water balance studies of lake systems commonly chose to ignore the groundwater component as unimportant or calculated it as the residual of the water balance equation. Subsequent to these papers a general lack of knowledge on the physics of lake-aquifer interaction and the relationship between regional and local flow systems, prevented scientists from detailing accurately the groundwater component of a lake water budget. Indeed it was suggested by Winter (1978) that the construction and placement of wells to measure potentiometric heads across a groundwater system relative to lake level, could lead to misinterpretation of the interrelationship of lakes and groundwater. In a water budget for Lake Joondalup, Congdon (1985) determined that groundwater inflow and outflow were very difficult and expensive parameters to measure and would therefore be solved as the residual of the water balance equation. Many water balance studies on lake systems to this time have therefore calculated groundwater fluxes not based directly on some form of experimental measurement but estimated it as the residual of the water balance equation. Given that error analysis is often not accurately detailed, the validity of results which calculate the groundwater component as a residual is questionable (Winter 1981)

The available literature on groundwater flow near lakes is limited with much of the earlier modelling studies tending to focus on a specific waterbody or a series of similar lake types. Townley et al. (1993) suggest that, in general, existing modelling results are not easily transferrable to previously unstudied lakes and that their usefulness is in the methods described rather than the applicability of their results. A generalised classification system exists, however, for describing lake/aquifer interaction as developed by Born et al. (1979), based on a distinction between recharge, discharge and flow-through lakes. "A lake is classified as a recharge lake if the aquifer is recharged over the entire lakebed, as a discharge lake if the aquifer discharges groundwater into the 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" (Townley et al. 1993). For a pictorial representation of a flow-through lake see Figure 5.3a.

1.5.3 Water balance studies using the Langbein et al. (1952) method

Meyboom (1967) studied the seepage and evaporation into a series of lakes in the Hummocky Morraine region of western Canada. His hypothesis was that an investigation of seepage and evaporation would yield a strict hydrological distinction between lakes in recharge areas and those in discharge areas. The study was conducted so as to monitor local, intermediate and regional flow patterns as described by Toth (1963). This necessitated the installation of piezometer nests ranging in depth from 10-100ft. The distinction between these flow types is necessary in studies where lakes bisect deeply into the groundwater intersecting numerous aquifers.

The relative flux of groundwater inputing or discharging from the lake was calculated using the field technique developed by Langbein et al. (1952), utilising mass transfer theory. "The mass transfer theory states that exchange of water vapor between a water surface and the atmosphere is directly proportional to the vertical humidity gradient and the windspeed" (Meyboom 1967). The basic empirical mass-transfer equation can be written as:

$$E = N[u(e_s - e_a)] \quad \text{where,}$$

E - evaporation in feet per hour

N - a coefficient of proportionality, called the mass transfer coefficient

u - Windspeed, in miles per hour at a fixed height above the water surface in mid-lake.

e_s - saturation vapor pressure at the evaporating surface (mm Hg), corresponding to the temperature of the water surface in mid-lake

e_a - vapor pressure of the air (mm Hg), corresponding to the dew point.

With the evaporation calculated using the mass transfer equation, the Langbein et al. (1952) method calculated the flux of groundwater as the residual of the sum of:

Fall in lake stage = evaporation loss + seepage.

The fluctuations in lake stages were recorded manually as was the monitoring of the weather station and piezometer wells. Using this technique Meyboom (1967) showed four different flow regimes (see Figure 1.5.3) at the lakes corresponding to the spring, early summer, late summer and autumn/winter seasons. The data also showed diurnal fluctuations in seepage around the lake perimeter explained as a drawdown effect created by transpirational losses from phreatophytes fringing the lakes. This study calculated net seepage by monitoring just two parameters but the range of assumptions made using the Langbein et al. (1952) technique restricts this approach. Efficient data logging equipment means that these parameters can now be measured to a much higher degree of accuracy and on a continuous basis.

1.5.4 Principles of lake/aquifer interaction

For the purposes of this study the following definitions are given as:

Lake capture zone -	Zone within which any recharge will eventually flow through the lake. The systems are dynamic so that the dimensions of the zone are variable in space and time (see Figure 5.3a).
Lake release zone -	Contains any water that has passed through the lake. Similarly, the dimensions of the lake release zone are variable.
Streamline -	Illustrates the average flow path of a particle of water in the aquifer for a steady state system.
Localised flow -	Groundwater flow in the vicinity of the lake which is influenced by its presence, as opposed to shallow flow, which signifies groundwater that interacts directly with the surface waterbody.
Dividing streamline -	Separates the groundwater that enters the lake from that which bypasses it and in the case of a flow-through system, defines the release and capture zones of the lake terminating at a stagnation point.
Anisotropy -	The ratio of the hydraulic conductivity of the aquifer in the horizontal direction relative to that in the vertical.

Winter (1976) recognised the importance of being able to define the boundary between local and shallow flow systems and concluded that the continuity of the shallow flow system beneath a lake is the factor that controls the interaction of lakes and groundwater. Winter determined that the continuity of the boundary could be assessed by examining the flow around the point of minimum hydraulic potential along the divide, the stagnation point. The stagnation point as described by Anderson and Munter (1981) in two dimensions, is the point of minimum head along the divide separating the shallow flow system associated with the lake from a deeper flow system. The stagnation point is a zone where the vector flow of water is equal to zero (Townley pers. comm.).

A feature of the discharge class of lakes studied by Winter (1978, 1981, 1983) is the presence of a stagnation point at some depth below the lake bed (Townley et al. 1993). The water table mounds on both sides of the lake are higher than the corresponding lake level resulting in a stagnation point somewhere below the lake bed and a continuous boundary between the two flow systems. This results in the shallow flow system associated with the lake being isolated from the rest of the groundwater system (Winter 1978) (see diagram 1). It is recognised that when a stagnation point does not exist then the lake can lose water over all or part of its bed (Winter 1978) (see diagram 2).

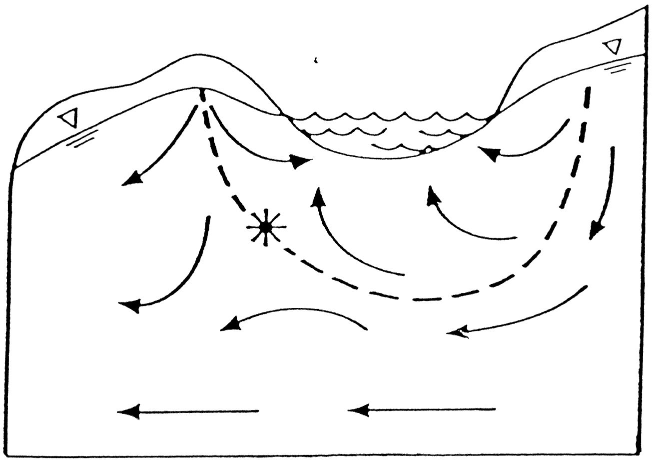


Diagram 1. Groundwater flow paths in the vicinity of a discharge lake. Asterisk shows the location of the stagnation point and the dashed line shows the location of the groundwater divide separating the shallow flow system from a deeper sytem.

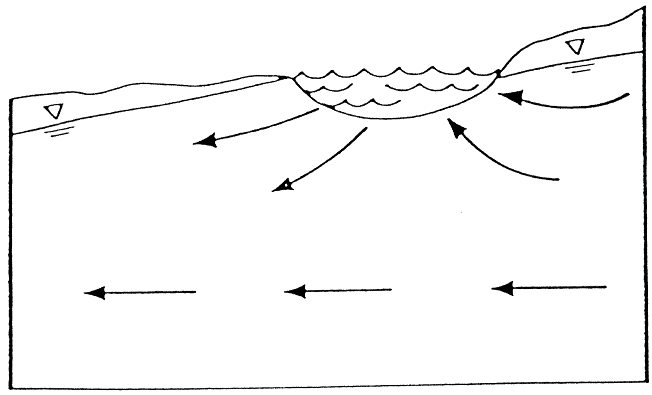


Diagram 2. Groundwater flow paths in the vicinity of a flow-through lake.

The two lines separating the flow systems then begin at the water table mounds on each side of the lake, and each one terminates at different points on the lake bottom. The part of the lake bed between these two termination points is the area of outseepage and this water becomes part of the regional flow. The major factors influencing the continuity of the shallow flow system boundary are described by Winter (1978) as, (1) height of adjacent water table mounds relative to lake level, (2) position and hydraulic conductivity of aquifers within the groundwater system, (3) ratio of horizontal to vertical hydraulic conductivity of the system, (4) regional slope of the water table, and (5) lake depth.

Anderson and Munter (1981) demonstrated seasonal reversals in flow at a lake caused by a transition from a flow-through to a discharge regime. In their discussion it is stated that the formation of a stagnation point at a flow-through lake is accomplished by the formation of a groundwater mound on the downgradient side of a lake. In reality the stagnation point described as occurring at a flow-through lake in this study, is in fact the stagnation point described by Winter (1976) associated with the continuous boundary of a discharging lake system. This study does not therefore associate stagnation points with true flow-through lakes. It is argued by Townley (pers. comm.), however, that at some point across the bottom of a flow through lake the two dividing streamlines will intersect and this region is denoted as a stagnation boundary, recognising the position of a stagnation point that cannot be accurately spatially defined (diagram 3).

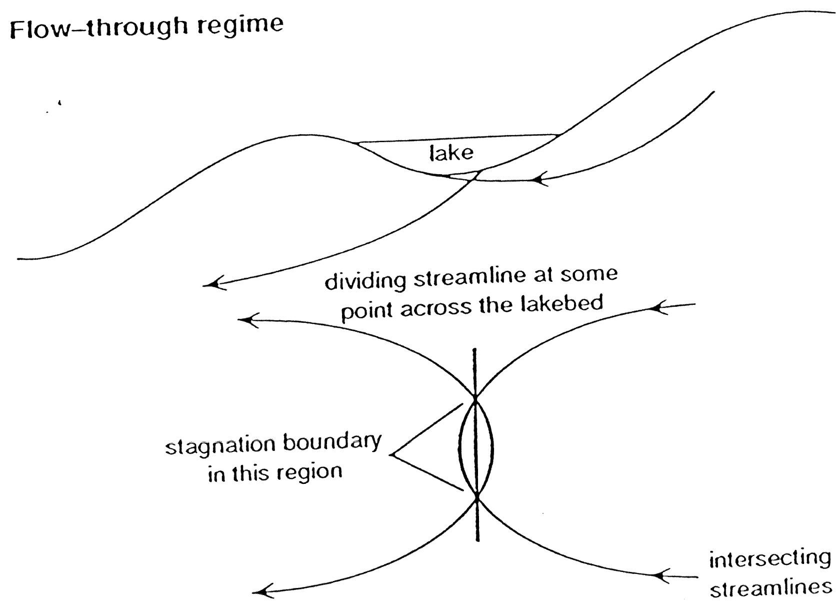
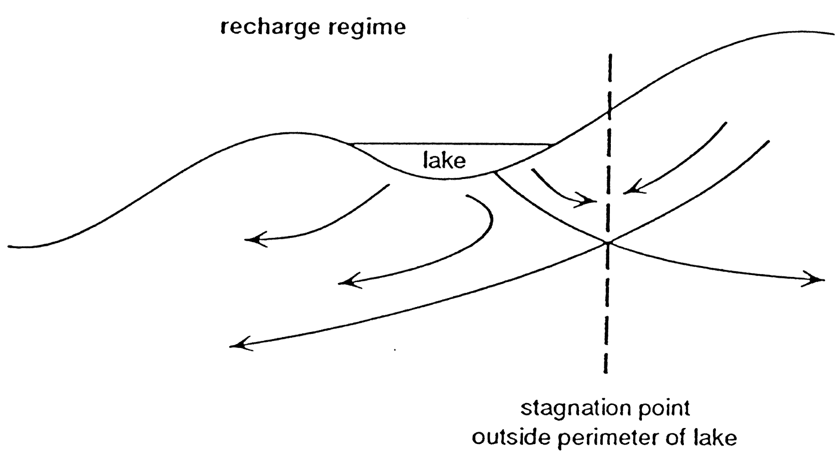


Diagram 3

Stagnation points can therefore form in any of the three flow regimes and will move in position depending on the relative groundwater fluxes. The position of the stagnation point is not restricted to the area below the lake bed but can move outside the lake boundaries as depicted below. A point depicting the position of the stagnation point in areal section is described as a saddle point as this point recognises the position of the dividing streamlines but cannot provide the position of the stagnation point in terms of relative depth (see diagram 4).



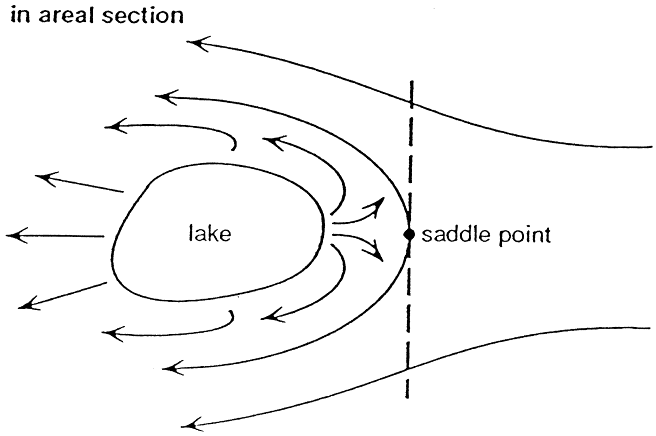


Diagram 4

1.5.5 Modelling of groundwater flow around lakes.

Recent advances in the understanding of flow-through lakes on the Swan Coastal Plain, have evolved over a number of years, culminating in the development of the FLOWTHRU model (Townley et al. 1992). The great advantage of this model is that, although developed as a general guide to surface water and groundwater interactions over a range of surface water conditions; its development was based on studies conducted on local flow systems such that some of the assumptions inherent in the model, ie. that the depth of the lake should be small in relation to the overall thickness of the superficial aquifer, are valid for the majority of wetlands in the Perth region. The FLOWTHRU model (Townley et al. 1992) resulted from the realisation that distinctions between the flow patterns of surface water bodies could be characterised in terms of simple geometrical ratios, or in terms of the directions of regional groundwater flow on either side of the water bodies (Nield et al. 1994).

The modelling of flow in two dimensional vertical section (FLOWTHRU) was preceded by studies of flow-through lakes in plan. The important concept from Townley and Davidson (1988) is the determination of the capture zone based on the geometry of the surface waterbody. Figure 1.5.5a shows that the width of the capture and release zones increases away from the perimeter of the lake, to a maximum value for w+, of twice the radius (a) of the waterbody.

In both the earlier studies in plan and the later work based on vertical cross-section, the need to specify how the lake receives or adds volumes of water to the aquifer was removed by imposing upstream and downstream gradients, ie a net flow into and out of the lake. The variables U_+ and U_- notate the flux per unit area of inflow (from right hand boundary of diagram) and outflow (through the left hand boundary) respectively (Figure 1.5.5b). Figure 1.5.5b shows how a decrease in the ratio of U_-/U_+ changes the flow pattern from a recharge to a discharge regime, with the stagnation boundary moving in a downstream direction around the lake boundary. On the same Figure (1.5.5b) the impact of a larger surface water body on the same area of regional flow does not show changes in the flow pattern, with the stagnation point remaining near the mid point of the lake.

Central to the understanding of flow near lakes is the analogy of a lake behaving as a thin layer of high permeability, which focuses the regional flow towards it (Townley and Davidson 1988). When a lake is in some sense large, the flow approaching the upgradient boundary of the lake rises and enters the lake and conversely the aquifer is recharged from the lake along its downgradient boundary. Seepage is recognised as being greatest at the shoreline decreasing approximately exponentially away from the shoreline (Townley et al. 1992). Extremely large lakes embedded in a regional flow are characterised by the fact that no underflow passes beneath the centre of the lake. Based on preliminary studies of flow in vertical section, Townley and Davidson (1988), suggested that earlier estimates of the top one third of the unconfined aquifer interacting with surface water bodies on the Swan Coastal Plain (Allen 1980, Davidson 1983), are underestimates. Based on a lake diameter of 2km, a lake depth of 2m and an aquifer thickness of 50m, the proportion of flow passing through the lake was calculated as being closer to 100%, even at anisotropy ratios of 10 and 20. This estimate did not account for the affect of a resistive lining on the lake bed.

1.5.6 Description of the steady state model FLOWTHRU

As discussed by Winter (1978), two dimensional models in vertical section require that the lake dimension perpendicular to the section has to be large, and that realistically they can only be applied to long linear lakes that are aligned perpendicular to the groundwater flow. This was recognised by Townley et al. (1992) but they indicated that preliminary results suggest the model also applies to large circular lakes. The remaining assumptions of the model can be summmarised as steady state saturated flow, a shallow waterbody (relative to the thickness of the aquifer) and homogeneity in hydraulic conductivity.

The FLOWTHRU (Townley et al. 1992) package is not a numerical groundwater model, but instead combines and displays a set of precalculated solutions. These solutions were obtained using the linear triangular finite element model, AQUIFEM-N (Townley unpublished). From these solutions a series of 39 flow regimes have been identified and it is from these predetermined regimes that a solution for any set of inputed lake geometries or boundary conditions are determined.

The range of flow conditions for which each regime occurs is described in terms of non dimensional ratios describing the geometry, physical properties, and flux boundary conditions of the system. It is these variables that can be manipulated by the user. As flow patterns were only devised for a limited number of lake/aquifer geometries and physical properties, there are a limited number of dimensional ratios from which to construct a model. The geometry is represented by the length of the aquifer body relative to aquifer thickness, physical properties are represented by the anisotropy ratio and the ratio of an equivalent sediment depth to the aquifer thickness, and boundary conditions are represented by a ratio of horizontal flows and a ratio of recharge to horizontal flow. Anisotropic systems can be converted to equivalent isotropic systems by scaling of the geometry.

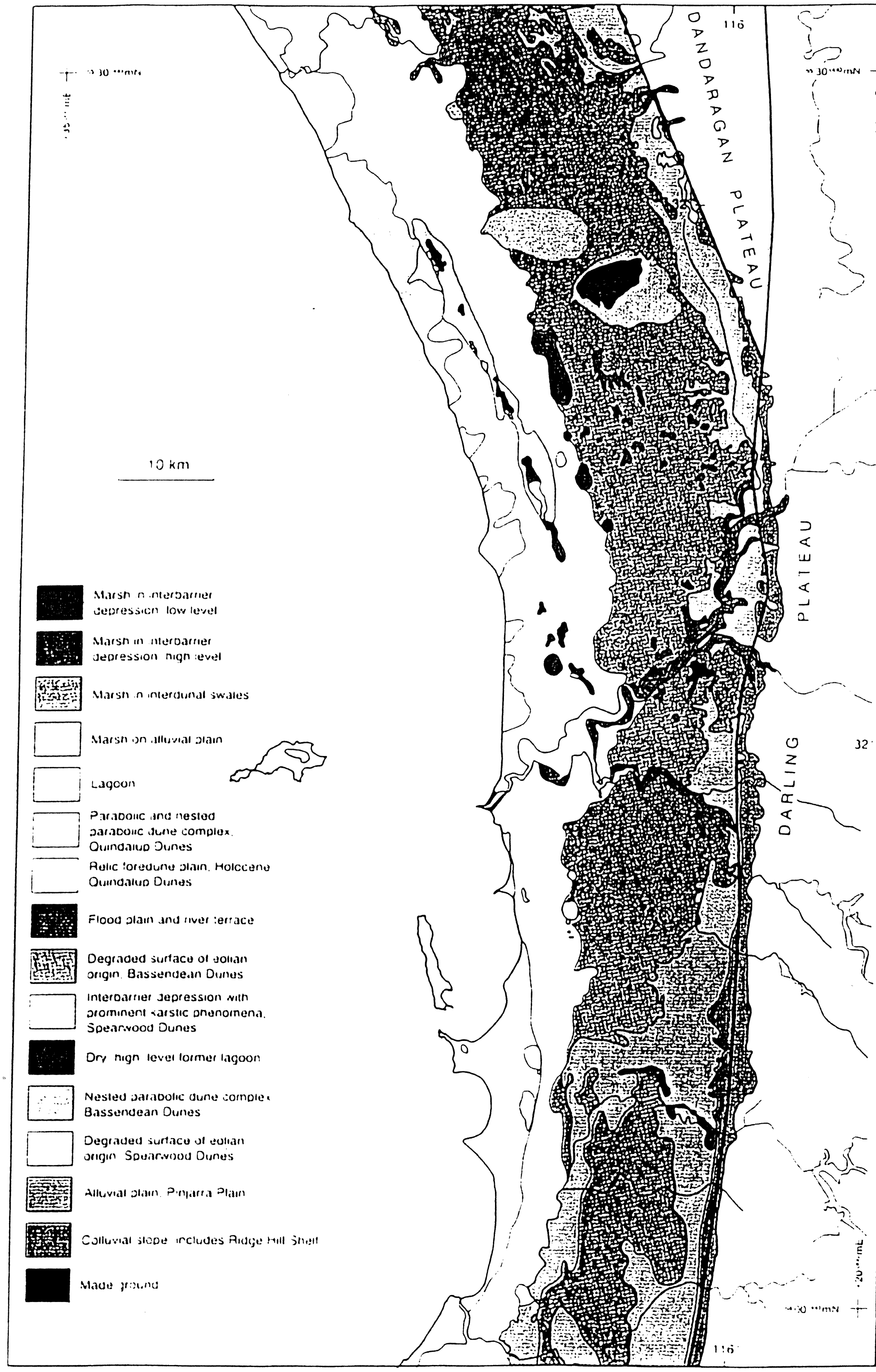


Figure 1.1a Generalised geomorphology of Swan Coastal Plain (after Davidson 1995)

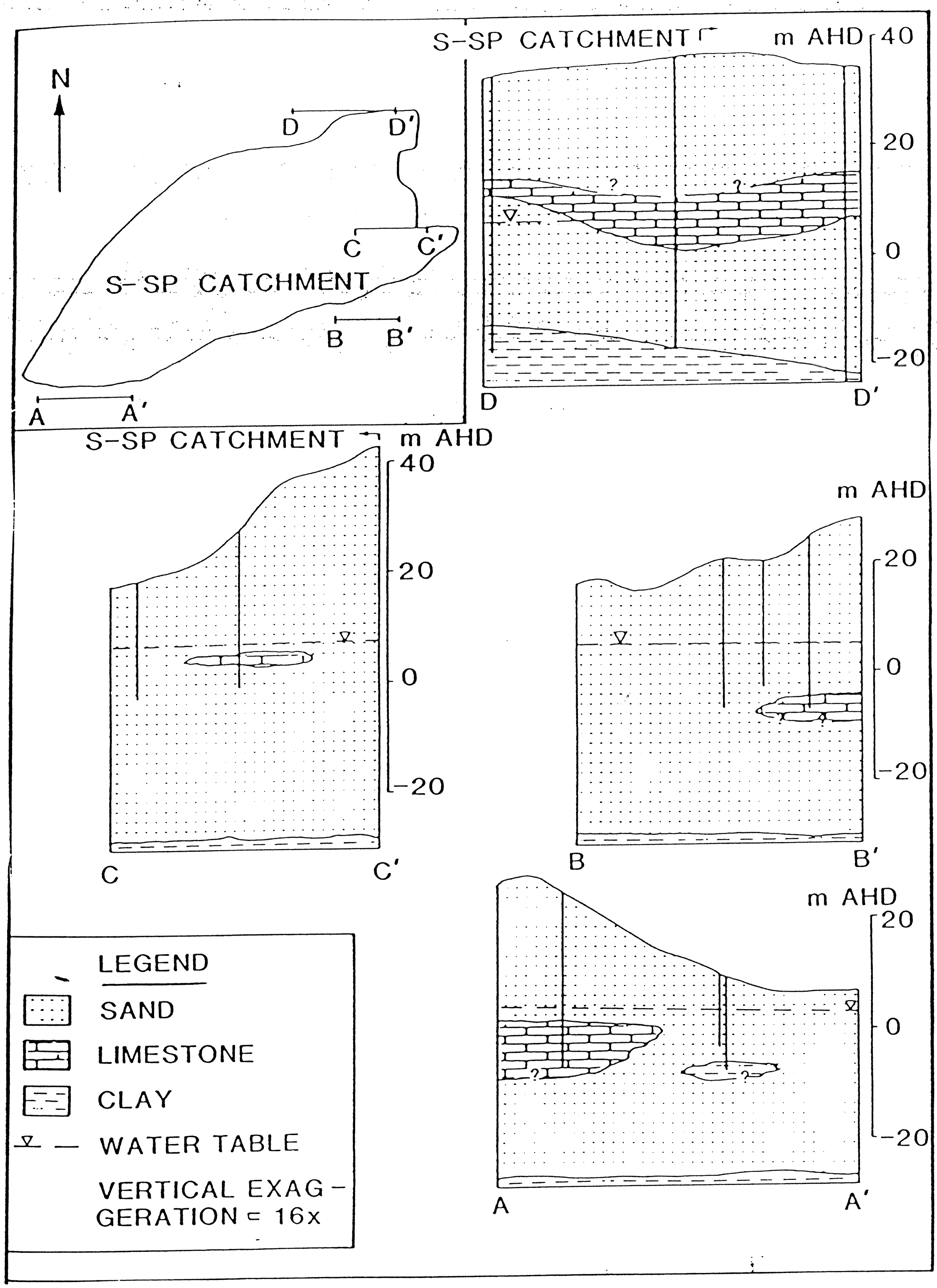


Figure 1.1b

Geological cross-sections across the Shenton Park catchment (after McFarlane 1984)

RUN-OFF EFFECTS	
It is well recognised that the creation of impervious surfaces eg. Roads, roofs etc. With low depression storages results in an increase in both amount and rate of surface runoff.	McFarlane (1984)
Lower frictional resistances of streets and conduits accelerates flow and increases peaks of urban run-off.	McPherson (1974b)
Depression storages of only 0.75 mm was reported for impermeable surfaces.	Carlson and Falk (1977)
Increased soil compaction is suggested to significantly increase run-off from permeable surfaces in urban areas.	Luovich & Chenogneva (1977) Carlsson & Falk (1977)
Lawns in urban areas calculated to have infiltration rates only $^1/_6$ that of forests	Felton and Lull (1963)
In cases of interbasinal transfer of water, if this water is discharged into the stormwater system, the increased annual streamflow has a marked effect on small sub-basins.	Kuprianov (1977)
In districts removed from oceans or estuaries almost all road runoff is discharged into adsorption basins, which may be either above the water table or are wetlands. These basins may have flood mitigation drains which connect them to an estuary or the ocean.	McFarlane (1984)
GROUNDWATER EFFECTS	
The decrease in soil infiltration, which occurs with an increase in shedding area is usually accompanied by an equivalent decrease in soil evaporation, transpiration and groundwater recharge.	Savini and Kammerer (1961)
A water table rise was reported due to urbanisation - significant factors reported as reduced transpiration losses, recharge from shedding area, addition of imported waters from septic tanks and garden watering.	Whincup & O'Driscoll (1978)
Interbasinal transfer of water for garden watering is recognised as a contributing factor in increased recharge.	Savini and Kammerer (1961)
Garden watering is a highly ineffective recharge source due to high evaporation and runoff rates.	Leonard (1981)

Table 1.2

Effects of urbanisation on catchment hydrology summarised from McFarlane (1984)

Trophic Classification	Total P (μgl)	Inorganic N (μgl)
1. Ultra-oligotrophic	<5	<200
2. Oligo-mesotrophic	5-10	200-400
3. Meso-eutrophic	10-30	300-650
4. Eu-polytrophic	30-100	500-1500
5. Polytrophic	>100	>1500

Table 1.4.3a Trophic state classification by total P and N of Vollenweider (1968)

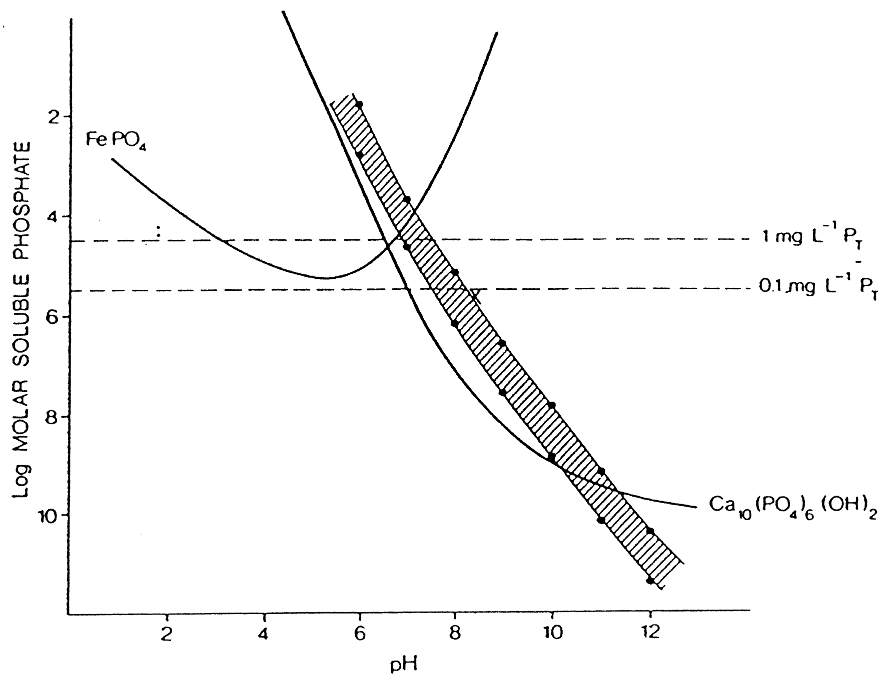


Figure 1.4.3b Solubilities of apatite-bound and iron-bound phosphorus in relation to pH (after Golterman, 1984)

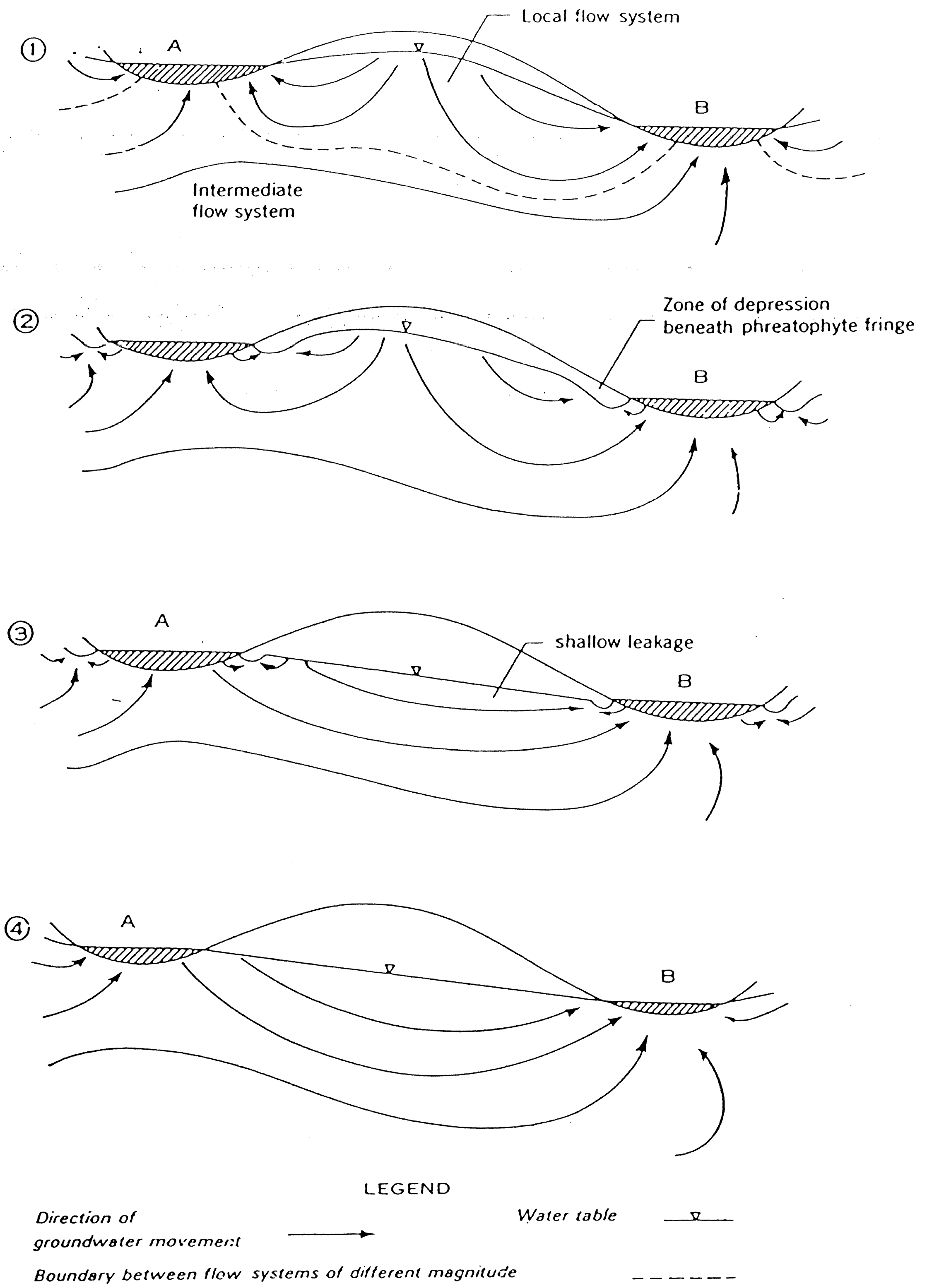
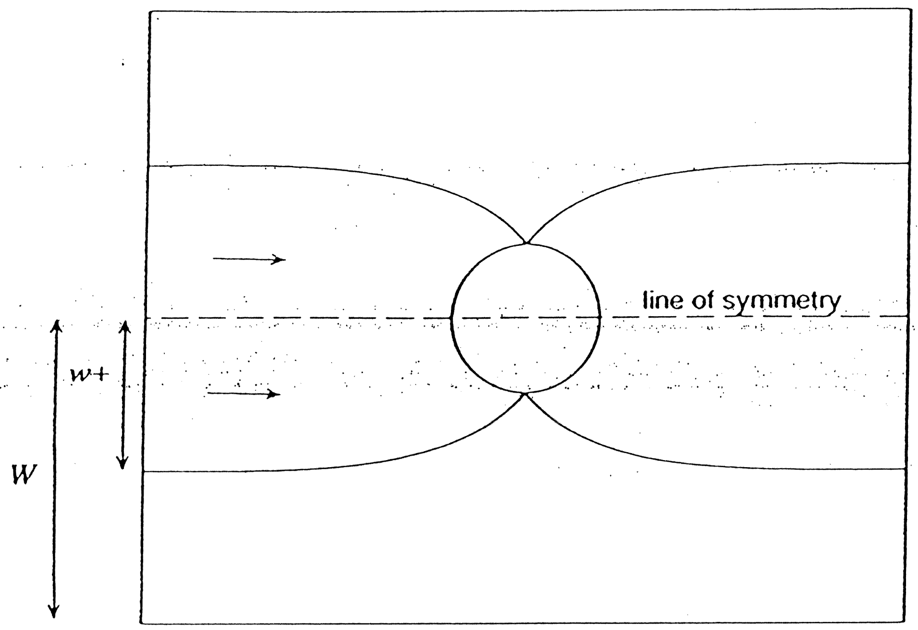


Figure 1.5.3 Four typical flow conditions near permanent lakes in hummocky moraine (1) Spring (2) Early Summer (3) Late Summer (4) Autumn and Winter (Meyboom 1967)



Definition of width of capture zone

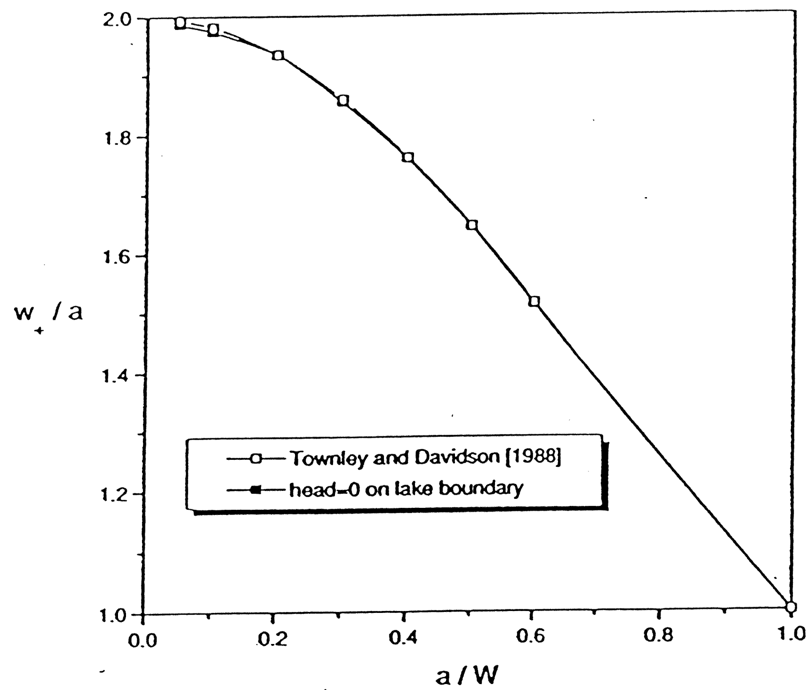


Figure 1.5.5a Width of capture zone for flow-through lakes with zero recharge

(from Townley et al. 1993)

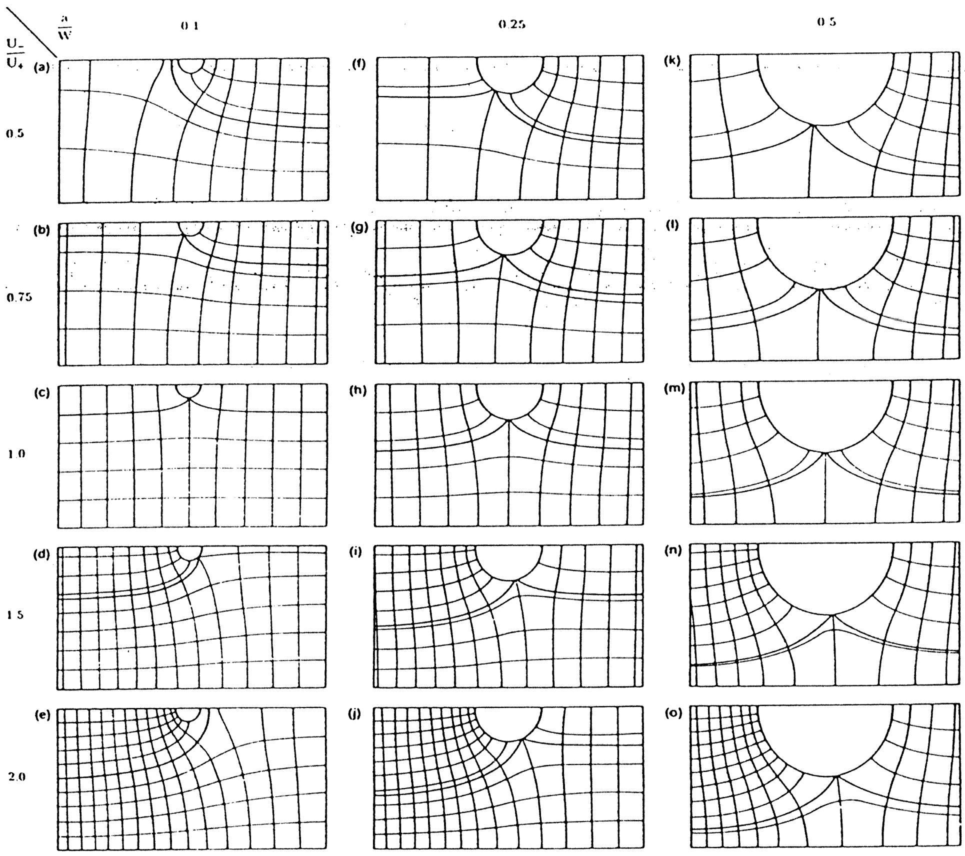


Figure 1.5.5b Flow nets near circular lakes with a/W = 0.1, 0.25 and 0.5, with five values of U_-/U_+, as shown. Contour intervals for ϕ and ψ are 0.2 $U_+ W/K_h$ (from Townley et al. 1993)

2. EXPERIMENTAL PROCEDURES

2.1 Measurement of rainfall.

The rainfall component of the water balance was studied over three sites, (1) at the university field station at Shenton Park, 2kms from the lake, (2) from a gauge in Kings Park, 3kms from the lake and (3) through a local resident, where the gauge was located 300m from the lake. Initially five years' worth of records from 1990-1994 from each of the sites were compared statistically to determine the variation in rainfall over the Shenton Park catchment. As the daily rainfall readings at two of the sites were made at 9am in the morning whereas at the third site (300m from lake) a reading was made at 5pm, the predictive ability of the Kings Park and Field station sites for the rainfall at Shenton Park Lake was only maximised to r^2=0.88. A more accurate indication of the variability of the rainfall across the catchment was achieved by using hourly rainfall records from the field station to calculate a daily rainfall over the same interval as the gauge at the lake. This was done for the months of July and August. It was found that on days when it was raining, the predictive ability of using the Shenton Park field station data for the rainfall recorded at the lake was significant at $r^2 = 0.96$. It is proposed that, due to the small size of the catchment, and based on the low variability between the rainfall records from the field station and lake, that the one gauge located within the catchment boundaries is sufficient to accurately predict rainfall over the entire study area.

The errors associated in recording rainfall as assessed by Winter (1981) depend on the gauge placement, gauge spacing and areal averaging technique. Based on the estimates by Winter (1981) the error in the recording instrument may range from 1-5% while the placement of the gauges, ie. height above the ground, is in the range of 5-15%. A site inspection of the rainfall gauge at the lake was made to check its location and ensure rainshadow effects, overhanging trees or splash effects would not create significant error. Despite the rain gauge being located approximately 2m above the ground, the site was recognised as being the most suitable on the block. The rainfall gauges at both the field station and Kings Park were deemed to be suitably located. The potential errors in the rainfall measurements are not expected to exceed 20%.

2.2 Shenton Park catchment: site description

The Shenton Park catchment is divided into two sub-catchments, designated O and P (Figure 2.2a). The catchment for drain 7, in which direct measurements of discharge were recorded, is shaded black. Fortunately, drain 7 had only recently been laid so that information pertaining to its catchment boundaries was available. By combining the information from local council drainage maps and water authority maps, the catchment boundary of drain 7 could be accurately determined. This drain covers an estimated 35% of the total area of the Shenton Park catchment.

The remainder of sub-catchment O is drained primarily by drain 8, another major drain, with drain 9 only receiving runoff from Excelsior Street and part of Derby Road (see Figure 2.2b). The drainage network of sub-catchment P was not intensively studied but has two major drains, drain 1 and 4. Drain 10 is the drain from the pump station at Aberdare Road.

It is evident from Figure 2.2a that the zoning is slightly different in sub-catchments O and P, with O having a higher housing density. It was noted by McFarlane (1984) that smaller block sizes, that is a higher housing density, resulted in larger runoff rates due to an increase in the area of impervious surfaces. The gradient of the land surface is also steeper in sub-catchment O and so it could be anticipated that runoff rates will be greater in this sub-catchment than in P. This is probably not the case, however, as the efficiency with which runoff is shed from impervious surfaces and into the drainage system (runoff coefficient of 0.9, WAWA Figure, pers. comm. K. Chinnery) should negate any effect of gradient between the two sub-catchments. Also the impact of slightly differing block sizes will be minimal on the overall discharge figures, as long as the land uses in the catchment are the same, so that there are no large areas where runoff rates are going to be significantly different. This argument is supported by the calculated WAWA Fi values for the two subcatchments (supplied by K. Chinnery) which are both recorded as 0.15 (see Table 2.2).

Table 2.2 shows the shedding areas in each of the subcatchments calculated by multiplying the catchment area by the Fi value and then by a runoff coefficient of 0.9. These values represent the area of land in the catchment which contributes to runoff. McFarlane (1984) calculated a shedding area in the Shenton Park catchment of 16.5% contributing to 100% of the runoff, which is 3.5 percent lower than the 20% calculated using the Fi values. It should be noted that the Fi values are only very approximate, so that the McFarlane result is recognised as being more accurate. What the shedding areas do provide is a comparison between the QE2 catchment and the Shenton Park catchment, both of which contribute water to the lake. Based on these figures, the shedding area contributing to the QE2 catchment is 0.09km^2 larger than that of Shenton Park. If 100% of the runoff from the QE2 catchment was pumped to Shenton Park Lake this catchment would supply 59% of the lake's water. This is not the case, however, and the real figure will be considerably lower. The estimate of the percentage of stormwater pumped interbasinally as a proportion of the total stormwater entering the lake was calculated as 48% from the McFarlane (1984) water balance.

2.3 Measurement of groundwater levels

2.3.1 Piezometer installation

Groundwater levels in the lake reserve were measured continuously by five water level probes with automatic loggers. Two probes were located up-gradient of the lake, two down-gradient with one probe in the lake itself (see Figure 2.2b). The loggers were all located at a depth so as to just intercept the water table. In all cases the piezometer holes were drilled through a medium grained sand so that the hydraulic properties of the soil at each of the holes were the same. The piezometer transect was aligned in the direction of regional flow as illustrated by McFarlane (1984). Holes 1, 3, 5 and 6 all contained water level probes, while the probe in the lake was housed in a protective plastic cover attached to the staff gauge. A further two holes were drilled on the up-gradient side of the lake, holes 4 and 2, but were never used due to the limited number of loggers available. Holes 3 and 5 were placed so as to provide data on the dynamic boundary between the lake and that of the surrounding groundwater. It is in this region that it was believed that the transition between flow-through and recharge patterns could best be investigated. Hole 4 was not deemed a suitable location for a logger as its close proximity to the lake meant that it was frequently flooded.

The configuration of piezometers as depicted in Figure 2.2b, aligned with the regional flow and both up-gradient and down-gradient of the lake, provides the most comprehensive coverage of groundwater fluctuation with limited instrumentation. It is noted that groundwater flow patterns at the perimeter of the lake, at points removed from the piezometer line, will not be adequately described from these records and will have to be described based on our current understanding of general flow patterns around lakes. Originally it was suggested that a second nest of piezometers might be installed perpendicular to the first to monitor groundwater seepage at the north-west and south-east corners. This was not possible due to budget constraints.

The loggers were installed on the 24/2/95, the logger in the lake on the 21/3/95. The final download of information was on the 31/8/95. This data series, therefore, provides a complete record of the transition between a summer and winter flow pattern. The loggers were set to record a water level reading every hour. The loggers were downloaded approximately every two months unless vandalism (hole 3) or a rising water table required them to be lifted and reset. The staff gauge and all of the holes were surveyed to above head datum (AHD). At each download the water level in the holes was measured using an electronic dipstick or, in the case of the lake, the level recorded from the staff gauge. These readings were used to check the accuracy of the final point of the logged data and to join each of the downloads to form a continuous data set, at the conclusion of the experiment.

2.3.2 Data conversion of groundwater record

Before installation the water level probes were calibrated using test tanks at CSIRO. The water level probe records a figure based on the level of water at a given height on the probe, but this is not in the form of a height as expressed in SI units. A calibration equation that is linear in form, had to be derived for each of the probes (and for all subsequent replacement probes) in order to process the raw logger data. This was the first step in processing the water level data. The second step was to convert all of the data to values expressed in AHD using the survey details. The time series data were then merged, using the water level measurements from the dipstick and the staff gauge (recorded at each download), as a check of accuracy, and as a means of aligning data sets where records were incomplete (i.e. missing data).

Processing of the data into spreadsheets and graphical form was completed using the graphics package Origin. Initially a temperature and conductivity probe was also located in the lake but this instrument failed after one month. The data is not presented here. This data would have had relevance to water quality studies but is not important to the water balance modelling.

2.4 Measurement of stormwater runoff - analysis of techniques

2.4.1 Estimates of runoff discharge via changes in lake volume

The hydraulic efficiency with which stormwater is transported ensures that in small catchments, such as Shenton Park, lake levels respond rapidly to runoff recharge and drainage of stormwater only continues for a short time following the conclusion of a rainfall event. The volume of stormwater runoff can therefore be calculated as the change in lake volume, if seepage to the groundwater is assumed to be equal to zero over this time period and direct precipitation into the lake is accounted for (note in early winter when the water table is low the effect of seepage may cause this first assumption to fail). This technique was used by McFarlane (1984) in calculating a rainfall/runoff relationship for the Shenton Park catchment.

In estimating the change in lake volume for Shenton Park Lake, only the change in height of the lake is required to be known as the area of the lake is approximately constant between 3.5 and 5.3m AHD. The McFarlane equation for the change in lake volume as a function of lake level is as shown below.

$$\Delta\text{Volume (m}^3\text{)} = 24847 \times \Delta\text{lake level (m)}$$

In the Shenton Park basin stormwater inputs can only be measured using this method until the lake water level reaches the height of the ocean outfall drain as any additional water will exit the system. The limitations of this technique are that if rainfall occurs over a long period then the assumption of zero seepage becomes less reasonable and there is potential for underestimation of the stormwater component. This technique is therefore restricted to rainfall events of short duration, if accuracy in the estimates is to be maintained.

In this investigation this technique was only used to calculate the proportion of stormwater contributed from the QE2 compensating basin. This pumped component of the water balance was calculated as the difference between the recorded increase in lake volume and the discharge from the Shenton Park catchment calculated using an area averaging technique from direct measurement of stormwater discharge (see section 2.4.2). It was found that direct precipitation into the lake could be discounted as negligible relative to the stormwater component.

2.4.2 Direct measurement of stormwater discharge.

Direct measurement of discharge can be achieved using one of two methods:-

The velocity-area method calculates discharge from the area of stream cross-section and the average stream velocity (equation 1). To calculate the stream cross-section in a pipe only the dimensions of the pipe (diameter) and the depth of water in the pipe are required. Both of the components depth and stream velocity are recorded continuously by the ultrasonic doppler instrument (UDI).

$$Q = VA \qquad (1)$$

Q = discharge (m^3/s or l/s)

V = average velocity (m/s)

A = cross-sectional area of the water (m^2)

The second method for calculating discharge is the slope-area technique, commonly used when reliable in-stream velocities are not available. It is questionable whether this is a direct or indirect method of measuring discharge with conflicting veiwpoints evident in the literature. Given the continuous depth records of the UDI and the certainty to which the other parameters of the Manning's equation are known, I suggest this method should be termed direct, recognising that the value of the Manning coefficient has to be a best guess estimate. Manning's formula is illustrated below (equation 2):-

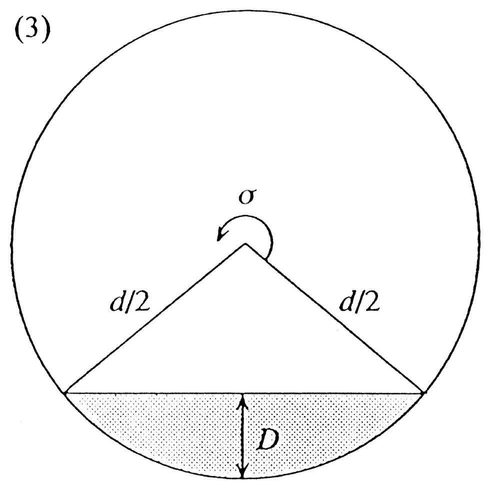
$$Q = \frac{1.49}{n} AR^{2/3}S^{1/2} \qquad (2)$$

where Q = discharge (m^3/s), n = "Manning's n", A = cross-sectional area of the flow (m^2), R = hydraulic radius (m), and S = slope. A value of n = 0.010 was used, for flow in a smooth concrete pipe as in Brater & King (1976), Linsley & Franzini (1971). The slope component is the average rise of the channel over the distance for which the depth has been described. Obviously in channels where the slope is changing over short distances the accuracy of this technique is limited. The hydraulic radius is calculated from the cross-sectional area of the flow and the wetted perimeter which is described further in the next section.

2.4.3 Calculation of cross-sectional area and the wetted perimeter

From the first series of data downloaded from the UDI it was apparent that many of the velocity recordings were spurious. This prompted a further investigation of the slope-area technique as a check for the discharge, as calculated by the velocity data from the UDI. In processing the data for both the UDI and the Manning's equation the cross-sectional area of the streamflow had to be calculated. This was derived by modifying an equation listed in Brater and King (1976) for the cross-sectional area of flow in a partially full circular conduit. The Brater and King (1976) equation calculated in degrees is as below (equation 3):-

$$A = \left(\frac{360 - \sigma}{360} \times \frac{\pi}{4} + \frac{\sin \sigma}{8} \right) \times \frac{D^2}{x^2} \quad (3)$$



A = cross sectional area of flow.

D = maximum depth of a partially filled conduit.

d = diameter of the conduit.

x^2 = D/d.

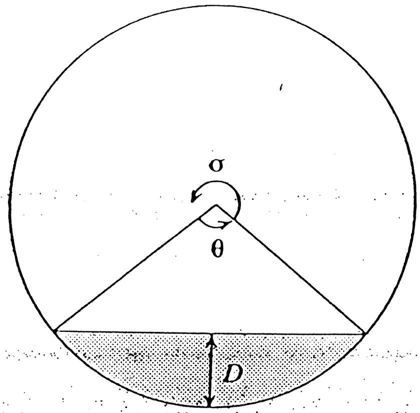
σ = angle between the radii subtending the water surface.

Also from Brater and King (1976) the expressions for the wetted perimeter p and the hydraulic radius R were taken.

$$p = \left(\frac{360 - \sigma}{360} \right) \frac{\pi D}{x} \quad (4) \qquad\qquad R = \frac{A}{p} \quad (5)$$

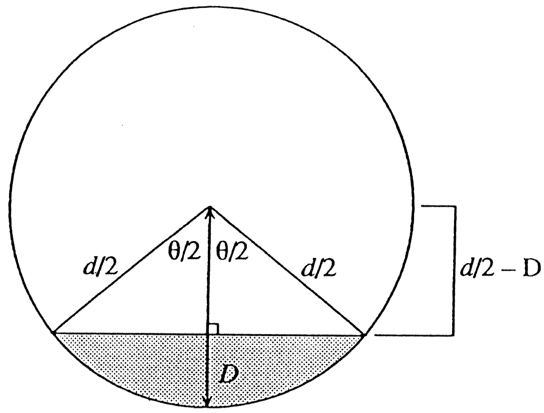
The variables are as listed above for equation (3).

The flow characteristics of drain 7, were that it flowed below half full at all times during the study. It is noted that these drains are designed to cope with peak discharges, in a wet catchment, for storm frequencies of return periods of perhaps one in fifty years and so the drain will only ever be partially full under normal operating conditions. In modifying the equations of Brater and King (1976), (3) and (4), a series of equations were derived, which were only suitable for determination of the cross-sectional area in flows below half pipe full (i.e. D < d/2). The equations use the reciprocal angle of the angle σ, symbolised by θ, which is equivalent to $360^\circ - \sigma^\circ$ as illustrated below.



The angle θ varies with depth but can be described in terms of d and D, such that the depth becomes the only variable, given that d is constant, see below.

$$\cos\frac{\theta}{2} = \frac{\frac{d}{2} - D}{\frac{d}{2}} \qquad (6)$$



$$\theta = 2\left[a\cos\left(\frac{\frac{d}{2} - D}{\frac{d}{2}}\right)\right] \qquad (7)$$

Equation (7) was then substituted into equation (3) for θ. The term d/2 is replaced by r (radius) and the angles are calculated in radians :-

$$A = \left[\frac{2a\cos\left(\frac{r-D}{r}\right)}{2\pi} \times \frac{\pi}{4} + \frac{\sin\left(2\pi - 2a\cos\left(\frac{r-D}{r}\right)\right)}{8}\right] \times \frac{D^2}{\left(\frac{D}{r}\right)^2} \qquad (8)$$

The same substitution was then made for equation (4) the wetted perimeter so that:-

$$p = \frac{2a\cos\left(\frac{r-D}{r}\right)}{\pi} \times d \qquad (9)$$

2.4.4 General description of the UDI.

The ultrasonic doppler instrument (UDI) combines velocity, depth and temperature sensors linked to a data logger. A power/communications cable runs from the UDI connecting it to a power source, a 12 SB10 battery, and allowing the logger to be downloaded to a laptop (see Figure 2.4.4).

During the experiment two doppler units were trialled, a prototype version and a production model. In both instruments water velocity is measured acoustically by doppler shift from particles carried in the water. Water depth in the UDI prototypes is measured by "time of flight" acoustics. A "ping" is transmitted from a sensor mounted at the rear of the doppler. This echoes from the water surface and the time taken to receive the signal is then calculated as a depth measurement. The process is described by the simple formula:-

$$2D = t \times c$$

D = Vertical depth of water above sensor	mm
t = Time between transmit and receipt of signal	sec
c = speed of sound in water	mm/sec

The production UDI uses a pressure transducer to measure the weight of water flowing over it. This is then converted to a depth reading and logged. The transducer needs to be equilibrated to atmospheric pressure and this is done through a drying tube housed beside the battery.

Temperature can be logged and used to refine the acoustic data which can be affected by variations in the speed of sound due to changes in water density. This facility was not used in the study trials.

2.4.5 The theory of doppler shift.

The principles of doppler shift were first described by Christian Doppler (1843), who observed that a change in the frequency of sound, light and radio waves was a function of their motion (Chalk 1995). When sound is reflected from a moving target the frequency of the sound is varied by the velocity of the target. This variation is known as doppler shift. The UDI uses particles in suspension, even air bubbles, as acoustic targets (or carriers). The UDI is an incoherent doppler instrument as opposed to a coherent (or profiling) doppler instrument in that it sends out a continuous signal and measures any signals anywhere and everywhere along the beam (WAWA 1994). Profiling instruments target specific locations and only measure these carriers' reflected signal. In a measuring cycle the transmitter produces a beam of ultrasonic sound at a fixed frequency known as a carrier frequency. This beam is received by a particle (or reflector) in the flow at a different frequency to the carrier frequency. Due to the doppler shift a reflected signal is picked up by sensors located at the front of the UDI. Two doppler shifts occur during this exchange, one shift as the signal is passed from the UDI to the reflector and a second shift when the signal is passed back from the reflector to the transmitter (UDI).

A measuring circuit detects the change in frequency caused by the doppler shift. A processing system accumulates and analyses the signals and calculates one median doppler shift from the range received (WAWA 1994). The velocity calculated from the doppler shift is as illustrated below

$$2V = D \times \frac{c}{f}$$

V = Water velocity	mm/sec
D = Doppler shift	Hz
f = Carrier frequency transmitted	Hz
c = Speed of sound in water	mm/sec

The velocity measured is the component along the beam. The beam is at an angle to the flow and so the velocity is adjusted by the angle cosine.

2.4.6 Programming the UDI.

The UDI can be programmed to record at different intervals depending on the specific requirements of the study and the available memory to store the data (the memory can be cleared each time the logger is downloaded). The recording characteristics can be modified by creating different 'schemes', which define the logging time and scan rate. The machine used was programmed with a log interval of 5 minutes (standard WAWA procedure, personal comm. Ian Tite), where the median doppler shift calculated in the minute between minutes 4-5 is logged by the UDI. The UDI has a scan rate or 'cycle', that is it emits a carrier signal and receives many hundreds of reflected signals, approximately every 15 seconds (± 0.25 sec's). This scan rate is automatically checked every time the logger is restarted after being downloaded.

2.4.7 Locating the instrument: site description.

The "UDI guidelines for operation and processing" (WAWA 1994) suggest that the sensor should be located near the downstream end of a straight and clean culvert, where linear flow conditions are maximised. Few potential sites were available in the drain 7 catchment that provided adequate access for maintenance and would not be subject to vandalism. The primary consideration in locating the UDI was that the discharge recorded by the doppler was the same as that entering the lake, i.e. there should be no further discharge into the drain downstream of the UDI. The location chosen was through a manhole on Derby Road. The manhole at this point opened at the intersection of three smaller drains that feed drain 7. The meeting of water at this junction caused considerable turbulence even at relatively low flows and so the UDI was located as far downstream as the length of the power/comm. lead would allow, approximately 13m. This location provided the best conditions of laminar flow and being located near the middle of the pipe, was well described using Manning's equation.

An expansion ring was used to secure the UDI in the drain and it was ensured that the instrument sat on the bottom of the pipe to avoid debris catching beneath it. The battery and drying tube were housed in a waterproof casing and hung below the manhole cover from a padlocked chain. The doppler was located so that the velocity sensors pointed downstream, to avoid debris collecting on the sensors and causing spurious readings. It is recognised that in "some" channels the sensor body may disturb the velocity distribution unacceptably when the sensor is aligned in this direction.

Maintenance of the logger required data to be downloaded once weekly at which time the battery was also replaced. The silica in the drying tube was replaced when required. Regular drain inspections were made to ensure that debris did not build around the UDI.

2.5 Measurement of the pumped component of stormwater in the lake water balance equation

The pump station at Aberdare Road maintains a maximum water level in the QE2 compensating basin by pumping excess stormwater to Shenton Park Lake. This station was monitored throughout the month of August to determine the level of water discharged to the Shenton Park basin. Two pumps operate at the station and these were fitted with loggers to record how many starts each pump had, on an hourly basis, and also to record the duration of the pump time. The pumps control the level of water in the basin automatically, operating by a float system. The pumps can run independently or in parallel depending on the pumping capacity required and the monitoring equipment could distinguish whether only one or both pumps were operating. This data with pump test results for discharge determined for both pumps operating singly or in parallel were used to estimate the volume of stormwater recharging the lake from the QE2 basin.

2.6 Estimation of runoff through ocean outfall.

This drain proved very difficult to monitor as access is virtually impossible with the nearest manhole located approximately 13m above the pipe. The only method suitable for estimating discharge was with the use of a DDT-200 Ultrasonic Depth Datalogger. This instrument has a number of distinct advantages over depth pressure transducers as used in the UDI. Firstly this instrument is easily installed by simply lowering the probe to just above the flow, which removes the necessity to actually climb into the drain. Secondly it is virtually impossible to foul the sensor with debris. The unit works on similar acoustic technology as the depth sensor on the prototype UDI. The logger has an internal battery which eliminates the need for weekly maintenance and only two downloads were required using a logging cycle of 15 minutes.

The sensor is located so that it hangs directly above the flow and records the depth of water by bouncing a signal off the surface of the water and calculating the return time. A problem was presented when it was discovered that beneath the manhole the pipe was not continuous. There was a sandy section (perhaps a sediment trap) so that the sensor during no flow may have returned distorted signals, not having a reflective surface with which to gain a return signal. This problem was overcome by building a temporary flume structure to join the pipe. The sensor was calibrated on installation and was again checked for accuracy on removal by comparing the last logged depth to a depth taken using a dipstick.

Results were calculated using a MICROCOM program using a Colebrook-White conversion for flow. The roughness coefficient was set at 1 and the pipe gradient was taken from WAWA plans to be 755. A height adjustment of -20.0 mm was taken from the initial calibration test.

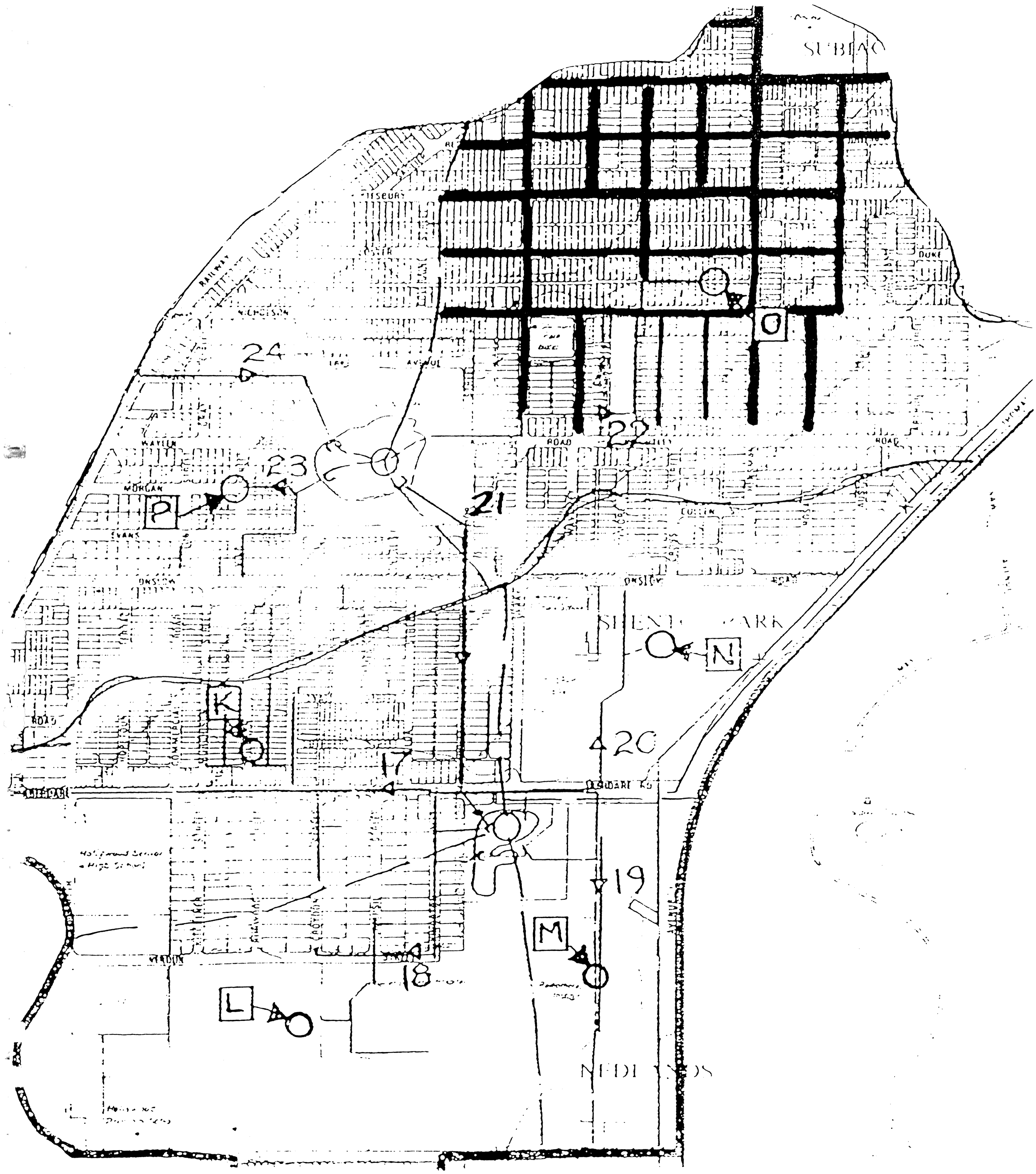


Figure 2.2a The Shenton Park catchment (north) and QE2 catchment (south) with sub-catchments O, P, K, L, M and N defined. Drain 7 catchment shaded black.

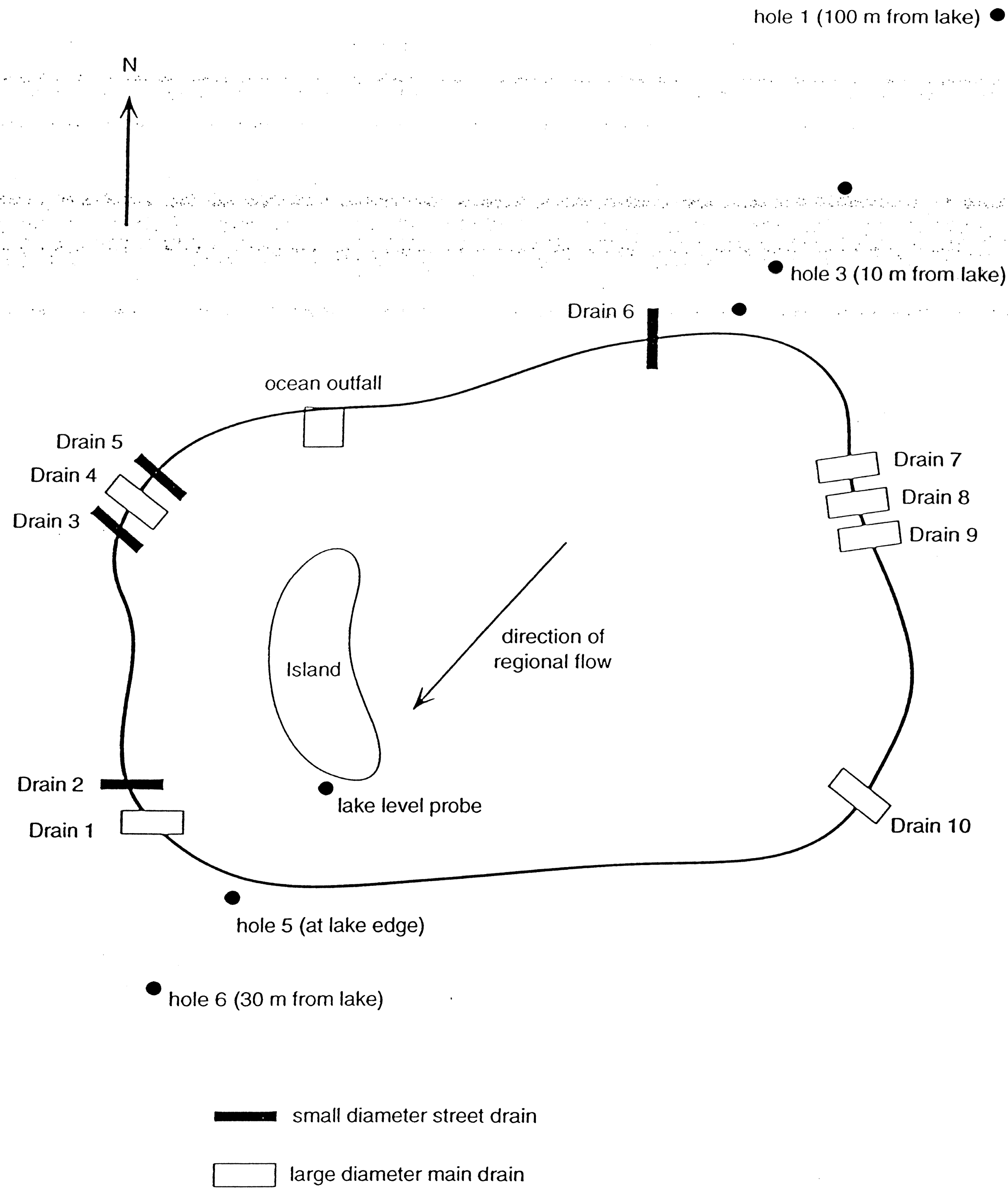


Figure 2.2b Shenton Park Lake study site.

Subcatchment	Area (km^2) (WAWA)	Fi value (WAWA)	Shedding area km^2
Shenton Park	1.5	-	0.2
O	0.89	0.15	0.12
P	0.61	0.15	0.08
Drain 7	0.52	-	-
QE2	1.39	-	0.29
K	0.40	0.13	0.05
L	0.50.	0.26	0.12
M	0.19	0.54	0.09
N	0.30	0.10	0.03

Table 2.2 Shedding areas in Shenton Park and QE2 catchments, calculated from WAWA Figures.

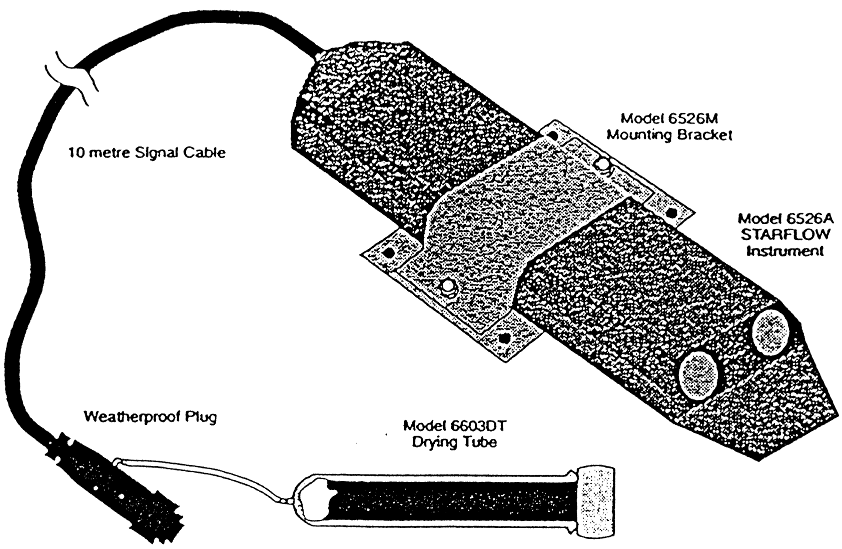


Figure 2.4.4 The UDI (Doppler instrument)

3. ESTIMATES OF LAKE WATER BALANCE - RESULTS AND DISCUSSION

3.1 The stormwater recharge component of the lake water balance

3.1.1. UDI depth and velocity distribution

Standard depth and velocity distributions as recorded by the UDI, in drain 7, are shown in Figures 3.1.1a and 3.1.1b. The variation in the depth record across the study period was well correlated to the rainfall as recorded on an hourly basis at the Shenton Park Field station and also over the daily record from the lake site. Calibration of the UDI in flow tanks at the Welshpool WAWA instrument facility and on site monitoring showed the depth record to be accurate to within 1-5mm. The only anomalies between the depth and rainfall records occurred when runoff was entering the drain from reticulation or, on one occasion, as the result of runoff from a building site.

Figure 3.1.1b clearly shows the periods of high constant velocity of over 1000mm/sec logged by the doppler instrument. These records are from periods of zero or very low flow where the acoustic signal emitted by the UDI is affected by erroneous signals received from the walls of the pipe. If a satisfactory depth/velocity relationship could be identified from the data these spurious readings could be easily edited from the data set.

3.1.2 Determination of a velocity/depth relationship

As a check on the validity of the logged UDI data it is important to establish that a velocity/depth relationship exists between the two records. This is best achieved by analysing a scatter plot of the data (see Figure 3.1.2). The region through which a linear relationship has been plotted, between 0 and 50mm, characterises what should be seen throughout the entire depth range, that is a series of closely spaced data. Above 50mm the scatter plot shows a series of constant velocities over the range of 150-350mm/sec. These points indicate intervals over which the doppler cannot calculate a median stream velocity as a result of large variation in the received signals; usually caused by excessive turbulence. The UDI program safeguards against recording values with large associated errors by establishing a maximum quartile range for the velocity distribution. For distributions in which the tails are excessively large so that the quartile range exceeds 100%, no median value is logged, rather the last accurate measurement is again recorded. This means that for each five minute interval where a suitable velocity distribution is not recorded by the UDI the last accurate value is redisplayed. This explains the constant range of velocities recorded at flows above 50mm.

A relationship determined over the depth range of 0-50mm cannot be extrapolated to 250mm, even if a linear relationship between depth and velocity in the pipe is expected, and be suitably justified. The velocity record is in this instance not an accurate predictor of discharge.

3.1.3 The rainfall/runoff relationship

The depth data was used as a predictor of discharge using the Manning formula by substituting it into the equations derived in the experimental procedures for the cross-sectional area and wetted perimeter of the flow (section 2.4.3). The variables in the Manning equation are as described in section 2.3.3, with a slope component of 0.027 taken from council drainage plans. The calculations were performed in an excel spreadsheet using the UDI depth data at five minute intervals. The discharges at five minutes were summed to give daily records. The days were not calculated from 00.00hrs but rather from 5pm so that they corresponded exactly to the rainfall data collected at the lake.

In estimating discharges, flows below 10mm were discounted as commonly water can become trapped in the recess in which the depth sensor is located and so a signal may be recorded for some time after the pipe is dry, until such time as the trapped water has evaporated. The difference to the overall discharge in discounting these small flows will be minimal in any case.

From the estimates of discharge over the months of July and August a rainfall/runoff relationship was determined for drain 7, (see Figure 3.1.3a). Using the known area of the drain 7 sub-catchment, 35% of the total, a discharge relationship was calculated for the entire Shenton Park catchment, not including waters pumped into the basin from QE2. The rainfall/runoff relationship, calculated from the UDI records and the linear relationship calculated by McFarlane (1984), are shown as Figures 3.1.3b and 3.1.3c. The accuracy of the relationship developed from the UDI records is limited to values under 30mm daily rainfall; the maximum rainfall that occured during the test period of July and August and therefore used as a predictor. The daily discharges as calculated over the entire study period from the first of February until the tenth of September are displayed graphically in Figure 3.1.3d.

3.1.4. Analysis of techniques in the estimation of stormwater runoff

A quadratic function was applied to the rainfall runoff data as the level of discharge is seen to increase with increasing daily rainfall (Figure 3.1.3b). This is as expected, for high daily rainfall is usually also associated with higher intensity falls that result in increased runoff rates. At lower daily rainfall some of the events will be high intensity over only a few minutes whereas others may be low intensity falls over the entire day. This results in larger variation of discharge at lower daily rainfalls. Although runoff coefficients are commonly set as 0.9 for impermeable road surfaces, at low intensity falls the effects of depression storage and evaporation do appear, based on this data, to significantly reduce discharge. It is also noted that at low rainfall levels the distribution across the catchment may be spatially more variable than when large events occur, increasing the error in only using one rainfall gauge.

Over the study period the runoff estimated from the UDI derived relationship was 33% lower than the discharge calculated from the McFarlane (1984) relationship. The overestimation of the McFarlane (1984) rainfall/runoff relationship is due to the inability to accurately predict discharge at low daily rainfall levels as shown in Figure 3.1.3d. The technique with which McFarlane (1984) developed the relationship was by calculating the increase in volume of the lake following rainfall; a technique which is quite effective over periods of high intensity rainfall, but, due to the effects of seepage, is less accurate when rainfall is of low intensity and distributed over a long period. In using a linear relationship to describe the data, the outlying large daily rainfall levels of 40-50mm dominate the fitting of the function and overestimates of low daily rainfall result. Note that both the McFarlane (1984) and the quadratic relationship produced here were significant at $r^2 = 0.957$.

At high daily rainfall values around 30mm the McFarlane (1984) and the UDI derived relationships are approximately equal. From McFarlane (1984) the runoff relationship is equivalent to 100% runoff from 16.5 % of the catchment (at a runoff coefficient of 0.9) while at 30mm the UDI derived relationship estimates a value of 17% (using a quadratic function means of course that this value changes depending on the level of rainfall). This shows very good agreement between the two methods over high rainfall events. These values are 4% higher than the calculated shedding area recorded by McFarlane (1984) using planiometric techniques, which suggests that estimates of shedding area from maps are not sufficiently accurate in estimating runoff for water balance studies of this nature.

- Over the study period from the start of February until the end of August an estimated 146,000m^3 of stormwater runoff entered the lake from the Shenton Park catchment.

3.2 Analysis of water level records

3.2.1 General trends in water level record

The water level records, Figures 3.2.1a and 3.2.1b are incomplete but provide enough data to make accurate predictions on the different flow regimes over the winter period.

The graphs show that until the break of season the groundwater gradient is as expected for a flow-through lake regime, the groundwater being highest in hole 1 and falling across the piezometer nest to a minimum level at hole 6. During the summer period the groundwater levels decrease due to losses via lake evaporation, evapotranspiration and pumping for irrigation. The rate of groundwater extraction appears to decrease from the months of February to May, based on the hole 1 record. One rainfall event occurs over the summer period, at the end of February, and a rise is seen in the lake level and the holes downgradient of the lake.

Over the winter the lake and groundwater levels respond rapidly to rainfall events. It is evident from the graphs that the catchment hydrology is dominated by the lake, being the principal recharge mechanism to the local groundwater system. If a significant volume of water was recharging the groundwater upgradient of the lake it would be expected that sometime after the rise in groundwater associated with seepage from the lake had finished, the water level in hole 1 would continue to rise due to water moving towards the lake from higher in the catchment. This is not the case.

The rises in water levels across the piezometers are in phase, the lag time in the responses being related to the distance of the probe from the lake perimeter. Both hole 5 and hole 3 are within 10m of the lake perimeter and show a rapid response to an increase in lake level. The responses in holes 6 and 1 are less pronounced.

In several instances over the time series, the groundwater record shows that the water level in hole 3 is higher than both the level in the lake and at hole 1, indicating the presence of a localised groundwater mound on the upgradient side of the lake. It should be noted that at all times during this study the water levels in holes 5 and 6 were lower than that of the lake so that no groundwater mound is present on the downgradient side.

The water level record in the lake and hole 5 both top out at certain points through the record. The proximity of hole 5 to the edge of the lake meant that when the lake flooded so too did this hole, if only temporarily. The probe in the lake recorded above the surveyed height of the weir on the ocean outlet but the record does not extend to the level of the grate which covers the top of the drain. What was noticed was that positioning of the probes in order to achieve a complete record was more difficult at the start of the year where water levels in the lake varied dramatically. Later in the year as the groundwater levels rose the variation in the lake level was restrained by the height of the surrounding groundwater and the bankfull level of the lake. It is important to note that the water level in the study area is limited to the height of the weir on the ocean outfall at 5.09m AHD which approximates the level in hole 1 from the end of July until the conclusion of the experiment.

3.2.2 Sensitivity of the groundwater record

The groundwater record of hole 3, even at the low resolution of Figure 3.2.1a, shows a distinct sinusoidal variation superimposed on the general decline from March through until early May. Figure 3.2.2a shows an enhanced plot of this data and from the accompanying time series it is clear that this variation is very regular. This groundwater response is due to the drawdown effect of a reticulation bore within the park reserve on the capillary fringe. This serves to demonstrate the very sensitive changes to groundwater level that are detected by the probes.

The accuracy of the surveying and water level records was checked by examining the lake water level data for a change in the seepage rate over the transition period when water was exiting the lake via the weir on the ocean outlet. The surveyed level to the bottom of the weir on the ocean outlet was 5.09m AHD. From an examination of Figure 3.2.2b a definite change in the seepage rate is seen to occur between 5.09 and 5.10m AHD so that the accuracy of the surveying and water level record is within one centimetre.

3.3 The pumped interbasinal component of stormwater to the lake water balance

Calculations based on the pump records at Aberdare Rd. (not provided) suggest that for any one rainfall event the proportion of stormwater supplied from the QE2 basin to the lake can vary between 0 and 60%. Commonly throughout the winter around 50% of the water entering the lake could be attributed to interbasinal pumping, based on the pump data collected from July and August and estimates derived as a residual of the increase in lake volume when runoff from the Shenton Park catchment is accounted for. This is in line with the 48% calculated by McFarlane (1984). The rates are highly variable, being dependent on the storage capacity of the QE2 basin prior to the storm. WAWA personnel suggested that in years where the water table was particularly high the pumps would switch on automatically for a short time each day just to maintain the level in the compensating basin; even when no rainfall was occurring.

Although the number of pump hours were accurately recorded, the pump ratings are not to be relied on as the pump tests were conducted over seven years ago. The pumps can also become blocked with debris and leaves so that the number of pumping hours is not a sufficient guide in calculating discharge. The best estimate that can be provided for an interbasinal pumping component to the stormwater balance is 48% of the overall budget as calculated by McFarlane, a figure that is also supported by this study's findings. It would have been preferable to have fitted another UDI unit to record continuous discharges in this drain.

3.4 Discharge to ocean outfall drain

Over the recorded period from the 3/8/95 to the 7/9/95, 19,500 m^3 of water left Shenton Park Lake via the ocean outfall (see Figure 3.4). This represents 30% of the stormwater that entered the lake over this period that failed to recharge the groundwater. In one day on the 6/9/95 10,000m^3 left the lake. Based on the runoff estimates on this same day a further 10,000m^3 of water would have flooded the surrounding parkland. These figures indicate the large component of stormwater that is not recharged to the groundwater and its significance to the lake water balance, particularly in the later months of winter. A 17% estimate of error is associated with these figures, calculated from the MICROCOM package.

3.5 Concluding comments

Velocity estimates from the UDI were seriously affected by turbulence effects in the pipe and perhaps also due to the positioning of the instrument facing downstream. Improved results may have been achievable by locating the velocity sensors pointing upstream if a simple filter could have been constructed over the drain inlet to prevent debris from accumulating on the sensors. The rainfall/runoff relationship constructed using the UDI depth records suggests that only direct measurements of discharge from pipes are sufficiently accurate in estimating the stormwater component of a water balance and that techniques based on estimates of shedding area are less reliable. In this study the accuracy to which the stormwater component of the lake could be calculated was limited by having no direct estimates of discharge of the pumped interbasinal water.

The water level in the catchment surrounding the lake indicates water levels in the area are limited to a maximum level of 5.1m AHD by the presence of the ocean outfall. This therefore limits the potential recharge to the catchment so that in late winter runoff waters pass in large volumes through this structure, resulting in the loss of a valuable resource.

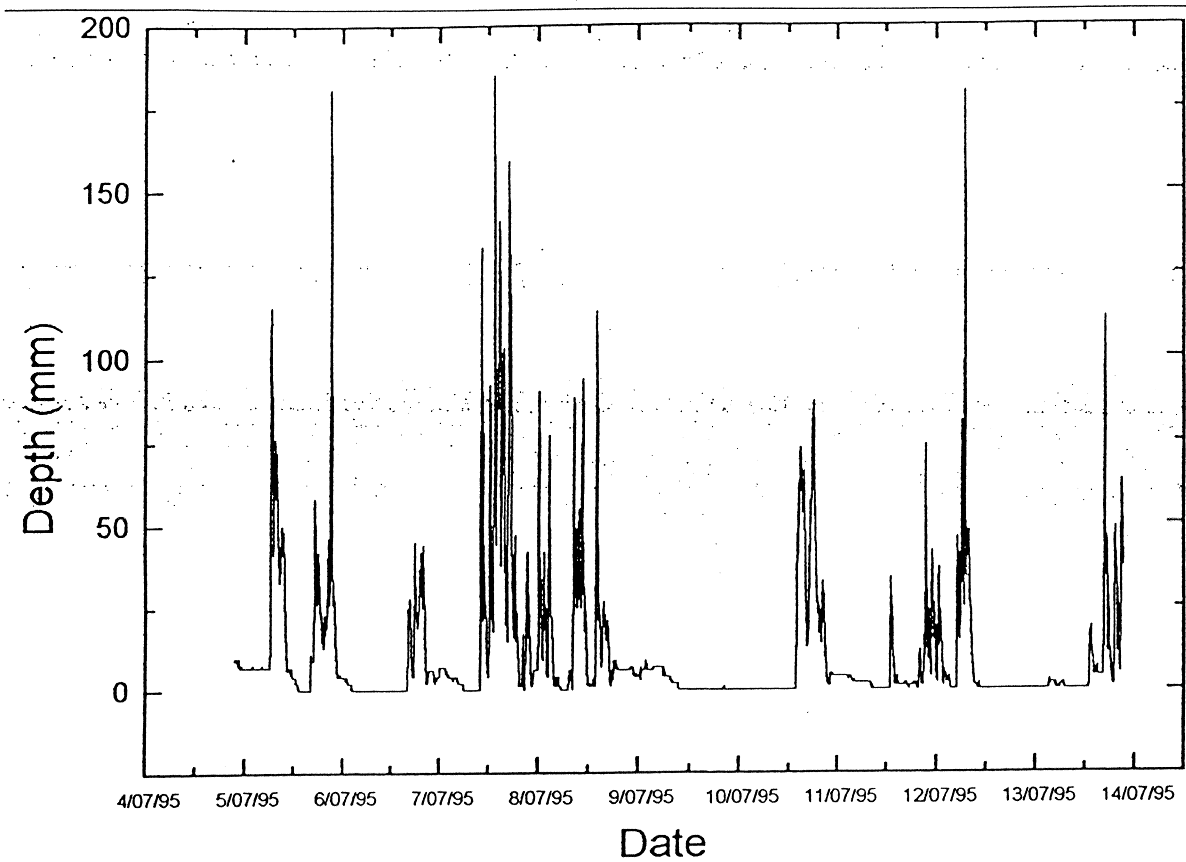


Figure 3.1.1a Depth distribution as recorded by UDI drain 7, July 1995

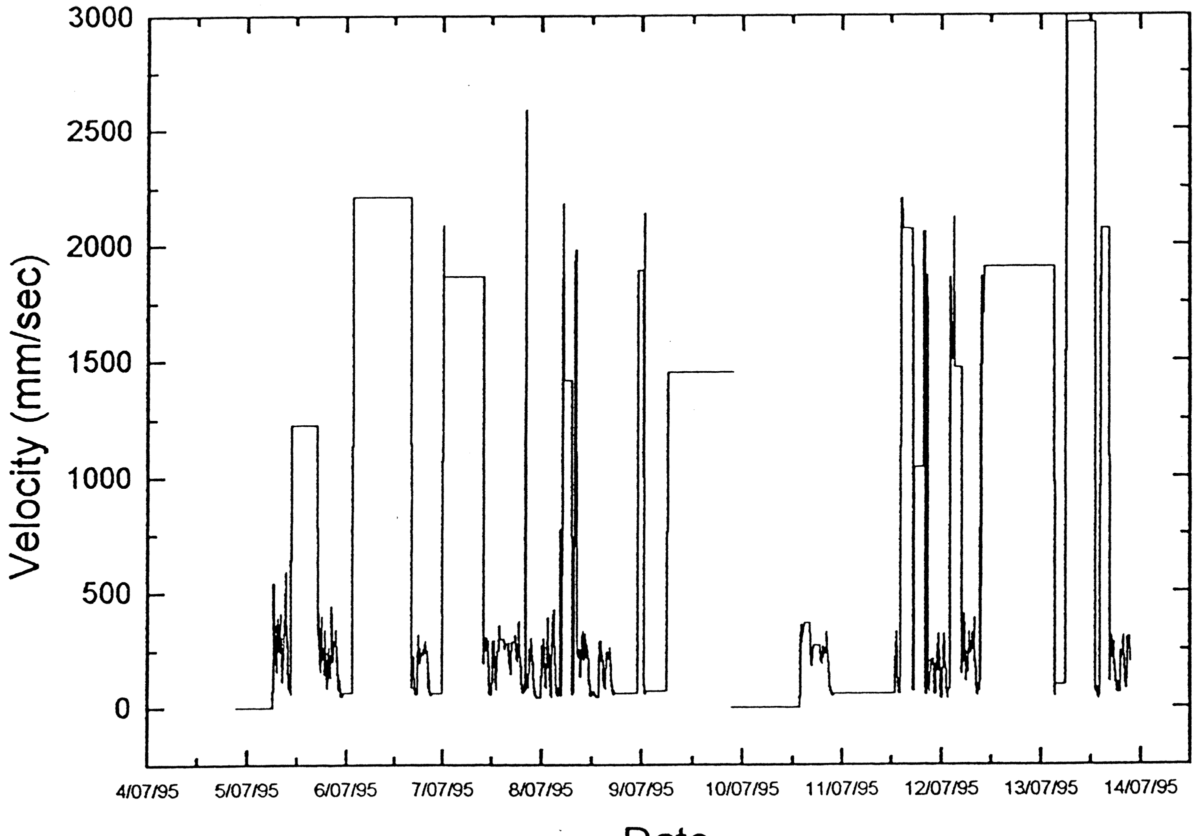


Figure 3.1.1b Velocity distribution as recorded by UDI drain 7, July 1995

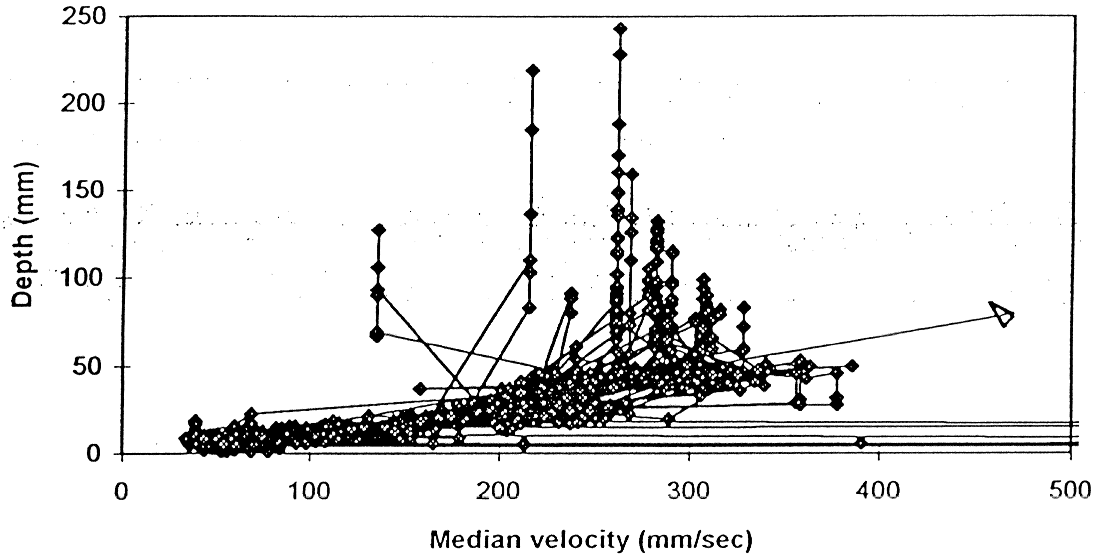


Figure 3.1.2 Scatter plot of depth vs velocity as recorded by UDI

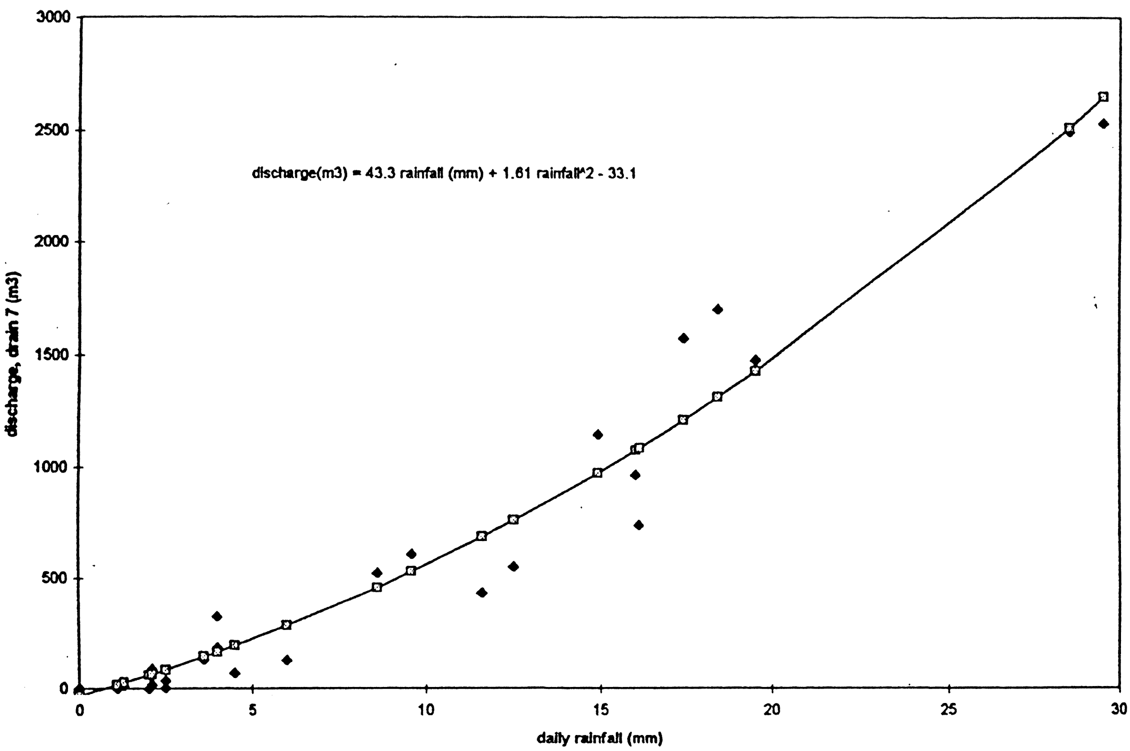


Figure 3.1.3a Rainfall/runoff relationship as calculated for drain 7

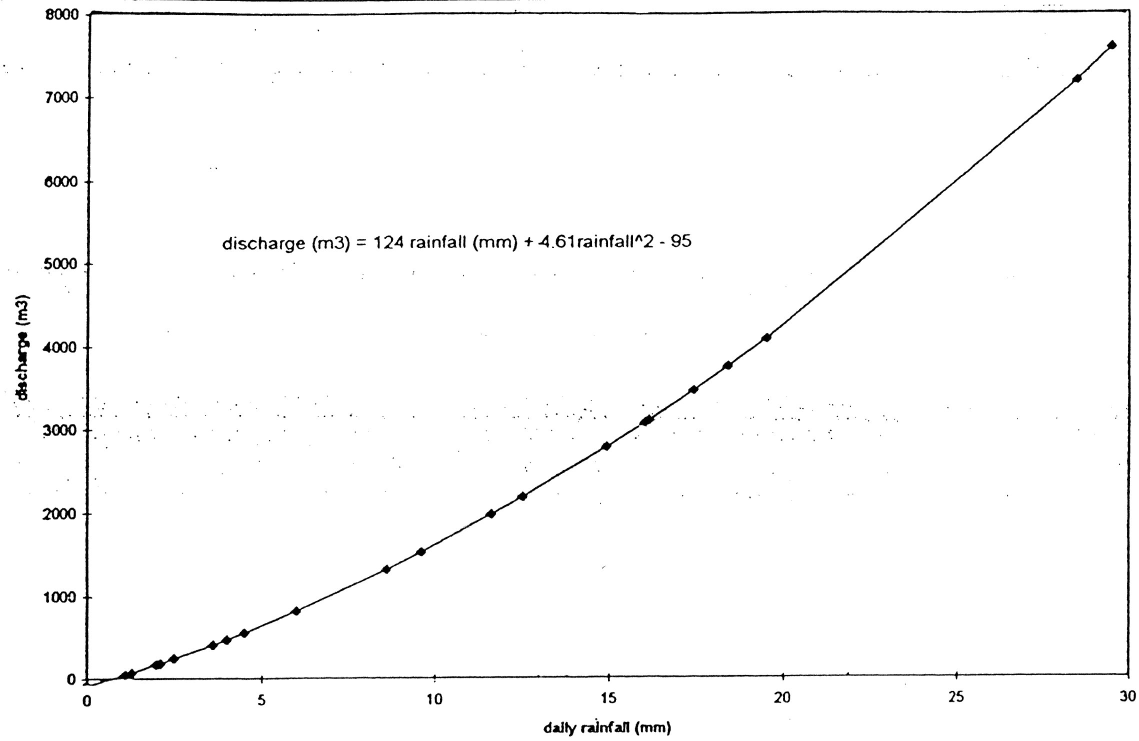


Figure 3.1.3b Rainfall/runoff relationship for the Shenton Park catchment derived from UDI results

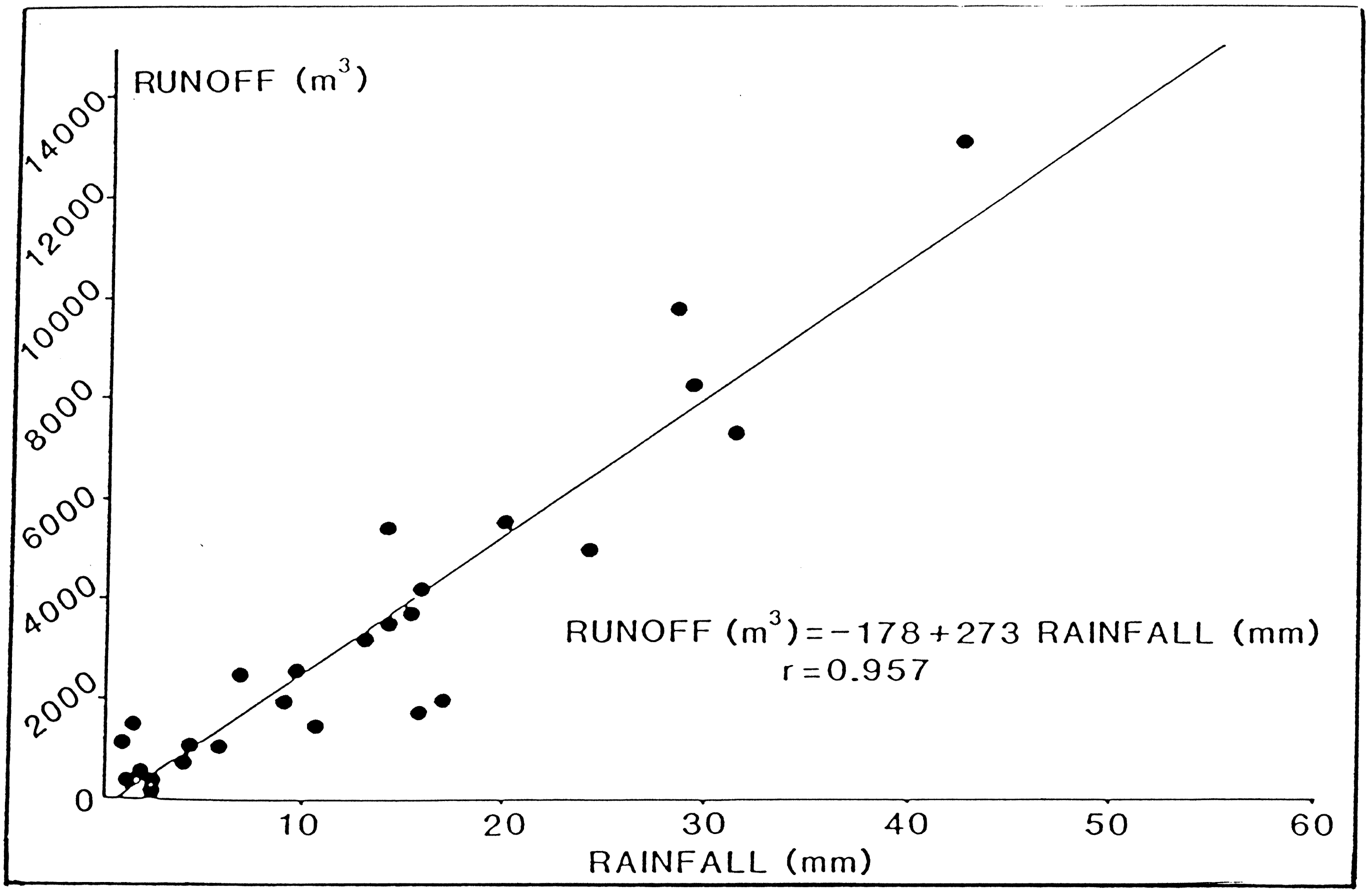


Figure 3.1.3c Rainfall/runoff relationship for the Shenton Park catchment as derived by McFarlane (1984)

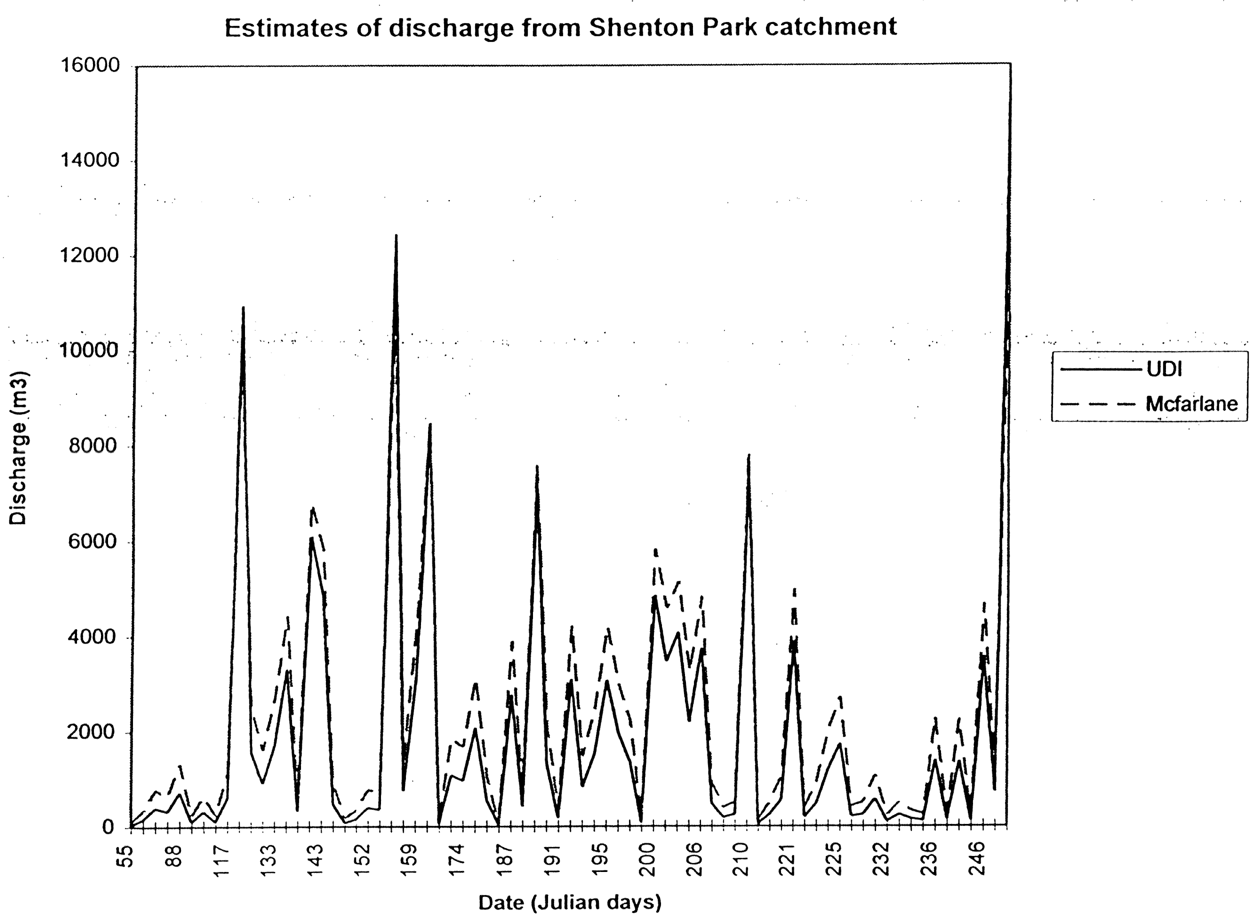


Figure 3.1.3d

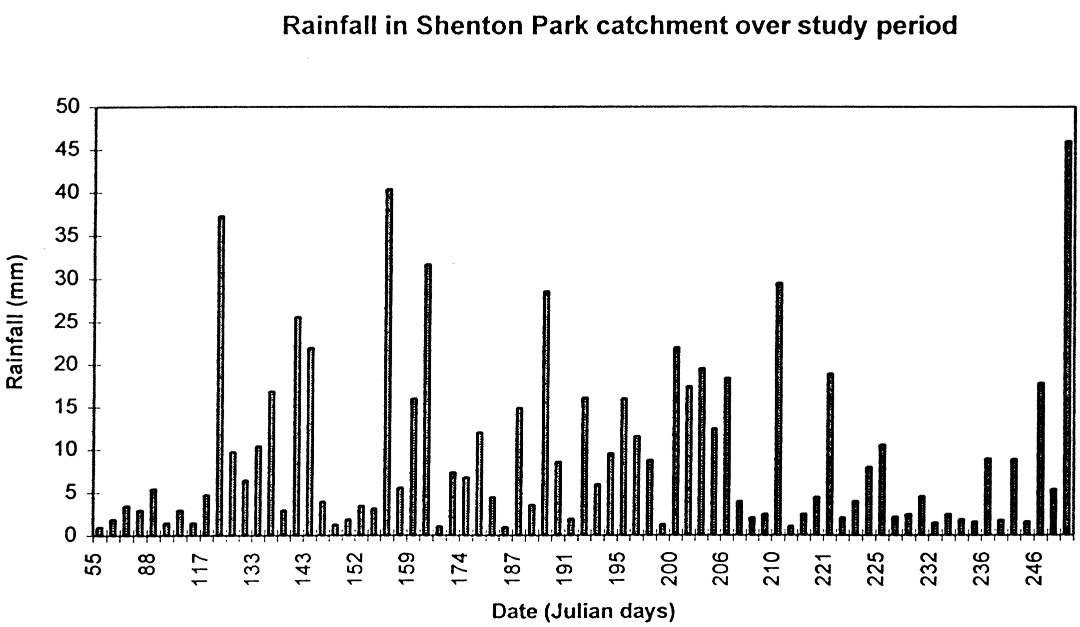


Figure 3.1.3e

Figure 3.2.1a

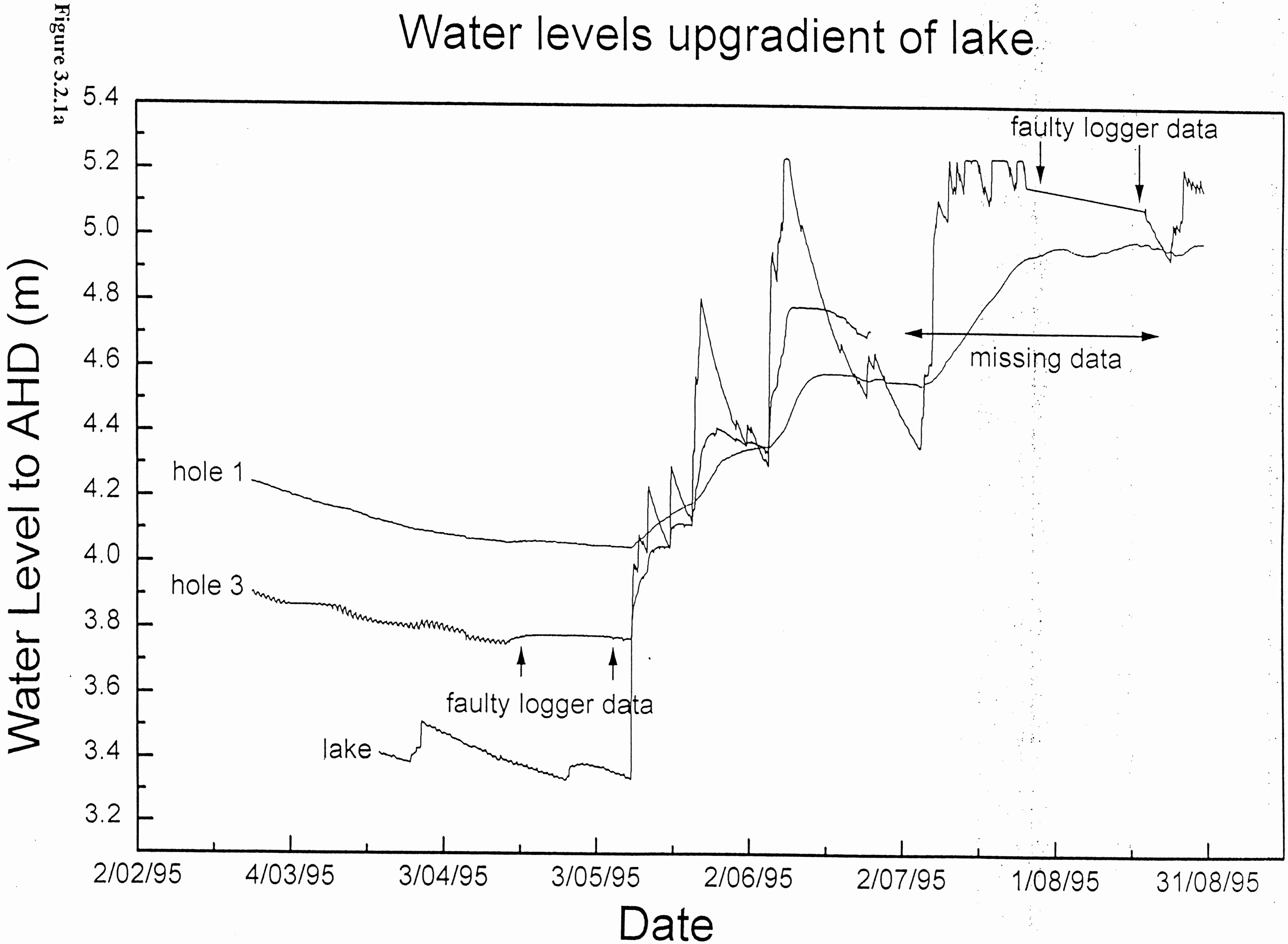
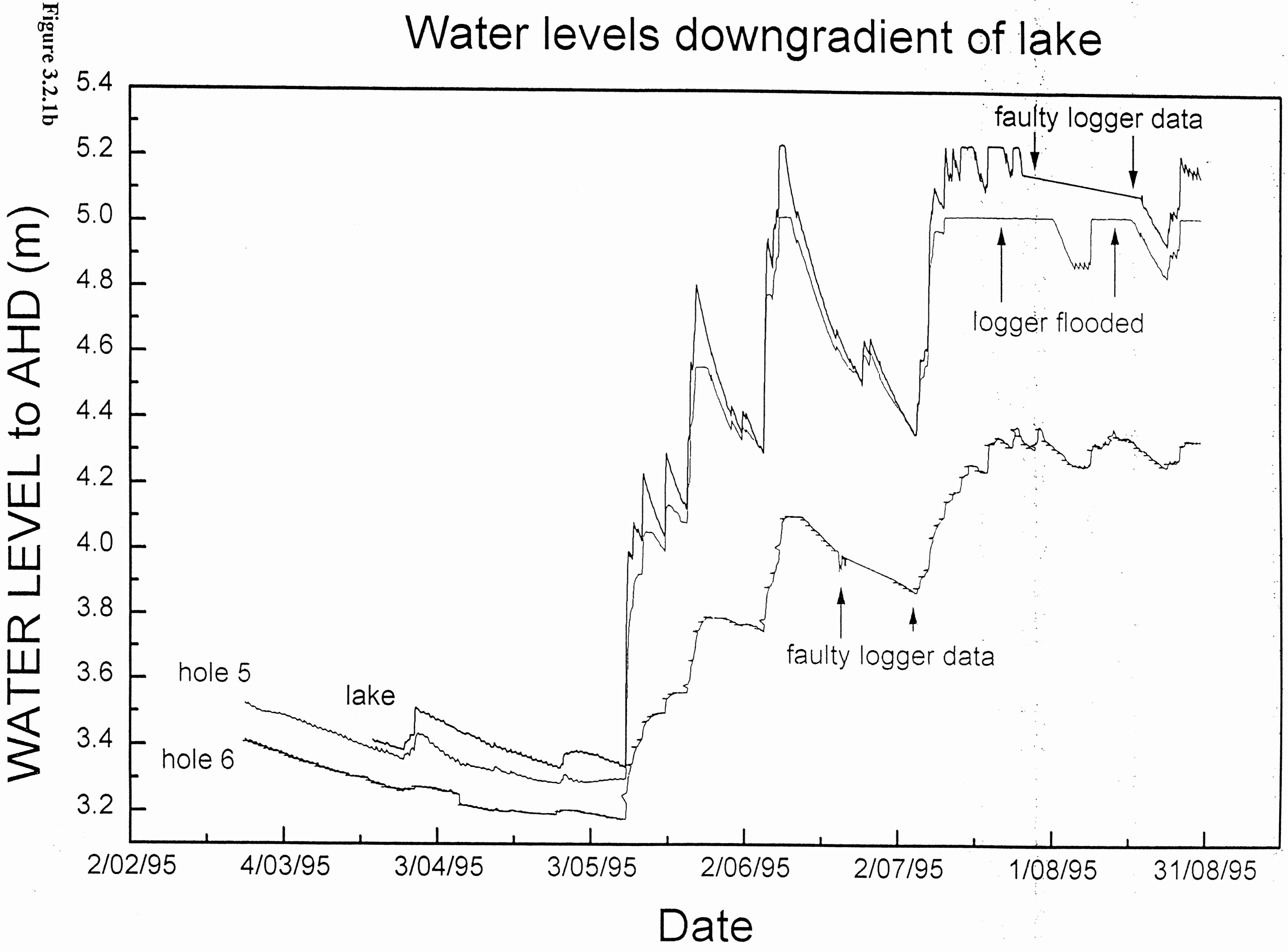

Water levels upgradient of lake
Water Level to AHD (m)
5.4
5.2
5.0
4.8
4.6
4.4
4.2
4.0
3.8
3.6
3.4
3.2
faulty logger data
missing data
hole 1
hole 3
faulty logger data
lake
2/02/95
4/03/95
3/04/95
3/05/95
2/06/95
2/07/95
1/08/95
31/08/95
Date

Figure 3.2.1b


Water levels downgradient of lake
WATER LEVEL to AHD (m)
5.4
5.2
5.0
4.8
4.6
4.4
4.2
4.0
3.8
3.6
3.4
3.2
2/02/95
4/03/95
3/04/95
3/05/95
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hole 5
lake
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faulty logger data
logger flooded
faulty logger data

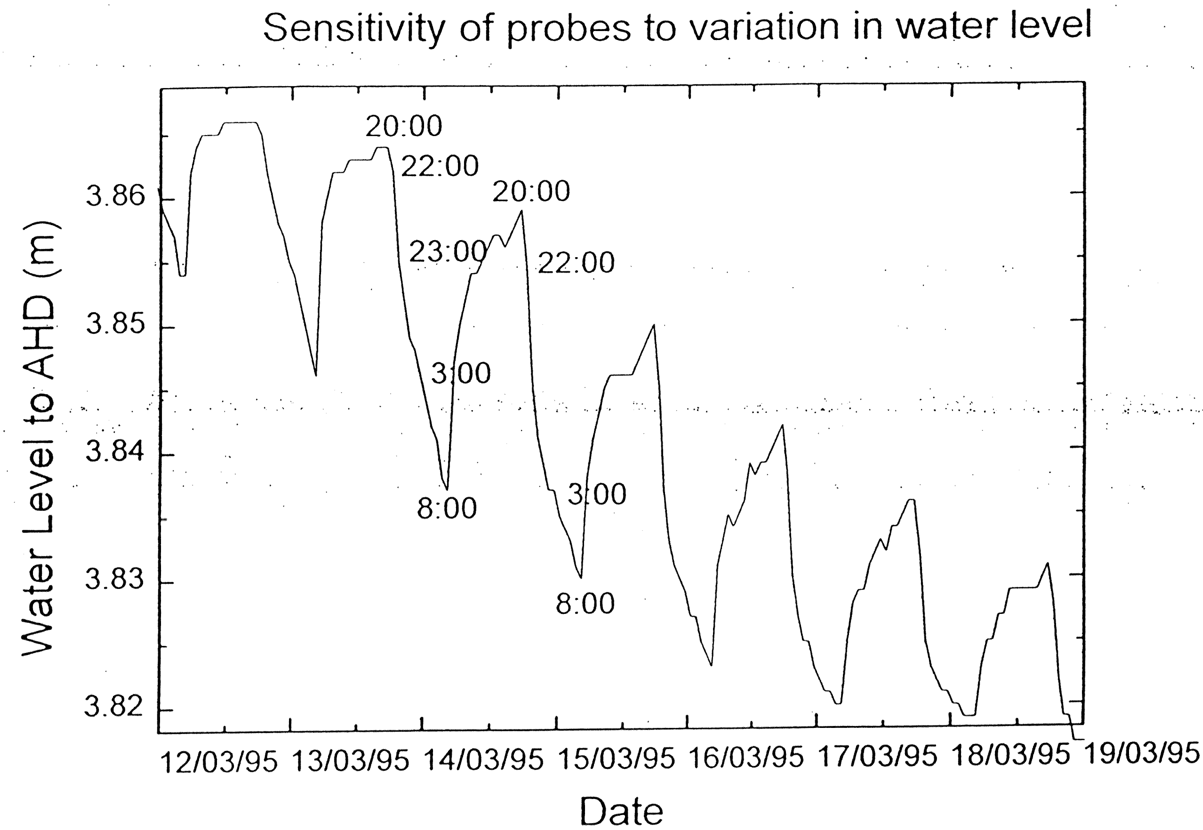

Figure 3.2.2a

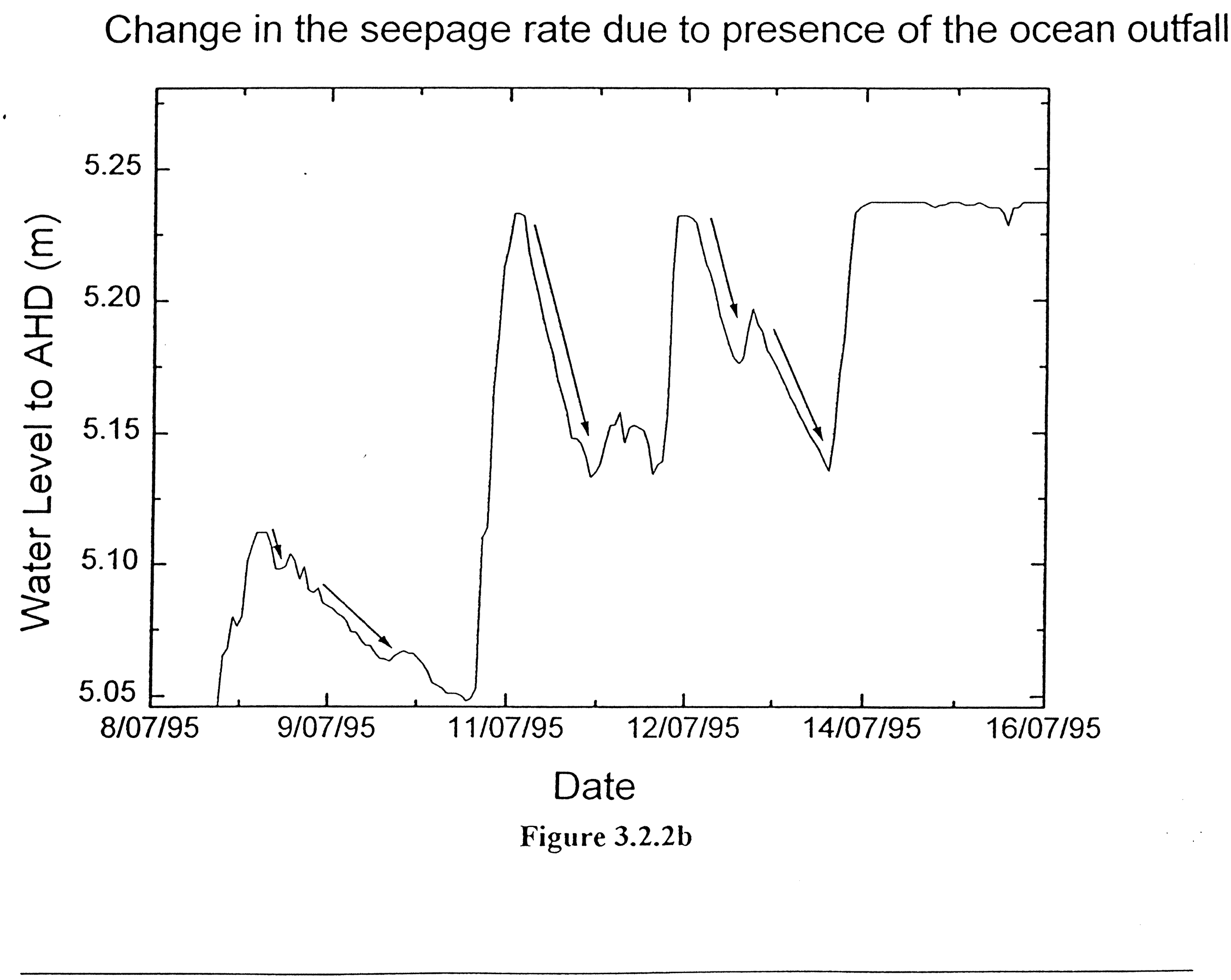

Figure 3.2.2b

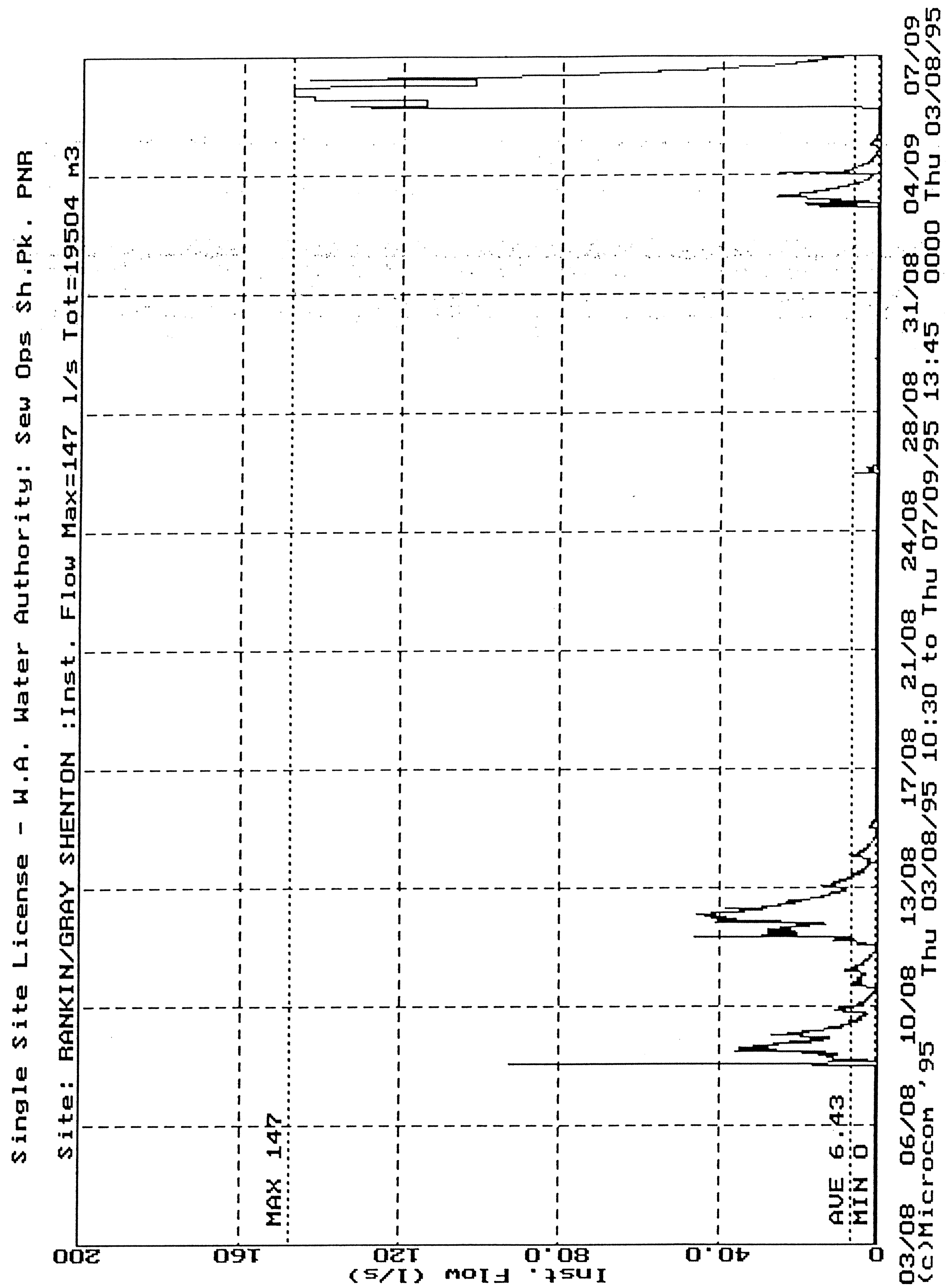

Figure 3.4 Instantaneous flow as recorded in the ocean outfall drain

4. MODELLING PROCEDURES

The flow dynamics and groundwater fluxes of the lake/aquifer system were plotted using a linear triangular finite element model, AQUIFEM-N (Townley, unpublished). A fine mesh grid was constructed by Tony Barr (CSIRO, Div. of Water Res.), approximating the geometry of Shenton Park Lake as a circular waterbody of diameter 200m, and the boundaries of the flow system, at 1km wide and 1.5 kms long. The solution technique used a time step series of one day, such that the heads are specified at 141 intervals. Forty nodes specify the perimeter of the lake while the grid is composed of 1000 elements and 550 nodes (Figure A.2.4). At any time the heads in the lake are equal so that the lake can be described by just one node on the perimeter of the lake (node 151). The AQUIFEM-N data set, from which groundwater responses were modelled, was constructed by Lloyd Townley using daily lake level records. Calibration and sensitivity analysis of the model was conducted by Lloyd Townley in conjunction with the author.

The model assumes that the aquifer properties are homogeneous. The net flux at the boundaries of the model are described by the transmissivity and the regional gradient so that:

$$Q = T^* \times gradient$$

Q = net flux at the boundaries

T* = transmissivity (confined aquifer) = aquifer thickness (B) x hydraulic conductivity

The system was linearised by assuming confined aquifer conditions, where B is equal to 20m. In reality the transmissivity in an unconfined aquifer is variable as B changes due to the rise and fall of the water table. It is reasonable, however, to suggest that a one metre rise in the water table is insignificant over the 20m aquifer such that T*=T (transmissivity of unconfined aquifer). A value of 0.3 was set as the specific yield of a sandy aquifer, as in Allen (1981). The remaining parameters as yet undefined, the gradient, hydraulic conductivity and the level of rainfall recharge (distributed spatially over the entire catchment) were varied over a series of runs, in a sensitivity analysis of the model. Below is a description of the variables in each run.

VARIABLE	RUN 1	RUN 2	RUN 3	RUN 4
Gradient	1:500	1:1000	1:500	1:500
Hydr. cond.	10 m/day	10 m/day	20 m/day	10 m/day
Rainfall Recharge	0	0	0	1.6mm/day

Estimates of two sands' hydraulic conductivities in the Shenton Park catchment by McFarlane (1984) of 7 and 20 m/day were used as a guide. A gradient between the holes 1 and 6 at the end of summer of 1:500 established a baseline for the model gradient, which was also supported by gradients calculated across the catchment by McFarlane (1984). The recharge parameter was evenly distributed across the catchment at a continuous rate of 1.6mm/day over a 75 day period from day 48. This is equivalent to 100% recharge to the groundwater of the rain falling between the start of June to the end of September over 18% of the catchment. McFarlane (1984) estimated that 28% of the Shenton Park catchment area was covered by roofs which had potential to recharge the groundwater. It is recognised, however, that infiltration rates are considerably smaller than the potential due to inefficient soak well systems (McFarlane 1984).

Calibration of the model was by comparison to the groundwater records of holes 1 and 6. The seepage fluxes for each run were also calibarated against estimates of the change in lake volume based on the McFarlane (1984) estimate of the true area of the lake (see section 2.3.1).

The FLOWTHRU package was used principally as a conceptual tool during this study. Its limited number of precalculated solutions mean that the range of parameters that can be modified is very much restricted before changes to the pre-set geometries of the lake are enforced. Classification of the flow regimes at Shenton Park Lake were, however, based on a working knowledge of the package and have been presented in the same form, with the lake illustrated as an infinitisimally thin layer on top of the aquifer and the system modelled in vertical cross-section (see section 5.3).

5. MODELLING RESULTS AND DISCUSSION

5.1 Description of model output

From the AQUIFEM-N model four series of plots were produced to describe the flow dynamics at the lake. Plot type 1 (see Figure A1.1) displays water level plotted against time; the models predicted groundwater responses to variations in the lake level. This plot was used to calibarate the model against the field measured responses from holes 1 and 6. Plot type 2 (see Figure A1.2) illustrates the change in water level as a function of distance from the lake. Plot type 3 (see Figure A1.3) displays the flux into the aquifer from the lake at three specially selected nodes. Plot type 4 (see Figure A1.4) is a areal plot of the aquifer, illustrating the direction and magnitude of the groundwater fluxes by a series of arrows (arrow length is proportional to the flux so that a 1mm arrow represents a flux of 0.25 m^2/day or as a function of aquifer width 0.25 m^3/day per metre width of aquifer). Periods of special interest in the time series were selected and plotted in areal (plot type 4) and in vertical cross-section (plot type 2). (In both instances the regional groundwater flow enters the system through the right hand boundary of the illustration). These included days which were representative of a certain flow regime or a series of days that showed the dynamic transition between regimes.

Plot type 1 displays the water level at nodes 129, 140, 151 (lake), 162, 172, 182, 192, all of which are aligned along the regional gradient such that the grid is symmetric about them (see Figure A2.4). These nodes represent distances from the centre of the lake of -144, -122, 100 (lake perimeter), 122 144, 167 and 189 metres, the negative values representing locations downgradient of the lake. In calibarating the model the piezometer records from holes 1 and 6, at 200m and -130m from the centre of the lake were compared to the modelled values at nodes 192 and 140.

5.2 Modelling results

5.2.1 Variation in the groundwater gradient

Only the runs in which a 1:500 gradient and a hydraulic conductivity of 10m/day were prescribed adequately model the lake. This is then runs 1 (no recharge) and 4 (recharge) (see appendices A1 and A2). A hydraulic conductivity of 20m/day produced seepage rates in excess of those calculated using the McFarlane (1984) equation (section 2.4.1) and similarly a gradient of 1:1000 failed to describe the water level records.

The modelled results (see Figure A1.1) show that from day 48 until the end of the modelled period the run using a gradient of 1:500 accurately describes the groundwater levels recorded by the water level probes (see Figures 3.2.1a and 3.2.1b). From day 0 to day 47, water levels in hole 6 downgradient of the lake, are adequately described by the model (although it is recognised any variation over a distance of only 22m is going to be minimal), but the upgradient water levels are underestimated by 0.5m at a distance of 100m from the lake.

What is not accounted for, in selecting one gradient over the entire season, is that the gradient across the aquifer is greater in summer than in winter, more closely approximating 1:200 (from groundwater level records). Indeed in the summer the water level gradient across the modelled aquifer is not sufficently described by just 1 value, but would be better described by a downgradient and upgradient component of flow about the lake. This situation is established when evaporation causes the lake to recede so that the gradient of the flow into the lake increases. Although the lake is shown by the piezometer records to be a continuously flow-through system in the summer, the size of the release zone will be small in comparison to the size of the capture zone so that the net flux into the lake is greater than the flow out. This factor was not accommodated for in the modelled runs and explains the discrepancy between these results and the groundwater records over this period.

By prescribing only one gradient over the study period, the modelled results over-estimate the net flux out of the lake in summer, so that the size of the release zone in the areal plots is exaggerated (see Figure A1.4). The position of the stagnation boundary in the areal plots over this time period is similarly inaccurate and will be further downgradient than illustrated in Figures A1.4-A1.8. Over the February rain event then, the lake will not transform into a fully recharging system as indicated in (Figure A1.6).

- After the first heavy rains the gradient decreases across the aquifer. The need to prescribe different gradients at the nodes on the upgradient and downgradient sides of the lake is removed in winter when the evaporation component becomes insignificant in the water balance.

5.2.2 Modelled results of seepage fluxes

The calculated fluxes from the lake to the aquifer, plot type 3 (see Figure A1.3), are taken at three nodes on the perimeter of the lake. Two of the nodes represent the extremes in the fluxes. These nodes are the nodes on the upgradient and downgradient perimeter of the lake along the line of regional flow, that is the line of symmetry across the grid in the horizontal plane (see Figure A2.4). The node which approximates the average of these values is the point equidistant around the perimeter between the extremes, that is the point on the perimeter of the lake along the line of symmetry in the vertical plane (see Figure A2.4). Summing of the fluxes at each of the 40 nodes around the lake perimeter is equivalent to forty times the value of the flux at that point based on a symmetrical grid, so this value is used to describe the average net flux to the aquifer. Integration of the average flux over the study period provides the groundwater seepage component of the water balance.

Seepage rates over a series of lake recessions calculated by the McFarlane (1984) equation (see section 2.4.1) were compared to average fluxes estimated by the model. The model assumes a surface area of the lakebed of 30,000m^2 whereas McFarlane's (1984) estimate is 24,847 m^2, a difference factor of 1.26. Table 5.2.2 shows that the modelled fluxes are also around a difference factor of 1.26 from those calculated using the smaller surface area, which is evidence of the accuracy to which the model predicts the seepage rates from the lake. Table 5.2.2 also includes the evaporation, expressed as a volume, to indicate the minimal over-estimate of seepage in the modelled results, by not allowing for evaporation effects at the lake. The evaporation estimates use pan values from Perth airport, not corrected by a lake coefficient.

5.2.3 Explanatory notes on flow dynamics

During the winter, as stormwater runoff to the lake continues to recharge the aquifer, the stagnation point (represented in areal section by a saddle point) moves outwards to a maximum range of 300m upgradient of the lake (see Figure A1.17). As the lake release zone increases the groundwater moving with the regional flow is directed around (and under) the recharge region as is indicated in the areal plots. As the lake recedes, the head difference between the lake and the aquifer also decreases so that the fluxes from the lake to the aquifer are lowered. One distinct feature well illustrated in the areal plots is that by the end of winter the fluxes from the lake are reduced as the head distribution between the lake and the surrounding groundwater becomes constant at a level of 5.09m AHD, equivalent to the height of the weir on the ocean outfall (Figure A1.17).

From plot type 2 (Figures A1.2 and A2.2) the same flow dynamics are described as in the areal plots only in vertical section. What is well illustrated is the movement of the stagnation point upgradient of the lake as the lake recedes following a recharge event (the point of minimum head indicating the position of the stagnation point). The streamlines display the gradient that is set up between the lake and the groundwater with the inflection point indicating whether water is entering or leaving the lake.

5.2.4. Potential for the establishment of a fully penetrating recharge system

One small modelling exercise was conducted using FLOWTHRU in order to determine the potential for a fully penetrating recharge system to be established at the lake. Based on a series of hydrograph records taken across the Shenton Park catchment by McFarlane (1984) and on the modelled results from AQUIFEM-N over the period of this study, the size of the release zone of the lake was estimated as being around a maximum length of 300m upgradient from the basin. At the anisotropic ratios of between 50 and 100 reported for a number of aquifers in the Perth region by Townley et al. (1993b) the dimensions of this release zone suggest that it is unlikely that the system ever becomes fully penetrating (ie, recharge of lake water to the base of the superficial aquifer). If the release zone could be shown to approach 400m upgradient then at the lower range of anisotropies there is the possibilty that the system may approach this flow regime. This is unlikely to occur given that the level of recharge to the aquifer is constrained by the presence of the ocean outfall. Almost certainly during the period when flooding was reported around the lake in the 1920's and 30's the recharge system in winter would have approached a fully penetrating system.

5.3 Discussion of modelling results

The areal plots graphically show the transformation from a flow-through regime in summer to a predominantly recharge regime in winter. When no rainfall recharge component is applied to the catchment (run 1), so that stormwater from the lake is the only recharge source to the aquifer, the lake forms a continuous recharging system throughout the winter. The most distinguishing feature when a rainfall recharge component is applied across the catchment is that a transition back to a flow-through regime occurs during the winter period, around day 105 (see Figure A2.3). Associated with this transition is an area around 150m upgradient of the lake where, although a mound is not shown to have developed, there is a significant decrease in the groundwater gradient. If a rainfall recharge coefficient above 18% were applied the flow-through regime would have become established earlier than the model predicts.

In a natural system on the Swan Coastal Plain with rainfall and recharge distributed evenly across the catchment it is presumed (pers. comm. L. Townley) that a flow-through condition would be maintained throughout the winter period. The recharging pattern modelled here may, however, exist in natural systems, in locations where the infiltration rates to the soil are low, ie non wetting sands, rocky hillslopes or where the intensity of rainfall events are such that large volumes of water run off the soil surface and pond in localised depressions, ie. in tropical climates.

Based on the modelling studies and analysis of the groundwater records, two primary flow regimes have been identified at Shenton Park Lake, with two likely further transition patterns. The classification of the flow regimes are as listed in the FLOWTHRU users manual (Townley et al. 1992) with Figures 5.3a to 5.3f representing the Shenton Park study site in vertical cross section aligned in the direction of regional flow. The illustrations are orientated so that the groundwater upgradient of the lake enters the system on the right hand side of the diagrams (ie, the regional groundwater flow enters the system through the right hand boundary of the illustration). The summer flow pattern (see Figures 5.3a and 5.3b) is the FT1, flow-through regime identified as being the natural flow pattern at the lake before urbanisation; note that the estimated anisotropy of the aquifer means that the dividing streamline does not penetrate to the bottom of the superficial aquifer.

The influx of stormwater recharge to the lake in winter produces a dynamic transition of the flow patterns at the lake to a partially penetrating recharge system; R1 (see Figure 5.3d). Between the establishment of the R1 regime from an FT1 condition a short transition period will exist where the lake recharges over a large area of its base with a small area of groundwater seeping into the lake on the upgradient boundary. This coincides with the movement of the stagnation boundary upgradient around the perimeter of the lake, until it finally moves outside the lake perimeter and a partially penetrating recharge system is formed. This transition period is well described by the FT4 regime (see Figure 5.3c).

The final flow type (Figure 5.3e) characterises the localised mounding effect (seen in the groundwater records) created by the seepage of lake water into the unsaturated zone in the banks surrounding the lake. The arrow moving from the mound into the lake characterises back seepage that must be occurring as the lake level drops rapidly relative to the movement of water under bank storage. This flow pattern coincides with lake recession after rainfall and drainage to the lake has ceased. The R7 flow regime (see Figure 5.3f) illustrates the fully penetrating recharge system that has been discounted from occurring based on assumptions on the aquifer anisotropy and the known size of the lake release zone.

DATE	15-17 MAY	17-21 MAY	23-30 MAY	11-19 JUNE
LAKE DROP (m)	0.187	0.168	0.391	0.441
EQUIVALENT VOLUME (m^3)	4646.4	4174.3	9715.2	10957.5
EVAPORATION (m^3)	352.8	193.8	223.6	524.2
AVERAGE SEEPAGE (daily/m^3)	43.4	42.3	59.7	50.6
MODELLED SEEPAGE (daily/m^3)	35	32	45	40
DIFFERENCE FACTOR	1.24	1.32	1.32	1.26

Table 5.2.2 Calculated seepage rates from lake level drop (based on recorded area of lake, McFarlane 1984) and from modelled results.

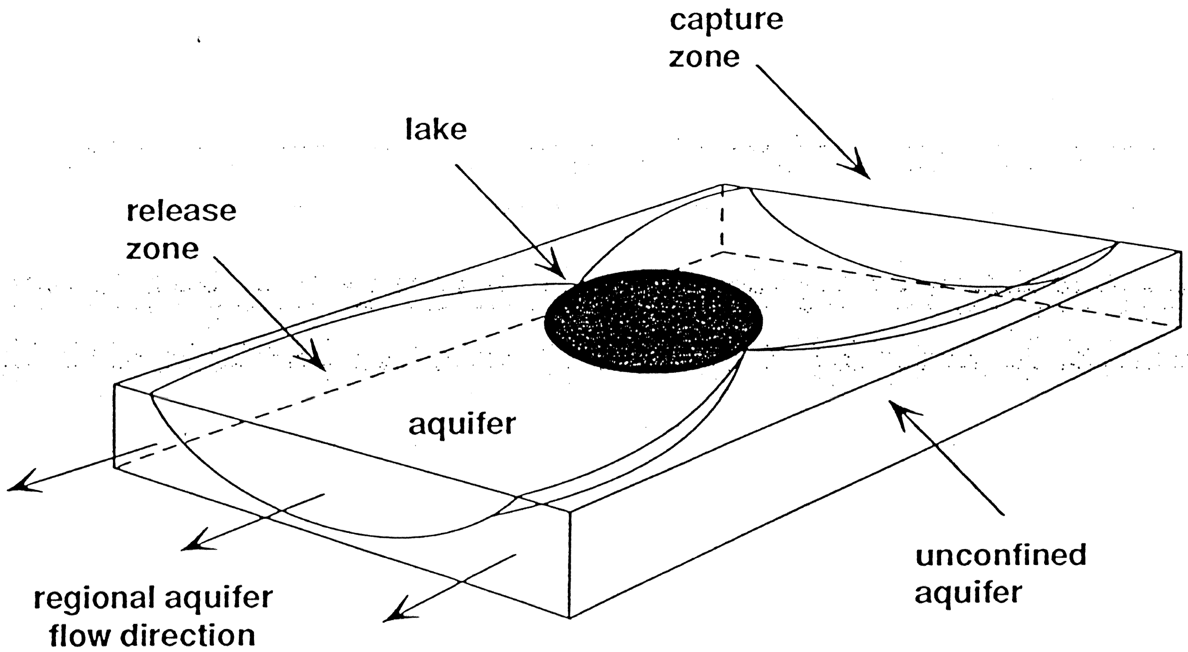

Figure 5.3a The flow-through regime at Shenton Park Lake (summer pattern) in 3d showing the release and capture zones of the lake

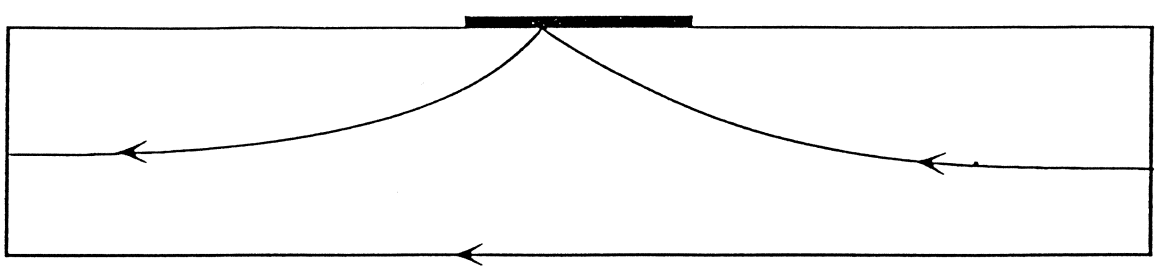

Figure 5.3b The FT1 flow-through regime (summer pattern)

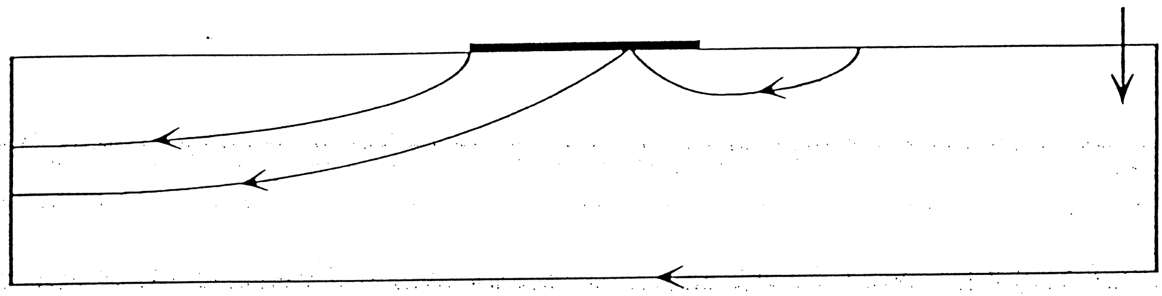

Figure 5.3c FT4 regime (transition pattern)

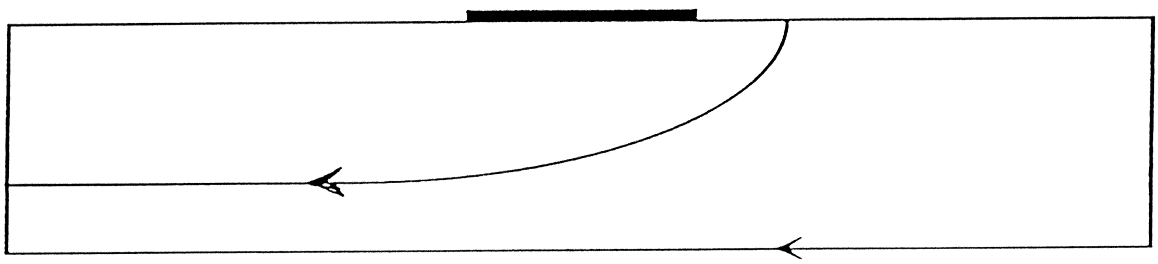

Figure 5.3d R1 (partially penetrating recharge regime)

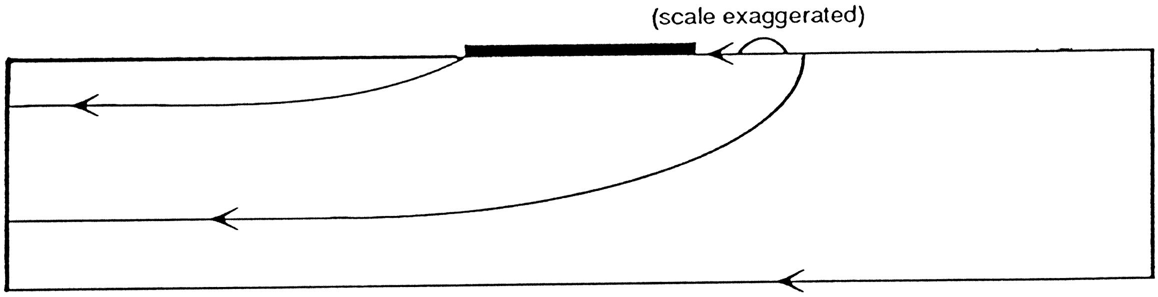

Figure 5.3e Representation of back seepage from localised groundwater mounds

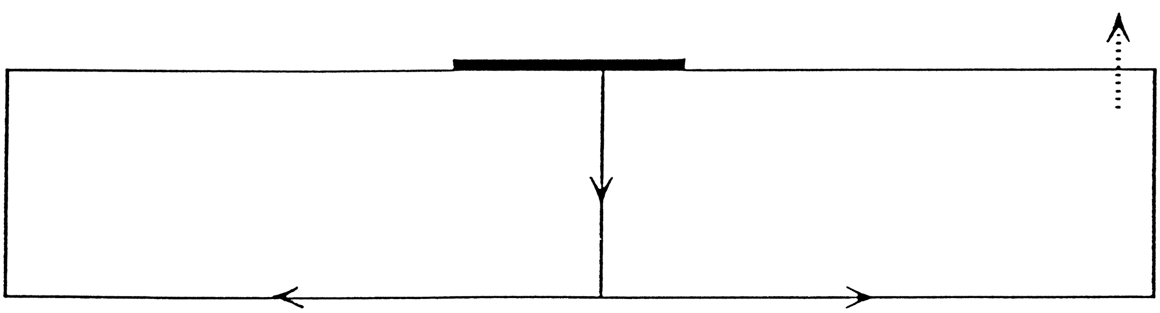

Figure 5.3f R7 (fully penetrating recharge system)

6. INFLUENCE OF STORMWATER ON LAKE AND GROUNDWATER CHEMISTRY

6.1 Sampling method

Sampling of lake water was conducted on three occasions to identify changes in lake chemistry from summer, through the break in season, and into winter. Stormwater samples were collected during the first flush rains and in the months of May and July. The sampling of stormwater in the drains (see Figure 2.2b) was limited to grab samples only, so that no attempt was made to estimate mean flow loadings. As peak loading values commonly occur just prior to, or at times of peak discharge, an attempt was made to sample at these times.

Following collection, the samples were refrigerated until analysed, when each sample was filtered through a 0.45μm filter. The samples were analysed at the CSIRO Division of Water Resources laboratories (see Table 6.1). As the sample for total N was filtered this concentration did not include colloidal material. Due to an oversight one sample was not analysed for total P.

6.2 Chemistry results

6.2.1 Major ion distribution

The Piper trilinear diagrams (Figures 6.2.1a, 6.2.1b and 6.2.1c) display the proportion of major ions (Na, K, Ca, Mg, Cl, SO_4, HCO_3+ CO_3) on a mass basis in the drain samples, in the lake itself and, as a reference, the proportions in rainfall (from Farrington et al. 1993). The underlying trend is that although there is variability in the individual drain samples, the ratio of the major ions in the lake is relatively constant throughout the study period. The drain samples show a distinct first flush effect (27/3/95), followed by a period in which there is variability across samples (9/5/95), and finally a very closely spaced series of samples in the middle of winter, presumably when much of the particulate matter across the catchment has been removed by previous runoff events.

From the cation plot (Figure 6.2.1b) it is noted that in the stormwater the proportion of Ca to Na+K is higher in the early rains of March and May, while the proportion of Mg is relatively fixed. A first flush effect is also seen in the anion plot (Figure 6.2.1a) with a higher proportion of sulphate. The proportion of the carbonate to chloride is also larger in the first two sampling events.

The diamond plot (Figure 6.2.1c) illustrates how the drain samples were closely grouped in the first and final runoff events, but that in the May sample the drains showed variation in their ionic composition reflecting differences across the catchment. It is noted that the ion balance of the lake changes slightly in the month of May where the proportion of carbonate is larger in the total anion count. This shift in the lake chemistry parallels what is understood in relation to the volume of water contributed by the main discharging drains into the lake. The three outlying drain samples in which there was also a larger proportion of carbonate in the anion balance at this time, include drain 7, which contributes 35% of the stormwater from the Shenton Park catchment. The one influencing factor that is not accommodated in this study is the chemistry of the pumped interbasinal water, which is estimated to contribute as much as 50% of the stormwater component of the water balance.

From table 6.2 the general trend over the winter period in the lake, is for the distribution of the major ions to move to those in rainfall as displayed on the Piper diagrams. Generally the distribution of major ions in the lake is not greatly different to that reported by McFarlane (1984) in the groundwater in the immediate vicinity of the basin. The exception is the sulphate/chloride ratio which is eight times greater in the groundwater. The concentration of chloride in the lake in summer is higher than the surrounding groundwater as would be expected in a system where little water is entering the waterbody and evaporation rates are high. The greatest impact of the influx of stormwater on the surrounding groundwater, is the decrease in the concentration of total dissolved solids.

6.2.2 Changes in lake nutrient balance with the influx of stormwater recharge

The concentration of total nitrogen entering the lake in stormwater during the early season rains was quite high (see Table 6.1). The highest recorded value of 5.45 mg/L was equivalent to that reported by Tan (1991), but also comparable to the concentrations in the lake water. As in Tan (1991), the total nitrogen levels decreased in the runoff in winter to an average of about 0.5mg/L. The high levels of total N in the first flush rains were due mainly to a large organic N component. A similar first flush effect was seen in the levels of orthophosphate in the drainage water but the concentrations were much higher than in the lake over summer. The levels of orthophosphate decline over the winter but it was suggested by Tan (1991) that the proportion of soluble P as a fraction of total P also decreases over this period, so that absolute concentrations of phosphorus input to the lake cannot be precisely estimated from the results of this study.

The concentration of orthophosphate in the lake is approximately 50% lower between summer and winter levels. The concentration of total nitrogen is more dramatically reduced from 6mg/L to 0.5mg/L. The largest component of nitrogen in summer is in the form of ammonium with the concentration of nitrate/nitrite surprisingly low. Indeed in all of the samples including those of stormwater runoff, this concentration was probably lower than expected.

6.3 Modifications to lake and groundwater chemistry resultant from stormwater recharge

6.3.1 Major ion distribution in stormwater

The first flush runoff events were typically composed of a larger proportion of major ion species only found in low concentrations in rainfall. The effect of the removal of the dust, fines and sediments from the shedding areas during the early rains results in the stormwater chemistry more closely resembling that of rainfall by mid winter. The bicarbonate ion concentration is the most variable between runoff and rainfall. The higher proportion of bicarbonate in stormwater indicates that it is removed from road surfaces during transportation. The low concentrations of total dissolved solids are similar to those reported by McFarlane (1984) and Appleyard (1994).

6.3.2 Nutrient loadings to lake and groundwater

The concentration of nutrients in the stormwater runoff is perhaps surprisingly low given the estimates of annual average application rates of nitrogen and phosphorus in an urban catchment of 80 kg ha^{-1} and 40 kg ha^{-1} respectively (Gerritse et al. 1990). The measured concentrations are, however, in the same range as documented in McFarlane (1984) and Tan (1991). The low concentrations are perhaps partly due to the timing of the fertiliser applications in the catchment. Much of the fertilisers applied to gardens is very quickly leached into the soil by frequent irrigating during the dry season, between October and April. The increased soil compaction of permeable surfaces in urban areas (Carlsson & Falk, 1977) and the sandy nature of the soils in Perth mean that erosion rates from urban catchments such as Shenton Park will be low. The level of particulates in the runoff will consequently be low so that the proportion of nutrients tranported in stormwater attached to particulate matter is also reduced.

The high concentrations of orthophosphate in stormwater during autumn and early winter, above the measured concentrations in the lake, suggest that a significant component of the phosphorus load is being assimilated in the lake sediments. Unlike the majority of lakes on the Swan Coastal Plain which have low flushing rates and high residence times (Davis et al. 1993), Shenton Park Lake is dynamic so there is potential for phosphorus input via stormwater to move rapidly to the groundwater. This is not likely to occur, however, due to the strong binding potential of iron oxides in the groundwater and sediments and the larger proportion of clay fines in the Spearwood soils group. This statement is supported by the findings of Appleyard (1994). The assimilation of phosphorus to the sediments does, however, create a large viable pool that can be released during the summer when anaerobic conditions in the sediments occur.

The large influx of stormwater, low in nitrogen, is seen to transform the lake from being hyper-eutrophic in nitrogen over the summer (from Vollenweider 1968) to an oligotrophic condition in the winter. The source of nitrogen to the lake is recognised as not being from stormwater through runoff from the surrounding catchment. During the summer, under what is presumed to be oxygen rich conditions in the lake, the proportion of nitrogen in the form of nitrate is surprisingly low. The largest component of total nitrogen is in the form of ammonium. There are two potential sources which could contribute large concentrations of nitrogen to the lake in the form of ammonium. Firstly, in the small volume of water at the end of summer, the local bird population, estimated to number around 150, could conceivably input significant quantities of nitrogen in the form of ammonium. This problem could well be exacerbated by the "generosity" of local residents feeding the bird life. The second potential source of ammonia is via groundwater from the former landfill site on the north east boundary of the lake where McFarlane (1984) measured concentrations of nitrate in the groundwater at 5mg/L (see Figure A3.1). When a flow-through system is re-established at the lake over the summer, the groundwater capture zone of the lake passes through this former landfill site. The groundwater has been demonstrated as being of low redox potential (FeS_2 was precipitated on the water level probes located outside the lake, indicating anaerobic conditions) such that reducing conditions and denitrification will be occurring; therefore if nitrogen is leached from this site it will be in a reduced form, ie NH_4^+.

Determining which nutrient is limiting the system in terms of algal growth is made complex by the fact a sample was not analysed for total phosphorus. Based on the studies by Davis et al. (1993) and Davis & Rolls (1987) organic phosphorus generally comprises 50-90% of the total P concentration of lakes studied such that the orthophosphate partition is between 10-50%. The value of 50% recorded for orthophosphate levels in one lake was exaggerated, however, due to the application of an organophosphate based insecticide to control midges, so that a more accurate estimate is closer to 10%. Using this estimate, the total P concentration of Shenton Park Lake based on the sample taken on 21/3/95 would approach 0.3mg/L. This represents a N:P ratio of 20:1 indicating P limitation. Based on this concentration the lake water is meso-eutrophic in P. In determining limitation on the basis of available nutrients as discussed in Rast et al. (1983), the low levels of nitrate may suggest that N was in fact limiting during summer, but this discounts the large nitrogen pool in the form of ammonium from which nitrate can be cycled. During winter the large drop in nitrogen in the lake means that on the basis of an orthophosphate fraction of 10% total P, the N:P ratio would approach 4:1. Not enough is known about the sinks of the two nutrients to state whether the system could transform to being nitrogen limited at this low ratio.

The concentrations of both calcium and carbonate are low in the lake waters (see Table 6.1). At these concentrations calcite would not be precipitated from solution, therefore the adsorption of phosphorus is due principally to binding on iron oxides in the sediments. The lake is low in both calcium and carbonate as the sands overlying the Tamala limestone were eolian deposits rather than marine deposits so the $CaCO_3$ concentration in the sands on which the lake sits is not alkaline.

6.3.3 Influence of stormwater recharge on groundwater chemistry

The low sulphate/chloride ratio in the lake over the entire study period relative to the groundwater explains what is seen in the distribution of sulphate across the catchment, where a plume of low sulphate concentration water indicated the extent of the release zone of the lake (McFarlane 1984) (see Figure A3.2). Although it appears that the oxygenated waters of the lake induce oxidation of sulfides to form sulphate in the groundwater, the distribution from McFarlane shows that these ions are then transported from the site in solution.

Sample Description		EC	pH	HCO_3	PO_4-P	NO_3-N	SO_4-S	CL^-	NH_4-N	K	Na	Fe	Mg	Ca	Tot. N	Org. N
		mS/cm	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L
SPL 27.03.95	D1	0.308	7.05	55.4	0.626	0.019	6.54	39.9	0.080	14.6	26.2	0.693	3.89	14.9	5.45	5.37
	D4	0.208	7.02	34.6	0.125	0.025	6.15	23.1	0.384	7.27	15.8	0.533	2.61	11.8	2.67	2.29
	D5	0.193	7.04	33.8	0.087	0.016	5.59	21.1	1.110	4.54	14.0	0.418	1.86	12.5	3.10	1.99
	D7	0.168	7.01	37.9	0.097	0.014	3.62	17.5	0.165	6.60	11.9	0.350	1.87	10.8	1.57	1.41
	D8	0.120	6.73	17.7	0.076	1.200	3.58	11.7	0.011	3.15	7.12	0.213	1.44	9.36	1.31	1.30
	D9	0.448	6.33	87.7	0.192	0.013	12.7	48.9	0.021	20.9	33.0	0.646	4.40	37.4	4.96	4.94
SPL 09.05.95	D1	0.084	6.57	12.4	0.185	0.217	1.36	11.1	0.221	5.51	5.14	0.076	0.724	3.42	0.62	0.399
	D4	0.071	6.55	7.86	0.138	0.248	1.53	9.14	0.294	4.19	4.69	0.075	0.672	3.24	1.06	0.766
	D5	0.077	6.51	7.91	0.148	0.152	1.48	10.5	0.120	4.53	4.32	0.121	0.863	3.47	0.59	0.470
	D6	0.068	6.84	14.9	0.133	0.211	0.957	7.28	0.209	2.04	4.43	0.075	0.638	4.29	0.43	0.221
	D7	0.071	7.01	15.9	0.089	0.119	0.878	7.27	0.192	3.30	4.10	0.074	0.618	4.29	0.36	0.168
	D8	0.070	6.60	8.66	0.088	0.438	1.30	8.69	0.200	3.16	4.64	0.097	0.611	3.87	0.11	-
	D9	0.102	6.89	22.8	0.122	0.062	1.66	11.0	0.417	4.99	5.19	0.131	1.26	6.20	1.20	0.783
SPL 05.07.95	D1	0.111	6.70	6.88	0.062	0.100	1.40	22.0	0.109	2.26	10.9	0.089	1.23	3.71	0.20	0.091
	D3	0.116	6.78	12.3	0.038	<0.005	1.26	21.5	0.012	1.62	10.9	0.090	1.45	4.48	0.37	0.358
	D4	0.103	6.47	7.78	0.046	0.093	1.45	18.6	0.238	1.79	9.33	0.112	1.12	3.87	0.67	0.432
	D5	0.101	6.71	9.20	0.025	0.167	1.79	16.5	0.536	1.61	8.60	0.113	1.07	4.48	0.96	0.324
	D6	0.113	6.89	12.9	0.019	0.046	1.22	20.8	<0.005	1.23	10.9	0.050	1.20	5.18	0.23	0.030
	D7	0.221	7.16	27.3	<0.005	0.099	3.17	39.7	0.016	2.50	24.3	0.245	2.22	8.78	0.71	0.694
	D8	0.187	7.19	23.3	0.010	0.154	2.94	32.6	0.279	2.08	18.7	0.225	1.91	7.83	0.83	0.551
SPL LAKE 21.03.95		1.36	8.06	182	0.027	0.050	13.4	281	3.25	16.2	174	0.118	12.8	52.6	6.18	2.93
26.05.95		0.361	7.62	51.6	0.020	0.046	5.32	64.3	0.135	3.53	40.2	0.162	3.82	15.8	0.43	0.295
28.06.95		0.365	7.56	42.5	0.012	0.120	4.63	69.2	0.125	3.21	41.8	0.358	3.82	13.5	0.46	0.335

Table 6.1 Chemistry of lake and stormwater, drain samples labelled D1-D9

Lake water samples (last three rows)

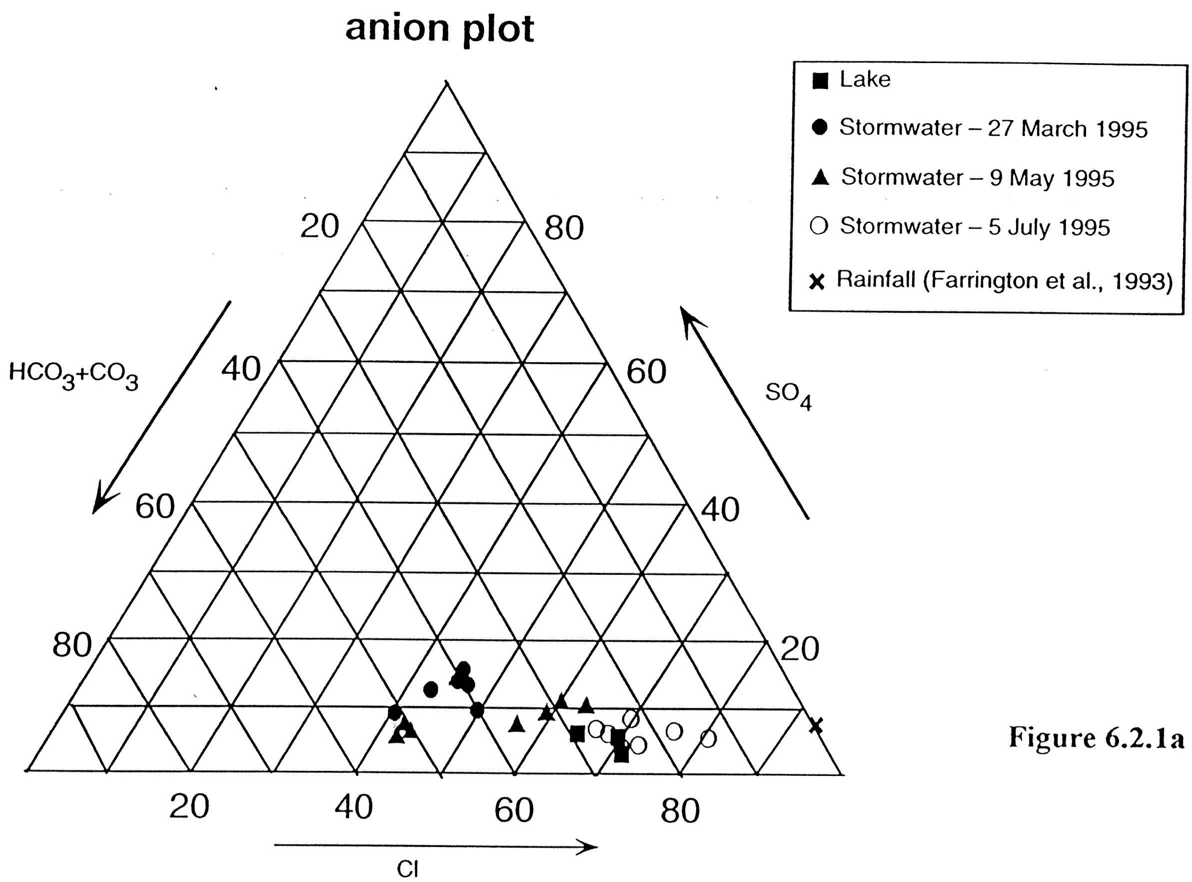

Figure 6.2.1a

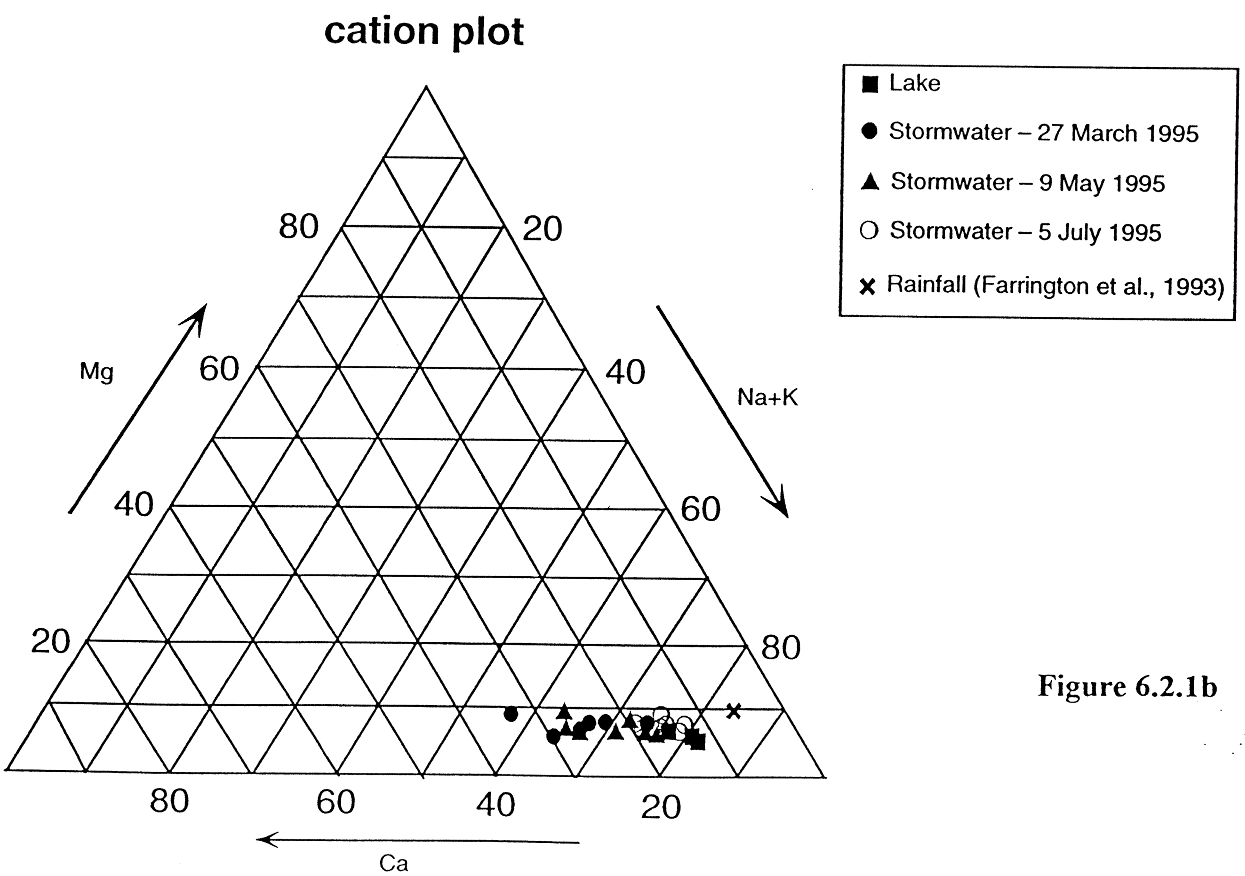

Figure 6.2.1b

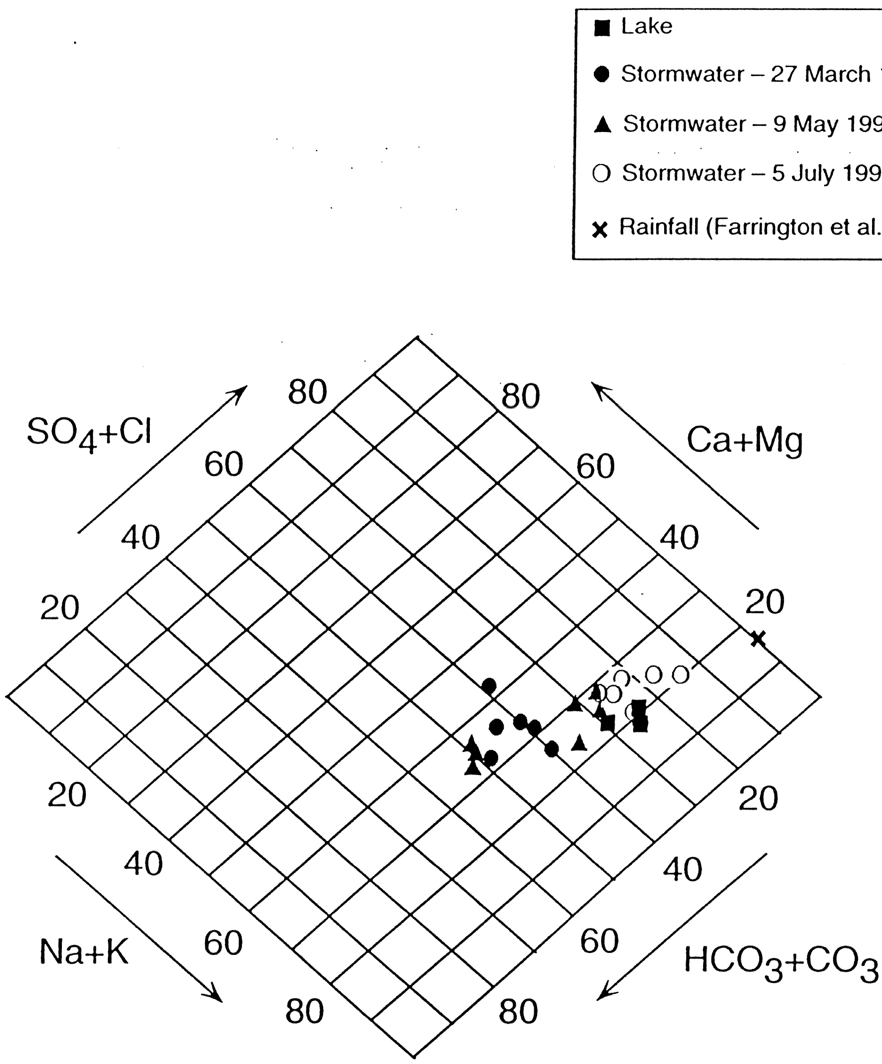

Figure 6.2.1c Diamond plot

Parameter (Shenton Park catchment)	Groundwater, Gerritse et al. 1990, adapted from Mcfarlane 1984)	Lake, summer (27/3/95)	Lake, winter (28/6/95)	Rainfall (Farrington et al. 1993, Floreat - Perth)
Na/Cl (g/g^{-1})	0.58	0.62	0.60	0.56
K/Cl (g/g^{-1})	0.051	0.058	0.046	0.045
Na/K (g/g^{-1})	11.3	10.7	13.0	12.5
Ca/Cl (g/g^{-1})	0.32	0.187	0.195	0.09
Mg/Cl (g/g^{-1})	0.13	0.05	0.06	0.07
Ca/Mg (g/g^{-1})	2.5	4.1	3.53	1.35
SO_4-S/Cl (g/g^{-1})	0.39	0.05	0.07	0.08
Cl (mg/L^{-1})	150	281	69.2	9.5
HCO_3-C (mg/L^{-1})	30	35.8	8.3	trace
TDS (mg/L^{-1})	480	576.8	145.4	17.6
pH	6.8	8.06	7.56	5.17

Table 6.2 Major ion composition of lake and groundwater

7. GENERAL DISCUSSION AND MANAGEMENT IMPLICATIONS

The dynamics of discharging stormwater runoff into Shenton Park Lake have been modelled to determine the extent of its influence on the lake hydrology and the regional groundwater. The dimensions of the lake release zone are known to be dependent upon a number of aquifer properties, including the regional gradient and the aquifer transmissivity, as well as the lake geometry (area) and the volume of recharge (Townley and Davidson 1988). The stormwater recharged to the groundwater is therefore not confined to the upper regions of the aquifer by 'density' effects as reported in Appleyard (1994) but is constrained by the same aquifer properites listed above. The influence of density affects are considered to be insignificant over the range of TDS concentrations reported in this study. The principle modification to catchment hydrology, with the channelling of large volumes of runoff to compensating basins and wetlands, is to change the system from a natural flow-through regime, where recharge is spatially uniform across the land surface, to a situation where one large point source of recharge dominates the flow dynamics in the lake/aquifer system.

With over 200,000m^3 of stormwater recharged to the aquifer over this study period at Shenton Park Lake, the potential for contamination of groundwater at these sites is high. From the findings of Appleyard (1994) and McFarlane (1984) on pollutant levels in the groundwater, stormwater recharge has no detrimental impact on groundwater quality in urban catchments, due principally to the low level of contaminants in the stormwater and the filtering effect of the wetlands and compensating basins in removing nutrients and heavy metals. Insecticides, pesticides and heavy metals were not studied in the chemistry component of this investigation but only heavy metals appear to be in significant levels in urban runoff (Appleyard 1994), and pose only a threat to wetland biota from assimilation in the lake sediments, not to groundwater supplies. The potential for the spillage of hazardous wastes into the drainage system remains, but this is recognised as being of low risk except under conditions of intentional illegal dumping. Unfortunately, due to the limited recognition by the general public of the fate of stormwater (O'Loughlin et al. 1992), inadvertent contamination of wetlands will occur with the washing and disposal of pollutants into the stormwater system by local residents.

Given the high dissolved oxygen concentrations of the recharged stormwater, it is reasonable to conclude that over time the redox potentials of water within the area of the lake's release zone may increase, resulting in conditions more favourable for the leaching of nitrates as reported in Appleyard (1995). From the results of Tan (1991), McFarlane (1984) and those reported in this study, increases in nitrate leaching to the groundwater as a result of recharged stormwater is not likely, given the low concentrations of total nitrogen in the runoff waters. It is suggested, therefore, that concentrations of nitrate in the groundwater under urban catchments will be maintained if nitrate concentrations in runoff water remain at present levels, and that the rates of denitrification of fertiliser leached directly through the soil are as extensive as those reported in Gerritse et al. (1990). One possible effect of high dissolved oxygen concentrations in the recharge waters is the oxidation of sulfides in the sands, probably largely from pyrite (FeS_2) and their subsequent leaching. The release zone of Shenton Park Lake can be distinguished by the distribution of the sulphate ion in the groundwater (Figure A3.2). Within the lake release zone sulphate concentrations are lower than in the surrounding groundwater, indicating that the leaching of sulphate by the recharge waters is on a much larger scale than oxidation effects on soil held sulfides. Although no significant changes were seen in the proportion of major ions in the lake before and after stormwater recharge, the concentration of total dissolved solids was seen to decrease dramatically. Thus TDS (as in Appleyard 1994) and sulphate are identified as tracers for estimating the release zones of lakes in which the stormwater component of the lake water balance is significant.

Although groundwater quality is not adversely affected by the recharge of stormwater, the function of the wetlands is to act as a large settling pond in which nutrients and heavy metals are assimilated to the sediments. In terms of nutrient inputs to Shenton Park Lake, only phosphorus in the stormwater is at high concentrations relative to those in the lake. This then represents the lake's main source of phosphorus, presuming that the phosphorus input from the local bird population is largely recycled. The total P concentrations were not measured but it seems likely that, as in Tan (1991), the proportion of soluble to organic P is larger in the early rains. The fact that the lake's P concentrations did not rise suggests that either the large component of soluble phosphorus is rapidly assimilated to the sediments, or perhaps the drain samples collected by point sampling are not indicative of the mean flow weighted loadings.

The estimates of the N:P ratio show that over the winter the ratio may be lowered to levels where nitrogen limitation is approached. In a situation where the N:P ratio approaches nitrogen limitation in the lake at the start of summer, increases in the temperature and pH of the lake, would be predicted to result in cyanobacterial growth (Balla et al. 1994). There is no evidence of this at Shenton Park Lake, however, so that it is presumed that nitrogen concentrations increase rapidly over the early part of the dry season, but that the lake aerators may also prevent cyanobacterial outbreaks. If the lake continues to receive high concentrations of phosphorus at the start of winter then rates of assimilation may drop as the sediments become saturated, leading to remobilisation of this large pool over summer, and promotion of blue/green algal blooms. Saturation of the sediments could also conceivably lead to increased leaching of phosphorus to the groundwater, given the high flushing rates of the lake in winter.

The former landfill site has not been actively studied as a potential pollution source but, based on the nitrate plume identified in Figure A3.1, it is a probable source of nitrogen additions to the lake. The modelling of the flow dynamics has shown that the potential for leaching of nutrients from this site and into the lake is probably largely restricted to the drier months of the year, because over the winter period the net groundwater flux is away from the lake. At this time oxygenated stormwater may cause nitrate leaching upgradient of the polluted area. During winter some leaching may occur to the lake during the short term transitions back to a flow-through system. The presence of the localised groundwater mounds may also result in back seepage to the lake of contaminated waters, although it is likely that these features are only very minor so that their influence is not significant. In early summer the leaching of nitrogen from the former landfill site may have a greater proportion in the form of nitrate as the redox potential of the water could be influenced by the mixing with stormwater recharge. By the end of summer it is likely that the groundwater moving into the lake is of low redox potential so that the nitrogen is in the form of ammonium.

McFarlane (1984) concluded that the major discharge source in the catchment was the ocean outfall drain, being larger than the transpiration losses due to phreatophytes and that due to extraction by private and public bores. It was also suggested that the amount of recharge to the groundwater via the lake was likely to be restricted by the presence of this drain, a finding that is supported by the results of this study. The rate of lake recession in summer is therefore influenced by this reduction in stormwater storage to the groundwater during the winter. This impact may, however, be minimal as the depression effect in the regional groundwater is very localised (groundwater levels adjacent to the lake limited to 5.1m AHD) and instead the recharge to the lake in summer is determined by groundwater gradients on a much wider scale than within this single catchment. The causes of low lake levels are likely to be increased extraction rates by private and public bores and evapotranspiration from the phreatophyte fringe. In 1994-1995 this was exacerbated by very low rainfall over the summer. The lake appears to have reverted back to its original status of a sumpland (Spillman 1985) with seasonal drying; only when rainfall occurs over summer will lake levels be maintained. It is concluded that low lake levels in summer are now a natural occurrence at this site, the unfortunate consequences being low water quality conditions, creating health related problems for the wetland fauna, and a wetland that is is aesthetically displeasing.

If the potential for groundwater storage (via stormwater recharge) could be maximised within the Shenton Park and surrounding catchments, then the rate of lake level decline could be decreased by higher rates of groundwater seepage into the basin over summer. The optimal solution would be to minimise large point sources of recharge and instead establish a network of smaller compensating basins throughout the catchment as is the practice in many newer suburbs. By distributing recharge in this way the system more closely approximates that found in an uncleared (vegetated) catchment. It is recognised that in an established suburb such as Shenton Park this may not be feasible due to the lack of available sites for locating basins. Greater recharge to the groundwater could also be achieved by extending the perimeter of the lake and also deepening it to increase the volume from 50,000m^3 to a size that could better accommodate the volume of winter runoff. By increasing the storage capacity of the lake the water volume stored before the start of summer would, of course, be greater, but the same problems would remain, only on a slightly larger scale.

If the goal is to try to maintain higher lake levels in the summer, water could be pumped from aquifers deeper than the superficial aquifer that supplies the lake, as is the practice at Lake Monger. This can be seen as a waste of a potentially valuable resource and installation and running costs of such an operation may be prohibitive. By maintaining high water levels, the concentration of nutrients by evaporation would be decreased and the lake would undoubtedly remain more aesthetically pleasing as well being more conducive to local wildlife. Lining the lake bed with a less permeable clay base would mean that runoff from summer rainfall events would be retained for a longer period in the basin. If the lake levels were artificially maintained by groundwater pumping then lining of the lake would also increase the efficiency of this practice. In years where very little rainfall occured over the summer the reduction in the net flux of seepage from the aquifer and into the lake, due to the lower permeability of the bottom sediments, may result in slightly increased rates of lake recession over the same ranges of evaporation. Lining of the lake is not a viable solution to the lake's management problems as high infiltration rates in winter are important in order to maximise recharge to the aquifer so that if the lake were to be lined an even higher percentage of runoff water would be lost to the ocean outfall and increased flooding would occur.

If low lake levels are an inevitable consequence of the long dry season then there are a number of steps that could be taken to improve water quality and the aesthetic appeal of the lake. Excessive nutrients levels are compounded in that no fringing vegetation except introduced species of willows have the potential to compete for available nutrients in the water column and sediments. A remodelling of the lake should include removal of the perimeter stone wall and an attempt should be made to grade the banks so that they can be planted with native wetland species, including rushes. In the grading process it may be possible to incorporate some of the bottom sediments of the lake, thus deepening the lake and removing nutrients that could potentially be remobilised. This will only be possible if the sediments are not toxic to the growth the fringing vegetation. If this is not the case, the sediments may have to be removed from the site. Sloping and vegetated banks will provide less of a stark contrast in summer when the lake levels are low. Fringing vegetation and replanting of reeds will provide shelter and a food resource for the native birdlife and will replace the bread resource that contributes to the lake's nutrient load. If this work was to be conducted, the problems posed by the potential release of pollutants from the former landfill site would need to be further investigated to avoid the risk of contamination of the site subsequent to the earthworks. The feeding of birds at the lake should be discouraged by informing the public by means of appropriate signs. This will help reduce the levels of nitrogen and phosphorus input to the lake, especially when water levels are low. The source of foodstuffs for the native birdlife is recognised as being limited by the absence of a well developed ecosystem that cannot be accommodated without sufficient fringing vegetation. Finally, the water quality of the lake is such that in winter continual aeration of the lake is an expensive and unnecessary exercise, other than to serve as an aesthetic attraction.

SUMMARY OF MANAGEMENT IMPLICATIONS IDENTIFIED IN THIS STUDY

- Low lake levels over summer are now a natural occurrence at this wetland.
- Resultant low water quality is due to concentration of nutrients via evaporation.
- Stormwater runoff is identified as the major input of phosphorus to the lake.
- Two potential sources of nitrogen addition over the summer have been identified - (1) excretia from the local birdlife, exacerbated by the local populace feeding the birds. (2) leaching of nitrogen from a former landfill site on the northern boundary of the lake.
- Concentrations of nutrients in the sediments and water column can be reduced by the planting of fringing vegetation and native rushes at the lake.
- A remodelling of the wetland should include removal of the stone perimeter wall and grading of the banks, possibly by incorporating sediments from the bottom of the lake.
- Sloping vegetated banks will provide a more aesthetically pleasing landscape and increase the recreational value of the lake reserve over the summer months. Fringing vegetation will also provide for the establishment of a more diverse ecosystem and increasing shelter and food resources for the local fauna.
- Lining of the lake with an impermeable clay base would not reduce rates of lake seepage in summer and would create a flooding problem in winter.
- Feeding of the birdlife should be discouraged as, in the reduced volume of water in summer, nutrient input from bread substantially reduces the water quality.
- The higher water quality of the lake in winter renders aeration of the lake unnecessary.

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APPENDIX 1 MODELLING RESULTS FROM AQUIFEM-N (RUN 1)

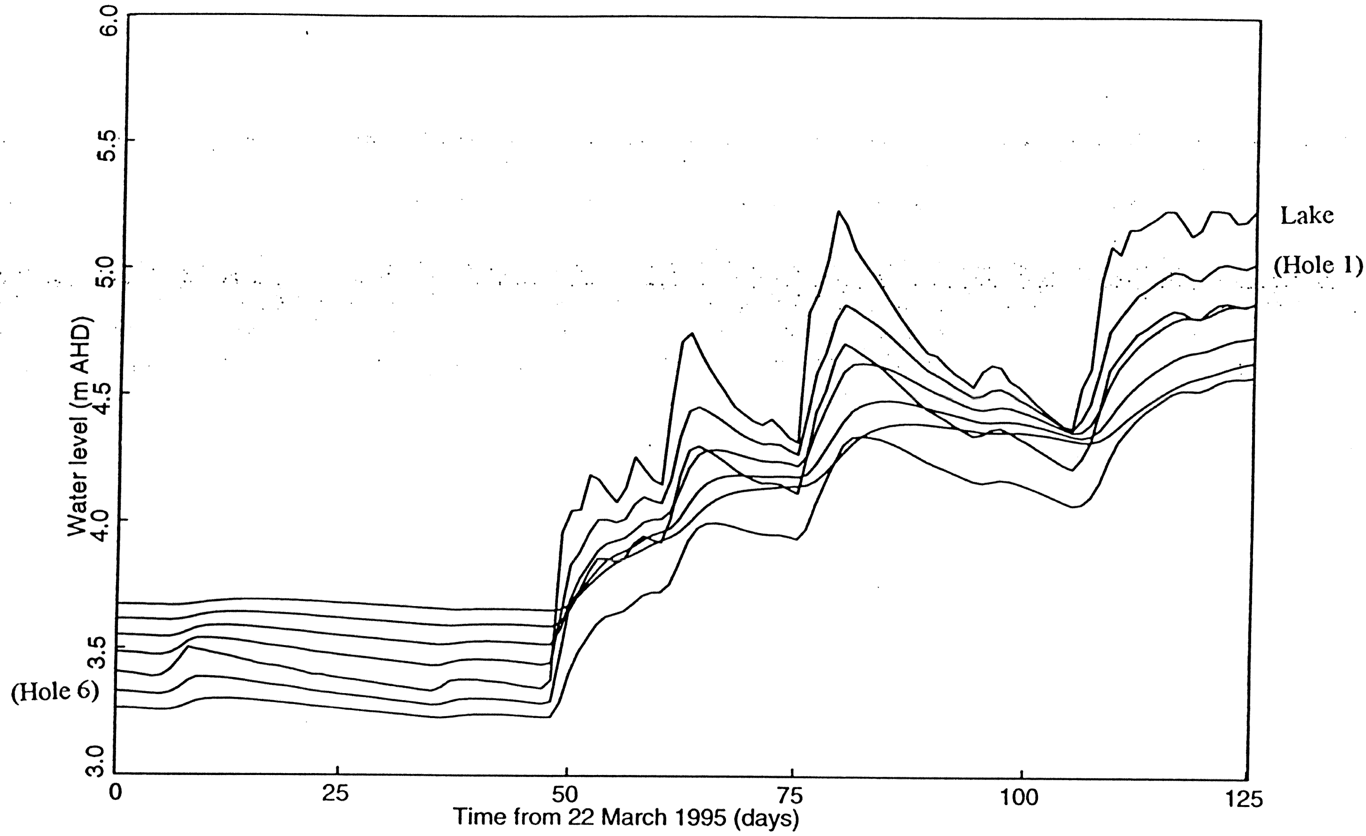


Figure A1.1, Variation in water level over time.

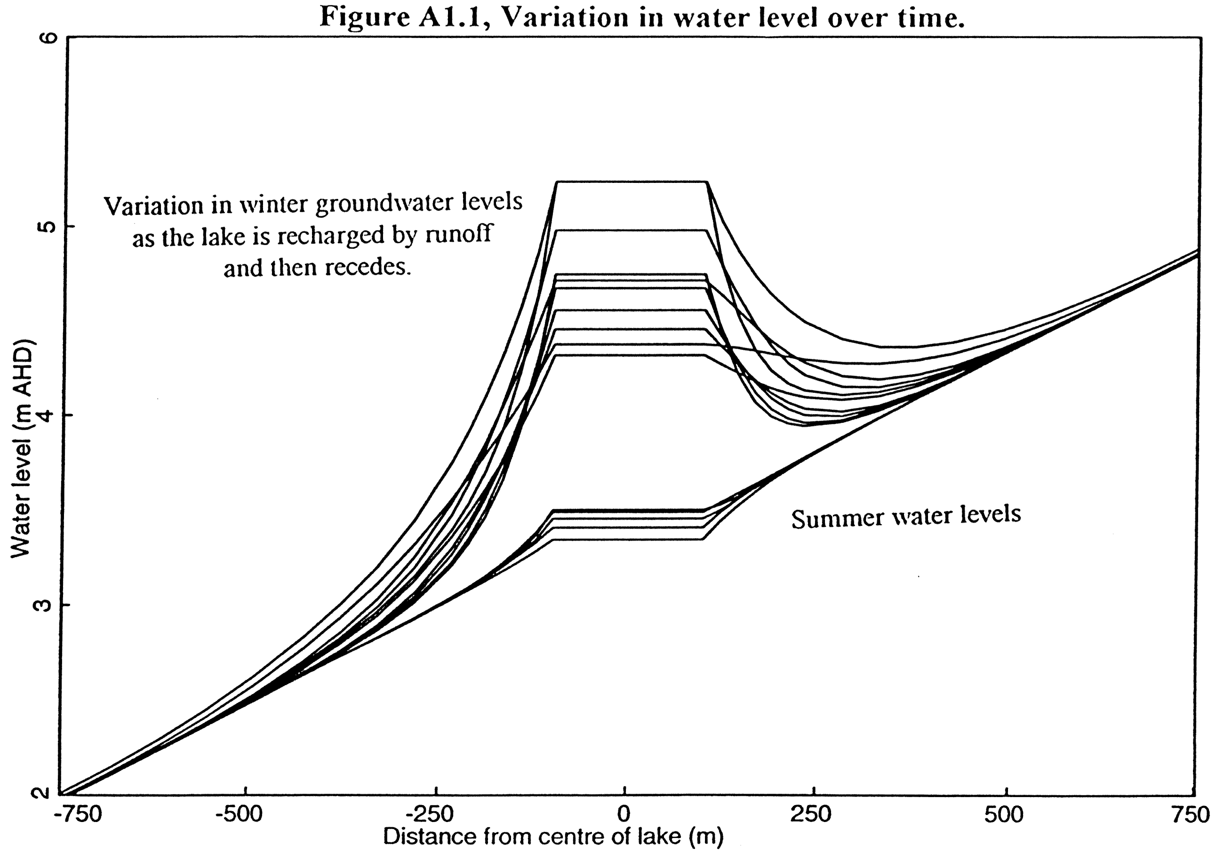


Figure A1.2., Water levels as a function of distance from the lake

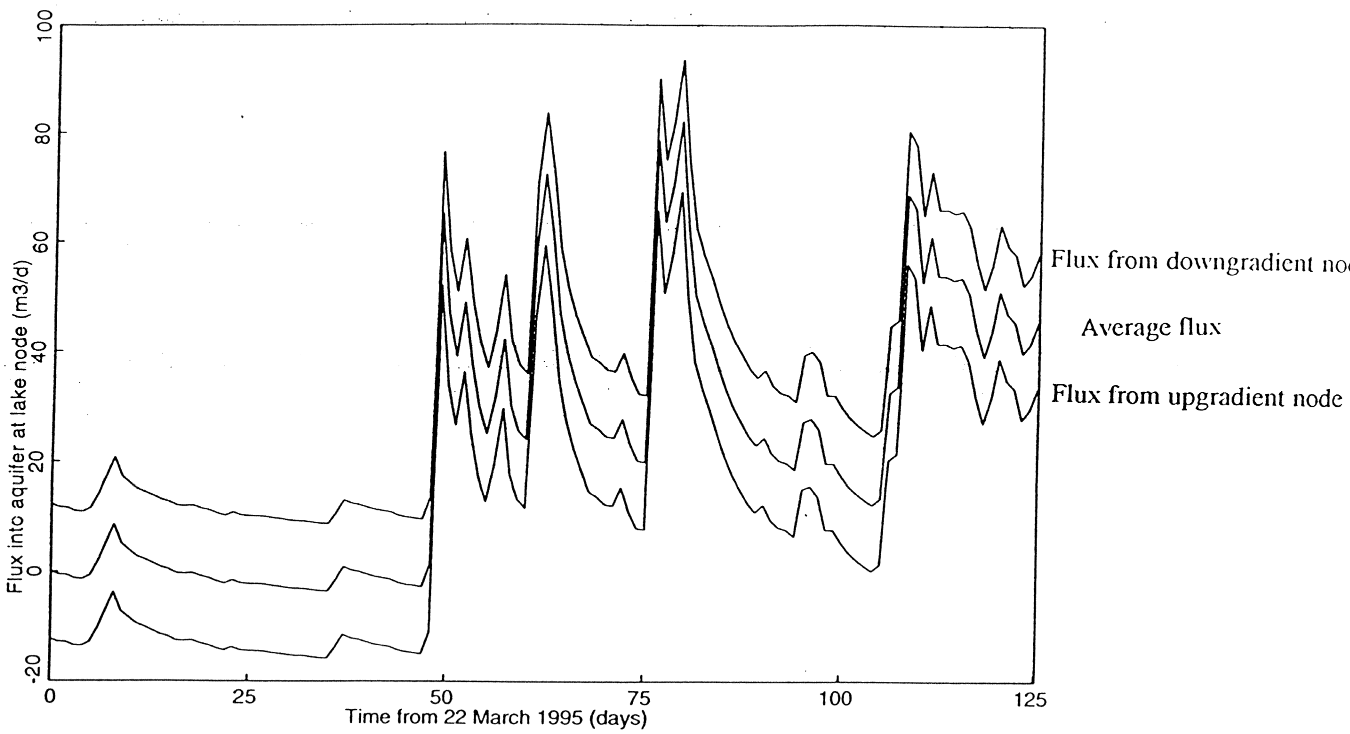


Figure A1.3., Seepage flux to the aquifer

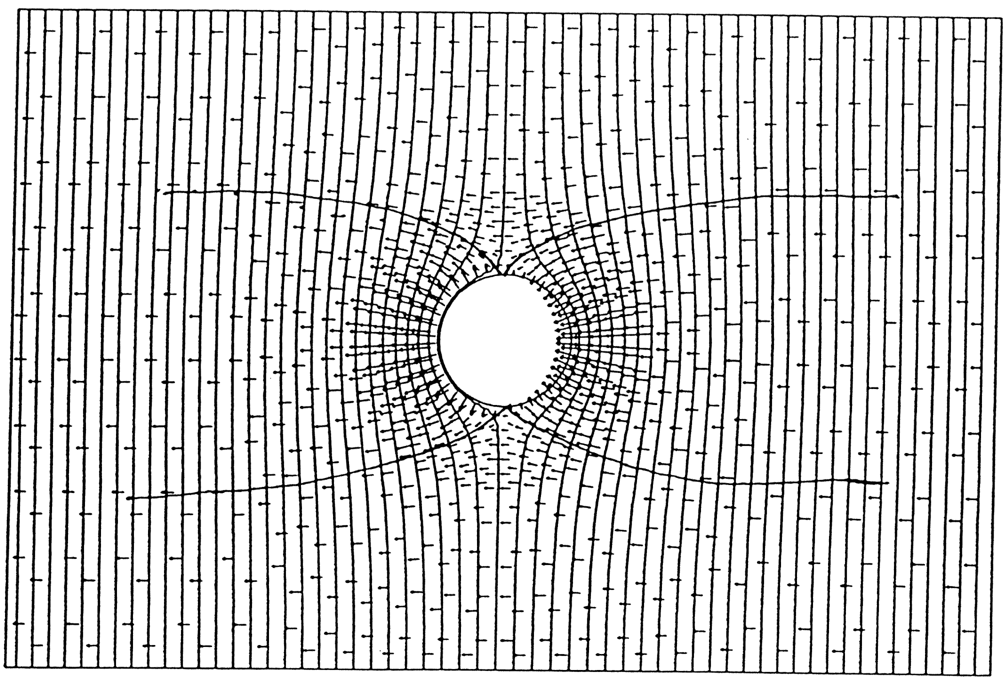


Figure A1.4., Day 0. Areal plot

Typical summer flow pattern with stagnation boundary located near the centre of the lake. The lake capture zone is on the right hand side of the diagram, the lake release zone downgradient of the lake exiting the system on the left.

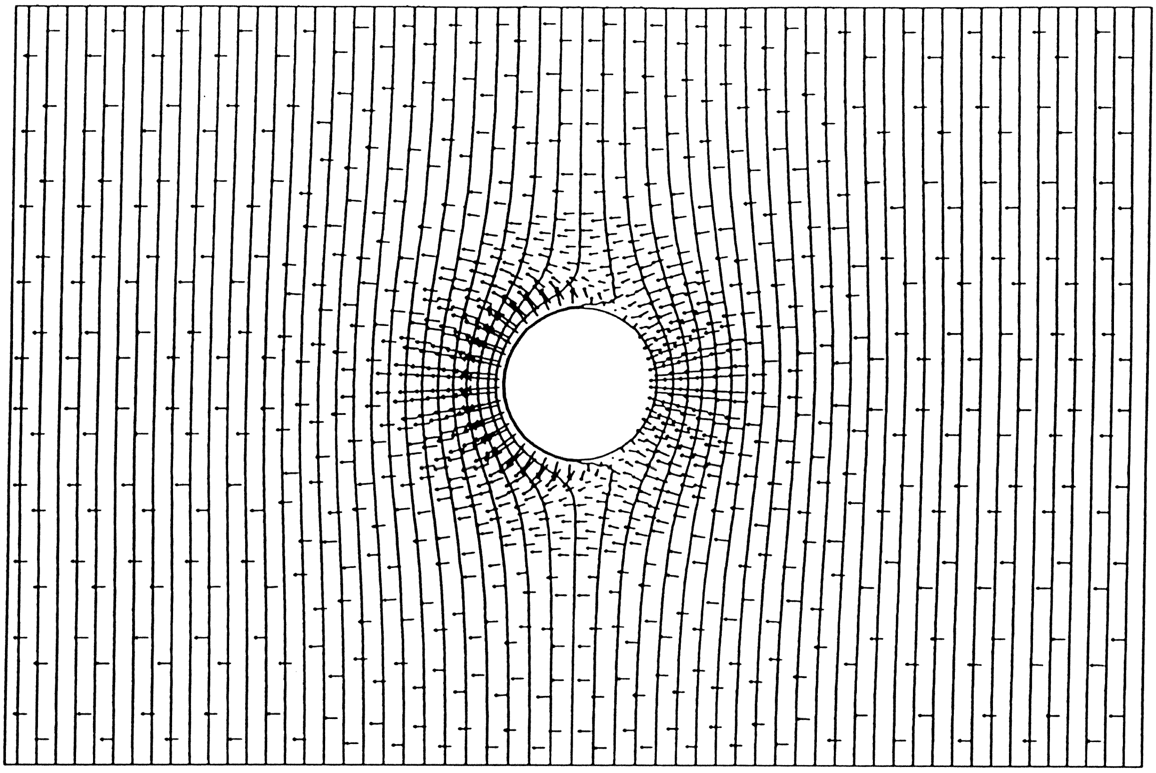


Figure A1.5., Day 7.

Stormwater has recharged the lake resulting in an increased flux to the aquifer and movement of stagnation boundary upgradient around lake perimeter.

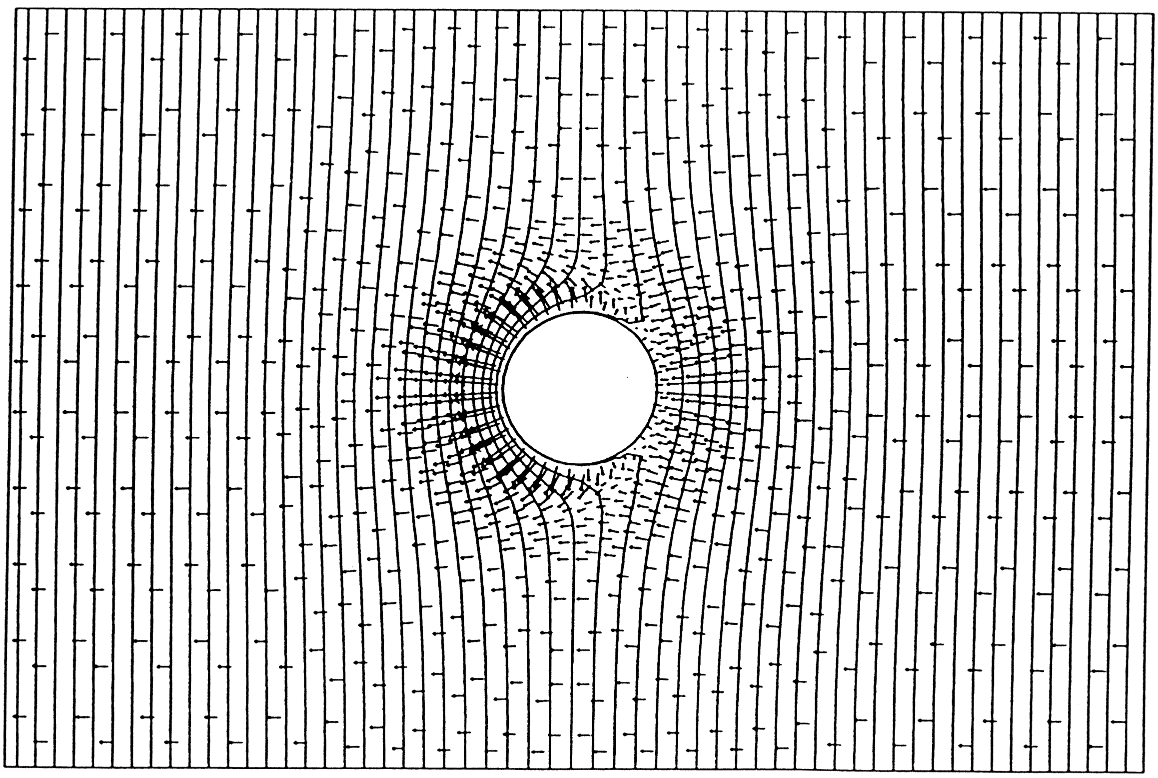


Figure A1.6., Day 8

The lake borders on a recharge system, only a small groundwater flux into the lake

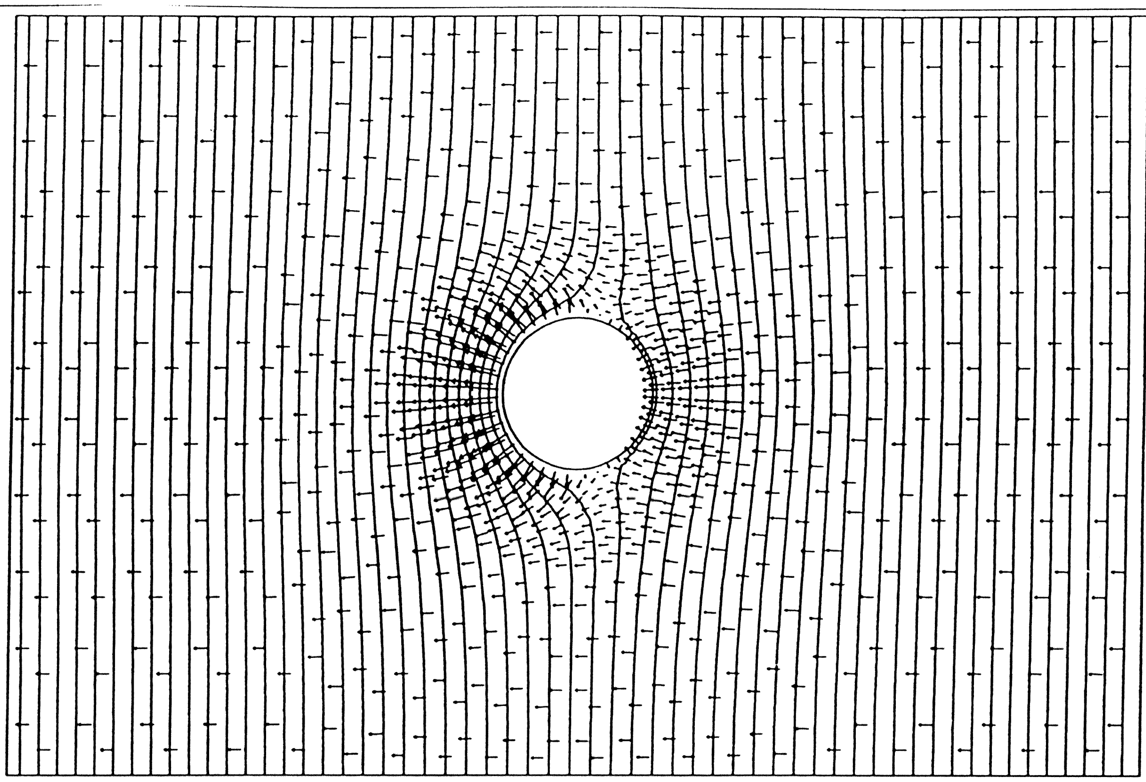


Figure A1.7., Day 10

System has reverted back to a flow-through regime

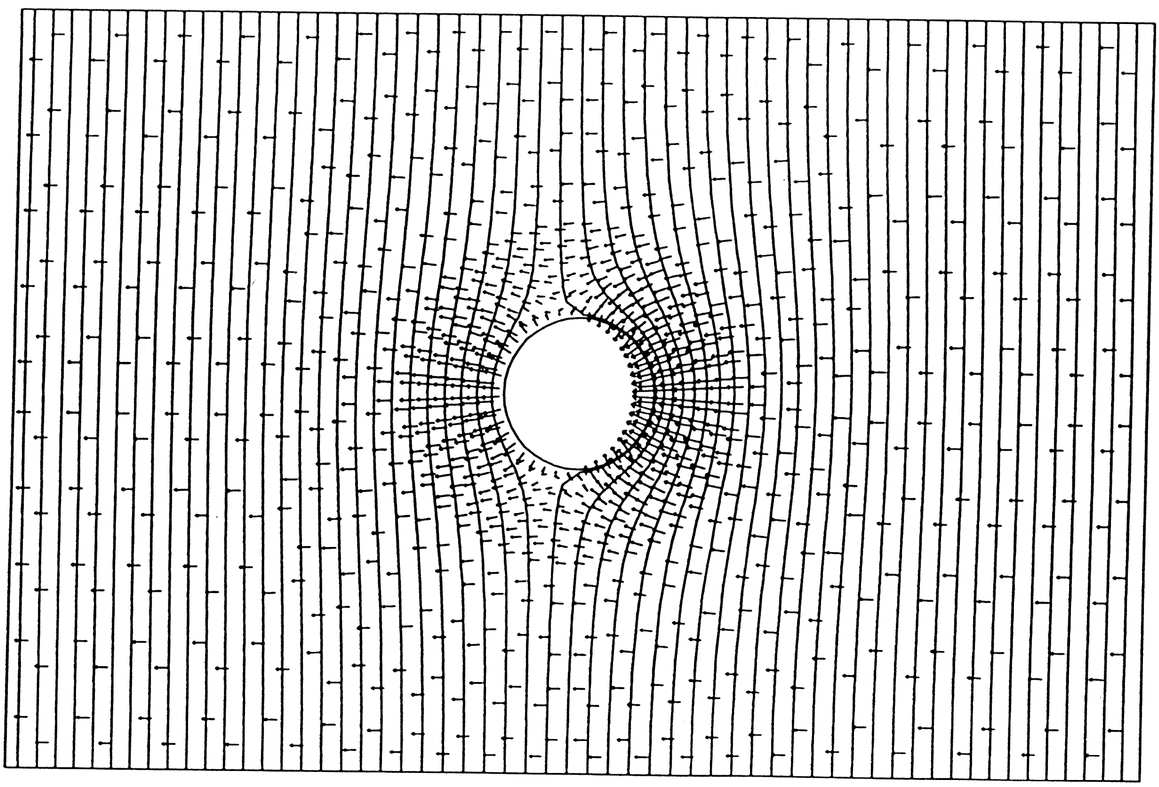


Figure A1.8., Day 47

Flow-through regime, stagnation boundary slightly downgradient of the centre of the lake

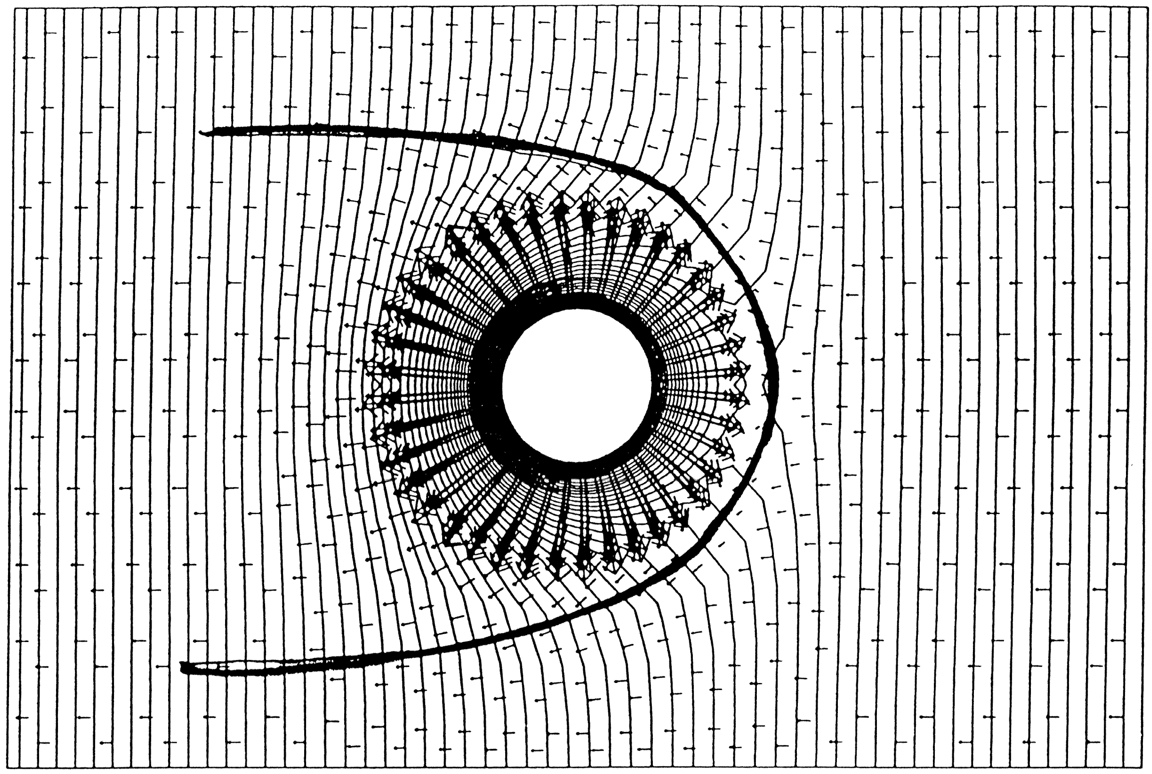


Figure A1.9., Day 63

Recharge regime (release zone of lake illustrated), large seepage flux to the aquifer, stagnation point 100m upgradient

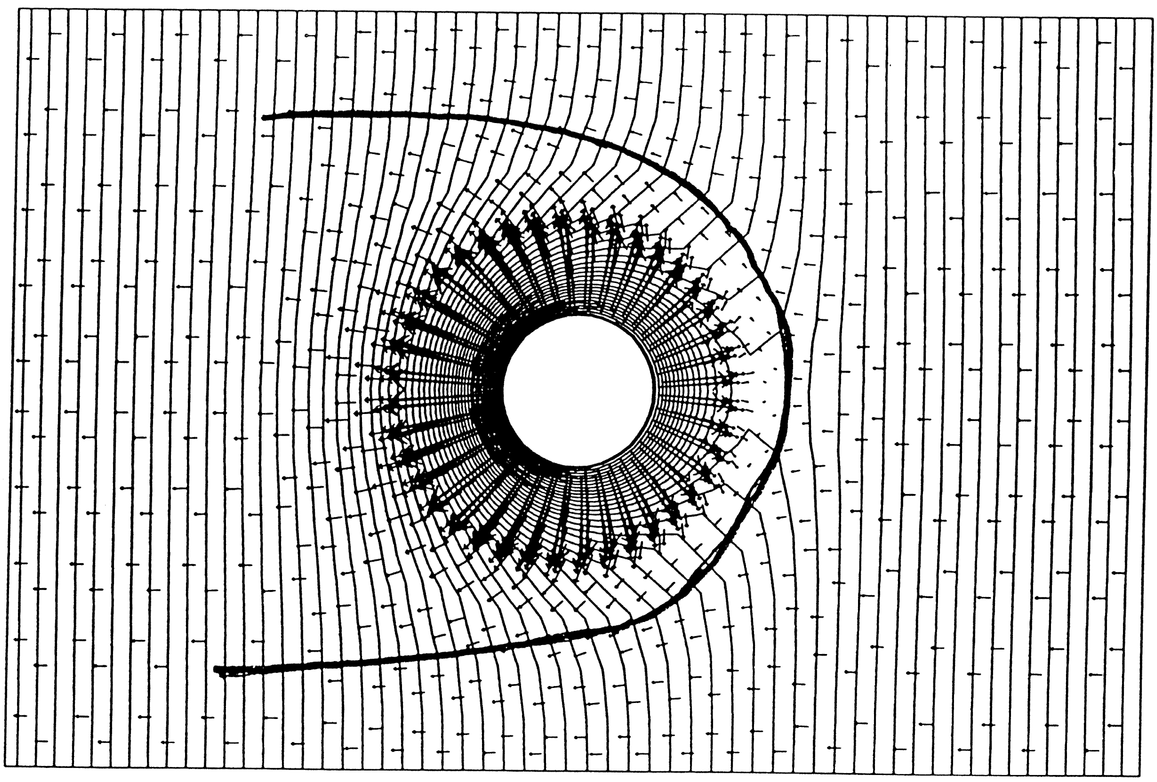


Figure A1.10., Day 64

Seepage flux is lower as indicated by the decrease in the shading around lake perimeter. Stagnation point is at approximately the same range

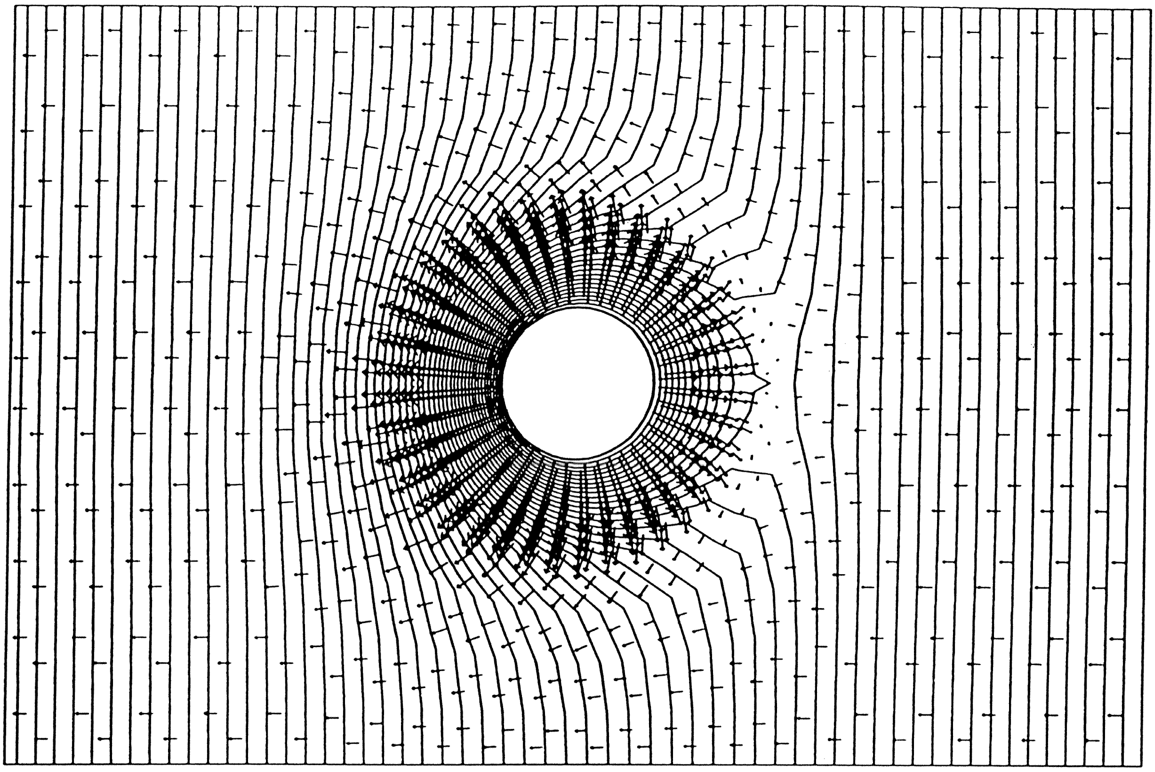


Figure A1.11., Day 66

As lake recedes seepage fluxes decrease, stagnation point continues to move outwards

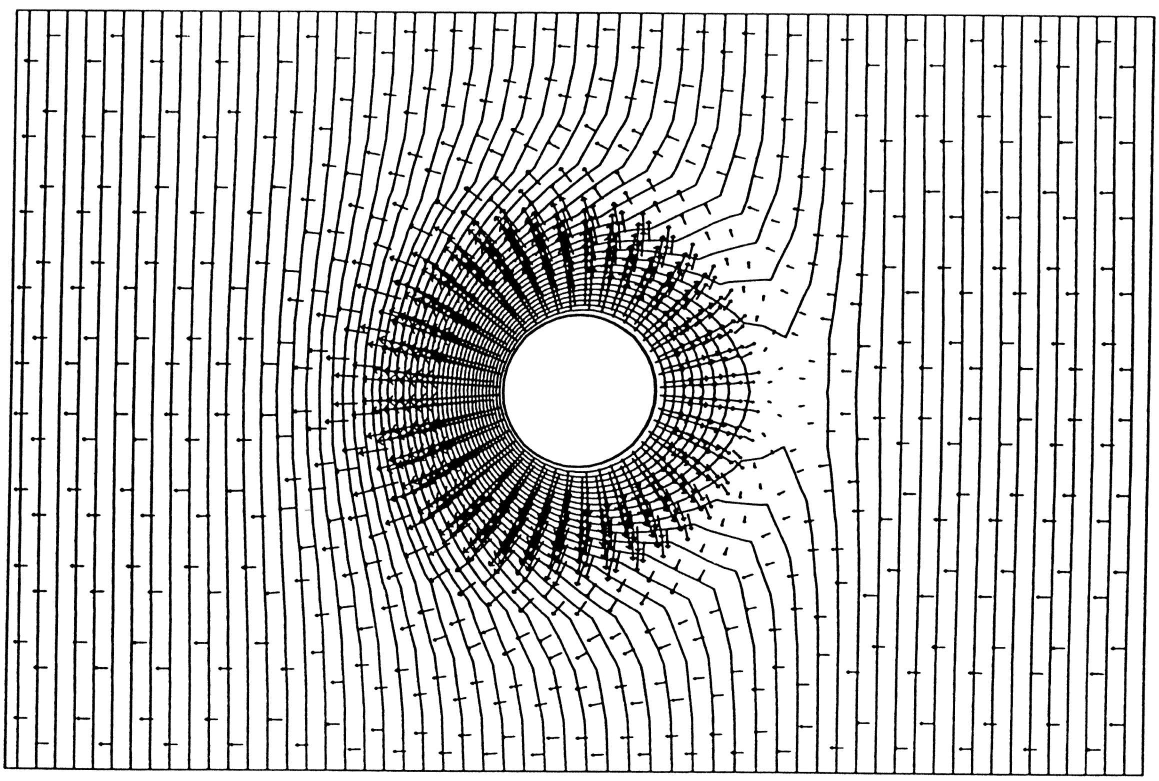


Figure A1.12., Day 68

Lake continuing in recharge regime, stagnation point 200m upgradient of lake perimeter

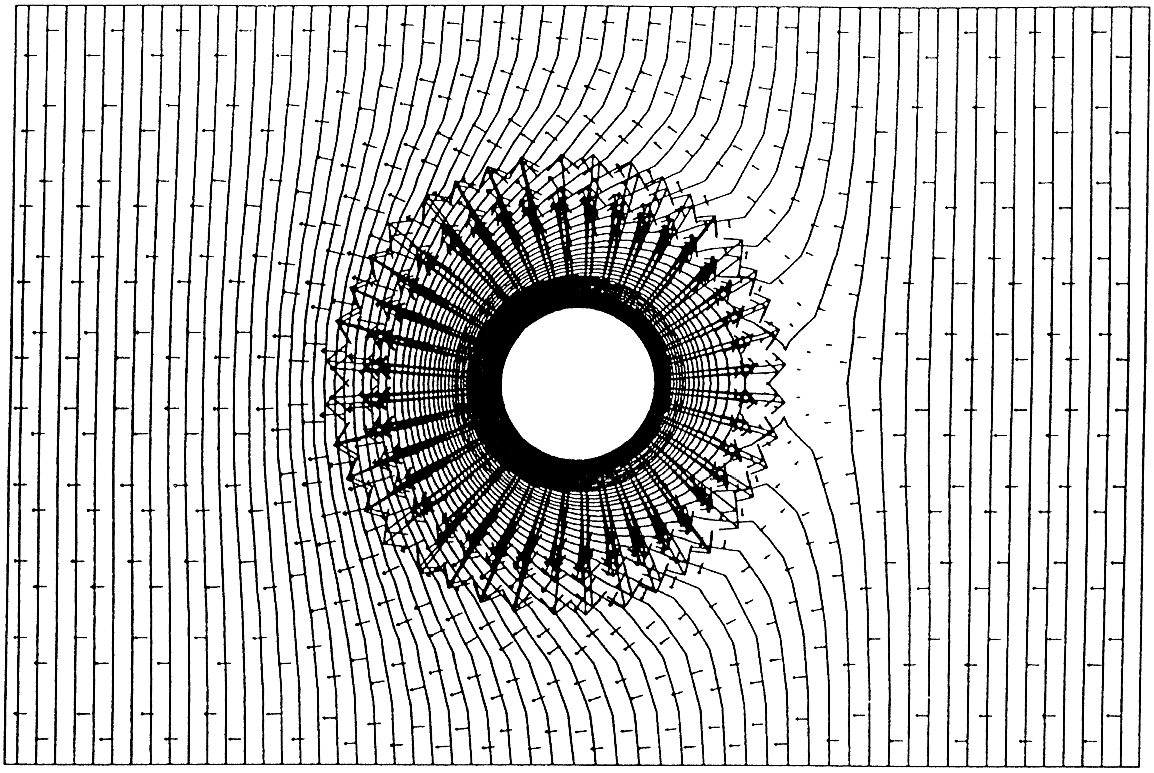


Figure A1.13., Day 79

Large runoff event, high seepage rates over a period in which the lake reaches its bank full level. In this recharge event some water is diverted to ocean outfall.

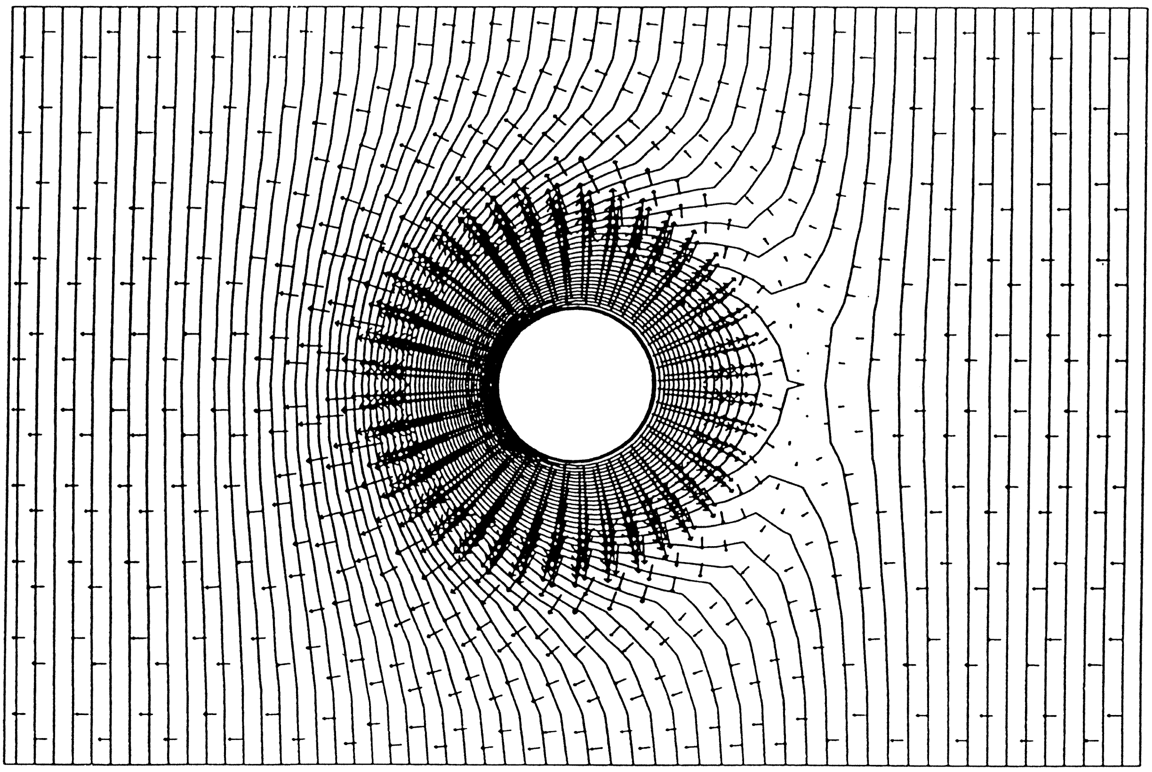


Figure A1.14., Day 83

Lake recedes and seepage flux to the aquifer is lowered

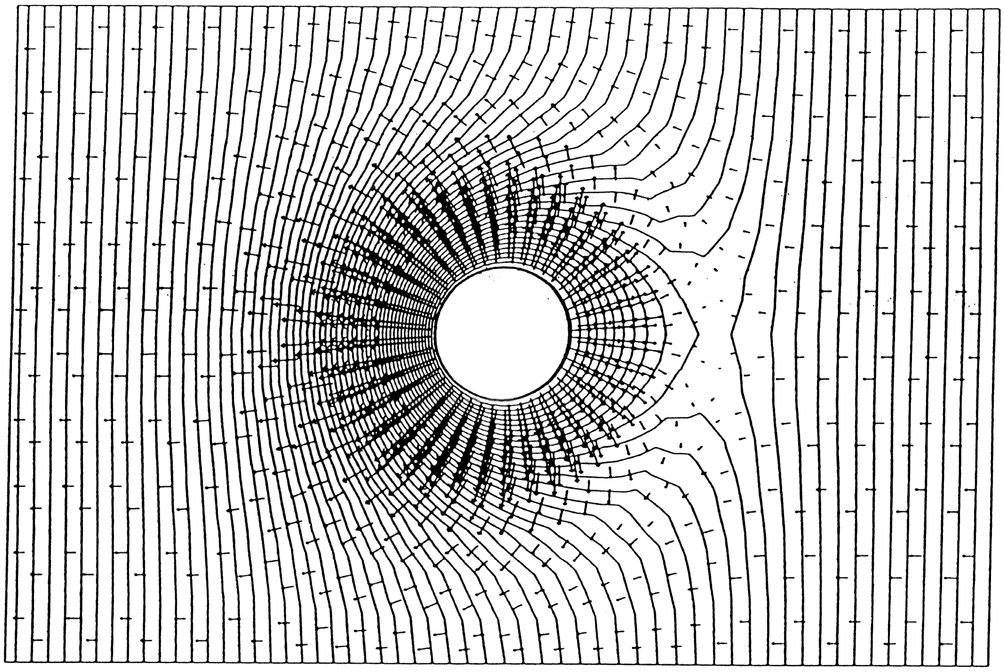


Figure A1.15., Day 88

Lake continues to recede, stagnation point is relatively constant at 200m upgradient.

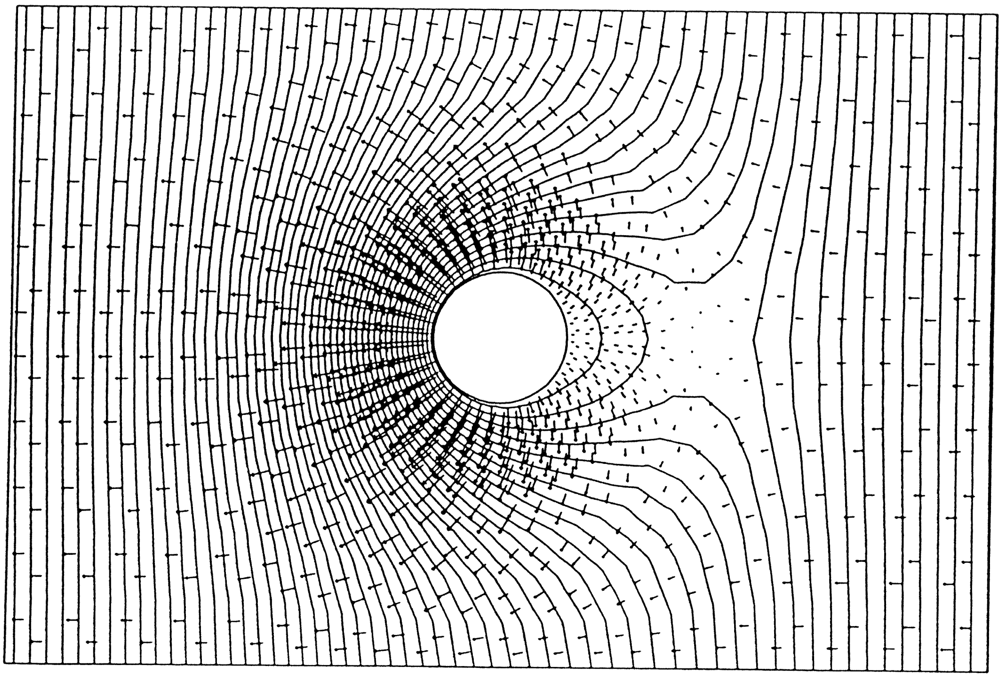


Figure A1.16., Day 105

The flux to the aquifer upgradient of the lake is shown to be very small although the lake continues to act as a recharge system.

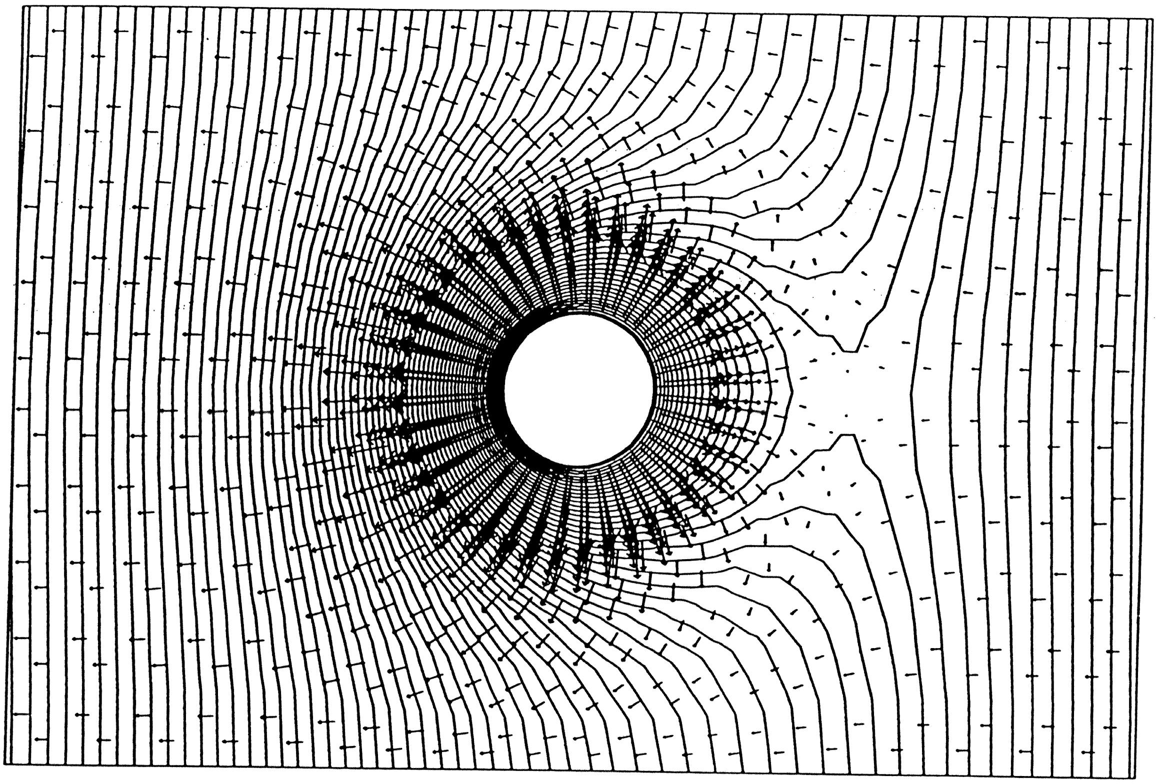


Figure A1.17., Day 125

The groundwater surrounding the lake has reached the same level as the ocean outfall, so that subsequent runoff events result in minimal recharge due to the increased discharge to ocean outfall. Stagnation point has reached a maximum distance of 300m upgradient of lake.

APPENDIX 2 MODELLING RESULTS FROM AQUIFEM-N (RUN 4)

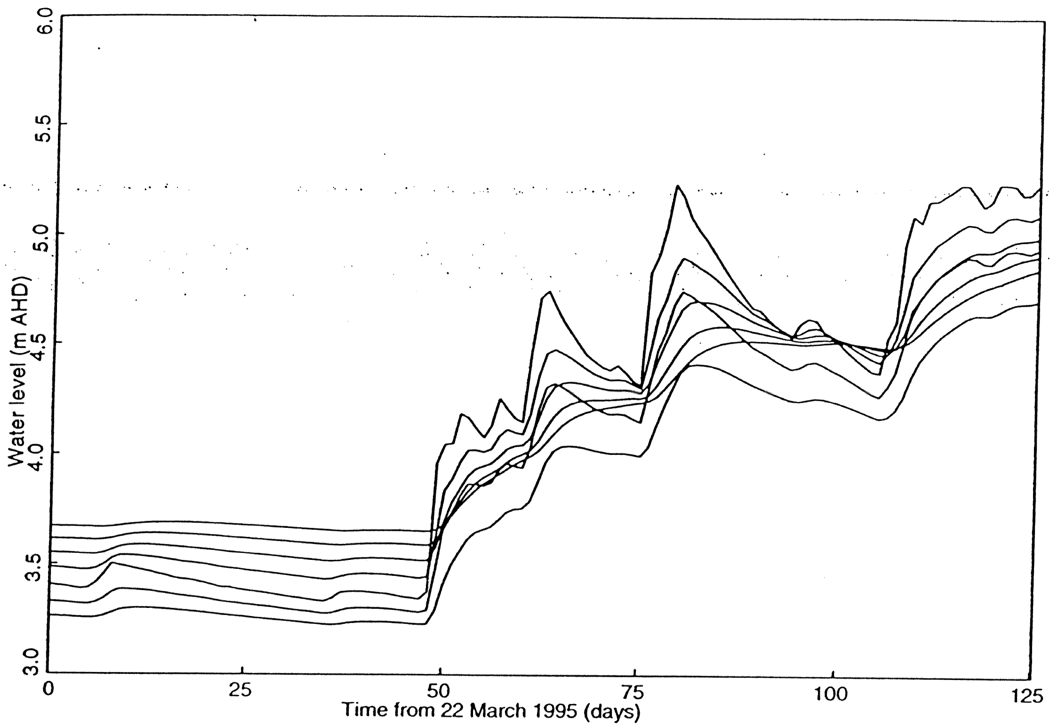
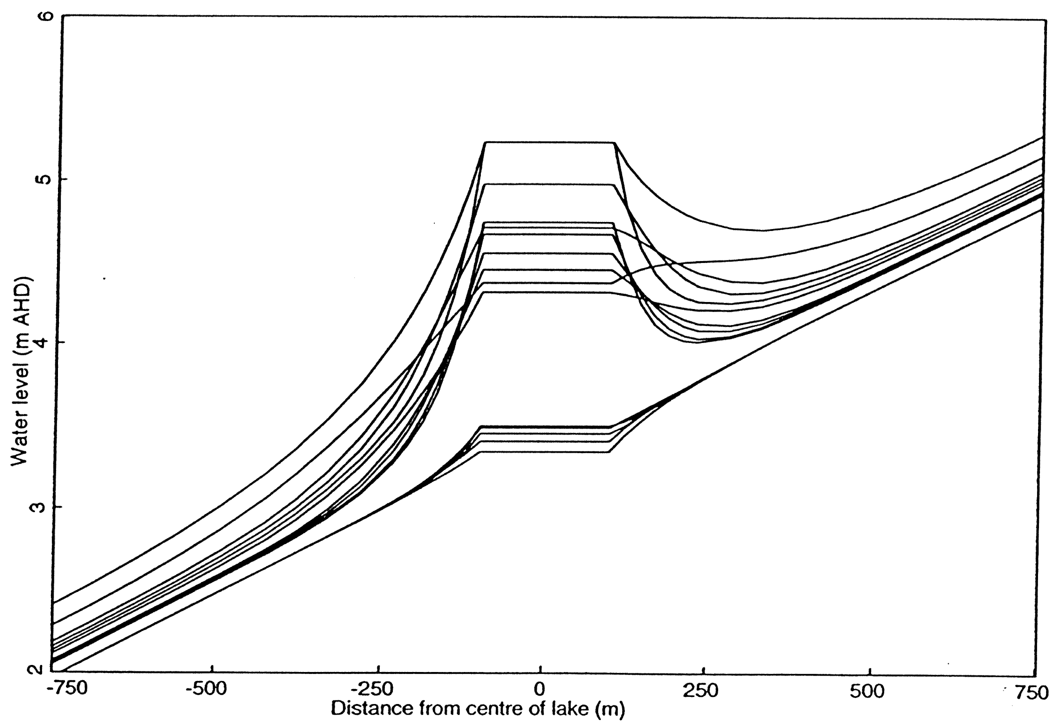


Figure A2.1., Variation in water level over time



- Establishment of flowthrough regime day 105

Figure A2.2., Water level as a function of distance from the lake

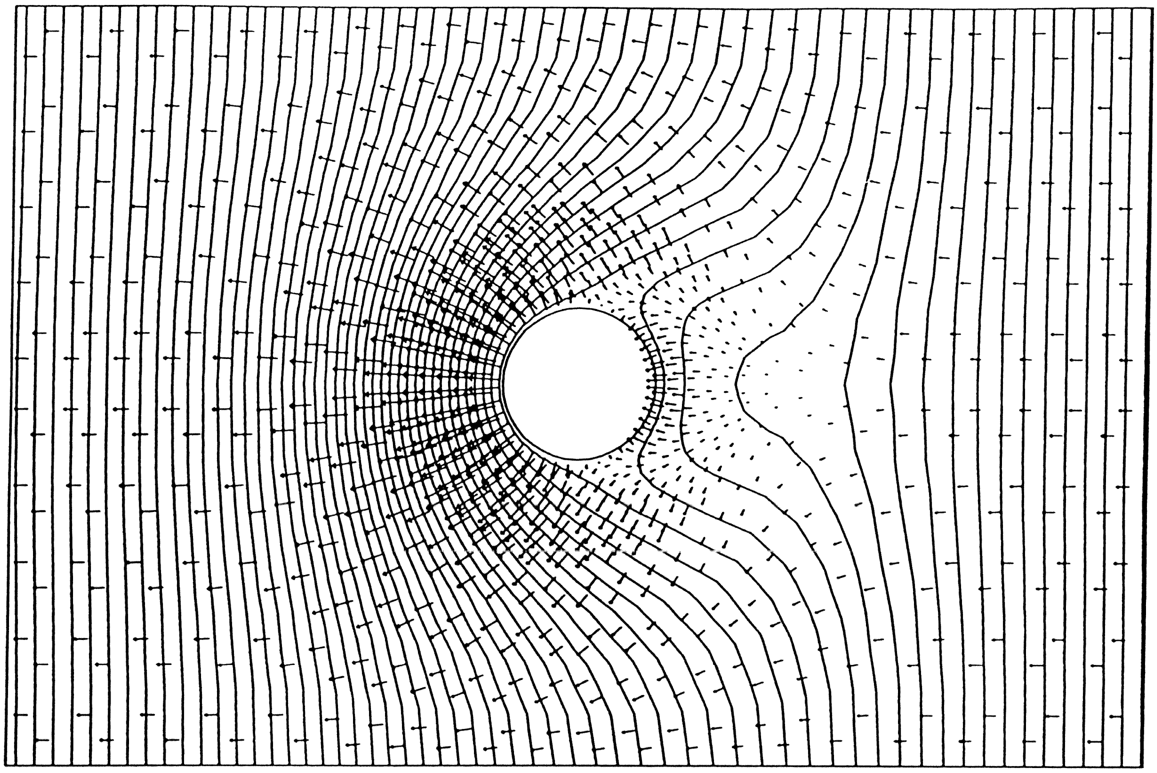


Figure A2.3., Day 105

Flow-through regime, note pronounced reduction in the groundwater gradient around 250m upgradient of lake perimter

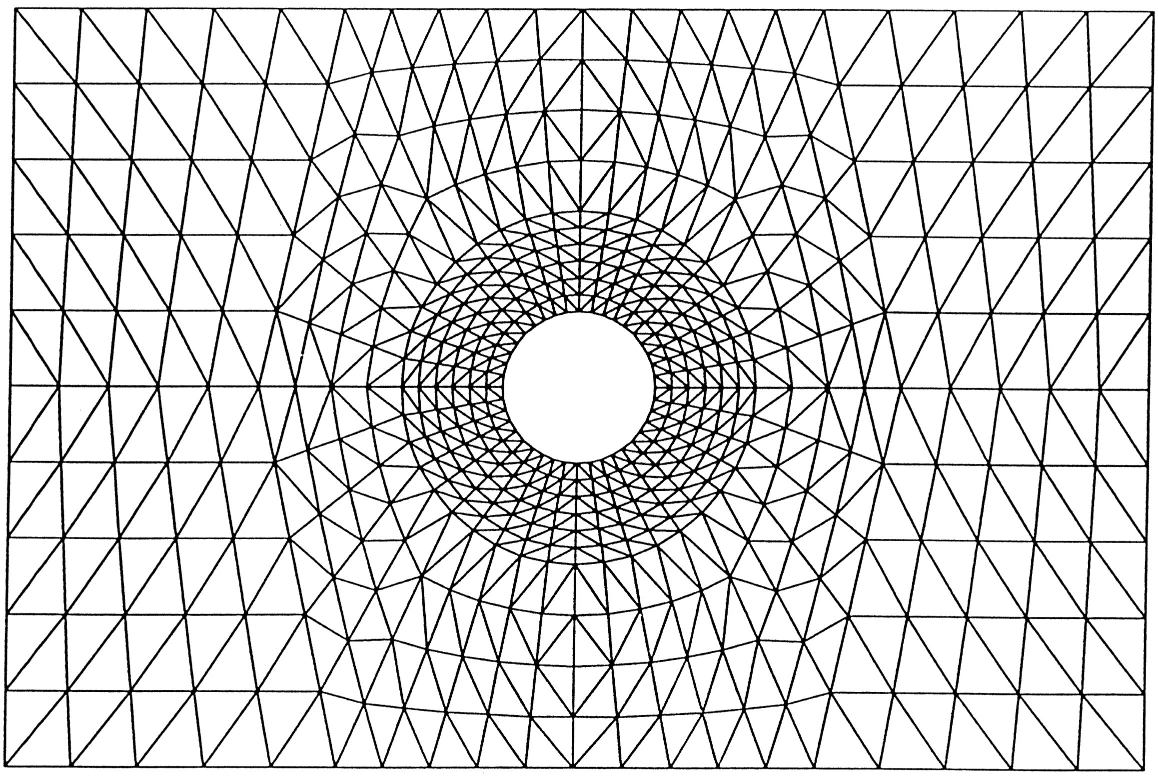


Figure A2.4., Fine mesh grid, lake diameter approximated to 200m, aquifer length 1.5km, aquifer width 1km

APPENDIX 3 NITRATE AND SULPHATE DISTRIBUTIONS

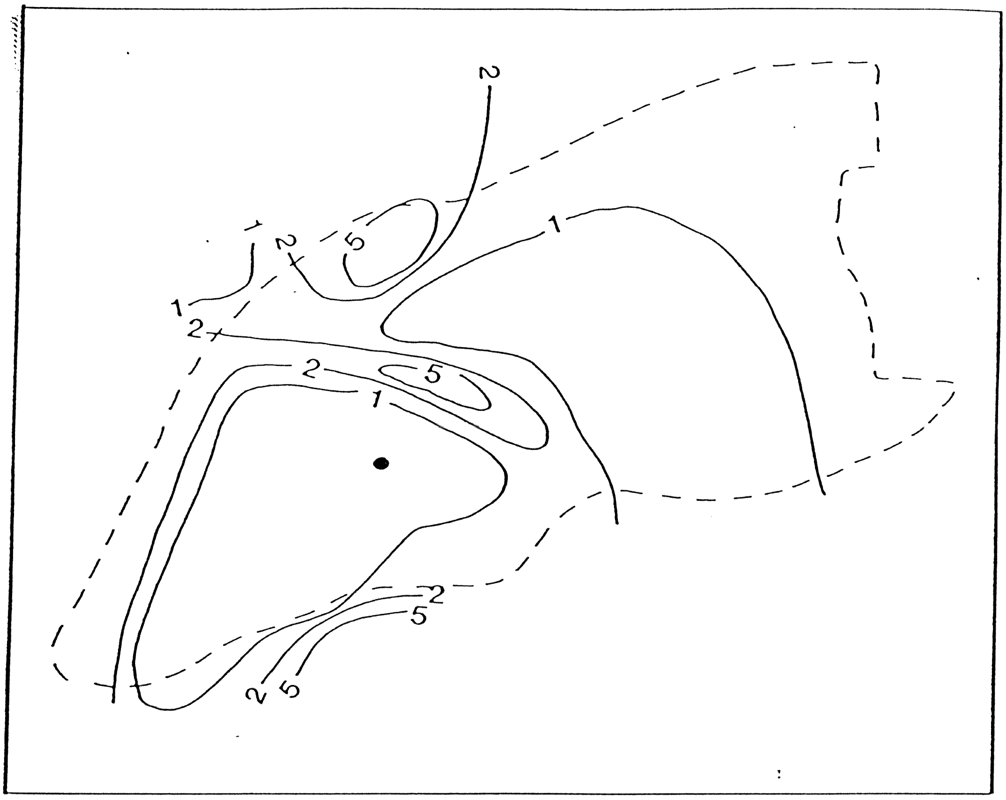


Figure A3.1 Nitrate concentration (mg/L) in the groundwater in the Shenton Park catchment (after McFarlane 1984)

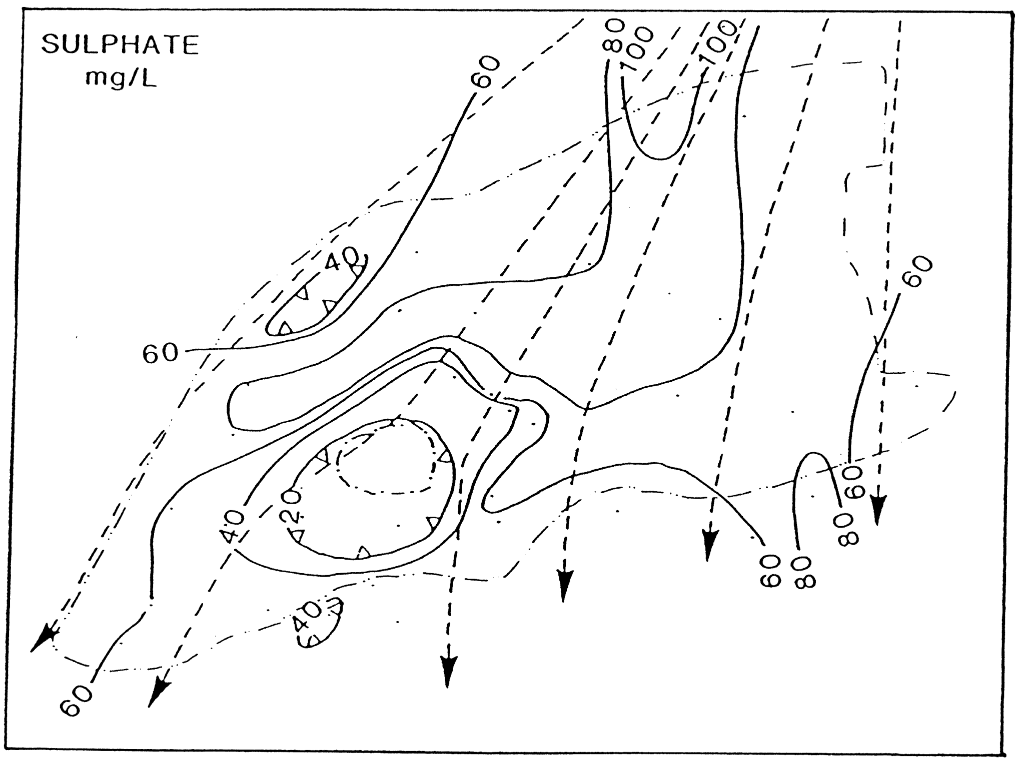


Figure A3.2 Sulphate concentrations (mg/L) in the groundwater in the Shenton Park catchment (after McFarlane 1984)