THE EFFECT OF URBANIZATION ON GROUNDWATER QUANTITY AND QUALITY

IN PERTH, WESTERN AUSTRALIA

by

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ABSTRACT

Hydrologic changes commonly attributed to urbanization include increases in rainfall, potential evaporation, runoff and stream flooding, and decreases in soil infiltration, recharge, groundwater levels and base flow to streams. Pollution of groundwater and surface waters by sewage and industrial effluent have also been reported. Pollution of surface waters by stormwaters is relatively common but no effects on groundwaters have previously been reported.

In Perth, Western Australia, groundwater levels have been noted to rise, rather than fall, following urbanization. Groundwater levels were low during 1977-79, the second driest three-year period recorded. During these three years, an additional 27,000 private bores were installed in the Perth area and limited saline intrusion was reported from around the Swan-Canning estuary.

To help understand the effects of urbanization and groundwater extraction on groundwater levels in Perth, water balances were carried out on two urban areas with differing housing- and bore- densities. Measurements were also made in native woodland areas. The complexities of urban hydrology meant that the water balances were necessarily approximate. However when annual water balances were calculated, net recharge, estimated by two independent methods, was in good agreement.

Both direct- and indirect- recharge were found to be much greater in the urban areas than in the woodland areas. Direct recharge is enhanced in urban areas by summer irrigation of lawns and gardens. Irrigation appears to encourage the shallow rooting of urban vegetation and results in a lower soil water deficit at the start of winter than occurs in non-irrigated woodland areas. Deep drainage of irrigation waters only took place when irrigation rates were greater than 60 per cent of potential evaporation.
rates. Canopy interception losses in urban areas were only 20 to 40 per cent the losses of woodland areas.

Indirect recharge was non-existent in the woodland areas due to the high infiltration capacities of the eolian sands. In the urban areas, indirect recharge of waters running off roofs and roads contributed 70 to 80 per cent of all groundwater recharge, despite these surfaces occupying only 30 to 40 per cent of the ground surface area. Indirect recharge took place after all, except very light, rainfalls in the urban areas. Consequently the difference in response between urban and woodland areas was at a maximum for light showers and for rainfall after long dry periods.

In an urban area with large blocks (and therefore relatively low shedding area) and 45 per cent private bore use, groundwater resources were found to be fully committed during a period of average to slightly below-average winter rainfall. As groundwater levels do not appear to have dropped appreciably over the past 30 years in this urban area, it is likely that groundwater inflow from adjoining high recharge areas has offset increased groundwater withdrawals. The relatively high proportion of vegetated area and high rates of irrigation in this urban area resulted in annual evaporation being in excess of annual rainfall.

In an urban area with small blocks (and therefore relatively high shedding area) and four per cent private bore use, large amounts of indirect recharge occurred, even when winter rainfall was below average. Excess waters in this area were drained to an ocean outfall and net groundwater recharge was limited by the amount of storage in the aquifer around the recharge (absorption) basin. The smaller percentage of vegetated area and lower rates of irrigation in this urban area resulted in annual evaporation being lower than annual rainfall. Water availability, rather than potential evaporation rates, appears to be the main constraint to evaporation in Perth.
The quality of road runoff waters was tested in one urban area and found to be appreciably better than those reported for Sydney and Melbourne road runoff waters. This is probably due to the virtual absence of soil runoff in Perth and a low fallout rate of atmospheric pollutants. The addition of road runoff to absorption basins resulted in seasonal cycles in salinity and iron in the down-gradient groundwaters. The iron was released when reduced waters flooded the goethite-bearing vadose zone in the vicinity of the absorption basins.

Generally, urbanization has a favourable effect on groundwater quantity in Perth, as a result of increased indirect recharge. Large-scale extraction of groundwater by bores has the potential to reverse the rise in groundwater levels which follows urbanization. Groundwater levels can best be managed by controlling indirect recharge of roof- and road-runoff waters, inputs from septic tanks and groundwater extraction by bores and drains.

Urbanization effects on groundwater quality in Perth are complex. In areas with high housing- and low bore- densities, groundwater salinities are unlikely to be affected overall. However, salinities around absorption basins will be low in late winter and spring and high in autumn and early winter. In areas distant from absorption basins, salinities are likely to be higher following urbanization. In areas with low or moderate housing densities, groundwater salinities are likely to be increased overall, due to concentration-by-evaporation of summer irrigation water. Where stormwaters are added to wetlands located within Spearwood Sands, increased dissolved iron is likely in down-gradient groundwaters following the opening winter rains.

The different hydrologic responses to urbanization in Perth result from the addition of roof- and road-runoff waters to the groundwater. These additions are made possible by most of Perth being located on highly permeable eolian sands.
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CHAPTER 1

INTRODUCTION

In 1974, a UNESCO report concluded that there was a general scarcity of data on the changes to the hydrologic regime brought about by urbanization (UNESCO 1974). The report noted that most studies were limited to the analysis of effects on a single component (e.g. runoff) of the hydrologic cycle, and a call was made for more water-balance inventories. Koch (1972) and McPherson (1973) had previously advocated using water-balance inventories in urban planning. These inventories would describe the quantity and quality aspects of water from precipitation through to runoff and evaporation. Both these authors had noted that studying a sub-system in isolation from the total system does not enable an investigator to perceive the total impact of alternatives.

In 1975, the American Society of Civil Engineers prepared a state-of-the-art report on the effects of urbanization on low flow, total runoff, infiltration and groundwater recharge (ASCE 1975). The report concluded that, while much had been written on the effects of urbanization, there was remarkably little quantitative information on these effects for water resources management. Only about 10 per cent of the 900 references examined were deemed suitable for citation.

Several water balance inventories, which have included urban areas, have been carried out in the last 20 years. The areas involved include the world (Baumgartner and Reichel 1975), continents (e.g. North America - Malhotra and Bock 1972), countries (e.g. USSR - Bochkov et al. 1972; West Germany - Keller 1972; Sweden - Carlsson and Falk 1977), cities (e.g. Chicago - Sheaffer and Zeisel 1966, cited by McPherson 1978; Moscow - Lvovich and Chernogaeva 1977) and parts of cities (e.g. Long Island - Franke and McClymonds 1972; Kursk - Lvovich and
Chernishov 1977; Goteborg – Malmquist and Svensson 1977a; and Halifax – Waller 1977). Most of these water balance inventories were based on working assumptions rather than on detailed measurements.

A major finding of the UNESCO report was that the problems of urban hydrology are remarkably similar in all parts of the world (UNESCO 1974). However a review of the literature (Chapter 2) indicates that, while there are similarities in the physical effects of urbanization on the hydrology of an area, the chemical (or water quality) effects are not as uniform (although this aspect has not received as much investigation). The review also indicates that the major reasons for differences in the hydrologic effect of cities are differences in city size, climate, geology and wastewater systems. City size influences the degree to which some climatic parameters are modified. Size also influences the demand for surface- and ground-waters, both inside and outside the urban basin. Larger cities generate more wastewater as a consequence of increased water use and runoff. Cities in temperate climates differ from cities in arid and semi-arid climates in aspects such as the heating of buildings, irrigation during dry seasons and the need to use road salt to de-ice roads. The geology of cities influences the amount of water that can infiltrate the soils and therefore the amount of runoff that takes place on permeable surfaces. The options available for the disposal of runoff from impermeable surfaces, also depend upon the permeability of soils. The type of wastewater disposal system (combined* or separate) affects the water quality of any receiving water bodies (i.e. rivers, estuaries, lakes, oceans, and groundwaters).

To date, little work has been done on the impact of urbanization on water quantity and quality in Australia. Dunin (1974) noted that over 100 experimental catchments were then being monitored in Australia. Many of these experiments were aimed at determining the hydrologic response to land-use changes. Dunin

* Combined sewerage systems carry both domestic sewage and stormwater runoff from roads.
stated that the most notable deficiency in the experimental program was the absence of monitoring of the effects of urbanization. Australia is one of the most urbanized countries in the world, with about 70 per cent of its population living in urban centres of more than 100 000 people (Australian Bureau of Statistics 1982).

Perth, the administrative and political capital of the State of Western Australia, has a population of about 900 000, which is over 70 per cent of the total population of the State (1981 census). About 70 per cent of the Perth metropolitan area is located on very porous and permeable eolian sands, the remainder being located on less permeable alluvial deposits, lateritic gravels and weathered granite (Figure 1.1).

A shallow, unconfined aquifer is present in the sands and is used extensively for irrigation, domestic water supplies and industrial processes. An estimated $215 \times 10^6 \text{ m}^3$ was extracted from the aquifer during 1981 (Metropolitan Water Authority 1983). Over three-quarters of this extraction was made by over 60 000 private bores, while the other quarter was used for public water supplies. About 27 000 private bores were installed in the three-year period 1977-79 (Australian Bureau of Statistics 1979), the second driest three-year period ever recorded*. Groundwater- and wetland- levels were at their lowest recorded levels during this dry period and limited salt water intrusion was reported from around the Swan-Canning estuary. As private bores require licensing only if they are within Public Water Supply Areas, the precise number of bores and amount of groundwater extracted are unknown. Isolating the effects of extraction on groundwater levels is made difficult by changes in levels brought about by changes in land- and water- use during the various stages of urbanization.

The general aim of this thesis is to investigate the effects of urbanization (including groundwater extraction) on groundwater

* The driest three-year period was 1975-77.
FIGURE 1.1: Location of urban areas in relation to shallow groundwater.

(Inset: Location of Perth in relation to Western Australia.
quantity in Perth by carrying out water balances on representative areas and comparing these areas with native woodland areas. By examining specific water quality parameters, the study also aims to identify urban effects likely to diminish the suitability of the groundwater resource for its present uses. A general examination of groundwater chemistry is included to aid the understanding of the physical changes brought about by urbanization.

The study provides information on the effects of urbanization on the hydrological regime of a predominantly-residential city with a Mediterranean climate (i.e. hot, dry summers; cool, wet winters) and largely situated on porous sands. The sands enable rainfall and irrigation waters to infiltrate the soil without appreciable surface runoff (except on man-made, impermeable surfaces). They also enable a proportion of runoff from the impermeable surfaces to be added to the groundwater, rather than all runoff being discharged to river and ocean outfalls as occurs in many other urban areas (especially those cities located on alluvial clays). The practice of adding runoff to the groundwater in Perth is facilitated by a separate sewerage system. This direction of urban runoff to the aquifer has implications for groundwater quality.

The specific aims of the thesis are:

(i) to identify and measure the components of the water balance in two dissimilar urban areas and to compare the urban situation with native woodland areas;

(ii) to gain sufficient understanding of hydrologic processes in the study areas that the results may be extended to unstudied areas, and

(iii) to assess the impact of urbanization on those groundwater-quality parameters which are likely to diminish the
usefulness of the groundwater in the study areas.

The second aim (above) of the thesis has particular relevance in light of a recently commenced major study by the Metropolitan Water Authority on the effects of overpumping and pollution on groundwater in the Perth area.
CHAPTER 2

LITERATURE REVIEW

2.1 URBAN EFFECTS ON THE HYDROLOGIC CYCLE

Urban influences on the hydrologic cycle can be divided into those which affect climate, surface waters and groundwaters. As climatic and surface water changes can affect groundwaters, all three elements will be reviewed in turn. Greater emphasis is given to those aspects which have most relevance to the present study. Where corresponding data are available, comparisons are made between Perth and the cities mentioned in the literature (principally North American and European cities). Urban effects on water quality are discussed in Section 2.2.

2.1.1 Climatic effects

The differing microclimates found in urban areas result from four physical mechanisms (Munn 1966). These are:-

(i) A disturbed, natural-radiation balance resulting from vegetation being replaced by concrete and brick. In some older, residential areas, however, the tree canopy can resemble that of a forest;

(ii) The creation of wind obstacles changing wind flow and turbulence;

(iii) A change in the water vapour balance by replacing moist surfaces with dry; and

(iv) The emission of heat, water vapour and pollution to the atmosphere.

Landsberg (1970, and in Lowry 1967) published a table of average climatic changes induced by urbanization. This table is reproduced in part as Table 2.1. The changes noted in the table
can be taken only as a general guide because there is, as yet, insufficient evidence of the changes (UNESCO 1974) and there is likely to be significant variation between cities (e.g. Sekiguti and Tamiya 1970). Changnon (1970) emphasised the difficulties in differentiating orographic, maritime and gauge-exposure effects on precipitation from urban effects. Similar difficulties can be expected for other climatic elements.

Table 2.1

**Average changes in climatic elements caused by urbanization**
*(after Landsberg 1970)*

<table>
<thead>
<tr>
<th>Element</th>
<th>Comparison with rural environment</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Radiation</strong></td>
<td></td>
</tr>
<tr>
<td>global</td>
<td>15% to 20% less</td>
</tr>
<tr>
<td>ultraviolet, winter</td>
<td>30% less</td>
</tr>
<tr>
<td>ultraviolet, summer</td>
<td>5% less</td>
</tr>
<tr>
<td>sunshine duration</td>
<td>5% to 15% less</td>
</tr>
<tr>
<td><strong>Temperature</strong></td>
<td></td>
</tr>
<tr>
<td>annual mean</td>
<td>0.5 to 1°C higher</td>
</tr>
<tr>
<td>winter minima (average)</td>
<td>1 to 2°C higher</td>
</tr>
<tr>
<td><strong>Precipitation</strong></td>
<td></td>
</tr>
<tr>
<td>total</td>
<td>5% to 10% more</td>
</tr>
<tr>
<td>days with less than 5 mm</td>
<td>10% more</td>
</tr>
<tr>
<td><strong>Relative humidity</strong></td>
<td></td>
</tr>
<tr>
<td>winter</td>
<td>2% lower</td>
</tr>
<tr>
<td>summer</td>
<td>8% lower</td>
</tr>
<tr>
<td><strong>Wind speed</strong></td>
<td></td>
</tr>
<tr>
<td>annual mean</td>
<td>20% to 30% lower</td>
</tr>
<tr>
<td>extreme gusts</td>
<td>10% to 20% lower</td>
</tr>
<tr>
<td>calms</td>
<td>5% to 20% more common</td>
</tr>
</tbody>
</table>

The means by which the physical mechanisms identified by Munn (1966) are responsible for the climatic element changes shown in Table 2.1 are examined below. Wherever possible, specific
reference is made to Perth conditions.

Incoming radiation (particularly ultraviolet radiation) is decreased by smoke, dust and carbon dioxide over cities (Zuidema 1974). Outgoing longwave radiation is also decreased. Landsberg (1970) has estimated that urban atmospheres contain 10 times more condensation nuclei and particulates, and 5 to 25 times more gaseous admixtures than do rural atmospheres. These atmospheric contaminants have important effects on urban temperatures and rainfall as will be seen later.

Radiation records for Perth are very limited (2 sites) and no comparisons of urban and rural areas have been made. Gentilli (1980) noted that Perth has always been free from extreme pollution but has often been affected by smoke from the burning-off of vegetation and from domestic fires. Mackey (1963) reasoned that a concentration of pollutants during the hotter months (mid-October to mid-April) was unlikely in Perth owing to turbulence which results from high temperatures and wind movement. In the remaining months of the year the prevailing westerly winds tend to remove any pollutants while the high rainfall during this period also contributes to removal of pollutants.

Zuidema (1974) reviewed the possible causes of increased air temperatures in urban areas and concluded that the following factors were important:

(i) A reduced latent heat flux (i.e. reduced evaporation) due to the rapid drainage of water from paved areas;

(ii) The different physical properties (e.g. albedo, specific heat and thermal conductivity) and structures of urban surfaces. Most urban materials have greater heat capacities and thermal conductivities than non-urban materials, and, as a greater amount of heat is stored during the day, more is released at night. Munn (1966) observed that as walls capture more solar energy than
horizontal surfaces when the sun is low, urban areas will have lower albedos than rural areas;

(iii) Energy released by heating and combustion processes (e.g. residential heating, industry, vehicles);

(iv) A reduction in outgoing longwave radiation due to the presence of dust and carbon dioxide in the urban atmosphere. The atmospheric contaminants also increase the flux divergence of incoming shortwave radiation. The resultant 'heat island' is at a maximum at night and there is often a weekly cycle.

Barrett (1979) found a maximum increase in temperature of 9°C between the suburbs and the central business district in Perth. The heat island was best developed in the evening and in cool seasons, and was at its weakest under conditions of maximum temperature. Barrett attributed the heat island to re-radiation of stored heat in buildings and roads, but it is possible the effect could be enhanced by a reduced latent heat flux in the central business district (McFarlane 1981). There is little heating of buildings in Perth in comparison to North American and European cities.

Increased precipitation and cloudiness in urban areas have been attributed both to a greater abundance of condensation nuclei and to convection cells resulting from the urban heat island. Evidence for increased precipitation in urban areas has been given by Atkinson (1970), Changnon (1970), Kuprianov (1974) and Yperlaan (1977), amongst others. There is some evidence for increased precipitation immediately downwind of some urban areas (e.g. Sekiguti and Tamiya 1970), which has led to the notion that urban areas affect the distribution, rather than the absolute amount, of precipitation (Kuprianov 1977). Table 2.1 indicates that a proportion of the increased precipitation comes in the form of light showers, although Massing (1976) reported a 10 per cent increase in rainfall intensity for Hamburg, West Germany.
Swindell (1979) studied the daily distribution of rainfall over Perth for three 5-day periods and found wind direction and synoptic situation showed the closest relationship with rainfall distribution patterns. During convectional rain the distribution pattern was found to be of cell form, whereas during frontal rain, the distribution patterns were cells within streets (at the beginning) and bands (at the end). The cells, cell streets and bands did not always occur over the same places and therefore urbanization did not seem to be exerting a marked effect on distribution. No long-term study of rainfall has been made for Perth.

Decreased relative humidity in urban areas has been attributed to decreased sources of evaporation (Munn 1966), brought about by the removal of vegetation and the rapid removal of storm waters. An increase in air temperature may also be partially responsible for lower relative humidities (Zuidema 1974). In winter, the emission of water vapour by combustion processes may reverse (Munn 1966) or at least decrease the effect (Zuylen 1971, cited by Zuidema 1974).

As no previous comparison of relative humidity values has been carried out for different parts of Perth, a comparison of measurements from three stations is presented in the next chapter.

The decrease in average wind speed and gusts in urban areas has been attributed to the increased roughness and friction of urban areas. The decreased speed is accompanied by increased turbulence (Munn 1966). However, when regional winds are light, the urban heat island may generate its own mesoscale circulation (Munn 1966; Lowry 1967).

As no comparisons have been made of the effects of urbanization on average wind speeds and gusts in Perth, a comparison of two sites is carried out in the next chapter.
From Table 2.1 it can be deduced that two climatic element changes (the increase in temperature and the decrease in relative humidity) would tend to increase potential evaporation rates in urban areas, whereas two other element changes (the decrease in radiation and wind speed) would tend to decrease potential evaporation rates. The effect of a decrease in wind speed may be partially, if not wholly, offset by the increase in wind turbulence. Kuprianov (1977) suggested that potential evaporation rates are 5 to 20 per cent higher in urban areas as a result of increases in both amounts of heat and evaporating surfaces. However, the amount of water available for evaporation was considered to be less in towns as a result of rapid runoff. Consequently, Kuprianov (1977) considered that the difference in actual evaporation between urban and rural areas probably lay within the limits of observational accuracy. Oke (1979) measured the energy balance of an irrigated urban lawn in Vancouver, Canada, and found that advected heat from drier surfaces increased evaporation rates. Oke considered that the increase partially compensated for the reduction in vegetated areas in urban areas. Massing (1976) contended that the proportion of rainfall that evaporates is higher in urban areas than in non-urban areas as a result of elevated temperatures. Costin and Dooge (1973) and Kuprianov (1977) considered that in arid and semi-arid areas, urban evaporation is possibly greater than in non-urban areas due to the presence of artificial reservoirs and planted vegetation.

No measurements have been made of variations in evaporative potential or amounts in the Perth area. McFarlane (1981) has postulated that evaporation per unit of vegetated area is greater in Perth urban areas, as a result of garden irrigation during the hot summer period. Whether this increase would be sufficient to offset the reduction in vegetated areas was unknown. A comparison of Class-A pan evaporation from three sites in Perth is made in the next chapter.
It can be concluded from the above review that urbanization can alter various climatic elements by about 5 to 15 per cent when compared with non-urban areas. These effects are likely to be more pronounced in larger cities which have the ability to affect larger air masses. There is also some evidence that cities in temperate areas differ in some respect to cities in arid and semi-arid areas (e.g. heating of buildings, evaporative amounts).

There are some indications (given in this and the next chapter) that the effects of urbanization on climate are less for Perth than for some of the other cities reported in the literature. A possible reason for a reduced effect in Perth may be a lower concentration of condensation nuclei and gaseous admixtures in the atmosphere (and therefore less effect on rainfall and radiation/re-radiation). Lower concentrations would probably result from a low emission rate and the presence of a strong and regular wind (average annual speed = 16 km/h). Perth also has a low population density in comparison with other studied cities.

The changes in climatic elements will probably produce changes of a similar, or smaller, magnitude to the inputs and outputs to the groundwater in urban areas. The observation by Landsberg (1970) that much of the increased rainfall in urban areas is in the form of light showers may mean that groundwater recharge is not increased by a commensurate amount. A much more important effect on groundwater levels than climatic changes is likely to result from changes to surface waters, as will be seen in the next section.

2.1.2 Surface waters

It has long been recognised that during urbanization, the creation of impervious shedding surfaces (e.g. roads, roofs, paths) with low depression storages results in an increase in both the amount, and rate, of surface runoff. A depression storage of
only 0.75 mm was reported for impermeable surfaces in Lund, Sweden, by Carlsson and Falk (1977). McPherson (1974b) considered that the lower frictional resistance of streets and conduits accelerates the flow and increases peaks of urban runoff. Flow distances are usually shortened in urban areas (Carlsson and Falk 1977), also aiding increases in flow rates. Brater (1968) introduced the concept of an 'hydrologically significant impermeable area' (HSIA) for cities to try to explain the hydrographs obtained in urban areas. The HSIA was found to increase with population density and to be higher during wet seasons once soils became wet. Carlsson and Falk (1977) and Lvovich and Chernogaeva (1977) considered that increased soil compaction in urban areas results in significantly greater runoff from permeable areas. Felton and Lull (1963, cited in ASCE 1975) found lawns in an urban area had infiltration rates only one-sixth those of forests, due to surface modification and compaction. Kelling and Peterson (1975) measured reductions in water intake of about 65 per cent in lawns which had textural and compaction discontinuities as a result of building and lawn construction.

Cordery (1976a) produced a table of the effects of urbanization on flood peaks. This is reproduced as Table 2.2 with modifications to include extra data and data on increases in runoff volumes. The table shows that there is considerable variation between catchments. Some of this variation will be due to variations in the percentage of impervious areas between the catchments (Cordery 1976a). The greatest variation between catchments occurs in the degree to which discharge speed is increased. Leopold (1968) has produced a figure for predicting the increase in runoff that occurs with differing degrees of urbanization.

The runoff from many cities is discharged into streams and the increased volumes of runoff and peak discharge (shown in Table 2.2) have resulted in flooding. Goddard (1973, cited by McPherson 1974a) reported that 60 per cent of stream flooding damage in the USA occurs in urban areas. Under conditions of decreased soil
### Table 2.2

**Effects of urbanization on flood peaks and runoff volumes**  
(modified from Cordery 1976a)

<table>
<thead>
<tr>
<th>No. of times peak discharge is increased</th>
<th>No. of times speed of discharge is increased</th>
<th>No. of times runoff volume is increased</th>
<th>Reference</th>
</tr>
</thead>
<tbody>
<tr>
<td>1 to 3</td>
<td></td>
<td></td>
<td>Espey and Winslow (1974)</td>
</tr>
<tr>
<td>3 to 4</td>
<td></td>
<td></td>
<td>Wilson (1964)</td>
</tr>
<tr>
<td>1.3 to 2.3</td>
<td></td>
<td></td>
<td>Sawyer (1963)</td>
</tr>
<tr>
<td>1.42 to 2.18</td>
<td>1.54 to 1.96</td>
<td></td>
<td>Rao <em>et al.</em> (1972)</td>
</tr>
<tr>
<td>3 to 4</td>
<td></td>
<td></td>
<td>Waananen (1961)</td>
</tr>
<tr>
<td>2</td>
<td>5</td>
<td></td>
<td>Wiitala (1961)</td>
</tr>
<tr>
<td>6</td>
<td>3.03 to 12.5</td>
<td></td>
<td>Van Sickle (1962)</td>
</tr>
<tr>
<td>1.58</td>
<td></td>
<td></td>
<td>USGS (1969)</td>
</tr>
<tr>
<td>1.8</td>
<td>2.94</td>
<td></td>
<td>Carter (1961)</td>
</tr>
<tr>
<td>&lt;6.7</td>
<td></td>
<td></td>
<td>Anderson (in Waananen 1969)</td>
</tr>
<tr>
<td>3.3</td>
<td>2.29</td>
<td></td>
<td>James (1965)</td>
</tr>
<tr>
<td>(Dozens)</td>
<td>10</td>
<td>1.5 to 2.0</td>
<td>Kuprianov (1977)</td>
</tr>
<tr>
<td></td>
<td></td>
<td>4</td>
<td>Lvovich and Chenogaeva (1977)</td>
</tr>
<tr>
<td></td>
<td></td>
<td>18</td>
<td>Carlsson and Falk (1977)</td>
</tr>
<tr>
<td></td>
<td>3</td>
<td>3.3</td>
<td>Espey <em>et al.</em> (1969, in Lazaro 1979)</td>
</tr>
<tr>
<td></td>
<td>2 to 3</td>
<td>8</td>
<td>Spieker (1969)</td>
</tr>
</tbody>
</table>

2 ca. 8 2 Mode
infiltration (as a result of the reduction in permeable areas in cities), recharge to groundwaters decreases and there is a resultant decrease in base flow to streams (Savini and Kammerer 1961). James (1965) calculated base flow to be only 70 per cent following urbanisation. Stream flow during dry periods may therefore be decreased following urbanization.

Most cities can not meet their water demands from within their own basin and large transfers of water are required from adjoining basins, or from aquifers not hydraulically connected to the urban basin (Savini and Kammerer 1961). When this imported water is discharged into rivers via the sewerage or stormwater system, increased annual streamflow can result, which has a marked effect on small sub-basins (Kuprianov 1977). Where sewerage discharge forms a significant proportion of streamflow, stream volumes may be increased throughout the year, despite the decrease in base flow (Spieker 1969). In parts of Denmark (Yenson and Venon 1973, cited by Kuprianov 1977), the disposal of urban sewerage and runoff into the ocean, coupled with increased groundwater demand, has resulted in a decrease, rather than an increase, in annual streamflow. However, most urban areas produce an increase in streamflow, at least during the wettest season. ASCE (1975) considered it important to distinguish between intrabasin and interbasin transfers of water. Storm sewerage of an area (an example of intrabasin transfer) may not change the net water balance of an area although it may result in a redistribution of water from infiltration to runoff. Waananen (1969) considered that the water yield of urban areas increases as a result of increased precipitation. However, Spieker (1969) contended that, while urbanization changes the distribution and quality of water in time and place, the quantity of water remains essentially unchanged.

James (1965) drew attention to the enhancement of the effects of urbanization on runoff during dry months and dry years.
Lvovich and Chernishov (1977) found that in central parts of Kursk (USSR), rainfalls as low as 1 mm produced runoff, whereas 5 mm was required in suburban areas. Various workers (e.g. Espey et al. 1969, cited by Lazaro 1979; Hall 1977) have advocated the correlation of variables controlling the shape of the unit hydrograph with catchment characteristics as a means of quantifying the effects of urbanization on surface runoff. Hall (1977) found the percentage of impervious area an insufficient measure for correlation with unit hydrograph variables, and suggested the inclusion of variables describing channel conditions and the distribution of urban areas within the catchment. ASCE (1975) have noted that the spatial distribution of pervious and impervious sectors within urban catchments is important in the production of runoff. If the runoff from impervious areas is discharged onto adjacent pervious areas, it is possible for a high percentage of a catchment to be impervious yet not produce the expected volume of runoff.

The only study of rainfall and runoff in Perth that has been published is an analysis of rainfall and antecedent precipitation index (API) by Davies (1975). The study concluded that rainfall was almost completely random, with the best correlation being obtained for consecutive days (r = 0.26). This correlation reflects the frontal type of rain experienced in Perth, whereby rain continues for several days, followed by several dry days. Davies found that the API for Perth was most closely associated with medium intensity rain. It was postulated by Davies that light rain had an equal chance of falling on either a wet or dry catchment and therefore there was little association between API and light rainfall. Medium intensity rainfall, however, falls mainly in winter and after long wet periods when catchments are wet. The value of the API was found to decrease for higher intensity rainfall, as these storms can also occur during summer periods when catchments are relatively dry.

The large variation in values shown in Table 2.2 partially
reflects the geology (especially soil infiltration capacities) of the areas undergoing urbanization. As a consequence of the very high permeability of the eolian sands in the Perth area, the only areas likely to produce soil runoff are areas of fine textured soils near rivers and wetlands, areas of non-wetting sands (see Roberts and Carbon 1971) and areas receiving runoff from adjacent shedding areas (McFarlane 1981). For Perth then, urbanization produces an effectively infinite increase in the unit hydrograph and in the values shown in Table 2.2.

In Perth, the amount of runoff needing disposal is lessened by the almost complete absence of soil runoff. Also, laws prohibit the direction of roof runoff to the road stormwater system in all parts of Perth except the Central Business District. Perth also has a separate wastewater system. Consequently, the amount of water requiring disposal in the public stormwater system is less in Perth than in most other cities. In districts removed from the ocean or estuary, almost all road runoff is discharged into absorption basins, which may either be above the water table or are wetlands. These basins may have flood mitigation drains which connect them to the ocean or estuary. That proportion of urban runoff in Perth that is not diverted to the groundwater is discharged directly at ocean or estuarine outfalls. Therefore, Perth does not experience the stream flooding that occurs in many other cities, particularly those sited on alluvial flood plains.

2.1.3 Groundwater

Savini and Kammerer (1961) noted that the decrease in soil infiltration, which occurs following the creation of shedding surfaces in urban areas, is usually accompanied by an equivalent decrease in soil evaporation, transpiration and groundwater recharge. However, the authors did note one case where soil infiltration was increased as a result of lawn and garden watering with imported (interbasinal) waters. Leonard (1981) has observed that garden watering is a highly inefficient source of recharge as
much of this water is lost to evaporation and some is also lost to drainage. Hamilton and Owens (1972) reported a rise in groundwater levels in part of Denver, Colorado, following the construction of a hydraulic barrier to groundwater flow, and also following increased recharge from lawn watering. Thomas (1951, cited by ASCE 1975) suggested that infiltration and groundwater recharge would be equivalent to that occurring before urbanization, in areas where roof drainage is discharged onto the ground.

However, such cases are unusual and groundwater levels are generally reported to decline in urban areas as a result of decreased recharge and increased groundwater extraction. In some cases, the decline in groundwater levels is accompanied by an increase in soil compaction (e.g. Lindh 1974) and salt water intrusion (e.g. Franke and McClymonds 1972).

Savini and Kammerer (1961) identified four urbanization stages (pre-, early-, middle- and late-urban) and noted various hydrologic effects at each stage. Table 2.3 below summarises the effects acting on the water table at each of the changes in land- and water-use as identified by Savini and Kammerer.

Table 2.3 draws attention to recharge of groundwaters from over-loaded and leaking sewerage systems. Groundwater can also infiltrate the sewerage system as indicated by the American Public Works Association (1970, cited in ASCE 1975) which has estimated that groundwater infiltration averages up to 15 per cent of total sewerage flows. Water can also recharge groundwaters from leaking reticulation pipes. Table 2.3 neglects recharge of interbasinal waters as a result of garden irrigation and also recharge from roof runoff, as noted by Thomas (1951). Seaburn (1970) and Aronson (1978) have detailed the practice of returning stormwater runoff to the groundwater through more than 2,500 specially constructed infiltration basins in urban areas in Long Island, New York. These basins augment recharge to the groundwater system and
Table 2.3

Effects on the water table during changes in land- and water-use associated with urbanization

(summarised from Savini and Kammerer 1961)

<table>
<thead>
<tr>
<th>Change in land- or water-use</th>
<th>Effect on water table</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Pre-urban to early-urban transition</strong></td>
<td></td>
</tr>
<tr>
<td>Removal of trees and vegetation</td>
<td>raised (effect inferred)</td>
</tr>
<tr>
<td>Drilling of wells</td>
<td>lowered</td>
</tr>
<tr>
<td>Construction of septic tanks and sanitary drains</td>
<td>raised</td>
</tr>
<tr>
<td><strong>Early-urban to middle-urban transition</strong></td>
<td></td>
</tr>
<tr>
<td>Construction of houses and culverts, paving of streets</td>
<td>lowered (due to decreased infiltration)</td>
</tr>
<tr>
<td>Abandonment of some shallow wells</td>
<td>raised</td>
</tr>
<tr>
<td><strong>Middle-urban to late-urban transition</strong></td>
<td></td>
</tr>
<tr>
<td>Urbanization of area completed by addition of more buildings and streets</td>
<td>lowered</td>
</tr>
<tr>
<td>Abandonment of remaining shallow wells because of pollution</td>
<td>raised</td>
</tr>
<tr>
<td>Construction of sanitary sewerage system and treatment plants</td>
<td></td>
</tr>
<tr>
<td>Increased water use</td>
<td>lowered</td>
</tr>
<tr>
<td>Wastewater reclamation and utilization</td>
<td>raised</td>
</tr>
</tbody>
</table>
also eliminate the need for long, costly sewers. The effects on groundwater levels of sewing areas previously serviced by septic tanks on Long Island was investigated by Sulam (1979). It was possible to attribute a 2.6 metre decline in levels to seewing in the glacial sand and gravel aquifer. Massing (1976) pointed out that water levels in urban areas could also be temporarily or permanently affected by drainage schemes.

It is evident from the above that a number of factors affect groundwater levels at different stages of urbanization. Differences between cities can be partly attributed to differences in climate and geology. It was seen that soil infiltration (and groundwater recharge) could be increased in cities with a dry season which requires a period of irrigation of gardens and recreational areas, using interbasinal waters (Thomas 1951; Hamilton and Owens 1972; Leonard 1981). Geology affects the amount of infiltration that can occur in urban areas. Malmquist and Svensson (1977a) detailed a water balance inventory to Goteborg, Sweden, where soil infiltration could be ignored, presumably due to the impermeable nature of the soil. At the other extreme, Thomas (1951) and Hamilton and Owens (1972) give details of roof runoff being added directly to the soil without flooding. The infiltration basins in Long Island (Aronson 1978) are sited at areas with particularly high infiltration capacities within the urban area.

In Perth, groundwater levels have been noted to respond to some of the land- and water-use changes detailed in Table 2.3. Burton (1976) noted that rises of about one metre had occurred in groundwater levels in the Perth area during the previous 20 years, as a direct consequence of clearing native vegetation and subsequent urbanization. Evidence given of such rises included hydrograph analysis, the existence of drowned fence lines, examination of old maps and the experience of old residents. Whincup and o'Driscoll (1979) reviewed groundwater levels in the Perth Metropolitan Area and noted areas where water levels were
rising as a result of urban development, despite a period of well-below-average rainfall. Factors responsible for the rise were thought to be reduced transpirational losses, recharge from shedding-area runoff and the addition of imported waters from septic tanks and garden watering. Burton (1976) and Pollett (1981) have identified areas of Perth where groundwater levels have declined as a result of pumping and drainage. Ho (1981) outlined rises in groundwater levels beneath wastewater-renovation basins. There is also some indication that the sewerage of areas of Perth which had previously been serviced by septic tanks, has resulted in a lowering of the water table. Pumps in a drainage station at Perry Lakes had operated for an average of about 1000 hours per year before sewerage in 1974 but only about 70 hours per year afterwards (MWA, unpubl. data). The post-sewerage period has been below average in rainfall, however, and therefore the decreased pumping may not be a direct result of sewerage.

Groundwater levels in Perth appear to respond readily to changes in land- and water-use. By measuring all factors affecting groundwater levels, through carrying out water-balance inventories, the effects of any one factor (e.g. pumping) can be assessed. An alternative method is to compare areas where all factors are the same, except the one under investigation (e.g. Sulam 1979). However, the large number of factors which affect groundwater levels makes the delineation of suitable 'control' areas difficult.
2.2 URBAN EFFECTS ON WATER QUALITY

Cordery (1976a) observed that the limited amount of data then available on the effects of urbanization on water quality, suggested that the effects were very different for different localities. These urban effects have received increased attention in recent years (e.g. Whipple (ed.) 1975; UNESCO 1977), particularly with respect to stream water quality. Much less work has been done on urban effects on groundwater quality. Studies on both stream water and groundwater quality have tended to support Cordery’s observation about variability in the urban effects. For example, Lager and Smith (1974, cited by Gutteridge et al. 1981) found that the concentration of pollutants in stormwaters varied by factors of ten or more from area to area, within a single storm, and also from storm to storm. Kaufmann (1977) noted that the quality of the top 15 m of groundwater in the Las Vegas Valley was highly irregular, being influenced mostly by waste disposal.

This review provides a summary of the urban effects on water quality, emphasis being given to effects noted on groundwater and also, in view of the variability noted above, to local examples wherever possible. Effects on quality are discussed by source (i.e. sewage, road runoff, sanitary landfills and garden and recreational areas). The effects of industry and liquid waste disposal sites are not considered as they do not occur in the urban areas that were studied. Saline intrusion was considered by Savini and Kammerer (1961) to be one of the most common types of contamination in urban areas. It is also of concern in the Perth area, but no investigation was made of this process in the present study.

2.2.1 Sewage

Many studies (e.g. Soderlund and Lehtinen 1973; Lagrega and
Keenan 1975; Neil 1975; Radziul et al. 1975; Ragan and Dietemann 1975) have identified overflows of combined sewerage systems during rain storms to be a major source of surface water pollution in urban areas. Such observations have resulted in efforts to separate urban wastewater systems in some countries (e.g. Sweden - Carlsson and Falk 1977). While separate wastewater systems are now commonly installed in new urban areas (Ragan and Dietemann 1975), combined systems are still very common in many countries (e.g. most German cities - Massing 1976; 60% of public wastewater systems in the USA - McPherson 1974a).

Dion (1972) could find no appreciable changes to groundwater quality in shallow aquifers in an area undergoing urbanization in Idaho, USA. No faecal bacteria could be detected in the groundwater despite the abundance of septic tanks. However, Nightingale (1970) recorded a 58 per cent increase in nitrate levels in groundwaters in an urban area in California over an 18 year period. Reductions in the rate of increase in nitrates could be attributed to decreased use of septic systems. Kimmel (1972) also attributed high, historic nitrate values in groundwaters in Kings County, Long Island, to septic tank and cesspool effluents. The continuation of high values following sewerage suggested to Kimmel that leakage from sewers was a principal source of nitrate.

Eisen and Anderson (1979) considered increases in chloride and sulphate to be the principal products of urbanization which alter groundwater chemistry in the USA. The authors noted that similar increases in these anions have been observed in other cities by Schicht (1977) and Long and Saleem (1974). The fact that the chloride and sulphate were sometimes associated with ammonium and faecal bacteria, led Eisen and Anderson (1979) to suspect sewage as the source of contamination, although a contribution to chloride levels was thought to be made by salt spread on roads to assist snow clearance. Fusillo (1981) has found the groundwater in an urbanized basin in the USA to be of the magnesium bicarbonate type while surrounding, non-urban areas
were of the sodium-chloride-sulphate type. This effect is
directly contrary to that noted by Eisen and Anderson (1979).

As mentioned previously, Perth has a separate wastewater
system and therefore overloading of treatment plants during storms
does not occur. However, in parts of Perth, winter flows in
sewerage pipes can be three or four times greater than summer
flows, as a result of leakage into the sewers when ground water
levels are high (Robinson, pers. comm.). Further evidence of
groundwater infiltration into sewerage pipes comes from comparing
fluoride levels in public water supplies with those in the
sewerage system (Kavanagh, pers. comm.). As groundwaters have
only about 10 per cent of the fluoride concentration of public
supply waters, it has been possible to estimate the proportion of
groundwater in sewage. Up to a third of wastewater entering
treatment plants is thought to have originated as groundwater in
some areas.

In areas where groundwater infiltration of sewerage pipes is
known to take place, leakage of sewage into groundwater could be
assumed to occur when the pipes are above the water table. No
measurement of leakages has been made as the amount leaking from
pipes is considered to be negligible in comparison to the amount
flowing in the pipes (Robinson, pers. comm.), and input from
individual premises is difficult to measure. For several reasons,
the amount of groundwater leaking into the sewerage pipes is
considered to be greater than the amount leaking out (Kavanagh,
pers. comm.). As depths of sewage in the pipe are usually small,
there is a low head to drive the outward leakage, in comparison to
the groundwater head that drives infiltration when the pipes are
below the water table. In addition, the length of joint through
which the leakage takes place is less for sewage than for
groundwaters. Also the joints at the bottom of the pipe are often
covered with sediment, further inhibiting the leakage of sewage.
Sewage is therefore likely to enter the groundwater from sewerage
pipes only where large failures occur in pipe joints (e.g. where
tree roots have penetrated).

About 45 per cent of Perth is serviced by septic tanks (Houghton 1979) and studies of their effect on groundwater quality have been carried out by Whelan et al. (1979), Parker et al. (1979), and Whelan and Parker (1981). These studies have shown that the sandy soils in the Perth area quickly lose their ability to adsorb phosphate, and in many localities both nitrogen and phosphate enter the groundwater in the same concentration as they were in the effluent. Coliform bacteria have been found to reach the groundwater when septic tanks are close to the water table (Whelan and Parker 1981). The experimental areas studied in this project are sewered and therefore septic tank additions will be minor (household connection to the sewerage system is not compulsory).

Outside the studied areas, secondary treated wastewater and/or sewage is entering the groundwater in the metropolitan area at recharge basins (Mathew et al. 1982), treatment plants (Government Chemical Laboratories-GCL 1979) and liquid waste disposal sites (GCL 1980). Plumes of contaminated water from liquid- and solid- waste disposal sites have been found not to extend more than several hundred metres from the point of disposal in most investigations (GCL 1981), presumably due to dispersion and dilution of the contaminants. Nutrients in septic tank effluent also seem to be rapidly dispersed in the groundwater (Whelan and Parker 1981). Nitrate levels in the top 15 cm of groundwater in septic tank areas were found by GCL (1972) to be greater than 40 mg/L in May, but to be considerably lower in June and October following dilution by groundwater recharge during the winter rains.

2.2.2 Road runoff

As treatment of domestic and industrial wastes is now quite extensive in many countries, emphasis has shifted to the quality
of stormwater runoff (Colston and Tafuri 1975; Feuerstein and Friedland 1975; Randall et al. 1975; Massing 1976). The high cost of treating large volumes of water in a short period (or constructing large storage facilities) has led to ideas of identifying, diverting and treating the poor-quality stormwater that first comes off the roads (e.g. Angino et al. 1972; North Carolina Department of Natural Resources and Community Development - NCDNR & CD 1979). Cordery (1976b, 1977) and Cullen et al. (1978) recognised that many of the pollutants in road runoff waters were associated with the suspended solids (SS) or non-filterable residue fraction and have examined the practice of allowing pollutants to settle out as a means of improving water quality prior to its entry into lakes and rivers. Cordery (1976b) found that settling had little or no effect on nitrate and ammonia concentrations but that all other constituents were significantly reduced. Buckney (1979) found that suspended materials carried much of the nutrient load in a river system in South Australia.

Various studies (e.g. Weibel et al. 1964; Soderlund and Lehtinen 1973; Colston and Tafuri 1975; Cordery 1976b; Gutteridge et al. 1981) have compared the quality of stormwater with the quality of raw and treated sewage. Generally, stormwaters equal or exceed raw sewage in SS and metals, equal or exceed secondary treated sewage in biological oxygen demand (BOD) and contain 5 per cent to 40 per cent the nutrient concentrations of secondary treated sewage.

Table 2.4 details the mean and range of values of pollutants in stormwaters from selected catchments in Sydney (Cordery 1977), Melbourne (Gutteridge et al. 1981) and Perth (Bishaw 1980) and gives the quality of raw sewage and secondary treated effluent for comparison. Single residential catchments were compared in each city, but in Perth only about 20 per cent of the catchment (Ballajura Estates) contained completed, or partially-completed, dwellings.
Table 2.4

Comparison of the quality of urban stormwater with sewage and secondary treated effluent for 3 Australian residential catchments
(All values are mg/L except coliforms)

<table>
<thead>
<tr>
<th></th>
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</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Mean</td>
<td>Range</td>
<td>Mean</td>
<td>Range</td>
<td>Mean</td>
</tr>
<tr>
<td>SS</td>
<td>236</td>
<td>30-1400</td>
<td>233</td>
<td>8-3600</td>
<td>65</td>
</tr>
<tr>
<td>TDS†</td>
<td>118</td>
<td>30-650</td>
<td>247</td>
<td>85-980</td>
<td>11</td>
</tr>
<tr>
<td>BOD₅‡</td>
<td>18</td>
<td>2.5-7.1</td>
<td>11</td>
<td>7-74</td>
<td>25</td>
</tr>
<tr>
<td>TOC</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>NH₄</td>
<td>1.92</td>
<td>0.37-14.0</td>
<td>0.33</td>
<td>0.009-6.6</td>
<td>0.097</td>
</tr>
<tr>
<td>NO₃$^-$+NO₂$^-$N</td>
<td>0.96</td>
<td>0.51-1.60</td>
<td>1.8</td>
<td>0.04-8.8</td>
<td>0.081</td>
</tr>
<tr>
<td>Kjeldahl-N</td>
<td>2.5</td>
<td>0.09-38</td>
<td>0.991</td>
<td>0.450-2.0</td>
<td></td>
</tr>
<tr>
<td>Total-P</td>
<td>1.60</td>
<td>0.63-5.1</td>
<td>0.6</td>
<td>0.074-13</td>
<td>0.14</td>
</tr>
<tr>
<td>Ortho-P</td>
<td>0.15</td>
<td>0.08-2.2</td>
<td>0.03</td>
<td>0.002-0.166</td>
<td></td>
</tr>
<tr>
<td>Cu</td>
<td>0.029</td>
<td>0.009-0.32</td>
<td></td>
<td></td>
<td>0.37</td>
</tr>
<tr>
<td>Zn</td>
<td>0.496</td>
<td>0.01-2.75</td>
<td>0.19</td>
<td>0.29</td>
<td>0.19-0.24</td>
</tr>
<tr>
<td>Pb</td>
<td>0.206</td>
<td>0.03-1.37</td>
<td>0.02</td>
<td>0.04</td>
<td>0.02-0.04</td>
</tr>
<tr>
<td>Ni</td>
<td>0.026</td>
<td>0.01-0.04</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Cr</td>
<td>0.020</td>
<td>0.02-0.03</td>
<td></td>
<td></td>
<td>0.01</td>
</tr>
<tr>
<td>Faecal coliforms</td>
<td>41</td>
<td>9-93</td>
<td>35</td>
<td>7-176</td>
<td></td>
</tr>
</tbody>
</table>

† Total dissolved solids
‡ Five day biological oxygen demand
Bishaw (1980) compared values from the Ballajura Estates catchment with four catchments in the USA and concluded that phosphorus concentrations were markedly lower than in the USA studies, whereas all other parameters fell within the ranges listed. Table 2.4 shows that all values (SS and nutrients) for the Perth samples are lower than those for Sydney and Melbourne. Bishaw found that NO$_3$ + NO$_2$ values in particular exhibited a strong 'first flush' effect (i.e. attained high values early in the runoff event). The concentration of NO$_3$ + NO$_2$ (and its proportion of total N) was found to decrease as the winter progressed, indicating that NO$_3$ + NO$_2$ was being flushed from the catchment. The concentration of NO$_3$ + NO$_2$ in the rainfall was also found to decrease as the winter progressed. The relative concentrations of nutrients in rainfall compared with stormwaters (59% for NO$_3$ + NO$_2$, 86% for NH$_4$ and 38% for orthophosphate) obtained by Bishaw, indicate that the atmosphere is an important source of nutrients in this catchment. Newman and Bishaw (1983) considered that organic matter in soil, deposited on impervious surfaces, was also a major source of nutrients in the stormwaters.

The only other study of the quality of road runoff in the Perth area prior to the present study was carried out by Congdon (1979) who reported the following concentrations of nutrients in waters from a partially developed catchment: NH$_4$ - 0.022 mg/L, NO$_3$ + NO$_2$ - 0.028 mg/L, Kjeldahl N - 0.74 mg/L, total P - 0.057 mg/L and ortho P - 0.023 mg/L. These values are also appreciably lower than those shown for Sydney and Melbourne in Table 2.4. No study had been made of stormwater quality in a fully urbanized catchment in Perth prior to this study.

Table 2.4 does not contain details of hydrocarbons (oils, grease, etc.) in the stormwaters as these are difficult to determine accurately due to their uneven distribution (Cordery 1977). Matis (1971) noted that it is possible to smell and taste hydrocarbon fuels at concentrations below the level of detection
for gas chromatographs and advocated tasting of waters as the best confirmational technique. Cordery (1977) noted that oil slicks occurred during all the floods monitored in the three drains that were investigated in Sydney. However, Ellis (1977) found that substantial amounts of rubber, bitumen, grease and oils commonly occur as capsules of hydrocarbon coating an inorganic nucleus, and that the distribution of volatile solids in stormwaters showed similar changes in value as did suspended solids. Matis (1971) noted that many hydrocarbon fuels do not deteriorate in the groundwater system, although Williams and Wilder (1971) observed that bacterial breakdown, by normally dormant bacteria, can be significant if there are sufficient oxygen, mineral nutrients and fuel-water interfaces in the system.

Table 2.5 lists the most common sources of pollutants in stormwater runoff as an aid to understanding the reasons for the large variations in pollutant concentrations. Walling and Gregory (1970) found that SS concentrations increased 2 to 10 times (and up to 100 times) the SS concentrations found in undisturbed catchments in Exeter, UK. Fusillo (1981) also reported a four-fold increase in sediment discharge in an area in the USA that was undergoing partial urbanization (14% of the total area). In Maryland, USA, the amount of sediment produced during construction was found by Wolman and Schick (1967) to be related both to the intensity of construction and to the drainage area involved. The smaller the drainage area, the greater the relative increase in sediment.

Ellis (1977) examined the removal of particulate solids from roads by street sweeping. Only 15-20 per cent of the < 0.06 mm size fraction were removed by sweeping. Although this fraction comprised only 4-8 per cent of the total suspended solids, Ellis (1977) found that it accounted for 25 per cent of BOD, 30-50 per cent of algal nutrients, 30 per cent of heavy metals, and 10 per cent of total coliforms.
<table>
<thead>
<tr>
<th>Parameter</th>
<th>Sources (major source = *)</th>
<th>Principal Reference</th>
</tr>
</thead>
<tbody>
<tr>
<td>SS</td>
<td>*Construction sites</td>
<td>Walling and Gregory (1970)</td>
</tr>
<tr>
<td></td>
<td>Plant material and litter</td>
<td>Ellis (1977)</td>
</tr>
<tr>
<td>TDS</td>
<td>*Atmospheric (dryfall and</td>
<td>Malmquist and</td>
</tr>
<tr>
<td></td>
<td>rainfall)</td>
<td>Svensson (1977b)</td>
</tr>
<tr>
<td></td>
<td>*Road salt</td>
<td>Skakalski (1977)</td>
</tr>
<tr>
<td>Nitrogen</td>
<td>*Atmospheric (combustion</td>
<td>Hoeff et al. (1972)</td>
</tr>
<tr>
<td></td>
<td>processes)</td>
<td>Kelling and Peterson (1975)</td>
</tr>
<tr>
<td></td>
<td>Fertilizers</td>
<td></td>
</tr>
<tr>
<td>Phosphorus</td>
<td>*Plant material and litter</td>
<td>Waller (1977)</td>
</tr>
<tr>
<td></td>
<td>Animal faeces</td>
<td>Kelling and Peterson (1975)</td>
</tr>
<tr>
<td></td>
<td>Fertilizers</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Atmospheric (dryfall)</td>
<td>Barkdoll et al. (1977)</td>
</tr>
<tr>
<td>BOD$_5$/TOC</td>
<td>*Plant material and litter</td>
<td>Berg et al. (1977)</td>
</tr>
<tr>
<td>Copper</td>
<td>*Corrosion</td>
<td>Malmquist and Svensson (1977b)</td>
</tr>
<tr>
<td></td>
<td>Atmospheric</td>
<td>Horkeby and Malmquist (1977)</td>
</tr>
<tr>
<td>Zinc</td>
<td>*Atmospheric</td>
<td>Malmquist and Svensson (1977b)</td>
</tr>
<tr>
<td></td>
<td>Corrosion</td>
<td>Horkeby and Malmquist (1977)</td>
</tr>
<tr>
<td>Lead</td>
<td>*Atmospheric (automobiles)</td>
<td>Bryan (1974)</td>
</tr>
<tr>
<td>Nickel</td>
<td>*Atmospheric (oil combustion)</td>
<td>Horkeby and Malmquist (1977)</td>
</tr>
<tr>
<td>Chromium</td>
<td>*Atmospheric (industry)</td>
<td>Horkeby and Malmquist (1977)</td>
</tr>
<tr>
<td>Hydrocarbons</td>
<td>Spills</td>
<td>Matis (1971)</td>
</tr>
<tr>
<td></td>
<td>*Vehicle exhausts</td>
<td></td>
</tr>
<tr>
<td>Faecal coliform bacteria</td>
<td>Animals (e.g. birds,dogs)</td>
<td>Cordery (1976b)</td>
</tr>
<tr>
<td></td>
<td>Septic tank overflows,sewers</td>
<td>NCDNR &amp; CD (1979)</td>
</tr>
</tbody>
</table>
Long and Saleem (1974) attributed a major proportion of the 843 per cent increase in chloride levels in an urban area in the USA to road salt additions. As temperatures in Perth have never been lower than 10°C since records commenced in 1830 (Vollprecht 1969), road salt has not been required to de-ice roads in the Perth area and any increase in chloride must be from some other source.

Wood (1975) and Cullen et al. (1978) have shown that nutrient concentrations in Australian rainfall are at the lower end of the range of values found in rainfall in the USA. The few analyses by Bishaw (1980) show nutrient concentrations in rain to be low in the Ballajura Estates area also. It is significant that SS levels were found to be so low (see Table 2.4) in the Ballajura stormwaters considering that 8.6 per cent of the catchment area was comprised of houses under construction (Bishaw 1980). The lack of runoff from the sandy soils in the Ballajura area is probably the cause of these low SS values. The same feature would also tend to decrease the importance of fertilizers in contributing to nutrient loads in road runoff. There is some suggestion therefore that the quality of stormwater may be better in Perth than that found elsewhere.

2.2.3 Sanitary landfills

In a critical review of the groundwater pollution potential of sanitary landfills, Zanoni (1972) concluded that there were few case histories of serious, or even troublesome, contamination of groundwater which could be attributed to leachates from landfills. Zanoni attributed this benignity to the ability of most soils to remove pollutants and to the dilution of any remaining pollutants in the groundwater. The only cases where landfills may be a threat to groundwater quality were considered by Zanoni (1972) to be those located in coarse sands and gravels, those in fissured materials (e.g. limestone, shale) and those in which the landfill temporarily, or permanently, resides below the water table. Apgar
and Langmuir (1971) noted that saturation of refuse by groundwater allows the ready solution of inorganic materials (e.g. Fe, Mn) and also results in the production of highly undesirable organic and inorganic substances. Golwer et al. (1977) investigated three waste-deposit sites situated in areas underlain by porous, non-consolidated aquifers and found that the groundwater regained its normal condition after a few, or several hundred, metres of flow as a result of dilution and self-purification. Kaufmann (1970) found that the increase in dissolved chemical species in groundwaters adjacent to landfills was high but restricted to local areas. However, the effects of leachate from a refuse dump at Jaipur, India, was noticeable up to 450 m from the site (Olaninya and Saxena 1977).

Hughes (1967, cited by Zanoni 1972) stressed that it is dangerous to over-generalise in extrapolating the findings of one area to another, as climatic, hydrologic and geologic factors strongly influence the production and spread of contaminants from landfill sites. Fortunately, a comprehensive study of a landfill site has been carried out in the Perth area by Bestow (1977) and other, smaller investigations, have also been made (GCL 1976).

Bestow (1977) found that water within the Hertha Road landfill was polluted with Cl\(^-\), HCO\(_3\), Na\(^+\), K\(^+\), Fe\(^{2+}\), NH\(_4\), orthophosphate, phenol and surfactants. However, the presence of peat at the base of the landfill, together with dilution by the aquifer, resulted in the reduction of all the above pollutants to acceptable levels outside the landfill, with the exception of NH\(_4\) and phenols. GCL (1978) noted that the levels of inorganic nitrogen in aquifers immediately below landfill sites in the Perth area are generally of the order of 100 mg/L, which is 300 times greater than eutrophic levels (0.3 mg/L). Because of this feature, GCL (1978) recommended against the location of rubbish disposal sites adjacent to lakes and rivers.
2.2.4 Garden and recreational areas

The main effects of garden and recreational areas on water quality in urban areas are likely to be on salinity (through evaporation of irrigated waters) and the leaching of nutrients, pesticides and herbicides. No studies have yet been made on the effects of irrigation on groundwater salinity in the Perth area. However Bestow (1970) and Allen (1976) have noted saline plumes downgradient of wetland areas as a result of evaporative concentration.

The poor ability of sands in the Perth area to remove nitrogen and phosphorus from septic tank effluent has already been discussed in Section 2.2.1. Hingston (1959) and Ozanne et al. (1961) have shown that phosphate is readily leached through the sandy soils of the Swan Coastal Plain. Ozanne et al. found that soils containing less than 100 ppm native phosphorus or giving less than 6 per cent ignition loss showed the highest losses due to leaching. The Karrakatta soils of the Spearwood Dune system have less than 3 per cent ignition loss (McArthur and Betternay 1974) and would therefore be expected to retain little phosphate. GCL (1981) reported a case of sulphate levels of up to 300 mg/L in the groundwater in a market garden area on the outskirts of Perth. The most likely source of the sulphate was thought to be superphosphate, the sulphate not taken up by the plants presumably leaching to the groundwater.

Bestow (1977) reported trace amounts of pesticides (dieldrin, DDT and BHC) in the groundwaters in the Hertha Road area in Perth, which probably resulted from leaching from surface soils. Newman and Marks (1980) have shown that the sandy soils in the Perth area have a very limited capacity to immobilize heavy metals. Muller (1979) has shown that soils in urban areas may contain up to 10 mg/L lead as a result of atmospheric fallout of particulates from vehicle exhausts. Muller suggested that this lead may enter groundwaters where soils have a low exchange
capacity. Boujos (1974) found that lead concentrations in soil adjacent to a major road in Perth reached 80 mg/L, but that this concentration had decreased to 9 mg/L at a depth of 5 cm and to only 0.4 mg/L at 13 cm depth. Thus, lead appeared to be relatively immobile, despite the low cation exchange capacity of the sand.

2.2.5 Water quality in relation to use

In discussing urban effects on water quality, consideration must be given to the present, and possible future, uses of the water. In the urban areas studied in this project (and in most other parts of Perth) groundwater is used only for irrigation of gardens and recreational areas. Road runoff and groundwater also maintain water levels in urban wetlands. In other, smaller, parts of Perth, groundwater is used for industrial, market-garden and public water supply purposes. In some areas, consideration also has to be given to the quality of water flowing into estuaries (e.g. Jack 1977a; Taylor 1980).

The water quality parameters that are likely to affect the groundwaters in the studied areas are those of salinity and iron. Salinity has already been mentioned in relation to evaporation of irrigated waters and wetlands. Iron in groundwaters in the Perth area causes widespread brown staining of buildings, fences, paths and vegetation. GCL (1978) found that iron levels as low as 0.4 mg/L have produced noticeable staining on asbestos cement fences and other alkaline materials, within several weeks. It has been observed by Jack (1977b) that iron levels in excess of 0.5 mg/L caused interference tints on smooth- and glossy-leaved plants. Such a feature is known to be detrimental to shrubs and small trees (Elliot et al. 1981). In addition, Oropeza-Monterrubio (1979) has detailed a case where high iron levels may be partially responsible for turbidity in an urban lake in the Perth area. There are some indications that changes in groundwater chemistry, as a result of urbanization, may cause increased iron levels in
groundwater (Martin 1980; McFarlane 1981). Martin found that iron concentrations in groundwater were principally related to geochemical abundance of iron in the sands and to the oxidation state of the waters. Organic matter and bacteria were implicated in the dissolution of iron. The practice of adding road runoff waters to wetlands was thought by Martin (1980) to have the potential to increase iron concentrations in groundwaters.

The water quality parameters that are likely to affect the utility of surface waters (wetlands) in the study areas are nutrients, biological oxygen demand, heavy metals, and oils. In a study of the impact of urbanization on the nutrient budget of a lake in Wisconsin, USA, Watson et al. (1981) found the main effect to be an increase in phosphate loads from runoff. Bishaw (1980) found that road runoff provided a much larger total input of nutrients than rainfall to a wetland in the Ballajura Estates area of Perth. It was not possible to assess the amount of nutrient inflow in groundwater seepage for this wetland. The stormwaters in the Ballajura Estates area are also likely to contain heavy metals, BOD and oils. However no analyses have been made for these pollutants in Perth stormwaters.

Given the poor ability of the eolian sands of Perth to remove pollutants, it is surprising that few cases of pollution (other than industrial) have resulted in a decrease in the utility of the groundwater for irrigation and wetland maintenance. Bores have been polluted by saline intrusion in some areas adjacent to the ocean and estuary. Whether the intrusion was due principally to bore use or to a coincident period of dry years has not been determined. High and rising sulphate levels have also been recorded from market garden areas (Martin 1980; GCL 1981; Hirschberg 1981) which have caused concern. In addition, leaching of fertilizers from Bassendean sands into surface waters has been implicated in the eutrophication of surface waters south of Perth (Birch 1982). While some residents have ceased to use groundwater for garden irrigation due to iron staining or because of odours
from gases (e.g. \( \text{H}_2\text{S} \)), these are naturally occurring chemicals within the groundwaters. As urbanization appears to produce significant physical changes to the groundwater in Perth, it is possible that accompanying changes may be made also to its quality which may affect its utility.
CHAPTER 3

CLIMATE, STRATIGRAPHY AND HYDROGEOLOGY
OF THE STUDY AREA

3.1 CLIMATE

3.1.1 General

The climate of Perth is Mediterranean in type with a mild, wet winter and a hot, dry summer (Vollprecht 1969). In the Koppen classification system, Perth is classified as Csa (Gentilli 1971). Csa climates are identified as being warm with hot, dry summers. Table 3.1 provides a summary of the main climatic parameters for each month of an average year.

Things to note from Table 3.1 are:

(i) Over 90 per cent of the rainfall occurs in the seven-month period between April and October.
(ii) Rainfall exceeds pan evaporation for only four months of the year (May to August).
(iii) The annual pan evaporation rate is more than twice the annual rainfall rate.
(iv) Global radiation and hours of sunshine are at a maximum in December, whereas maximum daily pan evaporation occurs in January, and highest maximum temperatures occur in February. Vollprecht (1969) attributed the six-week lag between radiation and temperature to the continued heating of surfaces after the summer solstice (incoming—exceeds outgoing—radiation until after February).
(v) On average, wind speeds are higher in summer than in winter. This is due both to the prevalence of an afternoon sea breeze and to a higher average pressure gradient during this part of the year (Vollprecht 1969). Wind speed reaches a maximum in
Table 3.1

Average climatic data for Perth

(Sources: Bureau of Meteorology and Gentilli 1971)

<table>
<thead>
<tr>
<th>Month</th>
<th>Rainfall (mm)</th>
<th>Class-A pan evaporation* (mm)</th>
<th>Mean maximum temperature ($^\circ$C)</th>
<th>Mean minimum temperature ($^\circ$C)</th>
<th>Mean daily global radiation (cal/cm$^2$**)</th>
<th>Mean hours of daily sunshine</th>
<th>Mean wind speed (km/h)</th>
</tr>
</thead>
<tbody>
<tr>
<td>January</td>
<td>8</td>
<td>270</td>
<td>29.6</td>
<td>17.7</td>
<td>650</td>
<td>10.5</td>
<td>18.8</td>
</tr>
<tr>
<td>February</td>
<td>12</td>
<td>232</td>
<td>31.1</td>
<td>19.6</td>
<td>603</td>
<td>10.1</td>
<td>18.2</td>
</tr>
<tr>
<td>March</td>
<td>20</td>
<td>203</td>
<td>28.9</td>
<td>17.8</td>
<td>493</td>
<td>9.0</td>
<td>16.7</td>
</tr>
<tr>
<td>April</td>
<td>45</td>
<td>121</td>
<td>24.7</td>
<td>14.6</td>
<td>349</td>
<td>7.4</td>
<td>14.5</td>
</tr>
<tr>
<td>May</td>
<td>124</td>
<td>88</td>
<td>21.5</td>
<td>11.9</td>
<td>258</td>
<td>5.9</td>
<td>13.8</td>
</tr>
<tr>
<td>June</td>
<td>183</td>
<td>65</td>
<td>19.1</td>
<td>10.5</td>
<td>219</td>
<td>4.9</td>
<td>14.3</td>
</tr>
<tr>
<td>July</td>
<td>174</td>
<td>67</td>
<td>18.5</td>
<td>9.8</td>
<td>235</td>
<td>5.3</td>
<td>14.8</td>
</tr>
<tr>
<td>August</td>
<td>137</td>
<td>81</td>
<td>17.9</td>
<td>9.1</td>
<td>321</td>
<td>6.2</td>
<td>15.0</td>
</tr>
<tr>
<td>September</td>
<td>110</td>
<td>110</td>
<td>19.4</td>
<td>10.1</td>
<td>427</td>
<td>7.2</td>
<td>15.4</td>
</tr>
<tr>
<td>October</td>
<td>55</td>
<td>162</td>
<td>21.2</td>
<td>11.5</td>
<td>538</td>
<td>8.3</td>
<td>16.3</td>
</tr>
<tr>
<td>November</td>
<td>21</td>
<td>196</td>
<td>24.6</td>
<td>14.6</td>
<td>633</td>
<td>9.7</td>
<td>17.5</td>
</tr>
<tr>
<td>December</td>
<td>14</td>
<td>250</td>
<td>28.5</td>
<td>17.4</td>
<td>672</td>
<td>10.6</td>
<td>18.3</td>
</tr>
</tbody>
</table>

Annual Average | 873 | 1845 | 23.7 | 13.7 | 449 | 7.9 | 16.1

* Average of 15 years only (1967-81)  
** 1 cal/cm$^2$ = 4148 J/cm$^2$
January, which may help to explain the maximum daily rate of pan evaporation during this month.

Having briefly summarized the mean values of climatic variables in Perth, the next few sections examine the variability in space and time of some of these parameters. Variations during the last four years are examined in particular, as groundwater levels are known to reflect inputs and outputs of water during previous years. How typical (or atypical) the water balance period was is also examined. A knowledge of spatial variability is required to assess the applicability of climatic measurements made at Perth's Bureau of Meteorology to the experimental catchments studied in this project.

3.1.2 Rainfall

Whincup and O'Driscoll (1979) considered the spatial variability of rainfall in the Perth area to be great enough to affect local groundwater levels. Figure 1.1 showed the isohyets of average rainfall for the Perth area. Rainfall increases to the east (due to the orographic influence of the Darling Escarpment) and to the south (due to the increased influence of inter-anticyclonic fronts - Gentilli 1971). While this is the general trend, the spatial distribution of rainfall in any one year can be quite variable as seen in Figure 3.1. For example, the rainfall maximum on the plain occurred at four different stations during 1979-82. A similar observation was made by Swindell (1979) in a study of rainfall from individual storms. Figure 1.1 showed that there is an inflection in the 800 mm isohyet over Perth. However there is not sufficient evidence to decide whether rainfall distribution over Perth is influenced by urban factors, particularly as appreciable variations will result from orographic, maritime and gauge-exposure effects.

Figure 3.2 (a) shows the temporal distribution of Perth's rainfall from 1876 when records commenced. There does not seem to
LEGEND:  • RAINFALL STATION  • PEARCE AIRPORT

600 —— ISOHYET (mm)

(Figure 3.1: Spatial distribution of rainfall over Perth (1979-1982) and location of meteorological stations.
(Source: Bureau of Meteorology)
FIGURE 3.2 (a): Annual rainfall for Perth, 1876-1982. (Source: Bureau of Meteorology)

FIGURE 3.2 (b): Three-point moving average of Perth's rainfall.

FIGURE 3.2 (c): Ten-point moving average of Perth's rainfall.

FIGURE 3.2 (d): Sunspot cycle.

FIGURE 3.2 (e): Autocorrelation of Perth's rainfall. (Small numbers show rankings of all correlations over ± 0.1)
be any strong regular cycle for rainfall, as shown by auto-
correlation (Fig. 3.2 (e)). The highest correlation (0.189)
occurs for a lag of 11 years, the sunspot cycle (Fig. 3.2 (d)).
The correlation between rainfall and the sunspot cycle is not
obvious from Figure 3.2. Figures 3.2 (b) and (c) are 3- and 10-
year centred moving averages of the record. Figure 3.2 (b) shows
that the period 1975-77 was the driest three-year period on
record, while 1977-79 was the second driest (despite rainfall in
1978 being 6 per cent above average). Figure 3.2 (c) shows that
1971-80 was the driest decade on record, being 12 per cent below
average.

Figure 3.3 shows variations in the **distribution of rainfall**
throughout the year for the period 1979-82. Rainfall in 1979 was
36 per cent below average (the fourth driest year on record)
whereas 1980-82 were only 3 to 6 per cent below average. In 1980
and 1981 the distribution of rainfall throughout the year was
average, except for slightly drier summer periods (October to
March), and more pronounced 'breaks' to the rainy season in April
or May. The mean rainfall values for the summer months are
affected by heavy rains which occur in some years. The summer
rainfalls in 1980 and 1981 were closer to the modal values than to
the mean values for these months. In 1982, 13 per cent of the
annual rainfall fell in January (the average being only 1 per
cent) as a result of a cyclonic depression affecting the Perth
area. There was also a record dry period in the autumn of 1982
when 40 days passed without rain. Thus, while 1982 had only a 6
per cent below-average rainfall, the distribution of the rainfall
throughout the year was very atypical.

### 3.1.3 Evaporation

Evaporation records for Perth are very limited. Between
1953 and 1966, evaporation was measured in central Perth using a
below-ground Australian tank evaporimeter. Since 1967,
evaporation has been measured with an above-ground class-A pan
FIGURE 3.3: Distribution of rainfall throughout the year for the period 1979-1982.
(Source: Bureau of Meteorology).

LEGEND
- Actual Rainfall
-- Average Rainfall
evaporimeter, as recommended by the World Meteorological Organisation. Evaporation from the Australian tank evaporimeter lags behind that of the class-A pan, due to seasonal differences in heat transfers from the soil (to the Australian tank) and from the air (to the class-A pan). Direct comparison of the two evaporimeters is made difficult due to this lag, which can result in simultaneous readings being quite different (Nimmo 1964).

An examination of **spatial variability** of pan evaporation in Perth is difficult owing to the paucity of measuring sites. Since November 1981, daily class-A pan evaporation measurements have been made at Perth Airport (11 km ENE of central Perth—see Fig. 3.1). Class-A pan evaporation is also read during week days at the Upper Swan Research Station (USRS). Figure 3.4 shows the average daily pan evaporation values for these three sites for the 12-month period, November 1981 to October 1982 (data are available only until June 1982 for the USRS). The figure shows that evaporation was about 18 per cent higher at Perth Airport than at central Perth over the whole year, the difference being about 30 per cent in late summer. While mean pan evaporation was higher at Perth Airport for all months, the difference was only significant \((P < 0.05)\) for the six hottest months. Evaporation from the USRS was very similar to that from Perth Airport for the eight-month period for which data are available. Annual pan evaporation data are also available for three years (1974, 75 and 78) for the USRS. During these years, pan evaporation from the USRS was, on average, 12.5 per cent higher than that from central Perth. Possible reasons for the differences between central Perth, Perth Airport and the USRS will be given later, after spatial differences in temperature, relative humidity and wind speed have been examined. Congdon (unpubl. data) has obtained class-A pan evaporation measurements for four-week periods for central Perth and Edgewater (a suburb 22 km NNW of Perth and 5 km from the coast). There was no significant difference between the two sites over a 68-week period. No other measurements of class-A pan evaporation have been published for the Perth area.

Figure 3.5 shows the temporal distribution of class-A pan evaporation and rainfall values for Perth for the period 1967-82. The data are plotted in semi-log form so that percentage variations can be observed. Annual variability of evaporation is much less than the annual variability of rainfall for this period (coefficient of variation (CV) for pan evaporation = 7%, CV for rainfall = 20%).

Figure 3.6 shows the variations in the distribution of evaporation throughout the year, for the years 1979-82. Cumulative evaporation amounts were below average for all of these years, except for the first month or two of 1979 and 1980. Only 8 of the 48 months during 1979-82 had an above-average evaporation rate. This is surprising, as rainfall was below average for all of these years. Possibly, the present average values do not yet reflect long-term averages, due to the shortness of the record and to the unusual number of dry years since 1967. Annual pan evaporation was 8 to 11 per cent below average during the last four years.

As there were so few measurements of potential evaporation for Perth, a comparison of three factors (temperature, relative humidity and average wind speed) known to influence evaporation, is made in the following sections. Urbanization has been shown to increase temperature and decrease relative humidity and average wind speed in other studies (as discussed in Section 2.1).

3.1.4 Temperature

Temperatures at two times of the day (9 am and 3 pm) during 1981 were compared for three sites. Table 3.2 gives details of the sites, while their location was shown on Figure 3.1. The table indicates that there are problems of confounding due to decreasing degrees of urbanization coinciding with increasing distance from the coast. However these are the only sites for
FIGURE 3.6: Distribution of Class-A pan evaporation throughout the year for the period 1979-1982. (Source: Bureau of Meteorology).
which data are available.

Table 3.2

Details of sites used in climatic comparison

<table>
<thead>
<tr>
<th>Site</th>
<th>Land use</th>
<th>Distance from coast (km)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Central Perth</td>
<td>High density urban</td>
<td>11</td>
</tr>
<tr>
<td>Perth Airport</td>
<td>Suburban</td>
<td>20</td>
</tr>
<tr>
<td>Pearce Airport</td>
<td>Rural</td>
<td>26</td>
</tr>
</tbody>
</table>

Figure 3.7 (a) and (b) shows the average monthly temperatures at 9 am and 3 pm for central Perth in comparison to Perth Airport and Pearce Airport. Values are shown for the airports only if they are significantly different ($P < 0.05$) from Perth.

At 9 am, Perth Airport was significantly cooler than central Perth by about $0.6^\circ$C between April and November. Pearce Airport was cooler than central Perth (but not significantly so) between June and October. This lack of significance is possibly due both to a smaller data set (readings are only taken on weekdays at Pearce) and to the small difference between the sites. The higher 9 am temperatures at central Perth during winter and spring may be due to a maritime influence. If the difference during this period was due to back-radiation of heat from buildings and/or to decreased evaporation from soil and vegetation in the built-up areas, then it would be expected that Pearce Airport, with its rural land use, would be significantly cooler than either of the other sites.

At 3 pm, the airports were significantly warmer than central Perth in the summer and significantly cooler in the winter (Fig. 3.7 (b)). This is almost certainly a maritime effect as evidenced
FIGURE 3.7(a): Average monthly temperature at 9 am for three sites around Perth. (Source: Bureau of Meteorology).

FIGURE 3.7(b): Average monthly temperature at 3 pm for three sites around Perth. (Source: Bureau of Meteorology).
by the seasonal variation. The warmer 3 pm temperatures in central Perth during winter may be partially due to back radiation of stored heat from buildings and a reduced latent heat flux, as Barrett (1979) showed that the heat island in Perth was most pronounced during the cool seasons.

Perth experiences a sea breeze for 15 to 20 days each month during October to March (Hounam 1945) which heightens the maritime effect during summer. Vollprecht (1969) reported a 2.8°C increase in annual maximum temperatures from Fremantle (on the coast) to Guildford (20 km inland). This difference is at its greatest in January when Guildford’s maximum is 4.7°C higher than that of Fremantle.

3.1.5 Relative humidity

Figure 3.8 (a) and (b) shows the average monthly relative humidity of the three sites considered in the previous section. Both airports have higher relative humidities than central Perth at 9 am between April and November, although the differences are significant only towards the end of this period. The differences may be due to differences in water content in the atmosphere and/or differences in temperature (the airports have lower temperatures at this time of year). By converting relative humidities to absolute humidities for August, it can be calculated that lower temperatures are responsible for 37 per cent of the higher relative humidity at Perth Airport and 13 per cent of the higher relative humidity at Pearce Airport. Thus, availability of moisture appears to be the principal reason for the differences in relative humidity between central Perth and the airports during winter and spring. Perth Airport had a significantly lower 9 am relative humidity than central Perth in January and February. The reason for this difference is unknown (temperatures were not significantly different during these months) but it may be a maritime (estuarine) effect.
FIGURE 3.8 (a): Average monthly relative humidity at 9 am for three sites around Perth.
(Source: Bureau of Meteorology).

FIGURE 3.8 (b): Average monthly relative humidity at 3 pm for three sites around Perth.
(Source: Bureau of Meteorology).
During the summer, central Perth has a higher relative humidity at 3 pm than the airports, probably due to the influence of the sea breeze (which lowers temperatures and carries in more humid air). In January, the proportion of the difference in relative humidity that can be explained by temperature is 39 per cent and 49 per cent for Perth- and Pearce- Airports respectively. In the winter, the airports have a higher relative humidity than central Perth. In July, the difference between central Perth and Perth Airport can be completely accounted for by the difference in temperature but only 40 per cent of the difference between central Perth and Pearce Airport is due to temperature.

In summary, the airports tend to have higher temperatures and lower relative humidities in summer and lower temperatures and higher relative humidities in winter. Maritime influences appear to have most effect on spatial variability although water availability appears to influence relative humidities in winter.

3.1.6 Wind speed

Central Perth had a lower, average wind speed than Perth Airport for every month of 1981, although the difference was significant for only six of those months (Fig. 3.9). The average wind speed was 16 per cent lower over the whole year. The central Perth meteorological equipment is located immediately east of the central business district of Perth. Although there are more easterlies than westerlies at all times of the year in Perth, the strongest winds occur in winter from the west (Vollprecht 1969). The sea breeze in summer is a southwesterly wind. Increased deflection and turbulence as a result of the buildings in the central business district is the probable cause of the lower wind speed for central Perth. As wind speed is not recorded elsewhere in the Perth area, no further comparisons can be made.
3.1.7 Discussion of climatic variability

Of the two major climatic influences on groundwater levels in Perth (rainfall and potential evaporation) rainfall is the more variable with time, both seasonally and annually. From the limited evidence available there does not appear to be an appreciable urban effect on rainfall amount or distribution. The amount of spatial variability in rainfall highlights the need to have a fairly dense network of stations when undertaking hydrologic studies.

The spatial variability of potential evaporation in Perth is less clear, due to the paucity of measurements. On a microscale, variability in evaporative potential is likely to be quite high due to variations in site factors (e.g. high potentials downwind of car parks due to advected energy). In inland urban areas, evaporative potential appears to be enhanced by higher temperatures and lower relative humidities during summer. However, coastal urban areas are likely to receive more wind during this period. Evaporative potential in the central business district appears to be enhanced by higher temperatures and lower relative humidities during winter, but decreased due to less wind. There are insufficient data to reach firm conclusions on the spatial variability of evaporative potential for Perth, although it appears likely that there is a progressive increase in potential with increasing distance from the coast.
3.2 STRATIGRAPHY

The unconfined groundwater studied in this project occurs within the Late Tertiary-Quaternary 'superficial formations' (Allen 1976, 1981). The 'superficial formations' form a thin (to 100 m) cover over about 13 000 m of Phanerozoic sediments which fill the Perth Basin. Detailed stratigraphy of the Perth Basin is provided in Playford et al. (1976). The Swan Coastal Plain occurs on the eastern, onshore portion of the Perth Basin and extends more than 200 km north and south of Perth. In the Perth area the Plain is bounded to the east by the Darling Escarpment.

Figure 3.10 shows the generalized surface geology of the Swan Coastal Plain in the Perth area. Allen (1981) noted that the 'superficial formation' sediments form a complex and more variable sequence than that shown in Figure 3.10. As yet, no detailed division has been made of the 'superficial formations', apart from that based on surface geology. Each formation has associated with it a distinctive geomorphology, vegetation and soils (McArthur and Bettenay 1974), which are shown in Table 3.3. In the Perth area, the unconfined groundwater is contained mainly within the Tamala Limestone and Bassendean Sand Formations. These formations are represented by the Spearwood and Bassendean Dunes respectively in their unsaturated, upper parts, and will be considered in more detail. The experimental catchments studied in this project were located within the Tamala Limestone/Spearwood Dune System (Karrakatta soil series).

The Spearwood Dune System has been described by McArthur and Bettenay (1960) as a core of calcareous eolianite (the Tamala Limestone) with a hard capping of secondary calcite, overlain by yellow or brown sands. Part of the system was later recognized as being marine in origin (McArthur and Bettenay 1974). The dunes were considered by the authors to have been calcareous to the surface originally, but continued leaching has removed the carbonate from the upper sections and deposited some of it as hard
FIGURE 3.10: Generalised surface geology in the Perth area.
(Modified from Allen 1981)
### Table 3.3

**Geomorphology, vegetation and soils associated with the 'superficial formations'** (compiled from Playford et al. (1976) and McArthur and Bettenay (1974) unless otherwise acknowledged)

<table>
<thead>
<tr>
<th>Geological formation(s)</th>
<th>Lithology</th>
<th>Approximate age</th>
<th>Geomorphic element</th>
<th>Vegetation</th>
<th>Soil series</th>
<th>Groundwater occurrence</th>
</tr>
</thead>
<tbody>
<tr>
<td>Safety Bay Sand</td>
<td>Eolian and littoral calcarenite</td>
<td>Recent</td>
<td>Quindalup Dune System</td>
<td>Acacia-Callitris-Melaleuca</td>
<td>Quindalup</td>
<td>Locally important aquifer south of Perth (Kwinana-Safety Bay area)</td>
</tr>
<tr>
<td>Tamala Limestone</td>
<td>Coarse-medium grained eolian calcarenite and yellow sand</td>
<td>Late Pleistocene</td>
<td>Spearwood Dune System</td>
<td>Eucalyptus gomphophale forest or woodland</td>
<td>Karrakatta Spearwood Wonnerup</td>
<td>Large groundwater resources. Saturated aquifer thickness &lt; 30 metres</td>
</tr>
<tr>
<td>Bassendean Sand</td>
<td>Leached, siliceous, eolian grey sand</td>
<td>Early-, middle- (Late) Pleistocene</td>
<td>Bassendean Dune System</td>
<td>Banksia-Casuarina-Xylem eium scrub</td>
<td>Gavin Jandakot Joel Muchea</td>
<td>Large groundwater resources Saturated aquifer thickness &lt; 65 metres</td>
</tr>
<tr>
<td>Guildford Formation</td>
<td>Fluvial interbedded sand, clay and conglomerate, calcareous in places</td>
<td>Pleistocene</td>
<td>Pinjarra Plain</td>
<td>E. marginata-E. calophylla forest</td>
<td>Coolup Boyanup Wellesley Blythwood Belhus Dardenup Vasse Pytton</td>
<td>Limited capacity aquifers. (Wilde and Low 1978)</td>
</tr>
<tr>
<td>Ridge Hill Sandstone</td>
<td>Littoral ferruginous sandstone, basal conglomerate</td>
<td>Early Pleistocene- Late</td>
<td>Ridge Hill Shelf or Piedmont Zone (Wilde and Low 1978)</td>
<td>E. marginata-E. calophylla forest</td>
<td>Lotons Range Oakover</td>
<td>Local supplies of potable water (Wilde and Low 1978)</td>
</tr>
<tr>
<td>Yoganup Formation</td>
<td>Littoral sand, basal conglomerate, clay lenses, heavy minerals</td>
<td>Tertiary</td>
<td></td>
<td></td>
<td></td>
<td>Not known to be an aquifer (Allen, pers. comm.)</td>
</tr>
</tbody>
</table>
capping below the surface. The depth of sand overlying the limestone is greater in the east (Karrakatta soil series) where it has a well-defined $A_2$ horizon, than in the west (Spearwood and Wonnerup soil series). The sands are podzolized sands (McArthur and Bettenay 1960). Biggs and Wilde (1980) described the Tamala Limestone as consisting of coarse-to-medium grained calcarenite, composed mainly of fossil skeletal fragments (foraminifera and mollusc) and varying amounts of quartz. The Karrakatta soil series which overlies the Tamala Limestone has been described by Biggs and Wilde (1980) as being a moderately sorted, fine-to-coarse grained sand. Individual grains are frosted, sub-angular to sub-rounded and varicoloured depending upon the degree of leaching, iron-staining and humic content.

Killigrew and Glassford (1976) and Glassford and Killigrew (1976) drew attention to the presence of kaolin spherites in the fine sand fraction of the soils overlying the Tamala Limestone. These authors also drew attention to the differences between the sands and the insoluble constituents in the underlying calcareous eolianite (viz a relative enrichment in the sands of fines, microcline, heavy minerals, yellow grain-surface clay coatings and kaolin spherites). Because of these, and other factors, Glassford and Killigrew (1976) argued that decalcification of the eolianite could not be the mechanism whereby the overlying sands formed. The presence of goethite in the clay coatings of grains gives the sands, beneath the leached A horizon, a yellow colour.

Klenowski (1975) examined the geotechnical properties of the Tamala Limestone and identified five types of material; cemented rocks (calcareous quartz sandstone, limestone), calcretized rocks (caprocks, pinnacles, solution pipes), rocks crystallized in cavities (dripstones), sand and thin lenses of calcareous silt clay and marl. The degree and type of calcareous cementation of the limestone was found to be quite variable.

McArthur and Bettenay (1960) described the Bassendean Dune
System as a series of low hills of quartz sand interspersed with poorly drained areas. The authors considered that the sand probably originated along a coastline as a calcareous shoreline- and dune-sand, but that continued leaching has subsequently removed most of the carbonate and some of the iron. However, it was later recognised that some areas of Bassendean Sand may have been siliceous on deposition (McArthur and Bettenay 1974). The soils of the Bassendean Dunes are podzols. Humus podzols (the Gavin series) occur in the older and lower parts of the dunes while iron-humus podzols (the Jandakot series) occur in the younger and higher parts. Removal of iron from the upper soil horizons reaches a maximum in the Gavin series (McArthur and Bettenay 1960), particularly where organic matter is most abundant. Biggs and Wilde (1980) describe the Bassendean sands as poorly sorted, fine-to-medium grained, quartz sands. The grains are angular to sub-rounded and often frosted.

Figure 3.11 shows sections through the 'superficial formations' for the Northern and Southern Perth areas (Allen 1981). Allen (public lecture 1981) has recognized that a shallow-water marine, calcareous sandstone (the Joondalup Formation) underlies the eolian Tamala Limestone in the Perth area. Small marine units (the Peppermint Grove- and Rottnest- Limestones) have been previously recognized to interfinger with the Tamala Limestone (Playford et al. 1976). Allen (public lecture 1981) also considered that the yellow Karrakatta sand, which is associated with the Tamala Limestone, is a primary sand that was not formed by decalcification of the Spearwood Dunes.

The stratigraphy of the sediments which underlie the 'superficial formations' are outlined briefly below, as appreciable exchange of groundwater occurs between the unconfined aquifer and some of the underlying formations (Allen 1981, Fig. 16). Figure 3.12 shows the distribution of these underlying formations while Table 3.4 gives the major features of each. From the figure and table it can be seen that the base to the
FIGURE 3.11: Geological cross-sections through the 'superficial formations' in the Perth area.

FIGURE 3.12: Sub-crop map of formations underlying the 'superficial formations'. (after Allen 1981)
<table>
<thead>
<tr>
<th>Age</th>
<th>Formation</th>
<th>Maximum thickness (m)</th>
<th>Lithology</th>
<th>Groundwater potential</th>
</tr>
</thead>
<tbody>
<tr>
<td>Quaternary-Late Tertiary</td>
<td>'Superficial formations'</td>
<td>100</td>
<td>Sand, limestone, clay</td>
<td>Major unconfined aquifer; Fresh groundwater</td>
</tr>
<tr>
<td></td>
<td>Unconformity</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Late Tertiary</td>
<td>Rockingham Sand</td>
<td>110</td>
<td>Sand</td>
<td>Minor unconfined aquifer; Fresh and saline groundwater</td>
</tr>
<tr>
<td></td>
<td>Unconformity</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Early Tertiary</td>
<td>Kings Park Formation</td>
<td>240</td>
<td>Shale, calcareous and glauconitic siltstone, minor sand</td>
<td>Local unconfined aquifer; Fresh groundwater</td>
</tr>
<tr>
<td></td>
<td>Unconformity</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>Gingin Chalk</td>
<td>25</td>
<td>Fossiliferous and glauconitic chalk</td>
<td></td>
</tr>
<tr>
<td>Late Cretaceous</td>
<td>Molecap Greensand</td>
<td>15</td>
<td>Glaucotic sand</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Osborne Formation</td>
<td>150</td>
<td>Glaucotic shale, siltstone and sand</td>
<td>Semi-confining bed; local aquifer; fresh groundwater</td>
</tr>
<tr>
<td>Early Cretaceous</td>
<td>Leederville Formation</td>
<td>500</td>
<td>Sandstone, siltstone and shale, locally calcareous and glauconitic</td>
<td>Major confined aquifer; fresh to brackish groundwater</td>
</tr>
</tbody>
</table>
'superficial formations' under much of the Perth area is composed of low permeability shale formations (Kings Park- and Osborne-Formations) which have only relatively small, local, sandy aquifers associated with them. Much more extensive groundwater exchange is known to occur with the Leederville Formation, a major confined aquifer.
3.3 HYDROGEOLOGY

3.3.1 Groundwater systems

Figure 3.13 shows the configuration of the water table of the unconfined groundwater in the Perth area. In the Northern Perth Area (i.e. north of the Swan River) the groundwater forms a single flow system, while six separate flow systems have been recognized in the Southern Perth Area (Davidson 1981). The boundaries of the systems are the major rivers of the Plain (all of which are gaining streams) and the ocean. The base of the unconfined aquifer systems (i.e. the base of the 'superficial formations') rises from about 30 m below the Australian Height Datum (AHD) on the coast to 35 m above AHD in the NE and 25 m above AHD in the SE (Allen 1976). The base is traversed by relatively deep river channels.

3.3.2 Recharge

Recharge to the groundwater arises principally from rainfall infiltration of the highly permeable eolian sands. Estimates of net and gross recharge have been made by various workers. As these estimates are relevant to recharge in urban areas, they are reviewed in some detail.

Bestow (1970) used techniques of groundwater flow and chloride mass balance to estimate that net recharge in the southern part of the Gnangara Mound was 8.3 per cent of the average annual rainfall. Using techniques similar to those of Bestow, Balleau (1973) estimated that net recharge for a 400 km$^2$ area of the Gnangara Mound varied locally from more than 30 per cent of annual average rainfall near the crest of the Mound, to a net water loss in the vicinity of lakes and surface water discharges. Allen (1981) used the ratio of chloride contents in rainfall to chloride contents of groundwater to estimate that the net average recharge for the whole Gnangara Mound was 11.5 per
cent of annual average rainfall. Davidson (1981) has estimated net recharge of the Jandakot Mound to be 12.3 per cent of annual average rainfall using flow net techniques, and estimated a similar percentage using chloride concentrations.

Butcher (1979) monitored soil moisture profiles under pine and native woodland areas in the Northern Perth Area using neutron attenuation techniques to provide point estimates of gross recharge. Drainage below six metres (deep drainage) was estimated by subtracting average evapotranspirational values from rainfall inputs, once wetting fronts had passed six metres. Deep drainage, as a percentage of rainfall, was estimated to be 29 per cent for native woodland, 19 per cent for young, open Pinus pinaster stands and only 8 per cent for dense pine stands. In wet years (1973, 1974), drainage below six metres was estimated to be 40 per cent of winter rainfall for the native woodland areas, while in dry years (1976), deep drainage was negligible. Carbon et al. (1982) also measured soil moisture profiles under forests and pastures on the Gnangara Mound by neutron attenuation techniques and estimated deep drainage by using the redistribution sub-routine of a computer model developed by Carbon and Galbraith (1975). Deep drainage in upland Spearwood Dunes was estimated by Carbon et al. to be 34 per cent of rainfall in native forest and 11 per cent in pine forests. Grazed pastures had variable amounts of deep drainage (21 to 43%) depending on the species, and it was considered by Carbon et al. that the replacement of native forest with such pastures would result in a significant increase in deep drainage. Sharma et al. (1983) have shown an absence of recharge under dense pines using chloride concentration measurements in the vadose zone.

The only estimate of recharge in urban areas of Perth has been made by Williamson and Cole (1976). These authors estimated that 49 per cent of irrigation waters would become recharge. About 68 per cent of rainfall that falls on individual household blocks, and almost all winter runoff from shedding surfaces, was
considered to become recharge. Using the computer model of Carbon and Galbraith (1975), the authors estimated that deep drainage below the root zone of lawns increased significantly once the volume of applied water exceeded 60 per cent of pan-evaporation rates. Carbon (1975a) had previously found that it is possible to maintain over 95 per cent cover on a couch lawn by watering at only half pan-evaporation rates, provided adequate fertilizer was also applied.

Allen (1976, 1981) and Burton (1976) considered recharge in the Spearwood Dune System to be greater than in the Bassendean Dune system, as the water table is generally further below ground in the former. Cargeeg et al. (1983) found that chloride concentrations in upper aquifer waters were lower in the Bassendean Dune System than in the Spearwood Dune System. As only part of the difference could be accounted for by decreased chloride concentrations in the rainfall, the authors concluded that recharge rates must be higher in the Bassendean Dune system.

Pollett et al. (1979) developed a recharge-discharge model for calculating the net withdrawal function for use in a digital computer model of groundwater flow in the Northern Perth Area. The recharge-discharge model was developed from the consideration of work by Kovacs (1969) and was calibrated by use in the groundwater flow model. The recharge-discharge model that was developed showed net recharge to be at a maximum when the water table was at a depth of about 9 m. Net discharge was predicted to occur when the water table was less than about 3 m from the surface, while net recharge tended to zero at depths greater than 9 m. Pollett (1981) considered recharge during above-average winters to be very important for recharging aquifers, as groundwater levels were noted to decline during periods of average rainfall. Allen (1981) considered that recharge to the groundwater systems in parts of the Northern- and Southern-Perth Areas also takes place by upward groundwater flow from the underlying Leederville and Osborne Formations.
3.3.3 Discharge

Natural discharge of groundwater occurs by evaporation from soils, vegetation and wetlands, by outflow to rivers and the ocean, and by downward leakage to confined aquifers (particularly to the Leederville Formation). Bestow (1976) estimated the water balance for the Northern Perth Area and, more recently, Allen (1981) has estimated the water balance for both the Northern and Southern Perth Areas. Evapotranspiration was considered by Allen to account for about 82 per cent of all losses from the groundwater system, with abstraction by bores accounting for about 6 per cent. Other losses were considered to be baseflow to rivers (2%), outflow to the sea (6%) and leakage to underlying aquifers (4%).

3.3.4 Hydraulic parameters

The specific yield of the aquifers of the Northern Perth Area is considered to be about 0.3, while the yield of those in the Southern Area is considered to be only 0.2, due to the greater abundance of clay in the latter area (Allen 1981). The hydraulic conductivity of the calcareous sediments is considered to be about 100 m/d and of the sands, 30 m/d (Allen 1981). The high hydraulic conductivity of the limestone probably results from flow in fissures and solution channels while that of the sands results from their being well sorted and uncemented.

Pollett (1981) drew attention to the degree of anisotrophy in the aquifers on the coastal plain, as evidenced by the leaky artesian response obtained during pump tests. In the Southern Perth Area, horizontal hydraulic conductivities of 15 to 40 m/d were measured whereas vertical conductivities were considered to be much lower. The leaky artesian response of the aquifers has meant that the effect of groundwater extraction from the base of the aquifers is distributed over a much larger area than would
occur in a purely unconfined aquifer. Pollett (1981) measured water table declines of only 30 to 70 per cent of that predicted by computer modelling, and attributed the difference to the leaky artesian nature of the aquifers.

An idea of relative transmissivities of the aquifer can be gained by studying the hydraulic gradients as shown in Figure 3.13. Allen (1976) pointed out that low gradients across the coastal strip reflect the high transmissivity of the Tamala Limestone while the high gradients east of the Gnangara Mound reflect the low transmissivity of the Guildford Formation. The steep change of gradient that occurs immediately east of the coastal strip close to the contact between sand and limestone was believed by Allen (1981) to mark an area where groundwater moves locally through limestone caves. The relatively uniform gradient that extends south of the Gnangara Mound was interpreted by Allen (1976) to indicate that the Bassendean Sand has relatively uniform hydraulic characteristics.

3.3.5 Groundwater quality

A general appreciation of groundwater salinity in the Northern and Southern Perth Areas can be gained from Figure 3.14. Total dissolved solid (TDS) concentrations vary from about 130 to 1200 mg/L, with an average of about 500 mg/L (Allen 1981). Figure 3.14 shows that salinities increase away from the groundwater mounds towards river- and ocean- discharge boundaries. More saline plumes have also been noted down-gradient from wetland areas (Bestow 1970; Allen 1976).

The most comprehensive study of groundwater chemistry in the Perth area has been made by Martin (1980) and Martin and Harris (1982). These workers found that as the groundwater flowed from the Bassendean Sand to the Tamala Limestone (overlain by the Spearwood sands), the following changes occurred:
(i) \( \text{Ca}^{2+}, \text{HCO}_3^- \) and pH rose as a result of the groundwater encountering the Tamala Limestone.

(ii) \( \text{Eh} \) rose as a result of lower concentrations of total organic carbon (TOC) in the Spearwood sands (which are younger and also contain fewer wetland areas).

(iii) \( \text{Fe}^{2+} \) rose, despite the rise in \( \text{Eh} \), because of the greater abundance of iron in the Spearwood sands. Some control on \( \text{Fe}^{2+} \) concentrations in groundwaters in the Tamala Limestone was thought to be exerted by carbonate and hydroxide precipitation, and in the Bassendean Sand by precipitation of pyrite in highly reduced waters.

(iv) \( \text{NO}_3^- \) contents rose, presumably as a result of additions by fertilizers and septic tanks in the Spearwood Sands. Orthophosphate was not detected in waters in the Tamala Limestone, possibly due to adsorption of phosphate by goethite and/or precipitation as hydroxylapatite due to the presence of limestone.

(v) Mean \( \text{SO}_4^{2-} \) contents were similar between the two formations but there was evidence of large increases in values in individual bores in both systems.

Almost all of the work cited in the above review of groundwater in the Perth area has been carried out in non-urban areas. With the exception of the estimates of Williamson and Cole (1976), urban hydrology in Perth had not been studied before this project was undertaken.
CHAPTER 4

WATER BALANCES - SETTING AND CONCEPTUAL MODELS

4.1 SELECTION OF STUDY AREAS

The first specific aim of this research is to measure the components of the water balance in two, dissimilar urban areas. In selecting areas for such a study, a number of contrasting aspects are pertinent, i.e.

(i) **Bore densities**: Isolating the effect of private bores on groundwater levels is a general aim of the study and therefore the areas chosen must have significantly different bore densities.

(ii) **Housing densities**: Assessing the effect of urbanization on groundwater is also a general aim of the study and therefore a comparison of areas with different housing intensities would be advantageous, in addition to the comparison of urban with woodland areas.

(iii) **Aquifer lithologies**: As mentioned in the previous chapter, the groundwater in the Perth area is mainly contained within two lithologies, the Tamala Limestone and Bassendean Sand.

(iv) **Waste disposal systems**: As only about half of Perth is sewered, a comparison of a sewered area with an area with septic tanks would be of interest.

Ideally, studies should be made using paired catchments to test independently for each of the four contrasts mentioned above. With only two catchments to be studied, it would be difficult to separate effects, if all four contrasts occurred between the areas. As water balances involve all the components of the hydrologic cycle, the effect of different waste disposal systems
can be estimated from studying areas with only one type of system (i.e. the effect of adding/subtracting the sewage flow component to/from the soil and aquifer can be estimated if the magnitude of the flow is known). However, the effect of different aquifer lithologies cannot be taken into account as easily. This is due to the different hydraulic behaviour of the two lithologies, which would be hard to estimate. Cargeeg et al. (1983) have identified a difference in recharge rates in the two lithologies but this difference is likely to be a function of climate and land use and therefore be variable in both space and time.

In this study it was decided to choose areas which differ in the first two aspects given above, while being similar in the last two. As will be seen later, bore- and housing- densities are often related in urban areas of Perth.

As well as providing the above comparisons, there are a number of desirable features the experimental catchments should have, i.e.

(i) **Representativity**: It is important that the urban areas studied are representative of other urban areas, if the results are to be applicable to a larger area.

(ii) **Accessibility**: The areas should be readily accessible for logistical purposes, and also sufficiently close so that there are not appreciable climatic differences between them, necessitating duplication of instruments.

(iii) **Well-defined catchment boundaries**: The definition of catchment boundaries in urban areas is complicated by street drainage schemes and inter-basinal transfers of water. Experimental-area boundaries may be based on ground surface contours, street drainage catchments, reticulated water supply areas or on water table contours (i.e. isopotentials and flow lines). Because surface runoff is rare in non-urbanized areas on
the coastal plain, water balances that have been carried out in these areas (e.g. Bestow 1970; Balleau 1973) have used water table contours as boundaries. A brief estimation of the water balance of parts of Perth by consultants (Sinclair et al. 1981) adopted boundaries of reticulated water-supply areas, as data on mains water use could then be used directly.

Street drainage catchments were used in this study to simplify measurements of surface-water flows. However use of such boundaries required extra attention to reticulated water use and groundwater flow. Potential catchments were examined to see if their boundaries were suitably aligned to groundwater isopotentials and flow lines so that errors in calculating groundwater flow could be minimized. Some problems were encountered in defining drainage boundaries as there are no accurate maps of drain locations in the areas examined.

(iv) Well-defined, impermeable base to the aquifer: To avoid having to make measurements of leakage between aquifer systems, it is desirable to have a well-defined and impermeable aquifer base. As mentioned in the previous section, the Kings Park Shale and Osborne Formations are generally impermeable. However, Allen (1981) has reported some upward leakage from the Osborne Formation and from the boundary of the Kings Park Formation with the Osborne Formation. The most impermeable aquifer base appears to occur in areas where the Kings Park Shale is at its thickest, i.e. immediately north of the Swan River and west of central Perth (see Fig. 3.12).

Using the above criteria, the following two urban catchments were chosen (Fig. 4.1) - Masons Gardens catchment in Nedlands-Dalkeith (N-D), an area of large blocks and containing numerous bores, and the Shenton Park Lake catchment in Subiaco-Shenton Park (S-SP), an area of small blocks and containing few bores. Table 4.1 gives general particulars about the two catchments. Both catchments have centrally located wetlands which are surface
FIGURE 4.1: Location of the urban- and woodland- study areas and of groundwater absorption basins.
Table 4.1

Details of the urban catchments

<table>
<thead>
<tr>
<th>Catchment</th>
<th>Nedlands-Dalkeith</th>
<th>Subiaco-Shenton Park</th>
</tr>
</thead>
<tbody>
<tr>
<td>Area (ha)</td>
<td>68.07</td>
<td>156.87</td>
</tr>
<tr>
<td>Number of houses</td>
<td>406</td>
<td>1621</td>
</tr>
<tr>
<td>Housing density (houses/ha)</td>
<td>6.0</td>
<td>10.3</td>
</tr>
<tr>
<td>Bore use density* (from questionnaire, Section 5.5)</td>
<td>43%</td>
<td>4%</td>
</tr>
<tr>
<td>Aquifer lithology</td>
<td>Tamala Limestone (limestone, minor sand)</td>
<td>Tamala Limestone (sand, minor limestone)</td>
</tr>
<tr>
<td>Domestic wastewater disposal system</td>
<td>Sewered</td>
<td>Sewered</td>
</tr>
<tr>
<td>Distance from Perth regional meteorological office (km)</td>
<td>7</td>
<td>5</td>
</tr>
<tr>
<td>Distance from Indian Ocean (km)</td>
<td>4.5</td>
<td>5.5</td>
</tr>
</tbody>
</table>

* Percentage of gardens being watered with groundwater.
expressions of the water table. Street runoff waters are directed into these wetlands and they therefore act as absorption basins. Both catchments also have some inter-basinal transfers of street runoff waters (shown as adjoining basins in Fig. 4.1).

Figure 4.1 also gives the location of the two native woodland areas which were selected. Only soil moisture measurements were made in these areas (the sites are shown in Fig. 4.1). Groundwater levels were not measured in the native woodland areas as the depth of the water table was considerable (11 to 20 metres) and there were no private bores from which the measurements could be made.

The N-D catchment is typical of many established suburbs in coastal and river suburbs of Perth in having large (900-1000 m² or 1/4 acre) blocks, and established gardens and street trees. Such areas are much sought-after and many residents find it both advantageous and financially possible to install private bores to water the large blocks. The S-SP catchment is typical of older, inner-city areas in having small (400-600 m² or 1/8 acre) blocks, established gardens and street trees and a greater proportion of high density residences (flats, home units and duplexes). Because of the small blocks, few residents find a private bore an attractive investment. About one-third of Perth has features similar to those studied in this project. The other two-thirds of Perth differs by having less well established street trees and gardens and/or by being situated on Bassendean Sands.
4.2 CONCEPTUAL MODELS

To fit the measurements of water balance components into a logical framework, conceptual models were developed for the two catchments. The model for the S-SP catchment is shown in Figure 4.2. Similar models have been developed for studies in other urban areas, e.g. Long Island (Franke and McClymond 1972) and Halifax (Waller 1977). The model shown in Figure 4.2 has been set up so that it could be run as a lumped-parameter computer model (e.g. simcomp package modelling - Gustafson and Innis 1973) if required. The boxes represent temporary - (e.g. roof water) or permanent - (e.g. aquifer water) stores of water. Provided that measurements of water balance components are measured only over relatively long intervals (i.e. days, weeks), temporary storages can be ignored and only permanent storages require measurement. Flows between storages require either direct measurement (e.g. rainfall) or estimation based on a knowledge of flows likely for various storages and environmental conditions, e.g. groundwater flow may be estimated from a knowledge of hydraulic gradient and aquifer parameters. The sources and sinks in Figure 4.2 represent storages which do not need quantification. The N-D catchment does not have a flood mitigation drain or fountain and therefore this part of the model is not applicable in this catchment.

Conceptual models using lumped parameters appear more suitable than distributed parameter models in urban areas, owing to the scattered distribution of pervious and impervious surfaces. The model draws a distinction between concentrated waters from shedding areas (e.g. roofs, roads) and distributed waters (e.g. rain falling on pervious areas and irrigated waters). Similar types of concentrated waters (e.g. spoon drain* and path edge waters) are grouped together. Lloyd (1981) distinguished between direct recharge (i.e. water that has passed through the soil before entering the groundwater) and indirect recharge (i.e. water that has passed through a runoff or interflow phase before entering the groundwater). Distributed waters give rise to direct

* also called 'splash blocks'
FIGURE 4.2: Conceptual model of the urban water balance for the S-SP catchment.
recharge and concentrated waters give rise to indirect recharge.

The time and resources that were given to the measurement of each of the storages and fluxes shown in Figure 4.2 were determined by a number of factors which are outlined below:

(i) The relative magnitude of the flux/storage. At the beginning of the study, hypothetic water balances were carried out on each catchment using a set of 'best estimates', given the state of knowledge at that time. From the balances, flux magnitudes were ranked in order of size. Table 4.2 gives the ranking obtained for the main fluxes in the S-SP catchment water balance. The rankings were similar but not identical in the two catchments. For example, private bore extraction was ranked much higher in the N-D catchment (rank 13), as was groundwater inflow and outflow (2 and 4 respectively), throughfall (11) and direct recharge (8).

(ii) The effort that would be required to achieve a specific improvement in the estimation of a flux/storage amount (i.e. the cost-benefit of any measurements). This factor was assessed in a subjective manner, unlike (i) above. For instance, it was decided that a considerable effort be placed in measuring direct recharge (rain falling on pervious surfaces, irrigated waters) as this flux was considered to be poorly understood and quantified. In particular, the amount of irrigation water that was becoming recharge was poorly understood. Much less effort was placed on measuring indirect recharge from roofs and paths, due to the difficulty of measuring the advance of three-dimensional wetting fronts and also the possibility of making reasonable estimates from a knowledge of direct recharge amounts.

(iii) The equipment and expertise that were available to the study. Because of the amount of help received in the form of equipment loans and advice from various Government Departments, this factor did not prove to be a major limitation to the study.
Table 4.2

Hypothetical water balance of the S-SP catchment — ranking of major fluxes in order of decreasing size

<table>
<thead>
<tr>
<th>Rank</th>
<th>Flux (and process where appropriate)</th>
<th>Percentage of rainfall</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>Atmosphere → Rainfall (rainfall)</td>
<td>100</td>
</tr>
<tr>
<td>2</td>
<td>Soil → Atmosphere (evaporation, including transpiration by non-phreatophytes)</td>
<td>69</td>
</tr>
<tr>
<td>3</td>
<td>Imported MWA water → Reticulation system Residences</td>
<td>49</td>
</tr>
<tr>
<td>4</td>
<td>Rainfall → Soil</td>
<td>46</td>
</tr>
<tr>
<td>5</td>
<td>Absorption basin → Flood mitigation drain → Ocean</td>
<td>39</td>
</tr>
<tr>
<td>6</td>
<td>Aquifer → Downgradient groundwater (outflow)</td>
<td>36</td>
</tr>
<tr>
<td>7</td>
<td>Upgradient groundwater → Aquifer (inflow)</td>
<td>33</td>
</tr>
<tr>
<td>8</td>
<td>Aquifer → Absorption basin (net seepage)</td>
<td>31</td>
</tr>
<tr>
<td>9</td>
<td>Residences → Soil (irrigation with mains water)</td>
<td>29</td>
</tr>
<tr>
<td>10</td>
<td>Rainfall → Roofs, paths</td>
<td>29</td>
</tr>
<tr>
<td>11</td>
<td>Soil → Aquifer (direct recharge)</td>
<td>19</td>
</tr>
<tr>
<td>12</td>
<td>Residences → Sewerage system → Ocean outfall</td>
<td>19</td>
</tr>
<tr>
<td>13</td>
<td>Roofs → Soak wells → Aquifer (indirect recharge)</td>
<td>15</td>
</tr>
<tr>
<td>14</td>
<td>Rainfall → Roads, car parks</td>
<td>13</td>
</tr>
<tr>
<td>15</td>
<td>Rainfall → Trees, large bushes</td>
<td>11</td>
</tr>
<tr>
<td>16</td>
<td>Roofs → Spoon drains/splash blocks</td>
<td>10</td>
</tr>
<tr>
<td>17</td>
<td>Trees → Soil (throughfall)</td>
<td>9</td>
</tr>
<tr>
<td>18</td>
<td>Roads → Absorption basin (runoff, direct recharge)</td>
<td>9</td>
</tr>
<tr>
<td>19</td>
<td>Spoon drains → Aquifer (indirect recharge)</td>
<td>8</td>
</tr>
<tr>
<td>20</td>
<td>Aquifer → Phreatophytes → Atmosphere (transpiration)</td>
<td>6</td>
</tr>
<tr>
<td>21</td>
<td>Roads → Atmosphere (evaporation)</td>
<td>4</td>
</tr>
<tr>
<td>22</td>
<td>Roofs → Atmosphere (evaporation)</td>
<td>3</td>
</tr>
<tr>
<td>23</td>
<td>Absorption basin → Atmosphere (evaporation)</td>
<td>3</td>
</tr>
<tr>
<td>24</td>
<td>Aquifer → Government bores → Soil (groundwater extraction)</td>
<td>2</td>
</tr>
<tr>
<td>25</td>
<td>Spoon drains, paths → Atmosphere (evaporation)</td>
<td>2</td>
</tr>
<tr>
<td>26</td>
<td>Aquifer → Private bores → Soil (groundwater extraction)</td>
<td>2</td>
</tr>
<tr>
<td>27</td>
<td>Rainfall → Absorption basin</td>
<td>2</td>
</tr>
<tr>
<td>28</td>
<td>Tree foliage → Atmosphere (interception)</td>
<td>2</td>
</tr>
</tbody>
</table>
(iv) The cooperation of people in the study areas. Many of the measurements that were made required the cooperation of residents in the study areas (e.g. drilling of holes in lawns and gardens, modification of bores to allow water level measurements to be made, deployment of rain gauges and interception troughs, questionnaires). While cooperation of people in the study was generally excellent, there were limitations on the amount of inconvenience that people could be asked to endure.

As the study progressed, the relative importance of the fluxes listed in Table 4.2 needed to be reconsidered. It was often possible to include extra measurements in the program to take account of these reappraisals but it was probably inevitable that some fluxes did not receive attention commensurate with their importance. For some fluxes (particularly those of recharge), measurements could be made only to a certain depth or degree, after which assumptions about the movement of water had to be made. These assumptions were based on what was understood about conditions in the study areas. Whenever extrapolations or assumptions were made in the water balances, they are detailed so that account may be made for them if later measurements prove them to be unfounded.

The measurement of most water balance components was made at fortnightly intervals over a 86-week period (6th April 1981 to 29th November 1982). The period was chosen so as to include one summer and two winter periods. Some water balance components were measured more frequently (e.g. groundwater levels in the areas around the absorption basins were measured weekly, surface water levels were measured continuously and rainfall was measured daily) while others were measured less frequently (e.g. mains water use was measured monthly).
The next chapter details the methods and results of the measurements of each component of the water balance in the two urban catchments. These measurements are integrated and compared with measurements from the native woodland areas in Chapter 6.
CHAPTER 5

WATER BALANCES - METHODS AND RESULTS

5.1 RAINFALL

5.1.1 Rainfall inputs

Section 3.1 showed that there was appreciable spatial variability in rainfall in the Perth area during 1979-82. The nearest rainfall stations to the study catchments (Fig. 5.1) are at Floreat (3.5 km from the S-SP catchment), Mosman Park (4.7 km from the N-D catchment), Perth (5 km from the S-SP catchment) and Melville (5.8 km from the N-D catchment). To augment these readings, a rain gauge was sited within each of the catchments and read daily by volunteers (on whose properties the gauges were sited). The records from these two gauges showed that only the record from the N-D catchment was kept with sufficient accuracy.

Figure 5.1 shows the Thiessen network for the rainfall stations within the vicinity of the catchments. The gauge in the N-D catchment will be referred to as the Dalkeith rainfall station in the following discussion. The Thiessen network shows that the N-D catchment and the Kings Park woodland area fall within the Dalkeith polygon, while the S-SP catchment and the Selby Street woodland area fall within the Floreat polygon. All subsequent measurements were made with the assumption that rainfall inputs to these study areas equalled the rainfall as measured at these two stations. Any errors involved in this assumption are likely to be greater for the S-SP catchment than for the N-D catchment, due to the spacing of the gauges.

Figure 5.2 shows a histogram of rainfall over 14-day intervals for the Dalkeith and Floreat stations for the 603-day study period. The figures shows a close correlation between the two stations (r = 0.954 for a comparison of daily readings). A paired t-
FIGURE 5.1: Thiessen network for rainfall stations.

FIGURE 5.2: Histograms of fortnightly rainfall for the Dalkeith and Floreat rainfall stations.
test of daily rainfall showed that the Floreat station had a significantly lower rainfall (P < 0.05) than both the Dalkeith and Perth stations (7% lower than Dalkeith and 6% lower than Perth). The Dalkeith and Perth stations were not significantly different. Figure 5.2 shows the rainfall patterns for 1981 and 1982 to be very similar. In both years there were dry periods in early July and late August, followed by average or above-average rainfall in September. The unseasonal rainfall in January 1982 (mentioned in Section 3.1) is also evident from Figure 5.2.

5.1.2 Rainfall partitioning

Planimetry was carried out using 1:1500 aerial photographs (orthophotomaps compiled from 1974 photographs) to determine the proportion of different surface types (see below) in the study areas. The proportions so obtained represent the partitioning of vertically-falling rain. Due to the scattered nature of the different surface types over the catchments, the point counting method of planimetry was used in preference to other methods such as perimeter tracing and paper weighing. The grid size adopted was 15 m x 15 m (on the ground), resulting in over 10 000 points being counted for the two urban catchments. The grid of points was aligned at 60° to the network of streets so that positioning of the grid did not affect the proportions obtained.

Surfaces were divided into five types:

(i) **Roofs and paths:** Rain falling on these surfaces is discharged onto (or into) the soil adjacent to the shedding surface. Some paths may discharge part of their waters onto roads (via driveways) but no attempt was made to distinguish this type of path.

(ii) **Roads and car parks:** Both of these surfaces discharge runoff into drains and hence to discharge points some distance away from the point of rainfall capture (i.e. absorption basins located within the catchment boundaries).

(iii) **Trees and large bushes:** These surfaces were thought to have
enhanced rainfall interception qualities in comparison to open areas.

(iv) **Small bushes, lawns and bare ground:** Some studies (e.g. McIlroy and Angus 1964) have shown that interception is relatively minor for short grass. On the scale of the aerial photographs used, it was not possible to delineate small bushes from lawns and bare ground, so these surfaces were grouped together.

(v) **Free water surfaces** (i.e. absorption basins).

Table 5.1 below presents the results obtained from the planimetry for the two urban catchments and representative areas of native woodland.

<table>
<thead>
<tr>
<th>Area</th>
<th>Percentage of area</th>
<th>Percentage of area</th>
<th>Percentage of area</th>
<th>Percentage of area</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Roofs and paths</td>
<td>Roads and</td>
<td>Trees and</td>
<td>Small bushes</td>
</tr>
<tr>
<td></td>
<td></td>
<td>car parks</td>
<td>large bushes</td>
<td>lawns and</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td>bare ground</td>
</tr>
<tr>
<td>N-D</td>
<td>21.7</td>
<td>9.6</td>
<td>21.6</td>
<td>46.9</td>
</tr>
<tr>
<td>S-SP</td>
<td>28.0</td>
<td>13.1</td>
<td>11.0</td>
<td>46.3</td>
</tr>
<tr>
<td>Native woodland</td>
<td>0</td>
<td>0</td>
<td>56.6</td>
<td>43.4</td>
</tr>
</tbody>
</table>

Table 5.1 shows that the S-SP catchment has a 9.8 per cent greater shedding area (6.3% from roofs and 3.5% from roads) and a 10.6 per cent lower tree canopy area than the N-D catchment. The relatively low percentage of trees and large bushes in native woodland areas confirms that these areas are woodlands rather than forests.

To check the precision of the planimetric method, a 2.8 ha
area in the S-SP catchment was counted twice. The results of the two measurements are presented in Table 5.2. The table shows that the greatest amount of variation occurs in estimating the percentage of trees and large bushes. This result was not surprising as the delineation of tree boundaries on the photographs was complicated by shadows, particularly in native woodland areas. Increasing the precision of the planimetry (e.g. by closing up the grid) is probably not warranted, when more serious errors are likely to result from rain falling at an angle rather than vertically.

Table 5.2

<table>
<thead>
<tr>
<th>Percentage of area</th>
<th></th>
<th></th>
<th></th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Roofs and paths</td>
<td>Roads and car parks</td>
<td>Trees and large bushes</td>
<td>Small bushes, lawns and bare ground</td>
</tr>
<tr>
<td>First estimate</td>
<td>20.3</td>
<td>22.0</td>
<td>8.9</td>
<td>48.8</td>
</tr>
<tr>
<td>Second estimate</td>
<td>20.6</td>
<td>23.8</td>
<td>6.4</td>
<td>49.2</td>
</tr>
</tbody>
</table>

Tables 5.1 and 5.2 also indicate the variability of surface proportions within the urban areas, with a 75 per cent difference in road area and a 25 per cent difference in roof area between the whole catchment (Table 5.1) and a part (Table 5.2).

As planimetry is a rather tedious procedure, the relationship between block size and the proportion of shedding area was tested for the two urban areas.

Antoine (1964) has estimated the percentage of impervious surfaces for three different block sizes in the USA to be as follows:

560 m$^2$ blocks - 80% impervious
560 to 1400 m² blocks - 40% impervious
1400 m² blocks - 25% impervious

The relationship between block size and shedding area in the study areas was determined on both an individual-block, and whole-catchment, basis. Planimetry was carried out on 26 blocks (including road verge) in the two catchments. Figure 5.3 shows the relationship between block size and percent shedding area to be quite variable for the cases examined. The linear relationship fitted to the data is significant at $P < 0.001$, despite the low correlation coefficient ($r = -0.587$). The proportions of shedding area for the small blocks examined are considerably less than those reported by Antoine (1964) for the USA. All the blocks were then grouped together on the basis of being greater than, or less than, 900 m² and average percentages of surface area determined. These percentages are shown in Table 5.3.

Table 5.3
Percentages of surface areas in groups of individual blocks

<table>
<thead>
<tr>
<th>Block size (incl. verge)</th>
<th>Percentage of area</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Roofs and paths</td>
</tr>
<tr>
<td>Blocks &gt; 900 m²</td>
<td>28.2</td>
</tr>
<tr>
<td>(average area 1118 m²)</td>
<td></td>
</tr>
<tr>
<td>Blocks &lt; 900 m²</td>
<td>37.2</td>
</tr>
<tr>
<td>(average area 643 m²)</td>
<td></td>
</tr>
</tbody>
</table>

The table shows that the increase in the percentage of roofs and paths in small blocks relative to large blocks is largely offset by a decrease in the percentage of lawn area.

To find the relationship between block size and percentage of shedding area on a whole catchment basis, the number of blocks
FIGURE 5.3: Relationship between block area and percentage shedding area.

FIGURE 5.4: Histograms of block areas for the N-D (above) and S-SP (below) catchments.
in different size categories was determined from 1:2000 scale maps of property boundaries (i.e. excluding verges). Histograms of block sizes for the two study catchments are shown in Figure 5.4. Block size in the S-SP catchment is more variable than in the N-D catchment. The mean block size in the N-D catchment is about 80 per cent larger than the mean size for the S-SP catchment. This differential is retained when verge areas are added to the property sizes that are shown in Figure 5.4 (i.e. 140 m$^2$ for verges in the N-D catchment and 80 m$^2$ for verges in the S-SP catchment). For the two catchments then, the mean block sizes (including verge) and percentage of shedding area (Table 5.1) are N-D: mean block size 1102 m$^2$, shedding area 31.3 per cent; S-SP mean block size 605 m$^2$, shedding area 41.1 per cent. These percentages of shedding area on a whole-catchment basis are about 2.5 per cent higher than that which would be predicted by the function for individual blocks shown in Figure 5.3. This difference indicates that the area of roads in the catchments exceeds the area of public open space (i.e. parks, ovals and woodlands).

It can be concluded that in areas which are predominantly residential, a reasonable estimate of percentage shedding area can be obtained from a knowledge of average block size. The percentage is higher, however, when considered on a catchment, rather than on an individual block, basis.
5.2 ROOF AND PATH SURFACES

Evaporative 'losses' from the rainfall that falls on roof and path surfaces can occur at two sites - from the surfaces themselves and from the soil once the water runs off the surfaces. These losses will be considered in turn.

5.2.1 Losses from roof and path surfaces

Lvovich and Chernogaeva (1977) assumed that 20 per cent of summer and autumn precipitation was evaporated from roofs in Moscow, but that only 5 per cent of snowmelt water was lost to evaporation during winter and spring. No measurements of loss were made to support the assumptions. In Kursk, USSR, Lvovich and Chernishov (1977) assumed losses from roofs to be practically zero. There are no measurements of evaporative losses from roofs known to the author. Most engineering design specifications (e.g. Martin 1973) are concerned with predicting maximum, rather than actual, flows in downpipes.

Evaporative losses from roofs in Perth can be observed as vapour rising from warm roof materials. At such times, sensible heat stored in the materials is being converted to latent heat. Evaporative losses also take place after storms from water stored in depressions, particularly in partially- (or wholly-) blocked guttering and downpipes. Slightly blocked downpipes increase the likelihood of gutters overflowing during short, intense bursts of rainfall. Marsh (1974) has measured rainfall intensities up to 96 mm/hour over 15 seconds in Perth. The time of concentration for roof surfaces is likely to be less than 15 seconds and therefore overtopping of unblocked gutters will occasionally occur.

To estimate the magnitude of losses from roofs in the study areas, a house in the S-SP catchment was monitored. Three downpipes from the roof were modified so that they directed runoff into 200-litre drums. A rain gauge was sited at roof level on the
western (i.e. windward) side of the house. Later another gauge was sited on the eastern (leeward) side of the house, after it became apparent that updrafts over the roof were resulting in decreased rainfall on the leeward side (also noted by Moss (1960)). Figure 5.5 (a) and (b) shows plan and elevation views of the roof and the locations of downpipes and rain gauges.

As much of the rainfall in Perth is accompanied by southwesterly to north-westerly winds (Vollprecht 1969), the rainfall angle will affect the amount of water captured by the roof. Figure 5.5 (c) shows how the area of the three roof-catchment areas varies with the angle of rainfall (assumed to be driven by a westerly wind). Once the angle of falling rain is less than $30^\circ$ to the horizontal, some of the roof will be in a rain shadow and the amount of rainfall measured by the rain gauges will not truly represent the amount falling on the roof. The roof that was monitored was representative of old houses with relatively high pitched roofs. No corrections were made for rain shadow effects (i.e. all rain was assumed to fall at angles greater than $30^\circ$ to the horizontal).

Figure 5.6 (a) shows the percentage of rainfall intercepted as a function of gross precipitation for 21 storms, each less than 10 mm. Larger storms could not be monitored due to the limited capacity of the collecting drums. No measurements of rainfall intensity were made. The storms were divided into those which occurred when it was thought gutter storage was negligible (dry roof) and those where it was thought there was a carry-over of gutter storage from a previous storm (wet roof). The roofing material (corrugated iron) dried out rapidly between storms. The rain gauge on the leeward side of the roof recorded 9 per cent less rainfall than the rain gauge on the windward side for the few storms where both gauges were deployed. The difference varied from 4 per cent for large storms to 13 per cent for small storms.

Figure 5.6 (b) shows that the variability of interception
FIGURE 5.5 (a): Plan of instrumented roof.

FIGURE 5.5 (b): Elevation of instrumented roof showing the effect of updrafts on rainfall.

FIGURE 5.5 (c): Roof area as a function of rainfall angle.
FIGURE 5.6 (a): Percentage roof interception as a function of gross precipitation.

FIGURE 5.6 (b): Percentage roof interception as a function of gross precipitation and roof wetness.
loss from the roof was so great that the effect of roof wetness was not found to be an important factor for the small data set that was collected. The variability in interception is probably more apparent than real as significant errors due to updrafts are likely in the estimation of rainfall capture. Two hyperbolic functions were fitted to the data (with the constraint that they intercept the ordinate at 100 per cent) as it was considered a hyperbola would be the most likely form of the relationship between the two variables. There is likely to be a threshold rainfall amount before runoff occurs, but as it appears to be less than 0.5 mm, it was ignored. The first model has no constraint to intercept the abscissa while the second model intercepts the abscissa at infinite rainfall. The residual mean square (RMS) for the second model is slightly lower than for the first model, probably due to the greater number of degrees of freedom for this model (Model 2 is a 1-parameter model, Model 1 is a 2-parameter model). However, there is little difference between the two models, or between the functions fitted to the wet and dry roof data (Fig. 5.6 (b)). Both functions show that over 20 per cent of rainfall is lost for storms of less than 1 mm and that about 5 per cent is lost for storms of about 8 mm.

The relationship derived for interception loss by roofs is only approximate, as only one house was monitored and only 21 storms were examined. No account was made for differing roofing materials or guttering examined. As the measurements involved alterations being made to downpipes and also the frequent disturbing of residents, only one roof was monitored. Given the relatively small fluxes involved in roof interception, a major study to improve the estimate was not warranted.

No measurements were made of evaporative losses from paths. The proportion of paths in the total area of roofs and paths, as determined by planimetry, was small. This was partially due to the difficulty in identifying paths on the scale of the aerial photographs (1:1500).
In the water balance calculations (Chapter 6), interception losses from roofs and paths were calculated using the Model 1 equation, daily rainfall amounts in the two urban areas, and the proportion of roofs and paths as shown in Table 5.1.

5.2.2 Losses from beneath discharge sites

In the study areas, roof runoff either enters a soak well or is discharged onto the ground surface (usually via a spoon drain or splash block) when it leaves the downpipe. Table 5.4 shows the percentage of houses having each (or both) of these water disposal devices, as determined by surveying about 120 houses in each catchment.

<table>
<thead>
<tr>
<th>Catchment</th>
<th>Percentage</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Soak wells</td>
</tr>
<tr>
<td>N-D</td>
<td>60.6</td>
</tr>
<tr>
<td>S-SP</td>
<td>46.2</td>
</tr>
</tbody>
</table>

Once the percentage of houses with both disposal facilities is distributed to the other two categories, it is evident that the N-D catchment has about a 15 per cent higher frequency of soak wells than the S-SP catchment.

As the soak wells discharge their water beneath the ground surface, a higher proportion of groundwater recharge is considered likely to take place from these disposal devices, than occurs for spoon drains. Figure 5.7 shows a diagrammatic section through a
FIGURE 5.7: Diagrammatic section through soil profiles showing discharge from a spoon drain and from a soak well in relation to lawn root density.
soil profile which is receiving water from the two devices. The most common sizes of soak wells (61 and 122 cm deep) are shown. Also shown is the theoretical distribution of grass roots in the profile. Carbon (pers. comm.) determined that the roots of couch grass (Cynodon dactylon) can extend to 160 cm in sandy Perth soils. Assuming that the decrease in root density is exponential with depth (Gerwitz and Page 1974; Atkinson 1980) and the density of roots at 150 cm is 5 per cent that at the surface, the function describing the density of lawn roots with depth is:

\[ D_z = D_0 e^{-0.02z} \]

where \( D_z \) = density of roots at depth \( z \) cms
\( D_0 \) = density of roots at the soil surface
\( z \) = depth in cms.

In soak wells most water is likely to issue from the bottom, as the area of contact and hydraulic head is greater at the bottom than at the holes or slots in the sides of the tank. Figure 5.7 shows that the density of lawn roots at the depths of maximum discharge for the two sizes of soak well (61 and 122 cm deep) are 30 and 9 per cent of the density at the surface, respectively. The area under the curve more accurately represents the percentage of roots above these depths. About 74 per cent of roots in the top 150 cm of the soil profile are found above 61 cm, with only about 4 per cent of roots being located beneath 122 cm. This distribution of lawn roots probably approximates the relative loss of water by evaporation (e.g. Camillo and Schmugge 1983) in the top 1.5 metres of the soil profile for these two water disposal facilities.

No attempt was made to measure the movement of soil water beneath spoon drains and soak wells due to difficulties in gaining meaningful measurements in a soil water system that varies in three dimensions. There were also difficulties in gaining access to soak wells as they are often covered with paving. An
indication of deep drainage beneath spoon drains and soak wells could be gained by measuring the chloride concentration of soil water beneath them (e.g. Sharma et al. 1983). However, to determine recharge rates, a number of soil cores would be required due to the non-steady state conditions existing in the soil profile. The destructive nature of the method precludes a number of measurements at the same site, particularly when the site is small. The proportion of roof- and path- waters that becomes indirect recharge will be considered again in Section 5.7.7.2.
5.3 ROAD AND CAR PARK SURFACES

5.3.1 Method of determining the rainfall-runoff relationship

The distribution of road drains in the two catchments is shown in Figure 5.8 (a,b). Both catchments are located in old suburbs and the records of drain locations are not complete. The location of some of the drains shown in Figure 5.8 has therefore been inferred from the location of gully pits.

The central absorption basin in the N-D catchment is known as Masons Gardens, while that in the S-SP catchment is known as Shenton Park Lake. Both basins are wetlands which are surface expressions of the water table.

Both catchments have drains bringing (or importing) road runoff waters from an adjacent catchment (Fig. 4.1). In the N-D catchment, the imported waters enter by gravity drainage. In the S-SP catchment, the imported waters are pumped into the central absorption basin from an adjoining absorption basin. The pumps are started automatically once the water level in the adjoining basin reaches a particular height. The S-SP catchment also has a drain which exports water from the central absorption basin to an ocean outfall. This drainage occurs under gravity and commences once a particular height is achieved in Shenton Park Lake.

The method used to determine the rainfall-runoff relationship for each of the catchments was to measure the change in storage in the absorption basins immediately following a measured rainfall. Direct measurements of runoff in all of the drains entering the basins was not possible owing to some of the drain outlets in each basin being submerged. Direct measurements would also have required the installation of numerous water level/flow recorders, or the assistance of several people, as runoff into the basins was found to be very rapid.
FIGURE 5.8 (a): Distribution of road drains in the N-D and adjoining catchment.

(Sources: Nedlands City Council maps, field verification).
FIGURE 5.8 (b): Distribution of road drains in the S-SP catchment.  
(Sources: City of Subiaco and MWA maps, field verification).
Both basins were surveyed using a dumpy level, access to the deepest part of Shenton Park Lake being achieved by boat. The basin bathymetries are shown in Appendix I.

From the bathymetry maps, level-volume and level-area curves were calculated. These curves are shown in Figure 5.9. The figure shows that the area of Masons Gardens increases rapidly with rising water levels. The area of Shenton Park Lake changes little between 4 and 5 metres AHD due to a stone retaining wall around the basin between these levels.

Continuous water level recorders were installed in each basin (an Alpina Werk float recorder with mechanical timer in Masons Gardens and a Rimco bellows recorder with battery-driven timer in Shenton Park Lake). Incomplete records were obtained from the two recorders due to problems of mechanical breakdown and vandalism, and difficulties in collecting records when water levels in the basins were high. Sufficient data were collected over the 20 months of the study, however, to derive the rainfall-runoff relationships shown in Figure 5.10.

The method used to derive the rainfall-runoff relationships is likely to underestimate runoff, as some infiltration of water will take place while runoff is occurring. Water levels in Masons Gardens in particular were noted to decline once runoff ceased (e.g. 1 cm/h from 2.4 to 2.3 m AHD, 0.5 cm/h from 1.9 to 1.8 m AHD). Some of the runoff values for the larger, and usually longer, storms in the Masons Gardens catchments appear to be lower than expected (see Fig. 5.10), perhaps due to infiltration losses. These values result in a positive intercept on the runoff axis for the rainfall-runoff function. The method was considered not sufficiently accurate to estimate the depression storage for each catchment.

5.3.2 Analysis of runoff in the N-D catchment

The contribution of runoff from the adjoining catchment was
FIGURE 5.9: Level-area and level-volume curves for Masons Gardens (above) and Shenton Park Lake (below).
FIGURE 5.10: Rainfall-runoff relationships for the N-D (above) and S-SP (below) catchments.
estimated by two methods. The first method assumed the amount of runoff was proportional to the relative area of the adjoining catchment. Using this method it was estimated that 38 per cent of runoff originated in the adjoining catchment. The second method involved measuring the proportion of runoff that was contributed by the eastern drain (which brings the imported water in addition to some intra-basinal water) and then apportioning the runoff in this drain to the relative contributing areas. Flow in the eastern drain during a storm was determined from measurements of depth of flow in the drain. Flow velocities were estimated by the Manning equation, i.e.

\[ v = \frac{R^{2/3} S^{1/2}}{n} \]

where
- \( v \) = average velocity (m/s)
- \( R \) = hydraulic radius (m) = cross sectional area/wetted perimeter.
- \( S \) = slope of the energy line (here taken as the slope of the drain)
- \( n \) = Manning's roughness coefficient.

From the measurements it was determined that 43 per cent of road runoff enters Masons Gardens through the eastern drain. The adjoining catchment was assumed to contribute most (76%) of this water on the basis of contributing area. In all, then, 33 per cent of runoff into Masons Gardens was estimated to arise from the adjoining catchment. This estimate is 5 per cent lower than the previous one. Possible reasons why the adjoining catchment would contribute less runoff than predicted, on the basis of relative area, include:

(i) losses in the drainage system in transporting the water to the N-D catchment, and

(ii) less runoff generated due to the presence of a soakage
pit (Fig. 5.8) and to a significant proportion of the adjoining catchment being a golf course.

Assuming that 67 per cent of the runoff shown in Figure 5.10 arises from within the N-D catchment, this is equivalent to 100 per cent runoff from 4.6 per cent of the catchment area of 68.07 ha. Planimetry has already shown that the road and car park area of the N-D catchment is 9.6 per cent. Possible reasons for the difference include:

(i) Evaporation from road surfaces: As for roof surfaces, vapour was observed to rise from warm roads.

(ii) Evaporation from depression storages: Depressions occur within the roads and gutters, as well as in the drainage system itself. In both catchments many of the gully pits consist of sealed boxes with the drain outlet about 50 cm above the bottom of the pit. Water in the gully pits evaporates between storms.

(iii) Leakage from gully pits: The most recently installed set of drains in the N-D catchment (the NE drainage system) include gully pits with open bottoms. Infiltration of water from these pits will be similar to infiltration from soak wells. The construction of older gully pits is unknown, but some of these may also have unsealed bottoms.

(iv) Overestimation of the catchment boundary: With the lack of accurate maps showing the location of drains, it is possible that the catchment boundary was overestimated.

(v) Underestimation of runoff volumes: This problem has already been mentioned, as has the relatively rapid infiltration of waters in Masons Gardens.

It is likely that all of the above factors contribute to some extent to the discrepancy between runoff volumes and road
area. However, the relative importance of each factor is unknown.

To help estimate the amount of infiltration which occurs during runoff, cores were taken from eight sites in the Masons Gardens basin. The results of the coring are shown in Appendix I. All cores taken from sites below 1.8 m AHD contained sections of either saprolite or peat. Some cores in the central part of the basin also contained some clay. Two peaty sections were evident from cores taken in the eastern part of the basin, separated by sand. This sequence probably indicates that the original swamp was inundated by sand once the road drains were constructed. This was then followed by the present, swampy conditions. In the deeper sections in the west of the basin, saprolite largely replaces peat in the sequence. A market garden in the Masons Gardens area had to be abandoned once water levels rose in the 1920s (Subiaco Post 2.2.1982). The area was then used as a sanitary landfill site until about 1950, when the present park was created (N. White, pers. comm.). The existence of the peat, saprolite and clay in the basin almost certainly inhibits groundwater exchange with the surface waters until the sandy banks are inundated. Infiltration during runoff is therefore unlikely to account for all of the difference between expected and measured runoff to the basin.

In the water balance calculations in Chapter 6, runoff was assumed to be 10 per cent greater than that measured, to take account of basin infiltration losses. Twenty per cent of rainfall falling on roads was assumed to be added to the soil from open gully pits and leaking pipes. The residual amount of water was assumed to be lost by evaporation from roads and gully pits. Even with these adjustments, losses from the road drainage system are about 30 per cent of rainfall. The adjustments for infiltration losses from the basin and open gully pits are unlikely to be much larger than those assumed. Therefore there must be either additional leakage from the drainage system and/or significant evaporative losses from the roads and drainage system.
5.3.3 Analysis of runoff in the S-SP catchment

To determine the amount of water being contributed to Shenton Park Lake from the adjoining catchment, the hours of pumping were obtained on a weekly basis from the Metropolitan Water Authority. Discharge rates were obtained for the pumps while operating separately and in series.

The rainfall-runoff relationship obtained for the S-SP catchment (Fig. 5.10) is equivalent to 100 per cent runoff from 16.5 per cent of the catchment of 156.87 ha. This is more than from the road and car park area of 13.1 per cent. As losses will occur due to evaporation from roads and depression storages, in addition to runoff being underestimated by the method used, the discrepancy will almost certainly be greater. Only a very small drainage system in the S-SP catchment has gully pits with open bottoms (installed in May 1982), so leakage from gully pits is probably minor in this catchment.

Possible reasons for the discrepancy include:

(i) Underestimation of shedding area: A number of driveways and paths conduct water onto the roads in this catchment, but these surfaces were not discernable on the aerial photographs used for planimetry. The small blocks in this catchment also require many residents to park on the verge rather than in garages. Soil runoff was observed on some verges where the soil had become compacted by traffic. The verges in this catchment were also much narrower and it is common for street trees to overhang the roads (this occurs to a lesser extent in the N-D catchment). The area of roads will therefore be underestimated by planimetry. Throughfall will occur onto roads once canopy storages are exceeded.

(ii) Inclusion of drainage water from the adjoining
catchment in the estimate of road runoff: Pumping of water into Shenton Park Lake sometimes took place while runoff was occurring. This would tend to overestimate the runoff volumes. Care was taken when examining the water level record to exclude cases where pumping was thought to have been taking place. However it is possible that some imported waters were included in the runoff estimation.

(iii) Underestimation of the catchment boundary: The location of drains in the south-east of the S-SP catchment is particularly unclear and it is possible that additional runoff waters enter the catchment from a major road in this area.

Again, each of the above factors is likely to contribute to some extent to the discrepancy between runoff volumes and road area. The different response of the two catchments highlights the need to measure road runoff, rather than estimate values from the percentage of roads and car parks in the catchment with the incorporation of a loss factor. A more direct method of measuring road runoff would have been preferable, but was not possible given the number and location of drain outlets.

Soil cores were taken from eight sites within Shenton Park Lake to help understand the exchange that occurs between groundwater and surface water. The results of this coring are shown in Appendix I. All sites had an upper section of about 50 cm of saprolite. In the eastern part of the lake the saprolite was underlain by sand (probably brought in by drains) and a peaty section, followed by coarse, clean sand. The peat was absent from the cores in the east where it was replaced by a clayey sand which was organic-rich at the top. Shenton Park Lake was originally used as a sports field before rising water levels caused it to be abandoned. Like Masons Gardens it was used as a sanitary landfill site until about 1950 (W. Flood, pers. comm.). The saprolite, peat and clayey sand would all inhibit groundwater exchange to some extent. Water levels in the lake would have to exceed five
metres AHD before the sandy banks are inundated.

In the water balance calculations in Chapter 6, 5 per cent of tree canopies were assumed to overhang roads. The percentage of road surfaces was also increased by 2 per cent to take account of runoff from driveways and verges. Evaporation from roads and gully pits was taken as 10 per cent of rainfall falling on road surfaces. These adjustments were necessary to distribute the unaccounted water to its most likely origins, as was necessary in the N-D catchment.
5.4 BASIN WATER BALANCES

Separate water balances were carried out on the absorption basins in the two experimental catchments to ascertain the seasonal influences the basins have on groundwater quantity.

5.4.1 Water balance of Masons Gardens in the N-D catchment

The water balance equation for the surface waters in Masons Gardens is:

\[ R + P + G_1 = E + G_o + \Delta S \]

where 
- \( R \) = road runoff (divided into intra- and inter-basinal)
- \( P \) = direct precipitation inputs
- \( G_1 \) = groundwater inflow
- \( E \) = evaporation
- \( G_o \) = groundwater outflow
- \( \Delta S \) = change in surface water storage

Twelve piezometers were installed around the basin (9 shallow, 3 deep) in an attempt to calculate groundwater inflow and outflow to the basin. The piezometers were installed using a combination of hammering, sludging and jetting. The holes were cased with 50 mm PVC, slotted over the bottom two metres. The deep piezometers were placed with the slotted position about six metres below the water table. Water levels in the piezometers and in the basin were measured weekly. A wooden stake was driven into the ground beside the piezometers. The reduced level of the stake was surveyed for its use as a datum point.

The network was considered insufficient to measure accurately the vertical heads around the basin, so net groundwater movements could be deduced only by difference in the water
balance equation, i.e.

\[ G_i - G_o = E + \Delta S - R - P \]

Inputs of road runoff to the basin were estimated by the rainfall-runoff relationship shown in Figure 5.10, incorporating the adjustment for basin infiltration mentioned in the previous section. Direct additions of rainfall were estimated by multiplying the average basin area for the week (obtained from the level-area curve shown in Fig. 5.9) with the weekly rainfall. Interception by overhanging willow trees (Salix babylonica) was ignored, as most rainfall occurs during the winter when these trees are bare. Evaporation from the open water surface was estimated by multiplying the average basin area by the weekly pan evaporation, with the incorporation of a lake-to-pan coefficient (Hounam 1973). The only estimation of a monthly lake-to-pan coefficient for a fresh water body in the Perth area is that carried out by Hoy and Stephens (1979) for the Mundaring Reservoir. These lake-to-pan factors are probably not very applicable to Masons Gardens due to differences in heat storage abilities (Masons Gardens is very shallow while Mundaring Reservoir is deep). However it will be seen later that the evaporative flux constitutes only a minor part of the overall water balance and therefore any error introduced by the choice of pan factors is likely to be minor.

The water balance components were calculated on a weekly basis and the values are given in Appendix 2. These values were cumulated to provide an annual water balance for an hydrologic year (i.e. the period between low water levels) and a comparison of the 1981 and 1982 'winters' (the same 32 week period monitored in both years). An 'hydrologic year', rather than a calendar year, was calculated as this was considered to be more meaningful. These values (converted to depths by dividing the volumes by the N-D catchment area) are shown in Table 5.5.
TABLE 5.5

Water balance of the Masons Gardens absorption basin for extended periods
(All figures in mm)

<table>
<thead>
<tr>
<th>Period</th>
<th>Total runoff</th>
<th>Runoff from N-D catchment</th>
<th>Runoff from adjoining catchment</th>
<th>Direct rain additions</th>
<th>Evaporation</th>
<th>Net additions to groundwater</th>
<th>Change in surface storage</th>
</tr>
</thead>
<tbody>
<tr>
<td>Hydrologic Year</td>
<td>64</td>
<td>43</td>
<td>21</td>
<td>3</td>
<td>2</td>
<td>65</td>
<td>0</td>
</tr>
<tr>
<td>(5.5.81-7.3.82)</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Winter 1981</td>
<td>55</td>
<td>37</td>
<td>18</td>
<td>2</td>
<td>1</td>
<td>55</td>
<td>1</td>
</tr>
<tr>
<td>(6.4.81-15.11.81)</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Winter 1982</td>
<td>46</td>
<td>31</td>
<td>15</td>
<td>2</td>
<td>1</td>
<td>46</td>
<td>1</td>
</tr>
<tr>
<td>(5.4.82-14.11.82)</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>
Table 5.5 shows that there is a close seasonal and annual correspondence between runoff and net groundwater additions. The direct rainfall and evaporation fluxes are very minor in comparison to the runoff and net groundwater fluxes. Direct rainfall additions are greater than evaporative losses, despite annual pan evaporation being more than twice annual rainfall, due to the basin area being four to five times larger in winter than in summer. Changes in basin storage constitute a minor part of the water balance. There was about 16 per cent less runoff and net groundwater additions during the drier, 1982 winter.

Figure 5.11 shows four-weekly totals of each water balance component for the 86-week study period (the values for the last two-week period have been doubled to remain on the same scale). The figure shows that there was also a close monthly correspondence between road runoff and net groundwater addition (i.e. there was little lag). There were only five weeks (not consecutive) during the 1981-82 summer when there was a net groundwater inflow to the basin (Appendix 2(a)). For most of the summer months the fall in water levels in the basin (see Section 5.10) was greater than that which could be attributed to evaporation from open water. While the 1981-82 summer was not very typical (due to the unseasonal January rainfall) it was expected that there would be a net groundwater inflow for much of the summer, as was found by Allen (1979) in a water balance of Lake Jandabup, north of Perth.

Water table contours were constructed from the network of nine shallow piezometers and the water level in the basin, to examine groundwater flow directions throughout the year. Figure 5.12 (a) and (b) shows the water table contours for weeks when the basin appears to be recharging and discharging the groundwater respectively. In Figure 5.12 (a) all the piezometric pairs show a downward head and the regional groundwater flow direction (north to south) was locally reversed. In Figure 5.12 (b) all the piezometric pairs shows an upward head and groundwater is being
FIGURE 5.11: Four-weekly totals of water balance components for Masons Gardens absorption basin, N-D catchment.
FIGURE 5.12 (a): Water table contours around Masons Gardens basin during a recharge period.

LEGEND: ● DEEP PIEZOMETER ○ SHALLOW PIEZOMETER

8.11.1982

FIGURE 5.12 (b): Water table contours around Masons Gardens basin during a discharge period.
diverted towards the basin. For 35 per cent of the year, water levels in the basin are below the average water level in the east and west piezometers, indicative of a discharge state. The piezometric pair immediately upgradient of the basin show an upward head for about 75 per cent of the year, while the pair downgradient show an upward head for about 25 per cent of the year. The piezometric pair located farthest from the basin (and upgradient) showed almost no vertical component of head throughout the year.

The water balance calculations of surface waters indicated that the basin acts as a net discharge site for only about 10 per cent of the year, while groundwater potentials indicate the basin is a discharge point for about 35 per cent of the year.

Transpiration by the 3000 m² of willow foliage in the basin must be responsible for the discrepancy between these estimates of discharge period. In a more normal year, the proportion of the year when the basin is a discharge site will almost certainly be greater. A regional decline in groundwater levels during the summer period due to groundwater pumping could also cause water levels in the basin to decline faster than can be attributed to basinal evaporation. However, the presence of converging flow lines and upward heads indicates that the basin is a local discharge point during the summer months. By examining the water balance values and the intensity of the groundwater gradients around the basin (Fig. 5.12), it is clear that over the year as a whole the basin is a net groundwater recharge site. Water use by the willows in Masons Gardens will be considered again in Section 5.9.

### 5.4.2 Water balance of Shenton Park Lake in the S-SP catchment

The water balance equation for the surface waters in Shenton Park Lake is:
\[ R + P + G_1 + Pu + F = E + G_0 + D + \Delta S \]

where \( R, P, G_1, E, G_0 \) and \( \Delta S \) are as previously defined

\( Pu = \) pumped waters from an adjoining basin
\( F = \) fountain additions
\( D = \) drained water (to an ocean outfall).

The factors which have been discussed previously in the water balance of Masons Gardens were determined in an equivalent manner for Shenton Park Lake. Adjustments were made to runoff to take account of the factors mentioned in Section 5.3. Pumped water and fountain additions were calculated from pumping time and pump supply rates. The water for the fountain comes from the aquifer immediately south-east of the lake. From water level observations, it was evident that the lake lies within the cone of depression of the bore.

To determine the amount of water draining from the lake to the ocean outfall, a rating curve was determined for the box drain already mentioned. Velocity of flow in the drain was determined for different stages (levels) using a Marsh-McBirney Model 201 flow meter. Both cross-sectional area of flow and velocity of flow increased linearly with stage, so that flow (the multiple of area and velocity) increased non-linearly with stage. Despite a good relationship being established between stage and flow \((r = 0.998\) for 6 points), application of the rating curve to the continuous water level record of the lake resulted in excessively high estimates of outflow. It was concluded that this method of estimating outflow was inherently unstable, as a small error in the water level record results in a large error in the estimate of outflow. This was due both to the non-linear relationship between the two variables and to the long periods (i.e. days) over which drainage occurred.

The method finally used to estimate drainage was to derive a relationship between net groundwater outflow and two controlling
variables (lake level and groundwater gradient away from the lake) for periods when no drainage was taking place from the lake and to apply this equation during weeks when the drainage was known to take place. Drainage therefore becomes the unknown in the water balance equation during those weeks in which it took place. The relationship obtained between net groundwater outflow and the controlling variables was:

\[ G_0 - G_1 = -11014 + 330 L^2 + 12860 G \ (r = 0.715) \]

where \( G_0 - G_1 \) = net groundwater outflow \( (m^3) \)
\( L \) = average weekly lake level \( (m \ AHD) \)
and \( G \) = groundwater gradient away from the lake (as measured by the difference in level between the lake and the SW piezometer).

All the coefficients in the multiple regression are significant at \( P < 0.001 \). Net groundwater outflow was found to be non-linearly related to lake level. With increasing lake levels, the area of contact between the surface waters and groundwaters increases. The effect of lacustrine sediments on groundwater exchange also decreases as levels rise, as was mentioned in the previous section. Between 4 and 5 m AHD, the lake is contained within a stone retaining wall, and water level rises in this interval take place with little increase in lake area (see Fig. 5.9). However, the hydraulic head across the lake sediments increases rapidly when the water level rises between these levels.

As for the Masons Gardens basin, a series of piezometers were established around the lake (10 shallow, 4 deep) and read at weekly intervals. Again, this network was considered insufficient to separate groundwater inflows from outflows so only net groundwater movements were estimated.

Weekly water balance values are shown in Appendix 2. Table 5.6 shows the cumulative water balance values for the
hydrologic year 1981/82 and for the 1981 and 1982 winters. The hydrologic year for Shenton Park Lake was longer than that for Masons Gardens.

Table 5.6 shows that only about half of the total additions to the lake in 1981/82 were added to the groundwater. About 40 per cent of the added water was drained to an ocean outfall. Evaporation from the lake was about 70 per cent higher than rainfall, more closely reflecting the relative rainfall and pan evaporation values, as the surface area of Shenton Park Lake does not change seasonally by the same percentage as Masons Gardens.

Road runoff from the S-SP catchment was about 15 per cent lower in the drier, 1982 winter than in 1981. However the amount of water that was pumped from the adjoining catchment, and the amount of drained outflow, was similar for the two winters. Both these latter fluxes are dependent upon water levels in the two basins. As a consequence of the January 1982 rainfall, water levels in the basins were higher at the beginning of the 1982 winter than at the beginning of the 1981 winter.

Figure 5.13 shows the four-weekly totals of the water balance components for Shenton Park Lake. While net groundwater additions are again closely related to road runoff, the former was generally smaller in magnitude. There is some evidence of carryover in groundwater additions from the January runoff. Pumping and drainage outflow were mainly restricted to a 20-week period in both winters. Drainage of the lake can begin early in the winter (e.g. May 1981) if early winter rains are heavy. This outflow occurs despite groundwater levels being close to their end-of-summer low levels. Rainfall and evaporation fluxes show a definite seasonal cycle due to the relative constancy of lake area throughout the year.
**TABLE 5.6**

Water balance of Shenton Park Lake absorption basin for extended periods
(all figures in mm)

<table>
<thead>
<tr>
<th>Period</th>
<th>Runoff from S-SP catchment</th>
<th>Direct rain additions</th>
<th>Pumped water added</th>
<th>Fountain water added</th>
<th>Evaporation</th>
<th>Drained outflow</th>
<th>Net additions to groundwater</th>
<th>Change in surface storage</th>
</tr>
</thead>
<tbody>
<tr>
<td>Hydrologic year (5.5.81-2.5.82)</td>
<td>125</td>
<td>14</td>
<td>95</td>
<td>22</td>
<td>27</td>
<td>98</td>
<td>123</td>
<td>8</td>
</tr>
<tr>
<td>Winter 1981 (6.4.81-15.11.81)</td>
<td>103</td>
<td>12</td>
<td>92</td>
<td>3</td>
<td>11</td>
<td>94</td>
<td>96</td>
<td>9</td>
</tr>
<tr>
<td>Winter 1982 (5.4.82-14.11.82)</td>
<td>89</td>
<td>10</td>
<td>92</td>
<td>4</td>
<td>11</td>
<td>101</td>
<td>79</td>
<td>4</td>
</tr>
</tbody>
</table>
FIGURE 5.13: Four-weekly totals of water balance components for Shenton Park Lake, S-SP catchment.
From the water balance calculations there was only one week during the 1981/82 summer when there was a net groundwater inflow to the lake. This was due to the addition of groundwater from the fountain during the summer months. There were seven weeks during the summer when net groundwater outflow was less than the amount added by the fountain. Presumably net groundwater inflow would have taken place during these weeks had the fountain not operated. The unseasonable January rainfall probably also decreased the number of weeks when net groundwater inflow would have taken place.

Figure 5.14 (a) and (b) shows the water table contours around Shenton Park Lake during periods when the basin appears to be recharging and discharging the groundwater respectively. In Figure 5.14 (a) the three piezometric pairs closest to the lake show a very strong downward head while the most upgradient pair shows a weak upward head. As for Masons Gardens, the regional groundwater flow direction (north to south) is locally reversed. In Figure 5.14 (b) the upgradient piezometric pair closest to the lake has a strong upward head. Difficulty was encountered in drilling the deep piezometer of this pair, due to heaving of sand under the strong upward gradient. The two downgradient piezometers in Figure 5.14 (b) have downward heads while the northernmost pair shows almost no vertical head.

Water levels in Shenton Park Lake are higher than the eastern piezometer for about 85 per cent of the year. An upward head exists in the upgradient piezometric pair that is closest to the lake for about 60 per cent of the year while both downgradient piezometers show a downward head throughout the whole year. The northernmost piezometric pair shows almost no component of vertical head throughout the year, as was the case for the equivalent pair in Masons Gardens.
FIGURE 5.14 (a): Water table contours around Shenton Park Lake during a recharge period.

FIGURE 5.14 (b): Water table contours around Shenton Park Lake during a discharge period.
Shenton Park Lake appears to be a net recharge site for most of the year despite having a large expanse of water and about 2250 m$^2$ of willow foliage. Possible reasons for the lake not being a net discharge site for much of the summer include the addition of fountain waters and the presence of low permeability lacustrine sediments at the bottom of the lake. The sediments, in addition to the stone retaining walls, would delay the decay of any recharge mound that is produced by water additions to the lake. The lake sediments would also hinder the recharging of the groundwater during winter. Groundwater exchange is probably greatest in the west of the lake where lacustrine sediments are thinnest.
5.5 MAINS WATER USE

5.5.1 Background

Caldwell (1981) noted that, prior to the imposition of water restrictions in 1977, water usage per person in Perth was about 30 per cent higher than in other Australian cities with a comparable level of metering. Metered water use has yet to return to pre-restriction levels (Metropolitan Water Authority Annual Report 1981/82). Possible reasons for the decrease in metered water usage per person include the progressive introduction of a pay-for-use scheme and the increased use of groundwater by private bore owners. A survey by the Australian Bureau of Statistics (1979) estimated that about 27 000 of the 54 900 private bores then being used in Perth, had been installed in the three-year period 1977-79.

Previous studies of mains water use in Perth have been carried out by Fleay (1968), Williamson (1975), Tondut (1976) and Nichol (1979). Fleay (1968) found the correlation between annual mean daily consumption and mean maximum daily temperature to be 0.83. Williamson (1975) estimated that on an annual basis, about 55 per cent of mains water was used on gardens, the proportion being 70 per cent during the summer months. Tondut (1976) monitored the metered water consumption of 111 households over 16 days and found that households in new districts used considerably more water than those in intermediate and old districts. Tondut also found that consumption per person decreased as the number of people per dwelling increased. From the survey, Tondut estimated that about 67 per cent of mains water that is consumed annually is used on gardens (this rising to 85% on hot days). Nichol (1979) correlated the weekly water use of nine houses over a 12-week period with mean maximum temperatures and found a correlation of 0.85. Rainfall also seemed to affect consumption.

In an Australia-wide study, McMahon and Weeks (1973) found
that mean maximum temperature and/or monthly pan evaporation were the climatic variables that were most related to monthly water use in cities with hot, dry summers. The authors also found that the number of rainy days per month was more significant than monthly rainfall in predicting monthly water use. On a weekly basis, total pan evaporation was found to be the climatic variable that most influenced water use.

A major study of mains water use is presently being undertaken by the Metropolitan Water Authority but the results are not yet available.

5.5.2 Leaks in reticulation and sewerage systems

Leaks from the reticulation system are estimated by the Metropolitan Water Authority by measuring flows in the mains during the early morning hours in winter. Such leaks include losses to the soil as well as losses to the sewerage system (or to septic tanks in non-sewered areas). Losses to the soil are estimated to comprise about 75 per cent of total losses (P. Downing, pers. comm.).

Leakages in the N-D catchment area were estimated at 4.4 litres/hour/service (standard error (SE) = 0.6 L/h/serv.) and 2.1 L/h/serv. (SE = 0.1 L/h/serv.) for the S-SP catchment. When multiplied by the number of houses in each catchment, losses to the soil become 43 m$^3$/d for the N-D catchment, and 82 m$^3$/d for the S-SP catchment. As water mains are two to three metres beneath the ground surface, it is probable that tree roots will intercept some water before it recharges the groundwater. The ability of tree roots to increase root densities in areas of favourable water supply has been observed in the soils of the Perth area.

Only 10 per cent of reticulation losses to the soil were assumed to become recharge in the water balances. The relatively small quantities of water that are involved in the leakages make
any error in this estimate of little consequence to the overall balances.

As mentioned in Chapter 2, infiltration of sewerage system pipes is more likely than exfiltration. However, only a small proportion of sewerage pipes are below the water table in the two urban catchments. In areas where up to a third of wastewater is thought to be groundwater, a large proportion of the pipes are beneath the water table. In the water balance calculations in Chapter 6, infiltration of groundwater was considered to be greater in winter than in summer, and also greater in the S-SP catchment which has a greater proportion of its sewerage pipes below the water table.

5.5.3 Methods of estimating mains water use

An examination was carried out of annual metered water use during 1979/80 in rating districts which include the study catchments. The N-D catchment lies within the Nedlands and Dalkeith rating districts while the S-SP catchment lies wholly within the Subiaco rating district. Meters which had been installed for fewer than 12 months were excluded from the examination. Data from all meters in the Nedlands and Dalkeith districts were available while only about 20 per cent of meter data were available for the Subiaco district.

Figure 5.15 shows the distribution of metered water use for households in the three districts. The distributions are skewed, with modal values just above the basic water allowance of 150 m³, above which charges are made for excess water use. The means of the Nedlands and Dalkeith rating districts are not significantly different from each other at $P < 0.05$ whereas both these districts have significantly higher means than the Subiaco rating district at the $P < 0.001$ level.

The mean consumption and standard deviations for the three
FIGURE 5.15: Distribution of metered water use for households in the Dalkeith, Nedlands and Subiaco rating districts.
districts are shown in Table 5.7. These data were used to estimate the sample size that would be required to estimate the mean water consumption for the district, with an error of less than 10 per cent of the mean and with a probability of greater than 95 per cent. The sample sizes thus calculated are also shown in Table 5.7.

**TABLE 5.7**

<table>
<thead>
<tr>
<th>Rating district</th>
<th>Mean consumption ($m^3$)</th>
<th>Standard deviation ($m^3$)</th>
<th>No. of meters required for sample</th>
</tr>
</thead>
<tbody>
<tr>
<td>Nedlands</td>
<td>370.7</td>
<td>232.3</td>
<td>107</td>
</tr>
<tr>
<td>Dalkeith</td>
<td>385.1</td>
<td>283.9</td>
<td>144</td>
</tr>
<tr>
<td>Subiaco</td>
<td>271.4</td>
<td>176.9</td>
<td>117</td>
</tr>
</tbody>
</table>

Considering that the N-D catchment lies within both the Nedlands and Dalkeith rating districts, it is evident from Table 5.7 that about 120 meters need to be read in each of the study catchments to estimate mean consumption with a 10 per cent error.

A survey by questionnaire was carried out to help determine water use (both mains and groundwater) in the two catchments. At the same time, permission was obtained to read about 130 meters in each catchment. A copy of the questionnaire which was distributed is shown in Appendix 3.

At the time of the survey there was some concern by bore owners that the Government would require private bores to be licensed within the near future. There was therefore a strong possibility of a bias in the survey due to bore owners failing to complete the questionnaire. To ensure a good response, the questionnaire was distributed to letter boxes on a Wednesday and personally collected on the weekend when people would be more
likely to be at home. All houses were visited on the Saturday and revisited on the Sunday if no response had been obtained. Further collections were made on the following Wednesday and weekend. Using this method, a 78 to 80 per cent response to the questionnaire was obtained. Table 5.8 shows that the main reason for non-response to the survey was the absence of householders.

TABLE 5.8

<table>
<thead>
<tr>
<th>Questionnaire - Details of non-respondants</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
</tr>
<tr>
<td>N-D catchment</td>
</tr>
<tr>
<td>----------------</td>
</tr>
<tr>
<td>Number of houses surveyed</td>
</tr>
<tr>
<td>Percentage of houses in the catchment surveyed</td>
</tr>
<tr>
<td>Percentage response to survey</td>
</tr>
</tbody>
</table>

Reason for non-response (per cent):

<table>
<thead>
<tr>
<th>Reason</th>
<th>N-D</th>
<th>S-SP</th>
</tr>
</thead>
<tbody>
<tr>
<td>Householder absent/house unoccupied</td>
<td>13.5</td>
<td>16</td>
</tr>
<tr>
<td>Refusal</td>
<td>3.5</td>
<td>3.5</td>
</tr>
<tr>
<td>New owner/householder sick</td>
<td>3</td>
<td>2.5</td>
</tr>
</tbody>
</table>

All houses in the N-D catchment were surveyed to obtain complete information on groundwater use in this catchment. More details on groundwater use are given in Section 5.6.

To determine the amount of mains water used outdoors, the frequency and length of watering was determined for different types of outdoor areas (i.e. lawns, gardens, road verges and vegetable gardens) in addition to the average number of sprinklers used during watering. Information on the time of day that watering was carried out was obtained to help estimate evaporative losses. Factors thought to influence mains water use were also surveyed (e.g. number of people, presence of babies, rainwater tanks, swimming pools and bores).
5.5.4 Results

Table 5.9 shows the results obtained from the questionnaire which are relevant to the estimation of mains water use. The averages and percentages shown have been calculated only from those households which answered that particular question. Where possible, comparisons have been made with statistics for Perth or for WA.

The study catchments have fairly similar occupancy ratios to the rest of Perth but, in comparison with the state population, the areas have a higher proportion of old people. The N-D catchment had about a 20 per cent higher occupancy ratio than the S-SP catchment. Most houses in both areas were connected to the sewerage system. The N-D catchment exceeded the S-SP catchment in frequency and duration of watering, in addition to the number of sprinklers operated during watering. Almost all of the fixed reticulation systems are operated from bores. While bore use in the N-D catchment was approaching 50 per cent, only about 5 per cent of houses surveyed in the S-SP catchment used groundwater for garden irrigation. Estimates of outdoor water use (all sources) during summer show that households in the N-D catchment use more than three times as much water as do households in the S-SP catchment. Bore users in the N-D catchment used about 2 1/2 times as much water as non-bore users.

The meters read during the study period were located on about seven streets in each catchment. Care was taken to locate the streets in different areas of the catchments and also to include a representative distribution of block sizes. The decision to read meters in particular streets, rather than at random throughout the catchment, was taken so as to reduce travelling times and costs. Meters in the catchments were read at four-weekly intervals, alternate catchments being measured each fortnight.
### TABLE 5.9

**Questionnaire results on mains water use**

1. Occupancy ratio (people/household): N-D 3.09; S-SP 2.55; Perth 2.86.

2. Percentage of people in each age group:

<table>
<thead>
<tr>
<th>Age last birthday</th>
<th>N-D</th>
<th>S-SP</th>
<th>Western Australia</th>
</tr>
</thead>
<tbody>
<tr>
<td>0 - 2</td>
<td>3</td>
<td>3</td>
<td></td>
</tr>
<tr>
<td>3 - 4</td>
<td>2</td>
<td>3</td>
<td></td>
</tr>
<tr>
<td>5 - 14</td>
<td>18</td>
<td>11</td>
<td>19</td>
</tr>
<tr>
<td>15 - 24</td>
<td>15</td>
<td>16</td>
<td>18</td>
</tr>
<tr>
<td>25 - 64</td>
<td>44</td>
<td>53</td>
<td>46</td>
</tr>
<tr>
<td>&gt; 65</td>
<td>18</td>
<td>14</td>
<td>8</td>
</tr>
</tbody>
</table>

3. Wastewater disposal system (per cent)

<table>
<thead>
<tr>
<th></th>
<th>N-D</th>
<th>S-SP</th>
<th>Perth</th>
</tr>
</thead>
<tbody>
<tr>
<td>Sewerage system</td>
<td>97</td>
<td>94</td>
<td>55</td>
</tr>
<tr>
<td>Septic tank</td>
<td>1.5</td>
<td>1</td>
<td>45</td>
</tr>
<tr>
<td>Don't know</td>
<td>1.5</td>
<td>5</td>
<td>0</td>
</tr>
</tbody>
</table>

4. Method of watering lawns and gardens (percentage of householders using the method)

<table>
<thead>
<tr>
<th>Method</th>
<th>N-D</th>
<th>S-SP</th>
</tr>
</thead>
<tbody>
<tr>
<td>Fixed reticulation, manually operated</td>
<td>25</td>
<td>9</td>
</tr>
<tr>
<td>Fixed reticulation, time clock operated</td>
<td>22</td>
<td>0</td>
</tr>
<tr>
<td>Movable sprinkler, manually operated</td>
<td>56</td>
<td>66</td>
</tr>
<tr>
<td>Movable sprinkler, time clock operated</td>
<td>3</td>
<td>1</td>
</tr>
<tr>
<td>Trickle irrigation</td>
<td>7</td>
<td>8</td>
</tr>
<tr>
<td>Hand-held hose</td>
<td>53</td>
<td>78</td>
</tr>
<tr>
<td>Can or bucket</td>
<td>11</td>
<td>16</td>
</tr>
<tr>
<td>Other method</td>
<td>1</td>
<td>2</td>
</tr>
</tbody>
</table>
TABLE 5.9 (cont'd)

5. Number of times area is watered per week (during summer)

<table>
<thead>
<tr>
<th>Area</th>
<th>N-D</th>
<th>S-SP</th>
</tr>
</thead>
<tbody>
<tr>
<td>Lawn</td>
<td>3</td>
<td>3</td>
</tr>
<tr>
<td>Garden</td>
<td>4</td>
<td>4</td>
</tr>
<tr>
<td>Road verge</td>
<td>3</td>
<td>2</td>
</tr>
<tr>
<td>Vegetable garden</td>
<td>5</td>
<td>4</td>
</tr>
</tbody>
</table>

6. Time (in minutes) taken to water each area, each time it is watered

<table>
<thead>
<tr>
<th>Area</th>
<th>N-D</th>
<th>S-SP</th>
</tr>
</thead>
<tbody>
<tr>
<td>Lawn</td>
<td>90</td>
<td>59</td>
</tr>
<tr>
<td>Garden</td>
<td>48</td>
<td>39</td>
</tr>
<tr>
<td>Road verge</td>
<td>41</td>
<td>32</td>
</tr>
<tr>
<td>Vegetable garden</td>
<td>35</td>
<td>29</td>
</tr>
</tbody>
</table>

7. Number of sprinklers used during watering

<table>
<thead>
<tr>
<th>Area</th>
<th>N-D</th>
<th>S-SP</th>
</tr>
</thead>
<tbody>
<tr>
<td>Lawn</td>
<td>6</td>
<td>2</td>
</tr>
<tr>
<td>Garden</td>
<td>3</td>
<td>1</td>
</tr>
<tr>
<td>Road verge</td>
<td>3</td>
<td>1</td>
</tr>
<tr>
<td>Vegetable garden</td>
<td>1</td>
<td>1</td>
</tr>
</tbody>
</table>

8. Time of day that watering occurs (percentage of householders)

<table>
<thead>
<tr>
<th>Time</th>
<th>N-D</th>
<th>S-SP</th>
</tr>
</thead>
<tbody>
<tr>
<td>No fixed time</td>
<td>7</td>
<td>10</td>
</tr>
<tr>
<td>5 am to 8 am</td>
<td>18</td>
<td>11</td>
</tr>
<tr>
<td>8 am to 12 noon</td>
<td>3</td>
<td>3</td>
</tr>
<tr>
<td>12 noon to 5 pm</td>
<td>6</td>
<td>3</td>
</tr>
<tr>
<td>5 pm to 9 pm</td>
<td>54</td>
<td>65</td>
</tr>
<tr>
<td>9 pm to 5 am</td>
<td>11</td>
<td>2</td>
</tr>
<tr>
<td>Don't water</td>
<td>1</td>
<td>6</td>
</tr>
</tbody>
</table>
TABLE 5.9 (cont'd)

5. Number of times area is watered per week (during summer)

<table>
<thead>
<tr>
<th>Area</th>
<th>N-D</th>
<th>S-SP</th>
</tr>
</thead>
<tbody>
<tr>
<td>Lawn</td>
<td>3</td>
<td>3</td>
</tr>
<tr>
<td>Garden</td>
<td>4</td>
<td>4</td>
</tr>
<tr>
<td>Road verge</td>
<td>3</td>
<td>2</td>
</tr>
<tr>
<td>Vegetable garden</td>
<td>5</td>
<td>4</td>
</tr>
</tbody>
</table>

6. Time (in minutes) taken to water each area, each time it is watered

<table>
<thead>
<tr>
<th>Area</th>
<th>N-D</th>
<th>S-SP</th>
</tr>
</thead>
<tbody>
<tr>
<td>Lawn</td>
<td>90</td>
<td>59</td>
</tr>
<tr>
<td>Garden</td>
<td>48</td>
<td>39</td>
</tr>
<tr>
<td>Road verge</td>
<td>41</td>
<td>32</td>
</tr>
<tr>
<td>Vegetable garden</td>
<td>35</td>
<td>29</td>
</tr>
</tbody>
</table>

7. Number of sprinklers used during watering

<table>
<thead>
<tr>
<th>Area</th>
<th>N-D</th>
<th>S-SP</th>
</tr>
</thead>
<tbody>
<tr>
<td>Lawn</td>
<td>6</td>
<td>2</td>
</tr>
<tr>
<td>Garden</td>
<td>3</td>
<td>1</td>
</tr>
<tr>
<td>Road verge</td>
<td>3</td>
<td>1</td>
</tr>
<tr>
<td>Vegetable garden</td>
<td>1</td>
<td>1</td>
</tr>
</tbody>
</table>

8. Time of day that watering occurs (percentage of householders)

<table>
<thead>
<tr>
<th>Time</th>
<th>N-D</th>
<th>S-SP</th>
</tr>
</thead>
<tbody>
<tr>
<td>No fixed time</td>
<td>7</td>
<td>10</td>
</tr>
<tr>
<td>5 am to 8 am</td>
<td>18</td>
<td>11</td>
</tr>
<tr>
<td>8 am to 12 noon</td>
<td>3</td>
<td>3</td>
</tr>
<tr>
<td>12 noon to 5 pm</td>
<td>6</td>
<td>3</td>
</tr>
<tr>
<td>5 pm to 9 pm</td>
<td>54</td>
<td>65</td>
</tr>
<tr>
<td>9 pm to 5 am</td>
<td>11</td>
<td>2</td>
</tr>
<tr>
<td>Don't water</td>
<td>1</td>
<td>6</td>
</tr>
</tbody>
</table>
TABLE 5.9 (cont'd)

9. Water in winter? (per cent)

<table>
<thead>
<tr>
<th></th>
<th>N-D</th>
<th>S-SP</th>
</tr>
</thead>
<tbody>
<tr>
<td>Never</td>
<td>11</td>
<td>23</td>
</tr>
<tr>
<td>Rarely</td>
<td>41</td>
<td>36</td>
</tr>
<tr>
<td>Sometimes</td>
<td>38</td>
<td>35</td>
</tr>
<tr>
<td>Often</td>
<td>4</td>
<td>0</td>
</tr>
<tr>
<td>Only for bore maint.</td>
<td>6</td>
<td>0</td>
</tr>
</tbody>
</table>

10. Ownership of rainwater tanks: N-D 10%; S-SP 11%

11. Ownership of swimming pool (per cent)

<table>
<thead>
<tr>
<th>Type of pool</th>
<th>N-D</th>
<th>S-SP</th>
</tr>
</thead>
<tbody>
<tr>
<td>Fixed, below ground</td>
<td>15</td>
<td>4</td>
</tr>
<tr>
<td>Fixed, above ground</td>
<td>1</td>
<td>1</td>
</tr>
<tr>
<td>Movable</td>
<td>1</td>
<td>2</td>
</tr>
<tr>
<td>No pool</td>
<td>83</td>
<td>93</td>
</tr>
</tbody>
</table>

12. Bore usage and plans for installation (per cent)

<table>
<thead>
<tr>
<th></th>
<th>N-D</th>
<th>S-SP</th>
</tr>
</thead>
<tbody>
<tr>
<td>Own and use bore</td>
<td>37</td>
<td>3.5</td>
</tr>
<tr>
<td>Use neighbour's bore</td>
<td>11</td>
<td>1.5</td>
</tr>
<tr>
<td>Plan to install a bore</td>
<td>5</td>
<td>2</td>
</tr>
<tr>
<td>Plan to use neighbour's bore</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>Financially constrained from</td>
<td></td>
<td></td>
</tr>
<tr>
<td>bore ownership</td>
<td>13</td>
<td>7</td>
</tr>
<tr>
<td>No plans to install a bore</td>
<td>34</td>
<td>86</td>
</tr>
</tbody>
</table>

13. Weekly sprinkler use (minutes) in summer
(Frequency of watering multiplied by the duration of watering and the number of sprinklers in use)

<table>
<thead>
<tr>
<th></th>
<th>N-D 1578 mins (non-bore owners - 901 mins) bore owners - 2308 mins</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>S-SP 480 mins.</td>
</tr>
</tbody>
</table>

14. Weekly outdoor water use in summer
(Weekly sprinkler use multiplied by an average sprinkler discharge rate of 12 litres/minute)

|        | N-D 19 m³; S-SP 6 m³.                                |
The meters in the study areas were read 23 times during the study, giving consumption values for 22 periods. Consumption values were correlated against site variables (e.g. number of people, block size, the presence of pools, rainwater tanks and bores) obtained from the questionnaire and from property maps. Average values for the whole survey group were also correlated against climatic variables for the consumption period.

Figure 5.16 shows how the correlation between metered consumption and three site variables (number of people, block size and bore ownership/use*) varies between seasons. Mean monthly temperatures are shown as a guide to seasonal variables. The correlation with bore ownership/use is negative.

The number of people in a house was positively correlated with consumption throughout all seasons in both catchments, but correlation coefficients were at a maximum during winter when indoor water use predominated. Correlation coefficients were not as high in the drier, 1982 winter. The correlations between individual age groups (e.g. babies aged 0-2 years) and consumption were usually positive and significant at $P < 0.05$. The correlation between people over 65 years and consumption was negative (and significant at $P < 0.05$) for all consumption periods in the N-D catchment and for 70 per cent of consumption periods in the S-SP catchment. As there was no correlation between people over 65 years and other variables such as block size and bore ownership in the N-D catchment, it was concluded that older people used less water.

Figure 5.16 shows that block size is not significantly correlated with water consumption in the N-D catchment, while it is significantly correlated during summer periods in the S-SP catchment. This result can be explained by the greater percentage of bore ownership in the N-D sample (52%) in comparison with the S-SP sample (6%). Also, block size is much more variable in S-SP

* including use of a neighbour's bore
FIGURE 5.16: Correlation between metered water consumption and three site variables for the N-D (above) and S-SP (below) catchments.
than in N-D, particularly when considered on a percentage basis.

Bore ownership had a high negative correlation with consumption during the summer periods in the N-D catchment but was generally not significantly correlated in the S-SP catchment (probably because few households have bores). The periods when bore ownership was not correlated with consumption in the N-D catchment indicates that the periods of non-watering during the 1981 and 1982 winters were about 20 and 18 weeks respectively. Thus bores appear to have been used for about 60 to 65 per cent of the year during 1981 and 1982.

Correlations between consumption and households having swimming pools and rainwater tanks were generally not significant in either catchment. There was also little correlation between consumption during winter and those people who indicated in the questionnaire that they watered their gardens during the winter period.

From the above it is evident that the important site variables which influence metered water consumption were the number of people in the household, the presence of a bore (affecting summer consumption) and block size (affecting summer consumption of non-bore owners).

Figure 5.17 shows mean consumption figures for the two catchments for the duration of the study, as determined from the four-weekly meter readings. Consumption by both bore- and non-bore owners is shown for the N-D catchment. The small number of bore owners in the S-SP sample (7) makes separation of consumption by bore- and non-bore owners invalid for statistical analysis so only total consumption is shown for this catchment.

Figure 5.17 shows that there are 16 to 20 weeks each winter when bore- and non-bore owners use similar amounts of water but that during the summer months, non-bore owners use significantly
FIGURE 5.17: Mean metered consumption and ratio of rainfall-plus-irrigation to pan evaporation (E pan) for the N-D (above) and S-SP (below) catchments.
more water. The inflection in consumption by non-bore owners in the N-D catchment during the 1981/82 summer coincided with the unseasonal January rainfall. Mains water use by bore owners also rises in summer. From the questionnaire it was determined that only 23 per cent of bore owners never use mains water on their gardens, 26 per cent rarely use it, 36 per cent sometimes use it, and 15 per cent often use it, which may explain the rise. It is also possible that other summer activities increase mains water use in summer in households which have bores (e.g. filling swimming pools, car and boat washing, watering potted plants).

Indoor water use can be estimated for the two catchments by taking averages of consumption during the winter periods. Using this method, indoor water use in the N-D catchment was 14.2 m$^3$/4 weeks/residence and in the S-SP catchment was 11.2 m$^3$/4 weeks/residence. Taking into account occupancy ratios in the two catchments, consumption was 1.15 m$^3$/person/week in the N-D catchment and 1.10 m$^3$/person/week in the S-SP catchment.

Figure 5.17 also shows the ratio of inputs from rainfall plus irrigation to pan evaporation for non-bore owners in the two catchments. Irrigation volumes were estimated from consumption in excess of indoor use, and irrigation areas from considerations of average block size and per cent shedding areas (Section 5.1). Figure 5.17 shows that inputs only exceed pan evaporation for 16 to 20 week periods each year. During summer, average inputs drop to 0.3 - 0.4 pan evaporation. Inputs, however, are unlikely to be evenly distributed over the non-shedding area of blocks.

Table 5.10 shows the estimates of indoor and outdoor mains water use during a 52-week period in 1981/82 for the two catchments. The table shows that 63 per cent of mains water use by non-bore owners in the N-D catchment was used outdoors compared with only 43 per cent in the S-SP catchment. When all users are considered, only 43 to 50 per cent of mains water that is supplied to the catchments is used outdoors. This estimate is lower than
those made by Williamson (1975) and Tondut (1976) who, respectively, considered that 55 and 67 per cent of mains water in the whole of Perth was used on gardens. In some respects the two study catchments are atypical from the rest of Perth (e.g. the N-D catchment has twice as many bores than the average and the S-SP catchment has smaller blocks than average) which may account for some of the differences in the estimates. More likely, the percentage of mains water used on gardens has changed since restrictions were introduced in 1977. The response of householders in the N-D catchment to the restrictions was to increase groundwater use (see Section 5.6) while some householders in the S-SP catchment ceased to water their verges.

### TABLE 5.10

**Indoor and outdoor use of mains water for a 12-month period in 1981/82**

<table>
<thead>
<tr>
<th>Catchment</th>
<th>User group</th>
<th>Indoor use in m$^3$ (percentage use)</th>
<th>Outdoor use in m$^3$ (percentage use)</th>
</tr>
</thead>
<tbody>
<tr>
<td>N-D</td>
<td>Non-bore owners</td>
<td>183 (37%)</td>
<td>313 (63%)</td>
</tr>
<tr>
<td>(12.7.81 to 13.7.82)</td>
<td>Bore users</td>
<td>182 (71%)</td>
<td>74 (29%)</td>
</tr>
<tr>
<td></td>
<td>All users</td>
<td>182 (50%)</td>
<td>181 (50%)</td>
</tr>
<tr>
<td>S-SP</td>
<td>All users</td>
<td>144 (57%)</td>
<td>107 (43%)</td>
</tr>
<tr>
<td>(27.7.81 to 27.7.82)</td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

Water consumption in the catchments was also calculated using median, rather than mean, values. Use of median values decreases the effect of occasional high consumption values on the overall record. Using median values, outdoor use in the N-D catchment during the 12-month period was 41 per cent (65% for non-bore owners and 23% for bore owners) and 40 per cent for the S-SP catchment. Median values therefore decrease the percentage of outdoor use even further. Estimates of indoor water use using
median values are 1.00 m³/person/week for the N-D catchment, and 0.97 m³/person/week for the S-SP catchment.

Table 5.11 shows the correlation coefficients between mean four-weekly, mains water consumption and selected climatic variables.

Penman evaporation is a measure of potential evaporation of the atmosphere based on a combination of energy balance and mass transfer equations, which avoids any need to specify surface conditions (Penman 1948, 1956). The equation uses standard meteorological data for its input.

\[ \lambda E = \frac{\Delta}{\Delta + \gamma} R_n (1-n) + \frac{\gamma}{\Delta + \gamma} (0.35 + \frac{u}{100}) D \]

where \( \lambda E \) is the latent heat flux, \( \Delta \) is the slope of the saturated vapour pressure curve at the mean temperature, \( \gamma \) is the psychrometric constant, \( R_n \) is the net radiation, \( n \) is albedo, \( u \) is wind speed and \( D \) is the vapour pressure deficit.

Radiation values were available from Murdoch University, 9 km south of the N-D catchment. The albedo of five lawns in both catchments (two measurements on each lawn) was determined to be 0.28 (SE = 0.01). The same albedo was obtained over three street tree canopies, so this value was used in all calculations of Penman evaporation. Monteith (1973) gives the albedo of grass as 0.24. Penman evaporation was highly correlated with pan evaporation (\( r = 0.854 \)) but was significantly higher (\( P < 0.001 \)). Penman evaporation was about 10 per cent higher than pan evaporation at moderate evaporation rates (e.g. 5 mm/d) but only 2 per cent higher at high evaporation rates (e.g. 10 mm/d).

Table 5.11 indicates that pan evaporation was the most highly correlated climatic variable that was examined, followed by mean maximum temperature, the number of days over 25°C, Penman
<table>
<thead>
<tr>
<th>Catchment</th>
<th>User</th>
<th>Pan evaporation</th>
<th>Penman evaporation</th>
<th>Mean maximum temperature</th>
<th>Days over 25°C</th>
<th>Days over 30°C</th>
<th>Number of rainy days</th>
<th>Rainfall</th>
</tr>
</thead>
<tbody>
<tr>
<td>N-D</td>
<td>Non-bore</td>
<td>0.956</td>
<td>0.926</td>
<td>0.940</td>
<td>0.926</td>
<td>0.828</td>
<td>-0.824</td>
<td>-0.673</td>
</tr>
<tr>
<td></td>
<td>Bore</td>
<td>0.922</td>
<td>0.889</td>
<td>0.885</td>
<td>0.841</td>
<td>0.839</td>
<td>-0.629</td>
<td>-0.538</td>
</tr>
<tr>
<td></td>
<td>All</td>
<td>0.954</td>
<td>0.926</td>
<td>0.937</td>
<td>0.838</td>
<td>0.807</td>
<td>-0.807</td>
<td>-0.647</td>
</tr>
<tr>
<td>S-SP</td>
<td>All</td>
<td>0.944</td>
<td>0.923</td>
<td>0.928</td>
<td>0.911</td>
<td>0.829</td>
<td>-0.789</td>
<td>-0.629</td>
</tr>
</tbody>
</table>
evaporation, the number of days over 30°C, the number of rainy days (i.e. days with > 1 mm rainfall) and rainfall. An examination of plots of mains consumption against each climatic variable indicated that the two rainfall variables (i.e. number of rainy days and rainfall) had non-linear relationships with mean consumption, which was partially the cause of the lower correlations (e.g. fitting a polynomial increased the correlation coefficient between rainfall and consumption by non-bore owners from 0.673 to 0.754).

To develop a predictive model of consumption from a knowledge of climatic variables, various combinations of variables were combined into multiple regression equations. Number of rainy days was found to be a non-significant variable if either mean maximum temperature or pan evaporation was included in the predictive model. Table 5.12 gives the simplest and best predictive equations which could be obtained for each user group in the two catchments.

The models shown in the table have considerable predictive value (85 to 95 per cent of the variability in consumption is explained by them). The predominance of pan evaporation and mean maximum temperature in the models is in accordance with the findings of McMahon and Weeks (1973), as is the greater significance of number of rainy days than monthly rainfall in predicting water use (Table 5.11). The relationship between depth of irrigation (from all sources) and pan evaporation is examined at the end of the next section.

In the water balance calculations (Chapter 6), actual - rather than predicted - mains water use was used. The predictive models show that mains water use can be estimated with some accuracy from readily available climatic data, once a relationship has been established and provided other conditions do not change (e.g. imposition of water restrictions).
TABLE 5.12

Predictive models of mains water consumption

<table>
<thead>
<tr>
<th>Catchment</th>
<th>User group</th>
<th>Predictive model</th>
<th>Multiple correlation coefficient</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Non-bore owners</td>
<td>Cons. = -38 + 0.216 Epan + 2.10 Temp.</td>
<td>0.977 9%</td>
</tr>
<tr>
<td></td>
<td>N-D Bore owners</td>
<td>Cons. = 9 + 0.083 Epan</td>
<td>0.922 9%</td>
</tr>
<tr>
<td></td>
<td>All users</td>
<td>Cons. = -16 + 0.125 Epan + 1.20 Temp.</td>
<td>0.975 9%</td>
</tr>
<tr>
<td></td>
<td>S-SP All users</td>
<td>Cons. = -7 + 0.076 Epan + 0.72 Temp.</td>
<td>0.963 9%</td>
</tr>
</tbody>
</table>

Cons. = mean consumption (m³/4 weeks)
Epan = pan evaporation (mm/4 weeks)
Temp. = mean maximum temperature (°C)
All coefficients in the models are significant at P < 0.01.
5.6 BORE WATER USE

In order to estimate the amount of groundwater extracted by bores in the study areas, it was necessary to determine the number of bores and their use and supply rates. An estimate of bore numbers was obtained from the questionnaire. However not all householders in the catchments answered the questionnaire so an additional survey was required. Fortunately there are a number of signs of bore ownership by which the remaining households could be assessed. The signs include iron staining of fences and paths and the presence of bore and well covers in the yards. Most fixed reticulation systems are also operated off bores.

The final estimate of bore use in the two catchments was N-D catchment - 45 per cent; S-SP catchment - 4 per cent. Figure 5.18 shows that the distribution of bores within the catchments exceeds 5 bores/ha. in some areas. The lowest concentration of bores in both catchments occurred where the water table was deepest. Bore construction costs are highest in such areas, both because of the extra drilling required and also the greater likelihood of encountering thick limestone sequences (see Section 5.10). The questionnaire indicated that 66 per cent of the bores in the N-D catchment had been installed after 1976, 59 per cent being installed in the three-year period 1978-80.

The amount of groundwater used during summer was assessed from the questionnaire using two methods. The first method involved estimating sprinkler usage as was explained in the previous section (Table 5.9). The second method involved surveying the duration of bore use and multiplying this value with an estimated supply for the bore. The average duration of pumping during summer for the two areas was: N-D catchment - 333 minutes/week, S-SP catchment - 210 minutes/week. Initially it was envisaged that the supply of each individual bore would be estimated from supply curves for the various pumps that were used in the areas. Therefore the questionnaire included questions on
FIGURE 5.18 (a): Bore distribution in the N-D catchment.

FIGURE 5.18 (b): Bore distribution in the S-SP catchment.
pump make and model, pump horsepower, the installation company and depth of the bore. The results from these questions are shown in Appendix 3. However the great variability in pumps installed in the study areas made this method impractical for the present study.

An average pump supply value was estimated from examining supply values as shown in the drillers' report for bores that are on record at the Geological Survey of Western Australia (GSWA). Figure 5.19 shows histograms of supply for 72 bores in the Nedlands-Dalkeith-Claremont area. Considering the highly skewed nature of supply values, the geometric mean of 5.6 m³/h may be more appropriate than the arithmetic mean of 6 m³/h. The supply rates obtained by drillers do not take account of head losses in the reticulation system (estimated at about 3.5 metres or 5 pounds per square inch). Consequently the mean supply rate for private bores was reduced to 5.0 m³/h (1100 gal/h).

Table 5.13 shows the estimates of groundwater use by bore owners in the two areas, calculated by the two methods outlined above. The duration of watering was taken as 32 weeks, from considerations of Figures 5.16 and 5.17.

The similarity of estimates for the N-D catchment does not mean that the methods of estimation were very accurate. The probable error in the estimates is 20 per cent, as shown by the S-SP estimates.

Table 5.10 indicates that the average outdoor use of mains water by each bore owner in the N-D catchment during the summer of 1981/82 was 74 m³. Thus total outdoor water use by bore owners was about 960 m³. Assuming this use occurred over the 32-week period 5th October 1981 to 17th May 1982, then the ratio of
FIGURE 5.19: Histogram of supply values for private bores in the Dalkeith-Nedlands-Claremont area.
(Source: GSWA files).
TABLE 5.13

Estimates of average groundwater use by privately-owned bores during summer in the two urban catchments

<table>
<thead>
<tr>
<th></th>
<th>Use</th>
<th>(m³/summer)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>N-D</td>
<td>S-SP</td>
</tr>
<tr>
<td>Estimates from sprinkler use (from questionnaire)</td>
<td>886</td>
<td>661</td>
</tr>
<tr>
<td>Estimates from bore usage</td>
<td>888</td>
<td>560</td>
</tr>
</tbody>
</table>

rainfall and irrigation inputs to pan evaporation for the whole summer period is 1.10. If the unseasonal January rainfall is excluded, the ratio is 1.02. This compares with the ratio for non-bore owners of 0.50 for the N-D catchment and 0.37 for the S-SP catchment (including the January rainfall).

Groundwater extraction by Local Government Authorities (LGAs) in the two study areas was estimated from time of pumping and supply data provided by the Authorities. In the N-D catchment only one bore was operated for 15 hours each week during summer. With a supply of 24.4 m³/h, weekly extraction was 366 m³/week. While this amount is about 13 times larger than from an average bore, the difference in number of bores means that extraction by the LGA was only seven per cent of total groundwater extraction. In the S-SP catchment, four bores were operated by the LGA, extracting an estimated 819 m³/week. In this catchment, LGA extraction represented 42 per cent of total groundwater extraction. To some extent, the smaller private garden areas in the S-SP catchment are offset by larger public open space areas which require watering by the LGA.
5.7 SOIL WATER AND RECHARGE

5.7.1 Background

Estimates of distributed water inputs to the soil are made in Sections 5.1 (rainfall), 5.5 (mains water), 5.6 (groundwater) and 5.8 (throughfall). This section aims at determining the proportion of this type of input which becomes direct recharge, the proportion that is evaporated and the proportion that is stored within the soil profile at different times of the year.

Lloyd (1981) reviewed the problems of estimating recharge and concluded that the estimation is fraught with difficulty and that there was no ready solution. Lloyd stressed that it was important to understand fully the recharge mechanism.

So as to gain an understanding of direct recharge processes in the study areas, soil moisture profiles were monitored using the neutron logging method. The Australian Water Resources Council (AWRC 1974) recommended the neutron logging method for estimating soil moisture in catchment studies.

There have been a number of different methods used to estimate the deep drainage component of soil water (i.e. drainage beneath the deepest monitored depth in the soil profile). These methods are reviewed briefly below.

Rose and Stern (1965) advocated obtaining the soil suction (h, the soil matric potential) at, above and below the terminal depth of the measured soil profile, so as to determine the matric potential gradient \( \left( \frac{dh}{dz} \right) \) for inclusion in the generalised form of the Darcy equation

\[
v = K + K \frac{dh}{dz}
\]
where \( v \) = drainage flux in the vertical direction (cm/s),
\( K \) = hydraulic conductivity (a function of water content) in the vertical direction (cm/s)
\( z \) = depth (cm).

The equation is a form of Darcy's Law with total potential divided into its gravitational (z) and matric (h) components. Rose and Stern (1965) noted that unless soil matric potentials are low, the matric potential gradient is usually far greater than unity. In such cases the gravitational component of drainage (K) is negligible in comparison to the matric or suction component (K \( dh/dz \)).

Black et al. (1969) reported that drainage in bare soil was related to the average water content of the soil profile, and that the drainage could be equated with the unsaturated hydraulic conductivity. Black et al. (1970) later investigated the drainage from the root zone of a sandy soil. Unit hydraulic gradients (i.e. little change in matric potential with depth) were found below the root zone and drainage could again be equated with the unsaturated hydraulic conductivity of the sand. In both the above cases, drainage was predominantly due to the gravitational component of the drainage equation and matric potential gradients could be ignored (the opposite of the example investigated by Rose and Stern (1965)).

Whisler and Bouwer (1970) compared a number of methods for calculating vertical drainage in soils. They concluded that numerical analysis techniques gave the best agreement with observations, but required considerable data.

Carbon and Galbraith (1975) developed a model which simulated the water balance for plants growing on coarse textured soils. The model divided the soil into discrete compartments, within which the soil properties are assumed to be homogeneous. The model requires inputs of eight physical parameters (e.g.
unsaturated hydraulic conductivity) and three biological parameters (e.g. total root length) for each soil compartment. Meteorological inputs are on a daily basis, unless an interception subroutine is used, when input is on an hourly basis. Such a model has many attractions for predicting the amount of distributed water additions that become recharge in the study areas. However, the degree of spatial and depth variability that was found in the sands of the study areas makes the data input requirements very high. In addition, the model was found by Carbon and Galbraith (1975) to be only barely satisfactory for low irrigation rates.

It was decided to measure deep drainage in the study areas by monitoring soil moisture profiles in representative sites. Deep drainage at the bottom of the profiles was assumed to be due principally to gravitational forces. Consequently deep drainage was equated with the unsaturated hydraulic conductivity. An examination of soil moisture profiles (Section 5.7.4) indicates that the assumption of gravitational drainage at depth within the sands is probably justified.

Klute (1972) surveyed the methods available for the measurement of hydraulic conductivity of unsaturated soils. Noting the difficulties involved in the direct measurement of unsaturated hydraulic conductivity, Klute included a review of methods of calculating it from other soil hydraulic properties which are easier to measure. Klute concluded that conductivity functions that have been calculated from moisture characteristic curves and saturated hydraulic conductivity (e.g. Millington and Quirk 1959, 1961) are suitable for field application, provided the water retention data and saturated conductivity adequately characterise the field soil.

Genuchten (1978) described a simple equation for the moisture characteristic curve, which enabled him to derive a closed-form analytic expression for relative conductivity (i.e.
conductivity relative to that at saturation) when substituted into the predictive models of Burdine (1953) and Mualem (1976). Genuchten compared the computed relative conductivities that were based on the model of Mualem (1976), with observed data for five soils with widely differing hydraulic properties. The method predicted the relative hydraulic conductivities well in all soils, except one with a high clay content. The advantage of the Genuchten procedure for estimating relative conductivities over the Millington-Quirk method (Millington and Quirk 1961) lies in the ease of computation which is afforded by the derivation of a closed-form analytical expression for relative hydraulic conductivity.

A method of estimating deep drainage in the soil profile, which does not require the measurement of soil hydraulic properties, consists of the development of an evaporation model when deep drainage is negligible. Such a model can then be used for the periods when deep drainage is taking place. In this method, deep drainage becomes the unknown in the water balance equation. As this method only requires changes in soil water content to be known, errors in estimating absolute water contents by the neutron method are averted.

Several evaporation models for different textured soils have been developed (e.g. Holmes and Robinson 1959; Holmes 1961; Zahner 1967; Black 1979). These models rely on there being a relationship between the ratio of actual to potential evaporation and soil moisture availability. For crops, this relationship changes as the crop matures (e.g. Holmes 1961; Denmead and Shaw 1962). Black (1979) found that below a critical soil water storage, the ratio of evaporation to potential evaporation rates decreased linearly with decreasing soil water storage. In this study, the evaporation model method of estimating deep drainage was used as a check on the unit gradient method.
5.7.2 Field methods

5.7.2.1 Selection of access tubes sites:

In selecting sites for the installation of neutron access tubes, account was taken of the proportion of areas affected by tree canopies (as determined by planimetry), differing watering regimes (as determined by the appearance of lawns and gardens) and the depth of the water table. The number of holes installed was determined by the constraints on logging time. If access holes were about six metres deep and logged at 30 cm intervals, AWRC (1974) estimated about 15 neutron access holes could be logged in one day with one neutron moisture meter. This number was consequently recommended by the AWRC for monitoring soil moisture in a catchment. However as soil moisture values were expected to be very variable in the urban areas, the number of holes that were monitored was almost double this value.

Twenty-eight access holes were installed overall to monitor soil moisture in the two urban catchments and in the two woodland areas. Twenty-six holes were installed about six months before the monitoring commenced and two further holes were installed about 20 weeks into the study, so as to increase the monitoring of soil moisture profiles under tree canopies. Three holes were installed in each of the two woodland areas and 11 in each urban catchment. The average logging depth of the holes was 680 cm (range 510 to 840 cm). Nine holes penetrated 1 to 2 metres below the water table. Nine holes were in unwatered woodland and road verge areas, six were in lawns and gardens watered with mains water and 13 in areas watered with groundwater. The proportion of areas watered with groundwater is over-represented, partially due to two householders installing bores after the access tubes were installed. However several of the groundwater users that were monitored had lawns similar to mains water users due to low water application rates. As most bore owners use time clocks for controlling watering, the estimation of irrigation additions for groundwater users was found to be much more accurate than for
mains water users. Two access tubes were in parks watered by the Local Government Authority in the S-SP catchment.

5.7.2.2 Installation of access tubes:

For the first nine holes, a stainless steel double tube was hammered 315 cm. into the ground. Using this method, samples suitable for estimating soil bulk densities were recovered. The double tube later became inoperable and all subsequent holes (and hole depths beyond 315 cm) were drilled by hammering in sections of mild steel tubing. As the outside diameter (OD) of this tubing was only 48 mm, some reaming occurred on the installation of the PVC casing (55 mm OD). A good contact was achieved between the PVC and the soil using this method, without undue compaction of the soil around the hole due to the friability of the sand. The top of the PVC tubing was sealed between measurements by both a tapered, solid plastic plug and an outer PVC cap. The solid plug prevented access to the hole by vandals while the PVC cap was added to provide an extra seal against water entry. All access tubes in urban areas were cut off at ground level to allow lawn mowing over them.

5.7.2.3 Logging and calibration procedures:

All access tubes were logged at 30 cm depth intervals each fortnight. The six woodland holes were usually logged on one day and the remaining 22 holes on the following day. As meteorologic readings were available on a daily basis (and the day that logging was carried out was recorded), the fact that the holes were not all logged on the same day was of little consequence in the correlation of soil moisture values with atmospheric inputs and outputs.

Two neutron moisture meters were used during the study - a Wallingford, which was used for about 90 per cent of the study, and a Troxler, which was used as a back-up instrument. The meters
were calibrated in the field by logging holes which had just been drilled (in both catchments) and for which the moisture profile had been determined by gravimetric methods.

Figure 5.20 (a) shows the relationship between count ratio (i.e. counts/standard counts) and volumetric water content for the Wallingford meter. Greacen et al. (1981) have noted that it is more accurate to regress the count ratio on water content than on the reverse. However as only one sample was taken for determining each water content, it is likely that there is more error in estimating water contents than in estimating count ratios and therefore the regression of water content on count ratio is justified. The agreement between the data points and the regressions are not as good for the few high water contents as they are for water contents below 0.10 cm$^3$/cm$^3$. The calibration holes were drilled into the water table and water contents rose rapidly once the water table was approached. Under such conditions it is very important that the probe be located precisely at the depth at which the samples were recovered. Samples recovered from below the water table could not be used for calibration purposes as compaction of the sands during drilling resulted in a change in water content. Figure 5.20 (b) shows that the relationship between count ratio and water content for the Troxler appears slightly non-linear. This feature was also evident when the two meters were cross-calibrated by logging six holes with both instruments and examining the correspondence. The best function for describing the cross-calibration was a quadratic equation:

\[
\text{i.e. Wallingford count ratio} = 0.0101 + 0.3459 \text{ Troxler ratio}^2 + 0.3308 \text{ Troxler ratio}^2
\]

\[r = 0.998, \quad n = 117.\]

Greacen et al. (1981) have noted that such curvilinear calibration curves are common when PVC access tubes are used. The quadratic relationship was used in estimating water contents for
**FIGURE 5.20 (a)**: Calibration curve for the Wallingford neutron moisture meter.

**FIGURE 5.20 (b)**: Calibration curve for the Troxler neutron moisture meter.
the ten per cent of the study in which the Troxler meter was used.

5.7.3 Soil characterisation

5.7.3.1 Bulk density, colour and texture:

Bulk density measurements were possible for only nine holes and only over the top 315 cm, as was explained previously. Figure 5.21 shows the mean bulk densities obtained from these holes (three holes were located in native woodland and six in the N-D catchment). The mean bulk densities for the holes increase rapidly between the surface and a depth of 150 cm but only slightly between 150 cm and 315 cm. The curve that was constructed to fit the data was extrapolated to eight meters and used in all conversions of gravimetric- to volumetric- water contents, and also for packing permeameters and soil columns during hydraulic parameter tests. Assuming the density of soil particles in the sand is 2.65 g/cm$^3$ (the density of quartz and feldspar), the porosity of the sands at the surface was estimated to be 0.49 and at eight meters to be 0.38.

As will be shown seen later, the extrapolation of bulk densities from 315 to 800 cm was probably not valid in some holes which showed a textural discontinuity at about 400 cm.
FIGURE 5.21: Bulk density in relation to soil depth as determined from nine soil profiles.
The sands in the vadose zone in all except two holes were yellow below a grey A-horizon. The two exceptions were holes located immediately east of the two absorption basins in which the sands were brown. All sands recovered from below the water table were white, with the sand colour changing from yellow to white about a metre above the water table. Limestone was encountered at the terminal depths of two holes. An organic-charcoal layer was intercepted at a depth of about two metres in one hole, probably indicating a buried A₁ horizon.

The texture of the soils recovered by the access tube installation program was examined in an attempt to relate soil hydraulic properties with parameters such as mean grain size and sorting. Masch and Denny (1966), amongst others, have reported some success in being able to predict the hydraulic conductivity of unconsolidated sands from relatively simple measurements of grain size and sorting. The spatial and depth variability of soil textures was also examined, along with mineralogy and chemistry, to see if regional variations in soil hydraulic properties could be predicted from a knowledge of how the eolian sands were deposited.

Dry sieving was carried out on samples from the first 26 holes drilled. Samples were taken at odd meter intervals (i.e. 1, 3, 5 etc.) and also from depths where higher than normal clay contents had been observed in the sands. Fifty gram samples were sieved for 20 minutes using sets of Endecott sieves to separate the 0.5 φ size fractions between -1.0 and +4.5φ (where φ = -log₂d, and d = effective grain diameter in mm). In all, 124 samples were sieved. The actual and cumulative percentages were plotted and the four moments of Folk and Ward (1957) were calculated (i.e. graphic mean, inclusive standard deviation, inclusive graphic skewness and graphic kurtosis). Table 5.14 gives the average and range of values of the four moments.
Table 5.14 indicates the Spearwood sands (Karrakatta soil series) of the study areas are mainly moderately well sorted, medium sands (but include some fine sands), are sometimes strongly fine skewed, and have a kurtosis (peakedness) similar to that of a normal distribution.

<table>
<thead>
<tr>
<th>Moment</th>
<th>Average value (φ units)</th>
<th>Range (φ units)</th>
</tr>
</thead>
<tbody>
<tr>
<td>$M_z$ (mean size)</td>
<td>1.503 (medium sand)</td>
<td>1.058 (medium sand) to 2.121 (fine sand)</td>
</tr>
<tr>
<td>$\sigma_I$ (sorting)</td>
<td>0.653 (moderately well)</td>
<td>0.501 (moderately well) to 0.864 (moderately)</td>
</tr>
<tr>
<td>$SK_I$ (skewness)</td>
<td>0.091 (near symmetric)</td>
<td>-0.241 (coarse skewed) to 0.597 (strongly fine skewed)</td>
</tr>
<tr>
<td>$K_G$ (kurtosis)</td>
<td>1.100 (mesokurtic)</td>
<td>0.840 (platykurtic) to 1.467 (leptokurtic)</td>
</tr>
</tbody>
</table>

TABLE 5.14
Averages and range of values of soil moments

Figure 5.22 shows the size distributions of samples from four profiles. The GB profile shows the most common trend that was found in the profiles. Mean grain size decreased (increasing $M_z$) with increasing depth and this was usually accompanied by a decrease in sorting (increasing $\sigma_I$) and fine skew (decreasing $SK_I$). Statistical analysis of all the profiles showed the decrease in grain size and fine skew with increasing depth was significant at $P < 0.001$. Increases in sorting, fine skew and peakedness with increasing grain size were also significant at $P < 0.001$.

The GS profile in Figure 5.22 shows a strongly bimodal distribution of grain size with an increase in the fine fraction (2.5 to 3.0 φ or 177 to 125 μm). Samples from two other holes
were also strongly bimodal and those from nine other holes showed
a slight bulge in distribution at this size interval (resulting in
them being classified as fine skewed). The fine sand fraction is
too coarse to have been produced by eluviation in the profile, so
is probably a primary feature of the sands. The KC profile shows
an apparent slight coarsening in texture at about five metres.
This feature was present in about seven holes and coincided with
zones which were noted during drilling as containing above normal
clay contents and having higher water holding capacities. The
clay content in these sections was too high for accurate
measurement by dry sieving, so wet analytical techniques were
carried out on two of the profiles (see below). The MA profile in
Figure 5.22 shows the fine skew that was mentioned previously, but
it also shows a coarsening of the sands beneath the water table
(last sample). Two of the seven holes that intersected the water
table showed this trend, while the other five showed a
continuation of the trend towards finer textures that was noted in
the vadose zone. The sample size is too small to be able to draw
any conclusions about the effect (if any) the groundwater may be
exerting on texture. The fact that all the saturated samples
recovered in the study areas lacked the yellow (goethite)
colouration, indicates some chemical influence of the groundwater.

Figure 5.23 shows the average grain size and sorting for the
26 holes investigated. Sands in the N-D catchment tend to be
coarser (lower Mz) and better sorted (lower \( \sigma_i \)) than those from
the other areas. Within the catchments, holes from adjoining
blocks (e.g. GA-GB-GC, DA-DB and HA-HB) are clustered together
indicating that there is some systematic variability in the sands.
Ranking the holes in the N-D catchment from the west indicates the
more westerly a sand, the coarser, better sorted (both \( P < 0.01 \))
and fine skewed (\( P < 0.05 \)) it tends to be. In the S-SP catchment
there was no significant trend with east-west location.

The soil textures of samples from the KC and MA holes were
determined by the international pipette method (Piper 1942). Ten
Figure 5.23: Average grain size and sorting for 26 profiles in the four study areas.
samples were analysed from the KC profile to cover the section of high clay while five samples from the MA profile included two from beneath the water table. Figure 5.24 (a) shows that the clay content at 4.5 metres in profile KC was substantial (over 14 per cent) and that the increase in clay at this depth was offset by a commensurate decrease in 'coarse' sand (> 0.2 mm). On the United States Department of Agriculture textural triangle, the samples at 4.5 and 4.8 metres are classified as loamy sands while the rest of the profile falls within the sand classification. The clay percentage was at a minimum immediately above the loamy sand horizon.

Mineralogical and chemical studies were carried out on the samples (Appendix 4). These studies suggest that the clay may originate by in situ kaolinization of feldspars, although a depositional origin cannot be ruled out. An examination of soil water profiles (Appendix 5) showed that the clay-enriched horizons were present at about 3.5 to 4.5 metres in about half of the 28 neutron access holes. The KC profile shows the horizon at its most pronounced. A linear regression between clay content and the summer minimum water content of the KC profile showed that 96 per cent of the variability in the residual water content could be explained by the clay percentage.

The clay-enriched horizon is related more to topography than to the water table as evidenced by its occurrence at a constant depth in the profile and its absence in holes that were drilled in topographically low positions (i.e. less than 15 metres AHD elevation).

The MA profile in Figure 5.24 (b) shows a trend towards increasing clay contents above the water table, and a decrease immediately below the water table. There is insufficient evidence to relate the decrease in clay content to the position of the water table.
FIGURE 5.24 (a): Textural components of the KC profile.

FIGURE 5.24 (b): Textural components of the MA profile.
While all the sands in the study areas are either medium or fine grained, there is a certain amount of variability in grain size and sorting. This variability is relatively systematic in that the coarsest textured sands occur at the top of profiles and in the N-D catchment. The coarser textured sands are better sorted than the finer textured sands.

5.7.3.2 Hydraulic properties:

In order to estimate deep drainage by the procedure developed by Genuchten (1978), saturated hydraulic conductivity and moisture characteristic data were required for the soils located at the depths at which drainage was to be estimated (i.e. at the bottom of the hole or at about 1.5 metres above the water table for holes that intercepted groundwater).

The saturated hydraulic conductivity of samples recovered from the drainage depths for all the soil profiles were determined using constant head permeameters. The samples were packed into cells (5 cm diameter, 8 cm long) at the estimated field bulk density and subjected (from below) to a constant head maintained by mariotte bottles. Flow from the permeameters was monitored at five minute intervals until constant outflow occurred.

The use of disturbed samples for estimating the hydraulic properties of field soils has often been questioned on the basis of loss of soil structures and the preferred orientation of minerals (e.g. Shaykewich 1970). However, Freeze and Cherry (1979) have described eolian sands as being quite homogeneous and about as isotropic as any deposits occurring in nature. From pump tests, the aquifers in the Perth area had indicated a strong anisotropy is present (Pollett 1981). It has been suggested by Allen (public lecture 1981) that the Tamala Limestone is marine in origin in the groundwater zone. Therefore there may be a difference in anisotropy between the eolian sands in the vadose zone and the marine sediments in the groundwater zone.
Figure 5.25 shows the relationship between saturated hydraulic conductivity and the graphic mean grain size (Mz) as determined by dry sieving. While there is a close relationship between conductivity and grain size (correlation coefficient for a linear regression $= -0.911$), the correlation between conductivity and sorting is poor ($r = -0.294$) and is not significant at $P < 0.05$.

The conductivity of the sands is seen to vary over almost an order of magnitude (i.e. 6 to 50 m/d). From pump tests, Allen (1981) has estimated the hydraulic conductivity of the sands to be about 30 m/d while Mathew et al. (1982), using permeameters, have measured the hydraulic conductivity to be about 16 m/d, which is very similar to that reported by Talsma (1974) for the same sands.

Masch and Denny (1966) investigated the relationship between hydraulic conductivity and grain size and sorting using river sand. Figure 5.26 shows the estimated conductivities for the study sands, as estimated by the Masch and Denny procedure, plotted against the measured conductivities. The measured hydraulic conductivities are considerably higher than predicted values for the coarser sands and slightly lower than predicted for the finer sands. Factors other than grain size and sorting that affect conductivity in the sands (e.g. roundness, sphericity and bulk density) may be the reason for the difference. As the sands have been transported by different media (i.e. water and air), the roundness and sphericity of the sand grains may differ. Masch and Denny did not report the bulk density used in their measurements.

The soils in the study areas were found to be draining profiles for most of the year, except for a short period when the wetting front passed (see Section 5.7.5). Consequently, moisture characteristics for the sands were determined under draining conditions. For the matric potentials between 0 and $-100$ cm, moisture retention values were determined by packing metre-long,
FIGURE 5.25: Saturated hydraulic conductivity as a function of grain size.

FIGURE 5.26: Measured and estimated saturated hydraulic conductivities for 26 soil samples.
acrylic columns (internal diameter = 3 cm) with samples from the depth at which drainage was to be estimated. The columns had been split in half and taped together prior to filling and were vibrated while the samples were added, so that the bulk density throughout the columns equalled that estimated using Figure 5.21. The coarser sands required less vibrating than the finer sands. The columns were wetted from above to saturation and allowed to drain while a free water surface was maintained at a constant depth at their base. Filter papers were attached to the top of the columns to restrict evaporation. After the columns were allowed to drain for one week, they were quickly laid on their side (to prevent drainage once the free water surface had been removed) and samples were taken at five-centimetre intervals for the determination of gravimetric water contents. The height of the sample above the free water surface was equated with the matric potential, for the determination of the moisture characteristic. This column method enabled the air-entry point to be determined accurately. Water retention at lower potentials was determined using pressure plates (−125 and −630 cm potentials) and pressure membranes (−15,850 cm) as outlined by Richards (1965).

Figure 5.27 shows the moisture characteristic curves for two soils in the study area. Soil CC is a fine-grained sand (Mz = 2.12) with a saturated hydraulic conductivity of about 7 m/d, while DA is a medium grained sand (Mz = 1.69) with a saturated hydraulic conductivity of 20 m/d. The fine grained sand had a higher water content at all matric potentials and a more gradual slope between the air-entry point and the lowest potentials. The air-entry point for the fine sand occurs at 10 cm lower potential than for the medium sand.

As the moisture characteristic curves had been determined for seven of the holes which intersected the water table, the laboratory measurements could be compared with the soil water profiles above the water table for periods when there was no downward flux of water in the profile. Figure 5.28 shows the soil
FIGURE 5.27: Moisture characteristic curves for a fine (CC) and a medium (DA) grained sand.
FIGURE 5.28: Comparison of laboratory and field measurements of moisture characteristic curves for six sands.
water profiles for six of these holes at the end of the 1981/82 summer. Good agreement can be seen between the laboratory and field measurements for three holes (DR, SL and RA). There was also good agreement for the profile not shown (MA). Two of the soil water profiles (ER and OR) show field values to be below the laboratory measurements. In the case of ER, the discrepancy is clearly due to the presence of tree roots as will be seen in Section 5.9. Extraction by roots is also possible in the OR profile, as the top of this profile is only two metres below the ground surface. In the AR profile, field soil water values were considerably above equilibrium conditions. This reflects downward water movement in this profile, the most heavily watered lawn investigated in this project.

For the calculation of the relative conductivities of the sands by the Genuchten (1978) method, the saturated- and residual-water contents of the soils needed to be specified. The saturated water contents were estimated by averaging water contents on the moisture characteristic curve, up to the air-entry point. The residual water contents were estimated by two methods. In most soil profiles, the water content at the drainage depth dried back to the same minimum value each summer. At such times, soil moisture storage in the profile also reached its summer minimum value and did not change until after the first substantial rains of winter. From the moisture characteristic curves, the water contents at these drainage depths corresponded to matric potentials of about -200 cm (i.e. slightly below field capacity, taken as -100 cm; Marshall and Holmes 1979). For these soils, residual water contents were taken as this summer minimum. For all of the woodland- and some of the urban- profiles however, water contents at the drainage depth were considerably lower than could be explained by drainage and almost certainly represented active extraction by plant roots. In some profiles the soil at the drainage depth had dried out to wilting point (-16,000 cm) by the end of summer. For these soils, the residual water contents were taken as the water content at -200 cm matric potential. A
few profiles did not dry out during the summer due to irrigation and for these soils the residual water contents were also taken as that at -200 cm matric potential. Field values of the residual water content were preferred over laboratory estimates wherever possible.

Figure 5.29 shows the relative hydraulic conductivity functions for the two soils shown in Figure 5.27. The medium sand (DA) has a higher relative conductivity than the fine sand (CC) for all relative water contents. This difference is heightened when it is recalled that the conductivities at saturation are 20 m/d and 7 m/d respectively.

5.7.4 Soil water redistribution

The redistribution of soil water in Spearwood sand has already been studied by Talsma (1974) and Carbon (1975b). Talsma found that gravity effects were dominant in redistribution and that increasing the initial moisture content, or the infiltration quantity, increased the contribution of gravity potentials relative to capillary (or matric) potentials. Both Talsma and Carbon found that the draining part of the moisture profile in Spearwood sand assumed the shape of the draining moisture characteristic.

Figure 5.30 shows the soil water profiles of a native woodland site (KA) for different times throughout the study. Only about one-third of all profiles measured are shown. The influence of the horizon with higher clay content at four metres on moisture retention is evident in the draining profiles. The maximum water content in the profiles often occurs immediately behind the wetting front. There is a general increase in water content with depth in the profile for much of the year, which reflects the higher water holding capacity of the finer-textured soils at depth. During spring and summer, the soil water profiles retreat almost parallel to this general trend in water content. Winter
FIGURE 5.29: Relative hydraulic conductivity functions for a fine (CC) and a medium (DA) grained sand.
FIGURE 5.30: Soil water profiles for a native woodland site (KA) over a 20-month period.
rainfall (May to September in 1982) was 15 per cent lower than during the same period in 1981 and the wetting front failed to reach the bottom of the access hole in 1982. The January 1982 rainfall was rapidly evaporated from the profile as the evaporative demand of the atmosphere was at a maximum.

The water contents at the bottom of the KA soil profile at the end of summer are lower than can be attributed to drainage (as estimated from the moisture characteristic curve for the soil at 700 cm depth) and almost certainly indicate that water extraction by tree roots extends to at least seven metres. The water content of the whole profile changes little at the end of summer and it is likely that at such times, moisture is extracted by plant roots at greater depths than was monitored.

Figure 5.31 shows soil water profiles under an urban lawn (site DA) that was watered (with groundwater) such that total water additions (rainfall plus irrigation) were 40 to 50 per cent of potential evaporation. This amount of watering was sufficient to maintain a reasonable cover of grass, but was not high enough to maintain the soil water profile that is established each winter. The water contents at the bottom of the profile at the end of summer are about the same as that which would be expected by drainage, with little plant root extraction. No urban soil profile (even those in non-watered areas) dried out to the same extent as occurred in the woodland profiles. There were some deep zones in urban profiles which indicated that extraction of water by roots was taking place, as will be seen in Section 5.9. However such zones were not as common as in woodland profiles. It is probable that the roots of many urban plants do not extend to the same depths as woodland plants, due to summer watering practices which often wet only the top metre of the soil profile. Also, plant communities in woodland areas have probably developed so as to maximize water use in the soil profile. The advance of the winter wetting front occurred much more rapidly in the urban sites than in the woodland sites. This was probably due both to
FIGURE 5.31: Soil water profiles under an urban lawn (DA) over a 20-month period.
the lower water deficit at the start of winter and to reduced interception and extraction of water in the urban sites.

Soil water profiles from other sites are shown in Appendix 5. In the most heavily-watered urban sites there was little variation in soil moisture throughout the year. A feature of the profiles which intersected the water table was that the water table rose before the wetting fronts reached the saturated zone. Such an event probably indicates that recharge had already reached the water table in an adjoining area. All holes which intersected the water table were close to an absorption basin and, as will be seen in Section 5.10, these basins exert a considerable influence on the water table.

5.7.5 Soil water storage

Soil water contents and storages were calculated using the computer program of Hignett (1980). Figure 5.32 (a) shows the soil water storage in the three holes in the Selby Street woodland area (Figure 4.1). The depth of the holes was 720, 630 and 720 cm for holes SA, SB and SC respectively. Hole SC contained coarser sand than the other two holes. Figure 5.32 (a) shows that soil water storages in all holes were less in the winter of 1982 than in 1981, while the summer minima were similar. Holes SA and SC showed very similar trends in storage, with storage being less in hole SC, possibly due to its coarser texture. Soil water storage was generally at a maximum when the wetting front was at the bottom of the hole (the time when the front first passed the bottom of the hole is shown with a dot). This coincidence is a result of the form of redistribution which takes place, as discussed in the previous section. Hole SB was located under a tree canopy and had lower changes in storage than either of the other two holes. The wetting front failed to reach the bottom of the SB hole in the winter of 1982.

Figure 5.32 (b) shows soil water storages for two holes
FIGURE 5.32 (a): Soil water storage in three native woodland soil profiles.

LEGEND:
- WETTING FRONT AT BASE OF MONITORED PROFILE

FIGURE 5.32 (b): Soil water storage in two urban soil profiles.
which were drilled into lawns on adjoining blocks. The lawn at DA was receiving water additions at about 40 to 50 per cent of potential evaporation rates during summer, whereas the lawn at DB was being watered at 70 to 80 per cent of potential rates. The DB site was also shaded for a longer time each day than the DA site (approximately 60 versus 10 per cent of the day). Soil water storage at DB altered little throughout the year, but this was due to high storage amounts in the top 90 cm of the profile during summer, while lower parts of the profile dried out. However, the soil profile did not dry out to the extent that occurred in hole DA and wetting fronts passed beneath the DB hole earlier than in the DA hole. The wetting front produced by the January 1983 rainfall failed to pass the bottom of the DA hole (indicating the soil moisture deficit in this hole exceeded 100 mm) whereas the wetting front passed the bottom of the DB hole (indicating a very low soil moisture deficit).

5.7.6 Deep drainage and evaporation

5.7.6.1 Unit gradient method:

The amount of deep drainage estimated by the Genuchten (1978) method of calculating unsaturated hydraulic conductivities, was found to be very sensitive to a number of factors. Estimates based on the Mualem model were considerably higher than those of the Burdine model and, in extreme cases, deep drainage was estimated to exceed the total water inputs to the profile. Both models were compared with data from the literature and, while the Mualem model gave a better overall estimation of unsaturated hydraulic conductivities, the Burdine model showed better agreement for the lower water contents. As water contents in this study only occasionally exceeded 10 per cent (volumetric water content), the Burdine model was preferred. Both models were very sensitive to the value selected for residual water content. In one hole, a change in residual water content of only half of one per cent resulted in a change in the estimation of deep drainage
of 50 per cent. Very high drainage values were also estimated when the wetting front was positioned at the drainage depth. At such times, the assumptions of unit gradient conditions (i.e. \( dh/dz = 0 \)) do not hold. The effect of hysteresis would also be evident on the moisture characteristic (and hence hydraulic conductivity) values at such times. Williamson (1978) drew attention to the effect of errors in estimating soil water contents on unsaturated hydraulic conductivity values. For water contents of 0.070, an error of 0.005 was estimated by Williamson to result in a 100 per cent change in conductivity. The influence of soil heterogeneity on soil hydraulic properties was taken into account in this study, through the use of samples from the depth at which deep drainage was calculated. However, the influence of soil anisotropy was not taken into account as disturbed soil samples were used.

5.7.6.2 Evaporation model method:

Evaporation models were developed for 18 of the 28 holes monitored, by solving the water balance equation for each profile when deep drainage was not taking place (determined by comparing water contents at the drainage depth with residual water contents). Runoff was taken as zero.

i.e. EVAPORATION = RAINFALL + IRRIGATION - CHANGE IN STORAGE.

The evaporation so derived was expressed as a ratio of Penman potential evaporation (pan evaporation did not give as good a relationship) and plotted against average soil moisture storage in the top three metres of the profile. This depth was found to give a better correlation than storage in the whole soil profile. Most models in the literature use the fractional extractive water in the root zone instead of storage. In this study, some root zones extended beneath the access hole. Linear functions were fitted to the data points, although some models in the literature (e.g. Holmes 1961; Zahner 1967) indicate that there is an initial
curvilinear increase in evaporation with water content.

Figure 5.33 shows the models developed for sites with low and high interception losses (determined by the presence of canopies to the west of the hole) and for woodland and urban areas. The data points used to calculate the lines are not shown for reasons of clarity. The slopes of the lines are greatest for sites with high interception losses ($r = 0.538$ between slope and percentage canopy in the western sector). These sites also had the lowest correlation coefficient which probably reflects variations in interception losses with direction and intensity of the rainfall, and the lack of interception with sprinkler additions. In general the models are barely suitable, with only 45 to 85 per cent of variation being explained. However, errors in estimating evaporation from sites should be partly compensatory, unlike the errors involved in the unit gradient method.

5.7.6.3 Comparison of methods:

Figure 5.34 shows a comparison of drainage estimations by the two methods outlined above. Only one-quarter of the estimates are within 20 per cent of the equivalence line. For most holes, the unit gradient method estimates larger amounts of deep drainage than the evaporation model method. This discrepancy was most marked in coarser sands which had high saturated hydraulic conductivities. The reason for coarser sands giving abnormally high drainage values (in some cases) may be the use of a common bulk density value in the laboratory measurements of hydraulic conductivity and moisture characteristics. It was noted earlier that coarse sands packed to the required bulk density relatively easily. If there are significant spatial variations in bulk density of the sands, there is a need for different calibration curves for the neutron moisture meters, to avoid the errors in the estimation of unsaturated hydraulic conductivity that was pointed out by Williamson (1978). The errors involved in estimating
FIGURE 5.33: Evaporation models for sites with high and low interception losses and for woodland and urban areas.
FIGURE 5.34: Comparison of unit-gradient and evaporation-model methods of estimating deep drainage.
changes in storage for the evaporation models is probably minor in comparison to the errors involved in estimating unsaturated hydraulic conductivities and absolute water contents.

Accordingly, the relationship between the amount of water input that becomes deep drainage at different depths in the soil (3 and 7 m) and for different rates of input was developed using the evaporation models. For the calculations, the water balance period was divided into four parts, depending upon whether irrigation or rainfall was the major water input. The beginning of winter periods was taken as the first heavy rains (normally mid-May) and the end of the winter periods as the time that deep drainage of winter rainfall had ceased (usually late November). The drainage which occurred in each period was then plotted against the ratio of inputs to potential evaporation, as shown in Figure 5.35. Mean evaporation models were used to calculate deep drainage in holes for which individual models had not been established. Both urban and woodland data were used as there appeared to be no significant difference between them. A few data points were omitted from Figure 5.35 due to atypical watering circumstances (e.g. washing of cars, boats) which would have obscured the overall relationship between input and deep drainage.

Linear functions were fitted to the data points where input exceeded 60 per cent of potential evaporation (the observed threshold for deep drainage to occur). Carbon (1975a) also found that water inputs had to exceed 60 per cent of potential evaporation rates before deep drainage became significant. The scatter of points in Figure 5.35 reflects errors in the evaporation models that were used and also site differences (e.g. interception of rainfall, shading, frequency and time of watering).

The drainage models indicate that about 15 per cent of inputs reach 3 metres when inputs represent 80 per cent of
FIGURE 5.35: Drainage models for low and high interception sites.
potential evaporation, but only 10 per cent reach 7 metres. When inputs represent 100 per cent of potential evaporation, the percentages reaching 3 and 7 metres decrease to 34 and 20 per cent respectively. The values for interception sites are particularly scattered. This scatter possibly reflects different seasonal effects of tree canopies. In winter, interception of rainfall is significant while in summer, interception of sprinkler additions is minor but shading of the sites is important. High interception sites appear to have a higher threshold value before deep drainage commences.

5.7.7 Estimation of recharge

5.7.7.1 Native woodland areas:

In the woodland areas, three neutron access holes were monitored in the Selby Street area (where the groundwater was 16 to 20 metres below the ground surface) and three in the Kings Park area (where the groundwater was 11 to 15 metres below the ground surface).

In the Selby Street area, 160 mm of water passed seven metres in the two low interception sites during the 1981 winter, while only 35 mm of water passed six metres in the high interception site. High interception areas constitute about 55 per cent of native woodland areas. During the 1982 winter, 80 mm of water passed seven metres in the low interception sites while the wetting front reached only 5.4 metres in the high interception site. The wetting front from the January 1982 rainfall penetrated to only 1.8 metres in the Selby Street area. There was little chance of significant recharge occurring at this woodland site during 1981 or 1982 given the great depth of the water table. All the incoming rainfall was either intercepted in tree and bush canopies (about 20% - see Section 5.8), or transpired and evaporated from the soil (about 80%). It is not known if the trees in the Selby Street woodland site were capable of reaching
the water table. Measurements by J. Dodd (pers. comm.) from a Banksia woodland north of Perth indicate that these trees can extract groundwater from at least seven metres when located in deep sands.

In the Kings Park woodland, 110 mm of rainfall passed seven metres in the two low-interception sites during the 1981 winter while only 50 mm passed seven metres in the high interception site. No water passed seven metres at any site during the 1982 winter. There is some possibility of recharge occurring at the KA site, at which the water table was only 11 metres below the surface. It is also possible that trees at the KA site could extract water from the water table, so this was possibly a net discharge area during 1981 and 1982. The reason that less deep drainage occurred in the Kings Park sites relative to the Selby Street sites may lie in the denser canopy in the former site and/or in the slightly greater clay contents and lower hydraulic conductivities (17 vs 30 m/d) of the sands in the Kings Park area. From the Thiessen network of rainfall stations, the Kings Park area would have been expected to receive more rainfall than the Selby Street area. Therefore the difference in the percentage of rainfall that became deep drainage in the two areas is even greater. The lack of significant recharge in the woodland sites during 1981 and 1982 highlights the importance of years of above average rainfall and areas with a moderately- (but not very) shallow water table for maintaining groundwater levels.

5.7.7.2 Direct recharge in the urban areas:

As few neutron access tubes exceeded 720 centimetres in depth, the amount of deep drainage that becomes recharge in urban areas where the water table was greater than this depth had to be estimated using assumptions about the location of tree roots and the magnitude of the soil moisture deficit. As mentioned previously, many urban soils drained back to water contents which were slightly below the field capacity of the soils, as determined
by their moisture characteristic curves. However, as the January 1982 rainfall fully penetrated only 40 per cent of the urban profiles, many urban soils do develop significant soil moisture deficits. Section 5.9 indicates that some street trees extract water from a depth of 18 metres.

In urban areas, the percentage of inputs reaching different depths was assumed to be as shown in Table 5.15. The uncertainty in extrapolation of recharge amounts past seven metres is considerable. The extrapolation is principally based on observations of the penetration of wetting fronts. Pollett (pers. comm.) has noted an absence of a seasonal cycle in groundwater levels in areas where the water table is at great depths below the ground surface. This absence may be due to an attenuation of the seasonality of recharge or the virtual absence of recharge. The water that does drain to depth will be subject to root extraction over the summer months by large trees (e.g. *Eucalyptus* sp.). It will also be subject to lateral dispersion under shedding areas. The general form of the table is similar to the recharge model developed by Pollett *et al.* (1979) for the northern Perth area.
### TABLE 5.15

**Percentage of rainfall and irrigation inputs becoming recharge for different depths of the water table**

#### (A) Low Interception Sites

<table>
<thead>
<tr>
<th>Rainfall + Irrigation Potential evaporation $\times$ 100</th>
<th>3</th>
<th>7</th>
<th>10</th>
<th>15</th>
<th>20</th>
<th>30</th>
</tr>
</thead>
<tbody>
<tr>
<td>(Measured)</td>
<td>60</td>
<td>80</td>
<td>100</td>
<td>120</td>
<td></td>
<td></td>
</tr>
<tr>
<td>(Estimated)</td>
<td>0</td>
<td>0</td>
<td>15</td>
<td>5</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td></td>
<td>0</td>
<td>80</td>
<td>100</td>
<td>120</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>0</td>
<td>0</td>
<td>10</td>
<td>5</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td></td>
<td>0</td>
<td>80</td>
<td>100</td>
<td>120</td>
<td></td>
<td></td>
</tr>
<tr>
<td>140 to 300 (estimated)</td>
<td>85</td>
<td>50</td>
<td>40</td>
<td>30</td>
<td>20</td>
<td></td>
</tr>
</tbody>
</table>

#### (B) High Interception Sites

<table>
<thead>
<tr>
<th>Rainfall + Irrigation Potential evaporation $\times$ 100</th>
<th>3</th>
<th>7</th>
<th>10</th>
<th>15</th>
<th>20</th>
<th>30</th>
</tr>
</thead>
<tbody>
<tr>
<td>(Measured)</td>
<td>60</td>
<td>80</td>
<td>100</td>
<td>120</td>
<td></td>
<td></td>
</tr>
<tr>
<td>(Estimated)</td>
<td>0</td>
<td>0</td>
<td>6</td>
<td>0</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td></td>
<td>0</td>
<td>0</td>
<td>8</td>
<td>4</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td></td>
<td>0</td>
<td>0</td>
<td>8</td>
<td>4</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td></td>
<td>0</td>
<td>0</td>
<td>15</td>
<td>8</td>
<td>2</td>
<td>0</td>
</tr>
</tbody>
</table>
The average depth of the water table in the two urban catchments is shown in Table 5.16.

**TABLE 5.16**

**Average depth of the water table in the two urban catchments**

<table>
<thead>
<tr>
<th>Depth of the water table (m)</th>
<th>Percentage of catchment N-D</th>
<th>Percentage of catchment S-SP</th>
</tr>
</thead>
<tbody>
<tr>
<td>0</td>
<td>0.2</td>
<td>1.9</td>
</tr>
<tr>
<td>0-5</td>
<td>5.2</td>
<td>10.0</td>
</tr>
<tr>
<td>5-9</td>
<td>11.2</td>
<td>11.5</td>
</tr>
<tr>
<td>9-13</td>
<td>13.0</td>
<td>20.4</td>
</tr>
<tr>
<td>13-17</td>
<td>24.2</td>
<td>17.4</td>
</tr>
<tr>
<td>17-25</td>
<td>35.0</td>
<td>18.1</td>
</tr>
<tr>
<td>25-39</td>
<td>11.2</td>
<td>20.7</td>
</tr>
</tbody>
</table>

About 30 per cent of the N-D catchment is within 13 metres of the water table compared with over 40 per cent in the S-SP catchment. Figure 5.18 showed that there was a concentration of bores in areas where the water table was close to the surface. As bore users used water at higher rates than non-bore owners, extra recharge is likely to result from this uneven distribution than would occur if bore owners were distributed evenly over the catchments.

To estimate irrigation inputs to the soils of the catchments, the distribution of sprinkler usage (Section 5.5) was divided into different groups so that irrigation and rainfall additions for the 1981/82 summer were percentages of potential evaporation, as shown in Table 5.17.

The data in Table 5.17 appear to contradict the findings in Section 5.6 that bore users in the N-D catchment irrigate at about potential evaporation rates while non-bore users irrigate at about 50 per cent of potential rates. The data in Section 5.6 were
based on mean values, however, and the presence of a strong positive skew in consumption increased mean values.

The use of mean values under such conditions would result in a misleading estimation.

TABLE 5.17

Percentage of householders in different irrigation groups in the two urban catchments

<table>
<thead>
<tr>
<th>Rainfall + Irrigation Potential evaporation x 100</th>
<th>N-D</th>
<th>S-SP</th>
</tr>
</thead>
<tbody>
<tr>
<td>0.4</td>
<td>41.5</td>
<td>64.4</td>
</tr>
<tr>
<td>0.6</td>
<td>30.0</td>
<td>21.2</td>
</tr>
<tr>
<td>0.8</td>
<td>11.5</td>
<td>6.4</td>
</tr>
<tr>
<td>1.0</td>
<td>3.7</td>
<td>2.1</td>
</tr>
<tr>
<td>1.2</td>
<td>1.9</td>
<td>0.4</td>
</tr>
<tr>
<td>1.4 to 3.0</td>
<td>11.5</td>
<td>5.5</td>
</tr>
</tbody>
</table>

Table 5.17 shows that about 70 per cent of householders in the N-D catchment and about 85 per cent in the S-SP catchment do not irrigate at rates that cause deep drainage. It is quite possible that proportions of such blocks are irrigated at above-threshold rates. The low irrigation rates are very similar to those measured by Moore (1977) for gardens in Metropolitan Adelaide and are generally below those recommended for irrigated crops.

In the estimation of direct recharge, the proportion of bore owners (i.e. heavier water users) was assumed to be twice as high in areas within 13 metres of the water table as in the rest of the catchment. The proportion of recharge was then calculated for different user groups and different depths to the water table for
the summer of 1981/82. The period of irrigation was assumed to be 30 weeks. Table 5.15 was also used to calculate recharge of rainfall during the winters of 1981 and 1982.

The calculations indicated that only about 10 per cent of total inputs became direct recharge during the winters of 1981 and 1982 in the N-D catchment while about 14 per cent became recharge in the S-SP catchment. The low proportions were the result of two factors. Firstly, about 30 per cent of total inputs fell on tree canopies in the N-D catchment and 20 per cent in the S-SP catchment. Recharge under canopies was restricted to areas where the water table was within 10 metres of the surface. Secondly, about 80 per cent of the N-D catchment is more than 10 metres above the water table while the proportion in the S-SP catchment is about 70 per cent. The difference in estimated recharge between the catchments may be less than that calculated, as a result of lower soil water deficits in the better-watered N-D catchment (which was not taken into account in the calculations).

In summer, only 6 per cent of total inputs of distributed water became recharge in the N-D catchment while in the S-SP catchment the proportion was only 4.5 per cent. These low values are the result of the high proportion of householders watering at below-threshold values, and the great distance of the water table from the ground surface.

On an annual basis, about 8.5 per cent of total inputs became direct recharge in the N-D catchment and about 11 per cent became recharge in the S-SP catchment. Estimates of net recharge on the Gungarri Mound as a whole have ranged from 8 to 12 per cent of annual rainfall. The comparatively low values for gross recharge in the urban areas may be the result of about 40 per cent of water additions taking place during the summer months in comparison with only about 14 per cent under non-irrigation conditions. Recharge may also be greater in the Bassendean Sands which contain less clay and have a higher water table than the
Spearwood Sands.

5.7.7.3 **Indirect recharge in the urban areas:**

The estimation of indirect recharge from concentrated water additions (i.e. additions from spoon drains, soak wells, etc.) for different depths to the water table also requires assumptions to be made about evaporative losses and soil water deficits. Lateral dispersion of wetting fronts was encountered in one access hole which had a vehicle parked over it for much of the 1981 winter. This profile (GR) failed to develop normal wetting fronts yet there was an increase in water contents in the profile towards the end of winter, presumably as a result of lateral dispersion. The deeper the water table, the more dispersion is likely to occur and consequently the greater the volume of soil in which the soil moisture deficit must be overcome before recharge occurs. Table 5.18 shows estimates of the amount of concentrated waters that become recharge. As no actual measurements were made of indirect recharge, these estimates are only approximate.

<table>
<thead>
<tr>
<th>TABLE 5.18</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Estimated percentage of water becoming indirect recharge</strong></td>
</tr>
<tr>
<td>Discharge device</td>
</tr>
<tr>
<td>Spoon drains, path edges</td>
</tr>
<tr>
<td>Soak wells</td>
</tr>
</tbody>
</table>

Section 5.2 showed that evaporative losses by lawn roots from soak well additions would be only 5 or 25 per cent those of spoon drain/splash block additions, depending upon the length of the soak well. This estimation was taken into account in
constructing Table 5.18. An attempt was made to estimate evaporative losses in drainage waters by measuring the chloride contents of soil solutions and comparing these values with the chloride concentration of rainfall as has been carried out by Sharma et al. (1983). However the values obtained in this trial proved to be quite variable. Such a technique could prove to be the best method of quantifying deep drainage in such three-dimensional wetting fronts, if the measurements of chloride concentrations can be made with sufficient accuracy.
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5.8 TREE CANOPIES

5.8.1 Background

Section 5.1 indicated that tree canopies constitute about 22 and 11 per cent of the surface area of the N-D and S-SP catchments respectively. They also constitute about 57 per cent of native woodland areas. The aim of this section is to estimate the interception losses that occur in tree canopies in the study areas, and hence the decrease in rainfall additions to the soil.

In a review of the interception processes, Blake (1975) noted that factors which were most important in determining the amount of interception were the amount and intensity of the precipitation and the antecedent moisture status of the canopy. Blake also noted that the commonly used linear regression model of interception may over-simplify the interception process and advocated the use of a curvilinear model to describe the frequency-intensity-duration relationship. In the Perth area, Butcher (1977) has measured interception losses in Pinus pinaster forests, and found losses of 10 per cent in open forests and 26 per cent in dense forests. Farrington (unpublished data) found interception losses of about 15 per cent in Banksia woodland areas, the losses being about double immediately under canopies.

The measurement of interception in the urban areas is complicated by the large variation in tree species and ages, and the effect on rainfall distribution of wind movement over high structures, as was mentioned in Section 5.2. From planimetry it was determined that street trees constitute about 21 per cent of the total canopy in the N-D catchment and about 30 per cent in the S-SP catchment. The distribution of street tree canopies was mapped onto 1:5000 aerial photographs. In the N-D catchment, almost all (99%) street trees were Tristania conferta (Queensland box tree or brush box), with a small number of Melia azederach (lilac) in the western part of the catchment. Both these trees
are also found in private gardens in the catchments. In the S–SP catchment, 46 per cent of the street trees were *T. conferta*, 26 per cent were *Agonis flexuosa* (peppermint) and there were 11 other species used as street trees. The trees which exceeded 3 per cent were *Eucalyptus* sp. (8%), *Callistemon* (Kings Park var. – 6%) and *Ceratonia siliqua* (carob bean – 4%).

*T. conferta* constitutes about 25 per cent of the total tree canopy (i.e. street and garden) in the N–D catchment and about 15 per cent in the S–SP catchment. The age of *T. conferta* trees is the same in both catchments (planted ca. 1932), making them the most common and uniform canopy type in the two catchments. As this species was considered fairly typical of the other evergreen trees in the urban catchments, it was used for most of the measurements of interception and groundwater extraction (Section 5.9). About half of the *T. conferta* canopies in both catchments (56% in N–D, and 48% in S–SP) are trimmed, so as not to interfere with overhead power lines. The trees respond to the trimming by increasing the density of branches, so the practice is not necessarily accompanied by a decrease in leaf area (see Section 5.9). In the native woodland areas, interception by *Banksia* spp. and *Casuarina* spp. was measured indirectly.

Two methods were used to estimate interception by *T. conferta* canopies – direct measurements using troughs and indirect measurements using paired neutron access holes. Only the latter method was used in the native woodland areas. The method are discussed separately below.

5.8.2 Direct measurements

Two compartmentalised troughs were placed immediately under a *T. conferta* canopy in the N–D catchment to provide direct measurements of interception. The tree chosen for study appeared fairly typical of the species. The troughs were arranged
radially with one end against the tree trunk and the other end just inside the canopy. Measurements of throughfall were made after each storm, or every 6 to 12 hours when the rainfall occurred as a semi-continuous series of storms or fell overnight. Gross precipitation was measured in a rain gauge situated about five metres from the tree.

The proportion of interception in each of the three compartments in each trough was weighted according to the proportion of the canopy (as measured from the trunk) that each compartment represented. Interception losses were almost always greatest in the inner compartment, but this compartment represented only 11 per cent of the tree canopy. In all, the two troughs collected about 2 per cent of the throughfall from the canopy, as estimated by relative areas.

The interception readings so obtained were divided into those which occurred when the canopy was completely dry before the rain (dry canopy) and those where it was only partially dry (wet canopy). A feature of the study was the speed with which the canopy dried out when windy and/or sunny conditions occurred between storms.

Figure 5.36 shows the data obtained from the trough measurements for 27 storms in the 1982 winter. Hyperbolic functions were fitted to the data points, with the constraint that the functions intercepted the ordinate at 100. While there may be a threshold precipitation before throughfall commences, the function shown in Figure 5.36 assumes no threshold. No attempt was made to measure the intensity of the rainfall although this is considered to be a major cause of the variation in the values shown in Figure 5.36. The lowest interception loss shown in Figure 5.36 corresponds to a particularly high intensity storm which fell on a dry canopy. Assuming the 27 storms represent the normal distribution of rainfall intensities experienced during winter in Perth, then the functions that are shown represent an
FIGURE 5.36: Canopy interception by *T. conferta* trees as a function of gross precipitation and canopy wetness.

FUNCTIONS

- **DRY CANOPIES**  \[ I(\%) = 26 + \frac{74}{(1+0.15P_g)} \]
- **WET CANOPIES**  \[ I(\%) = 30.7 + \frac{70.3}{(1+0.61P_g)} \]
- **PAIRED ACCESS HOLE DATA**
average interception loss for the particular *T. conferta* canopy studied. Also shown in Figure 5.36 are five data points obtained from comparing the change in storage in paired soil profiles (indirect measurements). These data points will be discussed in the next section.

The percentage interception that was measured by the troughs was surprisingly high. The values that were obtained for wet canopies are similar to functions shown by Blake (1975) for *Pinus radiata* forests receiving between 2 and 4 mm/hour, while those for dry canopies correspond to rainfall intensities of 0 to 2 mm/hour for *P. radiata* forests. The high interception values may be due to the nature of the rainfall in Perth, where rainfall is interspersed with periods of sun and wind which would dry out canopies.

### 5.8.3 Indirect measurements

The effect of interception on deep drainage was mentioned in Section 5.7. Two extra neutron access holes were drilled under *T. conferta* canopies about 20 weeks after the beginning of the water balance period to provide extra information on the effect of interception on deep drainage. In both cases, the extra hole was located adjacent (2-3 metres distant) to another neutron access hole which was not affected by interception losses. All four holes extended beneath the water table so the effect of interception on recharge could be assessed. Recharge under the canopies was only 24 and 33 per cent that of the unsheltered soil profiles. There was no evidence from the soil samples collected during drilling of the access holes that there were more roots in the soil under the canopies than in the soil around the paired access hole. The residual water contents after summer in the high interception holes did not appear to be lower than the adjacent, low interception holes. However, it is possible that more water was extracted by roots in the profiles located under the canopies. The advance of the wetting front was retarded by six to eight
weeks in the holes located under the canopies, resulting in the appreciable decrease in recharge.

The five data points from the paired holes shown in Figure 5.36 were obtained by comparing the changes in storage in the paired profiles during periods when the wetting front was advancing at the beginning of the 1982 winter and for the January 1982 rainfall. The rainfall values corresponding to these interception values were estimated by dividing the rainfall during the 14-day period by the number of rainy days. In general, these indirect interception estimates are lower than those measured in the troughs. This effect was most marked for the January 1982 rainfall. In one of the paired holes there was a greater increase in soil water storage in the hole located under the tree canopy than in the hole located in the open, following the January 1982 rainfall. These discrepancies probably result from reduced evaporation from the soil profiles located in the shade of the canopy. The January 1982 rainfall event was atypical for Perth in that it was continuous and with little wind or sunshine. Under such conditions, interception losses would have already been low before evaporation from the soil profiles commenced. During summer, the influence of shading would be at its maximum, which could explain why the difference between direct and indirect measurements of interception are at a maximum at this time of year.

Deep drainage at the two high interception sites in the native woodland areas was only 22 and 45 per cent that of the four low interception sites during 1981. During 1982, deep drainage took place in only two sites (both low interception) in the native woodlands. By comparing changes in storage in the soil profiles in the low and high interception sites, the interception values shown in Figure 5.37 were obtained. The considerable scatter of data points at 10 mm rainfall partly reflects the method whereby gross precipitation was calculated (i.e. total rainfall/rainy days). The amount of interception estimated by this method also
FIGURE 5.37: Canopy interception by Banksia-Casuarina stands as a function of gross precipitation.

\[ I(\%) = 17 + \frac{83}{1 + 0.76P_g} \]
depends upon the time of neutron logging relative to the time of rainfall. The longer the period between rainfall and logging, the greater will be the effect of evaporation differences (i.e. shading) between the sites. Interception by native woodland trees appears significantly lower than that for T. conferta. The leaf area index of woodland trees is generally low (e.g. about one on a per-tree basis; J. Dodd, pers. comm.) compared with T. conferta trees (about four on a per-tree basis; Section 5.9).

No attempt was made to measure stem flow in the study areas. Field observations during rainfall indicated that stem flow was minor in T. conferta and A. flexuosa trees, possibly due to the absorption of water by bark. Stem flow in T. conferta trees could be readily observed by the darkening of the bark, but it was rarely observed to extend to ground surface. Stem flow in smooth barked Eucalyptus spp. (e.g. E. citradora) was observed to be considerable, but was not measured.

For the calculation of interception losses in the water balances that are shown in the next Chapter, the functions shown in Figure 5.36 were used for daily rainfall figures. The wet canopy function was used for winter rainfall and the dry canopy function for summer rainfall.
5.9 PHREATOPHYTES

5.9.1 Background

Bouwer (1978) has defined phreatophytes as non-agricultural plants which take up groundwater. Phreatophyte water use has been estimated by measuring decreases in groundwater flow, falls in water tables, increases in salinity and by lysimeter and water balance methods (Bouwer 1978). Water use has also been estimated by studying the gas exchange of branches and whole trees (e.g. Greenwood and Beresford 1979; Anderson 1982).

The uptake and evaporation of soil water was discussed in Section 5.7. The aim of this section is to identify the plants in the study catchments which extract water from, or immediately above, the groundwater zone, and to arrive at an estimate of such water use. The soil water balance method that was used for measuring evaporation of soil water does not provide a means of estimating phreatophyte water use. It is of interest to know which plants are phreatophytes and how much water they extract, for groundwater management purposes. During the drought of 1977–79, a number of street trees died in the urban areas which were studied. There appeared to be no clear pattern in the death of the trees, even though it was commonly thought that the deaths resulted from the low water table that occurred during these years.

The methods of measuring phreatophyte water use that were outlined by Bouwer (1978) are generally not applicable in the urban study areas. It was not known which trees were phreatophytic. Also, with the exception of the Salix babylonica stands in the basins, trees in the urban areas are scattered. The problems of tree diversity (in species, age and watering history – which affect root development) and climatic variability, that were mentioned in the previous section, were even more apparent in the estimation of phreatophyte water use. In addition, garden
irrigation results in variations in soil moisture throughout the areas which will affect the amount of groundwater that is taken up by phreatophytes. The lithology of the vadose zone may also exert an influence on root penetration by the trees (e.g. root failure to penetrate limestone areas).

Water use by the relatively uniform street tree community was therefore measured to estimate water use by phreatophytes. One species (Tristania conferta) was studied in the N-D catchment while two species (T. conferta and Agonis flexuosa) were studied in the S-SP catchment. The proportion of the canopy that these species represent was about 25 per cent in both catchments, once the proportion of these trees in private gardens is taken into account. A few measurements were also made on S. babylonica trees. Given the tree, climate and soil variabilities given above, the estimation of phreatophyte water use can be only very approximate.

5.9.2 Identification of phreatophytes

To determine which T. conferta and A. flexuosa trees had contact with the water table, dawn readings of xylem pressure potentials (XPPs) were made during the summer of 1981/82 using a Scholander pressure bomb (see Richie and Hinckley 1975 for a review). The lower (more negative) the XPP, the more stress the plant is under. Mid-month readings were made on 29 trees (21 T. conferta and 8 A. flexuosa) which had different depths to the water table, watering regimes and canopy trimming. At least two readings were taken from each tree and three readings were taken if the first two readings differed by more than 1 bar (100 kPa).

There was no effect of trimming on XPP, but both watering and the depth of the water table had marked effects. Some difficulty was encountered in measuring the XPP of T. conferta trees due to the exudation of latex from laticifers in the cut stem. However, the potential of the latex was found to be higher.
than that of water in the xylem (as also found by Downton 1981) and it was possible to separate the two exudations.

Figure 5.38 shows selected dawn XPP values for 15 T. conferta trees in the N-D catchment over the six-month period, November 1981 to April 1982. The values for 25 and 39 m above the water table for watered trees represent the average of two trees at each elevation. February and March readings are not shown as the rainfall in late January resulted in a disruption to the overall trend in XPPs as the summer progressed. All data are given in Appendix 6.

Figure 5.38 shows that there was no discernable effect of height above the water table for trees located on a verge watered throughout the summer. There was a small increase in water stress for these trees between November and January but this was then followed by a small decrease by April. For well watered trees then, soil moisture appears to be sufficient to prevent a seasonal build-up of water stress. Measurements of XPP were also taken at mid-morning and mid-afternoon for these trees and, while there was a decrease in XPP during the day, the decrease was the same for all heights above the water table.

Figure 5.38 shows the development of considerable water stress in some unwatered trees. These trees were wilted. In November there was little difference between the XPPs of the watered and unwatered trees but by January, three trees had XPPs lower than -20 bars (-2000 kPa). The highest stressed trees were at 10.5, 12.8 and 17.0 metres above the water table, while the lowest stressed trees were at 9.8, 10.1, 10.8, 11.5 and 12.1 metres above the water table (all unwatered trees were located on the same verge, which increased in elevation). The late January rain resulted in February's values resembling those of November. By April, all trees except those at 11.5 and 12.1 metres appeared stressed. If the values for the tree at 10.5 metres above the water table are ignored, it would appear from Figure 5.38 that the
FIGURE 5.38: Dawn XPP values for unwatered (above) and watered (below) T. conferta trees in the N-D catchment.
T. conferta trees can extract groundwater from about 12.5 metres, at their present stage of development (i.e. 50 years old). Some of the trees which had contact with the water table early in the summer, appeared to lose contact in late summer and developed water stress. It may not be coincidental that this occurred in those trees which were closest to the absorption basin, which becomes a groundwater low by the end of summer (see Section 5.4).

The loss of contact with the water table was shown in the soil moisture profiles measured in an access hole located about five metres from a T. conferta tree in the S-SP catchment (Fig. 5.39). The figure shows that soil moisture depletion was concentrated in the capillary fringe above the water table. By February, the tree at this site had begun to wilt. The root zone of this tree, as identified by the zone of moisture depletion, is almost inundated when the water table is at its highest (October). Similar zones of depletion were found, to a greater or lesser extent, in six of the nine access holes which penetrated the water table. The access holes which showed zones of depletion were all located in areas receiving little or no water from sprinklers.

To determine more accurately the depth from which T. conferta trees could extract water, the girth of trees in unwatered sites was measured. It was considered likely that non-phreatophytic trees would have a smaller girth, due to the annual stress that they experience. Figure 5.40 shows the plot of girth against depth to the water table for 25 trees in the N-D catchment. Most trees situated within 13 metres of the water table had girths in excess of 80 cm, whereas no tree beyond 13 metres had a girth in excess of 80 cm. Accordingly, 13 metres was considered to be the depth from which T. conferta trees could extract groundwater.

Figure 5.41 shows dawn XEP values for eight A. flexuosa trees in the S-SP catchment (the values at 10, 17 and 20 metres are the average of two trees). The depth from which this species
FIGURE 5.39: Soil water profiles beside a *T. conferta* tree which is functioning as a facultative phreatophyte.
**FIGURE 5.40**: Tree girth as a function of height above the water table for 25 *T. conferta* trees in the N-D catchment.

**FIGURE 5.41**: Dawn XPP values for unwatered *A. flexuosa* trees in the S-SP catchment.
(at its present stage of development) can extract groundwater appears to be about 18 metres. The increase in water stress as the summer progressed, shown by the pair of trees at 17 metres, may indicate a gradual loss of contact with the water table. *A. flexuosa* trees did not develop XPP values as low as the *T. conferta* trees. *A. flexuosa* values were often above -5 bars (-500 kPa), a value only occasionally attained by *T. conferta* trees. Both *T. conferta* and *A. flexuosa* appear to be facultative phreatophytes (i.e. able to take advantage of shallow groundwater if present, but able to exist without it).

The effect of the stands of *S. babylonica* on the water table in the vicinity of the absorption basins has already been mentioned in Section 5.4. These trees have the ability to extract water from below the water table and therefore will be unlikely to lose contact as the water table falls. Janik (1979) has described pneumorrihæ ('breathing roots') in *S. pentandra*, but no such roots were noted in the basin areas. *S. babylonica* is a deciduous tree and therefore appreciable extraction of groundwater takes place only during the summer months. It is not known whether this tree is an obligate phreatophyte.

Amongst the *T. conferta* trees that had received adequate watering, those which would have been expected to be in contact with the water table did not have a higher XPP (or stomatal conductance - see later) than those not in contact. From a consideration of soil water potentials, the water table would need to be within three metres of the ground surface before groundwater would be absorbed in preference to soil water at one metre depth and at field capacity. It is also likely that the well-watered trees have failed to develop deep roots. Both gardens and street verges are usually well watered in the N-D catchment, whereas many verges in the S-SP catchment are unwatered. Therefore it is likely that groundwater extraction by facultative phreatophytes is more prevalent in the S-SP catchment.
The death of street trees during the drought years may have been caused by a sudden or prolonged isolation of phreatophytes from the water table. It was noticeable that the onset of stress in *T. conferta* trees which had recently lost contact with the water table was very rapid.

### 5.9.3 Water use by phreatophytes

To estimate water use by phreatophytes in the urban areas, a modified form of the Penman-Monteith equation (Monteith 1965, 1973) was used. Jarvis (1981) noted that under conditions of high vapour pressure deficit (D) and aerodynamic conductance, the radiation term in the Penman-Monteith equation can be neglected. Such conditions were found to exist in a woodland in the northern Perth area by Dodd (pers. comm.) in a comparison of the full equation with the simplified version. Jarvis (1981) considered forest plantations were rough enough to ensure that aerodynamic conductances were high. The roughness of urban areas is likely to be even greater than that of forests.

The modified Penman-Monteith equation that was proposed by Jarvis (1981) is:

\[
E_T = \frac{C_P \rho D g_c}{\lambda \gamma}
\]

where \(E_T\) = transpiration, \(C_P\) = specific heat of dry air, \(\rho\) = density of dry air, \(D\) = vapour pressure deficit, \(g_c\) = canopy conductance, \(\lambda\) = latent heat of vaporization, and \(\gamma\) = psychrometric constant.

To solve the above equation, only \(D\) and canopy conductance are required, the other terms being either constant or varying only slightly with temperature. Vapour pressure deficits were calculated from relative humidity and dry bulb temperature data for Perth using Tetens formula (Mills 1975). Canopy conductances for *T. conferta* trees were calculated from monthly measurements of
stomatal resistance and from estimates of leaf area index.

Stomatal resistances were measured for 18 *T. conferta* trees, twice a day (mid-morning and mid-afternoon) for one day each month during the summer of 1981/82. A Delta-T diffusion porometer was used for the measurements. This instrument measures the rate of humidification of a cup placed over plant stomata, the rate being related to stomatal resistance.

Tests were carried out prior to the study on the effect of leaf position in the canopy (i.e. sunlit/shaded, height) and leaf age on stomatal resistance. The influence of sunlight on resistance was found to be marked ($P < 0.001$), as was position in the canopy. It was concluded that the position in the canopy effect was due to the aspect of the leaves to the sun. Very little difference could be found between leaves of different ages (if they come from the same part of the tree), unless the leaves were very old or very young. It was decided that eight leaves be taken from each tree canopy to provide an estimate of stomatal resistance for the whole canopy. Four leaves were taken from high positions and four from low positions. At each level in the canopy, two leaves were taken from the sunlit side and two from the shaded side. The harmonic mean of these eight resistances was taken as the average resistance of the whole tree. This method of averaging removed any bias that would be introduced by leaves with very high resistances. The harmonic mean of resistances is the same as the reciprocal of the arithmetic mean of conductances.

Figure 5.42 shows the average stomatal resistances and XPPs of *T. conferta* trees experiencing three different watering regimes (dry = no watering, moist = light watering, and wet = medium-to-heavy watering) throughout the summer of 1981/82. The resistance values shown are the averages of mid-morning and mid-afternoon readings. The resistances for both the sunlit and shaded halves of trees are shown. During periods when the shaded half of the tree has the highest resistance, light intensity was probably
FIGURE 5.42: Average daily stomatal resistance (above) and XPP (below) for T. conferta trees experiencing three different water regimes and differing temperatures and vapour pressure deficits.
limiting the transpiration rates of the shaded leaves. This situation occurred throughout the year for the wet trees but only at the beginning and end of the year for the dry and moist trees (once the effect of the late January rainfall is taken into account in the February values). When the sunlit side of the tree has the highest resistance, environmental factors (e.g. high temperatures, low vapour pressure deficits) and/or low leaf water potentials are probably limiting transpiration rates. Jarvis (1975, 1976) has reviewed the effects of environmental and biological factors on leaf resistances.

It was noticeable that the leaves on the sunlit side of some dry trees during January had effectively ceased transpiring. The XPP of these trees reached -30 bars (-3000 KPa) at such times. High resistances were also noted for some leaves on well watered trees, after the commencement of the afternoon sea breeze. It was thought that this increase in resistance was due to the abaxial side of the leaves (which contains most of the stomata) being intermittently exposed to sunlight in the breeze.

The XPP values for moist and wet trees are shown in Figure 5.42 to change little throughout the summer. The XPP values for dry trees show a progressive increase throughout the summer (excepting February). However, the highest stomatal resistances for dry trees occurs in January, rather than at the end of summer. The maximum resistances correspond to maximum temperature and D values. The stomatal resistances of T. conferta appear very responsive to both environmental and biological factors. Jarvis (1976) noted that the correlation of stomatal resistances and leaf water potentials with environmental variables has met with only limited success to date.

Correlations between stomatal resistances and environmental variables showed D to be the most highly correlated variable for T. conferta trees, followed by temperature. Xylem pressure potentials were negatively correlated with resistance for two
months of the summer for dry trees and positively correlated for
two months with wet trees. It is possible that high transpiration
rates in wet trees cause the leaf potentials to decrease, but not
far enough to bring about stomatal closure.

To estimate the leaf area index of *T. conferta* trees, the
planar areas of those trees (both *T. conferta* and others) which
lay within 13 metres above the water table was measured by
planimetry. The planar area of *A. flexuosa* trees within 18 metres
of the water table was also measured. The proportion of trees
which were on unwatered verges was also estimated.

The area of leaves which occurs within the planar areas of
the tree canopy (i.e. the leaf area index [LAI] of individual
trees) was determined for six *T. conferta* trees, three from each
catchment and three with each type of canopy (i.e. trimmed and
untrimmed). The method used to estimate the LAIs relied upon the
clumpy nature of the canopy, each clump corresponding to a branch.
A typical branch was selected in the canopy and the number of such
branches in the canopy estimated by two observers. The estimates
almost always agreed to within 10 per cent. The typical branch
was then removed and its leaf area determined with a LI-COR
portable area meter used in conjunction with a conveyor belt.

As the branches were large (e.g. 5% of the total canopy),
the determination of leaf area was time consuming and so the
relationship between leaf area and oven dry weight was
investigated, as has been done in other studies (e.g. Monk *et al.*
1970). Small, young leaves were found to have an area/weight
ratio 50 per cent higher than large, old leaves. The method
finally adopted was to measure the leaf area of a sub-branch,
which contained both old and young leaves, to establish the
area/weight ratio for the whole branch. The area of the remaining
leaves on the branch was then determined from their oven dry
weight. The ratio of area/dry weight using this method varied
between 60 and 80 cm$^2$/g (mean = 70 cm$^2$/gm, SE = 5 cm$^2$/g). Table
5.19 shows the results obtained for the six trees measured.

Efforts were made to relate the LAIs with more easily measured tree dimensions (i.e. the dimensions noted in Table 5.19 plus volume and surface area of the canopy). The best correlation that could be obtained using the six trees examined was that for basal area of the canopy \( r = -0.656 \) which, however, was not significant at \( P < 0.05 \), given the small data set. The effect of canopy trimming probably disrupts any allometric relationship. Tree D had the highest LAI, but it had not been trimmed as recently as E and F. An average LAI of 4.2 was finally used for all the \textit{T. conferta} canopies in the study areas.

The LAIs were then converted from a per-tree basis to a catchment-area basis and the transpiration during the summer of 1981/82 estimated using the Jarvis (1981) equation. Only the stomatal resistance of unwatered trees below 1.3 m was used in the calculations. Table 5.20 summarises the estimates of phreatophyte occurrence that were used in the estimations.

The stomatal resistance of \textit{A. flexuosa} could be measured on only a few trees, due to their narrow leaves. The few measurements which were made indicated the resistances were similar to those of \textit{T. conferta} on the abaxial side of leaves. The stomatal resistance on the adaxial side of \textit{A. flexuosa} leaves was about twice that of the abaxial side, unlike the leaves of \textit{T. conferta} which have few stomata on their adaxial side. Consequently the LAI of \textit{A. flexuosa} trees were assumed to be 50 per cent more than that of \textit{T. conferta} to account for the extra stomata.

Measurements of stomatal resistances of \textit{S. babylonica} were also carried out on a few trees and compared with \textit{T. conferta} trees at the same site. The abaxial side of leaves of \textit{S. babylonica} had stomatal resistances only 70 per cent those of \textit{T. conferta} leaves. The adaxial side of leaves of \textit{S. babylonica} had
<table>
<thead>
<tr>
<th>Tree</th>
<th>Trimmed (T) or untrimmed (U)</th>
<th>Leaf area index</th>
<th>Total leaf area (m^2)</th>
<th>Tree girth (m)</th>
<th>Height of tree (m)</th>
<th>Basal area of canopy (m^2)</th>
</tr>
</thead>
<tbody>
<tr>
<td>A</td>
<td>U</td>
<td>4.7</td>
<td>143</td>
<td>1.05</td>
<td>9.9</td>
<td>30.7</td>
</tr>
<tr>
<td>B</td>
<td>U</td>
<td>4.6</td>
<td>238</td>
<td>1.43</td>
<td>10.9</td>
<td>51.5</td>
</tr>
<tr>
<td>C</td>
<td>U</td>
<td>2.1</td>
<td>113</td>
<td>1.45</td>
<td>10.9</td>
<td>53.3</td>
</tr>
<tr>
<td>D</td>
<td>T</td>
<td>7.2</td>
<td>175</td>
<td>0.89</td>
<td>5.9</td>
<td>24.4</td>
</tr>
<tr>
<td>E</td>
<td>T</td>
<td>3.5</td>
<td>169</td>
<td>1.55</td>
<td>7.5</td>
<td>47.9</td>
</tr>
<tr>
<td>F</td>
<td>T</td>
<td>3.1</td>
<td>100</td>
<td>0.85</td>
<td>5.9</td>
<td>32.0</td>
</tr>
</tbody>
</table>
### Table 5.20

**Estimates of phreatophyte occurrence in the urban areas**

<table>
<thead>
<tr>
<th>Estimate</th>
<th>N-D</th>
<th>S-SP</th>
</tr>
</thead>
<tbody>
<tr>
<td>Basal area of <em>T. conferta</em> within 13 m of water table (m²)</td>
<td>11700</td>
<td>13950</td>
</tr>
<tr>
<td>Basal area of <em>A. flexuosa</em> within 18 m of water table (m²)</td>
<td>0</td>
<td>10800</td>
</tr>
<tr>
<td>Basal area of <em>S. babylonica</em> (m²)</td>
<td>2925</td>
<td>2250</td>
</tr>
<tr>
<td>Basal area of all other trees, within 13 m of water table (m²)</td>
<td>40725</td>
<td>51300</td>
</tr>
<tr>
<td>LAI of <em>T. conferta</em> (on catchment basis)</td>
<td>0.072</td>
<td>0.037</td>
</tr>
<tr>
<td>LAI of <em>A. flexuosa</em> (on catchment basis)</td>
<td>0</td>
<td>0.041</td>
</tr>
<tr>
<td>LAI of <em>S. babylonica</em> (on catchment basis)</td>
<td>0.018</td>
<td>0.006</td>
</tr>
<tr>
<td>LAI of all other trees (on catchment basis)</td>
<td>0.251</td>
<td>0.137</td>
</tr>
<tr>
<td>Percentage of <em>T. conferta</em> trees which are unwatered</td>
<td>10</td>
<td>40</td>
</tr>
<tr>
<td>Percentage of <em>A. flexuosa</em> trees which are unwatered</td>
<td>0</td>
<td>40</td>
</tr>
<tr>
<td>Percentage of <em>S. babylonica</em> trees which are phreatophytes</td>
<td>100</td>
<td>100</td>
</tr>
<tr>
<td>Percentage of all other trees which are unwatered</td>
<td>10</td>
<td>30</td>
</tr>
</tbody>
</table>
stomatal resistances which were about six times those of the abaxial side of T. conferta leaves in a sunny position, the resistance becoming almost infinite in shaded positions. About two-thirds of S. babylonica leaves were considered to be shaded. To account for this extra transpiration, the stomatal resistance of S. babylonica leaves was assumed to be only 60 per cent that of T. conferta leaves on trees which had contact with the water table (i.e. November and December readings). The LAI of S. babylonica trees were assumed to be the same as T. conferta trees.

The percentage of each tree species which was reliant on groundwater was estimated from a knowledge of the watering regime in the two catchments. On some of the large blocks in the N-D catchment, the rear sections are not watered. In the S-SP catchment, many verges are unwatered. The estimates shown in Table 5.20 are necessarily approximate.

Table 5.21 shows the estimates of phreatophyte water use in the two catchments calculated using the modified Penman-Monteith equation and the data in Tables 5.20 and 5.21. The November and February water transpired by all but the S. babylonica trees is probably soil water rather than groundwater, as there was no difference in XPP and stomatal resistances between phreatophytes and non-phreatophytes in non-watered sites during these months. Transpiration by S. babylonica represents about 50 per cent of the total transpiration by phreatophytes in the N-D catchment. On a catchment basis, the percentage of potential evaporation that is transpired was only about 3 per cent in the N-D catchment and 3.5 per cent in the S-SP catchment. The size of the transpiration flux is therefore quite small in both catchments and therefore the large errors in estimating it are of probably little consequence to the overall urban water balances.
<table>
<thead>
<tr>
<th></th>
<th>Nov</th>
<th>Dec</th>
<th>Jan</th>
<th>Feb</th>
<th>Mar</th>
<th>Apr</th>
</tr>
</thead>
<tbody>
<tr>
<td>Average vapour pressure deficit (mb)</td>
<td>7.1</td>
<td>9.4</td>
<td>19.4</td>
<td>7.5</td>
<td>11.5</td>
<td>7.1</td>
</tr>
<tr>
<td>Stomatal conductance for <em>T. conferta</em> trees (m/s x 10^{-3})</td>
<td>1.34</td>
<td>1.34</td>
<td>0.41</td>
<td>2.64</td>
<td>0.87</td>
<td>1.03</td>
</tr>
<tr>
<td>Stomatal conductance for <em>S. babylonica</em> trees (m/s x 10^{-3})</td>
<td>2.23</td>
<td>2.23</td>
<td>2.23</td>
<td>2.23</td>
<td>2.23</td>
<td>2.23</td>
</tr>
<tr>
<td>N-D Catchment</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Transpiration by <em>T. conferta</em> (mm/d)</td>
<td>0.02</td>
<td>0.03</td>
<td>0.02</td>
<td>0.03</td>
<td>0.02</td>
<td>0.02</td>
</tr>
<tr>
<td>Transpiration by <em>S. babylonica</em> (mm/d)</td>
<td>0.09</td>
<td>0.12</td>
<td>0.25</td>
<td>0.10</td>
<td>0.15</td>
<td>0.09</td>
</tr>
<tr>
<td>Transpiration by all other trees (mm/d)</td>
<td>0.08</td>
<td>0.10</td>
<td>0.06</td>
<td>0.10</td>
<td>0.08</td>
<td>0.06</td>
</tr>
<tr>
<td>Total transpiration (mm/d)</td>
<td>0.19</td>
<td>0.25</td>
<td>0.33</td>
<td>0.23</td>
<td>0.25</td>
<td>0.17</td>
</tr>
<tr>
<td>Penman potential evaporation (mm/d)</td>
<td>9.0</td>
<td>9.2</td>
<td>11.2</td>
<td>7.2</td>
<td>7.6</td>
<td>5.2</td>
</tr>
<tr>
<td>Percentage of potential rate</td>
<td>2.1</td>
<td>2.7</td>
<td>2.9</td>
<td>3.2</td>
<td>3.3</td>
<td>3.3</td>
</tr>
<tr>
<td>S-SP Catchment</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Transpiration by <em>T. conferta</em> (mm/d)</td>
<td>0.04</td>
<td>0.06</td>
<td>0.04</td>
<td>0.06</td>
<td>0.05</td>
<td>0.03</td>
</tr>
<tr>
<td>Transpiration by <em>A. flexuosa</em> (mm/d)</td>
<td>0.05</td>
<td>0.07</td>
<td>0.04</td>
<td>0.06</td>
<td>0.05</td>
<td>0.04</td>
</tr>
<tr>
<td>Transpiration by <em>S. babylonica</em> (mm/d)</td>
<td>0.03</td>
<td>0.04</td>
<td>0.08</td>
<td>0.03</td>
<td>0.05</td>
<td>0.03</td>
</tr>
<tr>
<td>Transpiration by all other trees (mm/d)</td>
<td>0.12</td>
<td>0.16</td>
<td>0.10</td>
<td>0.16</td>
<td>0.13</td>
<td>0.10</td>
</tr>
<tr>
<td>Total transpiration (mm/d)</td>
<td>0.24</td>
<td>0.33</td>
<td>0.26</td>
<td>0.31</td>
<td>0.28</td>
<td>0.20</td>
</tr>
<tr>
<td>Percentage of potential rate</td>
<td>2.7</td>
<td>3.6</td>
<td>2.3</td>
<td>4.3</td>
<td>3.7</td>
<td>3.8</td>
</tr>
</tbody>
</table>
5.10 GROUNDWATER

5.10.1 Background

Previous sections have quantified the 'vertical' inputs and outputs to the groundwater in the study areas. This section examines changes in groundwater storage and groundwater inflows and outflows.

The estimation of the specific yield and transmissivity for the aquifer in the Perth area by pump tests has been hindered by the 'leaky-arterian' response (Pollett 1981). The hydraulic conductivity of a sandy section of the Tamala Limestone at Gwelup (9 km north of the S-SP catchment) has been estimated from pump tests to be about 25 m/d (Metropolitan Water Authority (MWA), unpublished data). Allen (1981) estimated the hydraulic conductivity of the sands on the Swan Coastal Plain to be about 30 m/d and of the calcareous parts of the aquifers to be about 100 m/d. Allen also estimated the specific yield of the aquifers to be about 0.3.

Pump tests have been carried out in the southern Perth area by ALCOA of Australia (Hazelgrove, pers. comm.). These tests have shown transmissivity to be very variable, but throughflow (transmissivity x hydraulic gradient) to be much less variable. Such a situation indicates approximately steady-state conditions. Davidson (1981) carried out flow net analyses in the southern Perth area. Transmissivities in the area were estimated from pump test analyses and from expected hydraulic conductivities for various lithologies (e.g. medium sand - 16.5 m/d, medium to coarse sand - 33 m/d, limestone - 100 m/d). Davidson acknowledged that this method of estimating transmissivities involved inaccuracies.

No pump test or flow net analyses have been carried out in the study areas. Bouwer (1978) noted that the specific capacity of wells has been used to determine the transmissivity
distribution in aquifers. Such a procedure is possible only if the wells are of a similar depth and construction (making well losses similar so that the only difference between wells is due to formation losses) and situated in the same aquifer.

There are some bore log and specific capacity data available for the study areas from records kept by drilling companies on the installation of private and public bores. Many of these bores have similar construction details.

5.10.2 Geological cross sections

Five geological cross sections across the Dalkeith peninsula were constructed using drill log information (Fig. 5.43). Three of these sections include the N-D catchment. As the logs were compiled by the drillers of the holes, it is possible they include inaccuracies.

The sections show that limestone is the principal aquifer material in the N-D study area. The thickest sand sequences appear in the low-lying area (i.e. around the absorption basin) and in the southern part of the catchment. The base of the aquifer was taken from a GSWA map (Fig. 5.44) which shows that a deep channel is evident in the S-SP catchment while a basement high extends towards the N-D catchment.

Only partial sections could be constructed for the S-SP catchment, as bore log information was very limited in this area (Fig. 5.45). Most sections are sandy, in contrast to the sections in the N-D catchment. There are, however, insufficient data on the deeper parts of the aquifer in this catchment.

5.10.3 Water level network

In order to estimate hydraulic gradients and changes in groundwater storage in the urban areas, a network of groundwater
FIGURE 5.43: Geological cross-sections across the Dalkeith peninsula. (Source: Drillers logs on file at GSWA)
FIGURE 5.44: Contours of the base of the 'superficial formations' in the study area.
(Source: GSWA).
FIGURE 5.45: Geological cross-sections across the S-SP catchment.
(Source: Drillers logs on file at the GSWA).
level sites was established. Dupuit-Forschheimer assumptions were adopted for the measurements from the network (i.e. equipotentials were assumed to be vertical and hydraulic gradients were assumed to equal the slope of the water table and to be invariant with depth). Measurements from the piezometric pairs around the absorption basin (Section 5.4) indicated that there was little upward component of hydraulic head, once sites were located more than about 50 metres from the basins. The groundwater level network included sites which were screened at different depths below the water table, but the isopotential maps produced from the network contained no anomalous sites. The adoption of Dupuit-Forschheimer assumptions therefore seems valid in all areas except those immediately adjacent to the basins.

There were no water level monitoring bores within the study areas before the study commenced. However one bore (GE4 - see Fig. 5.46) was located 200 metres south of the S-SP catchment boundary and had been monitored since June 1978.

Methods were developed whereby water level readings could be obtained from private bores and wells. These methods are outlined in Appendix 7. Not all bores were suitable for taking water levels and some owners refused permission for their bores to be used. These problems created little difficulty in the N-D catchment as there were a number of alternative bores available. However the scarcity of bores in the S-SP catchment meant that there were some areas of the study area with few monitoring points. The regular nature of the groundwater potentials made this lack of coverage of little consequence.

In all, 24 private bores were used for measuring groundwater levels, in conjunction with the 21 shallow bores that had been drilled around the basins and the single monitoring bore that was available south of the S-SP catchment. Water level probes were left in most of the private bores between readings, so that errors would not arise from the probe being lowered along
FIGURE 5.46: Thiessen networks for groundwater storage calculations in the N-D (above) and S-SP (below) catchments.
different routes each time a measurement was taken.

Figure 5.46 shows the Thiessen networks that were established for the two urban areas so that changes in groundwater storage could be estimated. Bores outside the study areas were chosen to obtain hydraulic gradients for calculation of groundwater inflows and outflows. Figure 5.46 shows there was an extra density of bores around the basins. These bores were established to measure the large changes in storages which occur in these areas, as well as to aid in the basin water balances (Section 5.4). One bore (11 VS) was lost about half way through the study. Fortnightly readings were obtained from all other bores throughout the 20 month study period, weekly readings being obtained from the bores near the basins. The reduced levels were obtained for all water level sites by levelling from bench marks of known elevation. Levels were taken to the top of the bore base plate (for submersible pumps) or to the top of the cement well liner (for centrifugal pumps). For holes which were drilled around the basins, the levels of stakes driven into the ground beside the holes were obtained (Section 5.4).

5.10.4 Water levels

Figure 5.47 shows the configuration of the water table in the N-D catchment at times of minimum and maximum groundwater storage (for storage calculations, see Section 5.10.5). The water table was only 60 cm above sea level in the southern part of the catchment in May 1981. Limited salt water intrusion was reported around the Dalkeith peninsula towards the end of summer during the drought years, 1977-79. At maximum storage, the hydraulic gradient south of the basin was considerably greater than that at minimum storage, and also greater than that north of the basin. The water table was 60 to 80 cm higher in October 1981 relative to May 1981. Flow directions had also altered slightly.

Figure 5.48 shows the configuration of the water table in
5.5.1981

LEGEND
- WATER LEVEL SITE
- 0.7 WATER TABLE CONTOUR (m AHD)
- FLOW LINE

5.10.1981

FIGURE 5.47: Water table contours in the N-D catchment at times of low (above) and high (below) groundwater storage.
FIGURE 5.48: Water table contours in the S-SP catchment at times of low (above) and high (below) groundwater storage.
the S-SP catchment at the same times as for the N-D catchment shown in Figure 5.47. Hydraulic gradients do not appear to change as markedly between seasons in this catchment (see Section 5.10.6.1). The water table was over 100 cm higher near the basin in October but only about 60 cm higher in the north of the catchment. Flow directions stay approximately the same, apart from distortions round the absorption basin.

Figure 5.49 (a) and (b) shows the hydrographs for the individual bores within the two urban catchments. In both catchments, fluctuations in groundwater levels were most extreme in the basin areas, with the levels being more subdued and delayed in areas where the water table was deep below the ground surface. The rise in levels following the January 1982 rainfall was delayed by two weeks in the bores furthest from the basin in the S-SP catchment. Maximum water levels were similar during 1981 and 1982 but the minimum water levels in 1982 were appreciably higher than in 1981 (about 150 mm in the N-D catchment and about 300 mm in the S-SP catchment).

All the bores near the basin in the S-SP catchment show a cut-off hydrograph peak, as a result of the flood mitigation drain which operates in this area. Water levels at the SE site are affected to some extent by pumping to operate the fountain in Shenton Park Lake.
FIGURE 5.49 (a): Individual bore hydrographs in the N-D catchment.
FIGURE 5.49 (b): Individual bore hydrographs in the S-SP catchment.
5.10.5 Groundwater storage

Groundwater storage and change in storage were calculated over 14-day periods for both catchments from a knowledge of Thiessen areas (Fig. 5.46), aquifer thicknesses (Fig. 5.44) and water levels (Fig. 5.49). The specific yield of the aquifer was assumed to be 0.3 (Allen 1981). The values obtained from neutron logging and moisture characteristic curves confirm this estimate for the sands.

It was necessary to incorporate the storage of surface water in the basins in the calculations of groundwater storage. For computational reasons, functions were fitted to relate storage volume (m$^3$) in the basins with reduced water levels (Fig. 5.9). These functions were:

\[
\begin{align*}
\text{N-D basin} & \quad \text{Volume} = 198.6 \text{ Level}^{3.029} \\
\text{S-SP basin*} & \quad \text{Volume} = -82734 + 24847 \text{ Level} \\
\end{align*}
\]

\[r = 0.997 \quad r = 0.999\]

* Volumes for levels between 3.5 and 5.5 m AHD only — i.e. for periods when the basin is largely retained within its confining walls.

Figure 5.50 shows the results of these calculations. As was shown by the hydrographs, groundwater storages had similar maxima during 1981 and 1982 but the minima in 1982 were higher (50 mm for the N-D catchment and 100 mm for the S-SP catchment) than in 1981. Groundwater storage increased more in the S-SP catchment than in the N-D catchment over the 20 month period. This may be due to imported waters being pumped into the S-SP basin when groundwater levels were high in the adjoining catchment during 1982. The slight rise in groundwater levels in the N-D catchment at the start of the study was the result of 16 mm of rainfall which fell three days before the water balances commenced.

The change in storage in both catchments was very similar. The January 1982 rainfall produced one of the largest increases (and
FIGURE 5.50: Groundwater storage (above) and change in storage (below) in the two urban catchments.
subsequent decreases) in storage in both catchments for a 14-day period. During periods of rapid increases in storage, the S-SP catchment responded more quickly than the N-D catchment. Towards the end of summer (i.e. April-May) there was little change of storage in the aquifers, which indicates approximately steady-state conditions.

5.10.6 Groundwater flow

5.10.6.1 Hydraulic gradients:

As mentioned previously, Dupuit-Forschheimer assumptions were involved in the estimation of groundwater flows into and out of the study catchments. Groundwater flow was calculated on a fortnightly basis using water level data to calculate hydraulic gradients and transmissivities for each period. Water level data from up to three pairs of bores which straddled the catchment boundaries were used to estimate hydraulic gradients. The aquifer base map (Fig. 5.44) was used in the estimation of transmissivities.

Hydraulic gradients into and out of the study areas were calculated by the following expressions (see Fig. 5.46 for bore locations).

**N-D catchment**

Gradient in = \[\frac{(7LS - 47MA) + (84FR - 155DR)}{165 + 190}\] / 2

Gradient out = \[\frac{(56GR - 13PR) + (155DR - 31DR)}{200 + 230}\] / 2

**S-SP catchment**

Gradient in = \[\frac{(KEMH - 51GS)}{700}\]

Gradient out = \[\frac{(RA - HA) \times 0.20}{775}\] + \[\frac{(4YS - 49YS)}{245} \times 0.35\]

+ \[\frac{(161HR - 23CS)}{245} \times 0.45\]
where 7LS is the water level in bore 7LS, etc.

165 is the distance, along the flow line, between bores 7LS and 47MA. The gradients are weighted according to the distance they represent.

Figure 5.51 shows the gradients for the two urban areas throughout the water balance period. In the N-D catchment, the gradient out of the catchment was greater than the gradient in for about 80 per cent of the year while in the S-SP catchment, the gradient out was always greater than that in. Gradients in the S-SP catchments were about four times those in the N-D catchment.

Gradients in the N-D catchment were at a maximum in about November and at a minimum in about June during 1981 and 1982. These times were about one month after maximum and minimum storage periods respectively (Fig. 5.50).

Gradients in the S-SP catchment were at a maximum in about April and at a minimum in about October during 1981 and 1982. These times coincide almost exactly with minimum and maximum storage periods respectively.

The reason for this different response between the catchments is not clear. Gradients would be expected to increase with groundwater storage (as seems the case in the N-D catchment) as the difference in potential between the groundwater and sea level increases. The anomalous behaviour of the S-SP catchment may be due to the flood mitigation drain which is operative at times of maximum groundwater storage. This drain flattens hydrograph peaks during periods of maximum storage and may therefore decrease hydraulic gradients at this time. The annual change in gradients in the S-SP catchment is considerably lower than that in the N-D catchment, when considered on a percentage basis (i.e. about 25 per cent versus 125 per cent).
FIGURE 5.51: Hydraulic gradients of groundwater flowing into and out of the N-D (above) and S-SP (below) catchments.
5.10.6.2 Hydraulic conductivity:

A number of different methods were used to obtain an estimate of the hydraulic conductivities of the urban areas. From a consideration of lithology, the expected hydraulic conductivity of the sandy S-SP area would be about 30 m/d (Allen 1981; Davidson 1981; MWA, unpubl. data) and of the calcareous N-D area, about 100 m/d (Allen 1981; Davidson 1981). Three other methods were used to obtain information on actual- or relative-hydraulic conductivities for the areas. These methods were flow net analysis, a comparison of specific capacities, and pump test analyses. The methods are outlined below.

(A) Flow net analysis:

Using the water level networks which had been established in the two urban areas, in addition to two MWA bores located between the areas, a flow net for April 1981 was drawn up for the area shown in Figure 5.52. Assuming flow at this time was approximately steady-state (as was indicated in Fig. 5.50), the relative hydraulic conductivities at A-A' and B-B' (i.e. $K_1$ and $K_2$) were calculated as shown below.

If $Q_{AA'} = Q_{BB'}$, then

$$K_1 \times b_1 \times \frac{i_1}{l_1} = K_2 \times b_2 \times \frac{i_2}{l_2}$$

where $Q_{AA'}$, = flow across AA' in m$^3$/d etc.

$b_1$ = thickness of aquifer at AA' etc.

$i_1$ = hydraulic gradient at AA' etc.

$l_1$ = length of isopotential at AA' etc.

$$\frac{K_1}{K_2} = \frac{b_2 \times \frac{i_2}{l_2}}{b_1 \times \frac{i_1}{l_1}}$$

$$= \frac{13 \times 0.0005 \times 1350}{33 \times 0.0011 \times 400}$$

$$= 0.6$$
FIGURE 5.52: Flow net for the study area, April 1981.
i.e. the hydraulic conductivity in the S-SP catchment is only about 60 per cent that in the N-D catchment, according to this method.

(B) Specific capacity:

Using data obtained by drillers in the Nedlands-Dalkeith-Claremont area, the distribution of the specific capacity of bores was calculated and plotted as shown in Figure 5.53. The sparsity of measurements means that only general trends in formation losses can be gained from Figure 5.53. The transmissivity in the central part of the N-D catchment appears less (lower specific capacity) than in surrounding areas. The zone of lower transmissivity parallels the ridge in the base of the aquifer, but is offset to the south west (Fig. 5.44). The low specific capacities in the central and southern part of the catchment coincide with the sandy sections of the aquifer (Fig. 5.43).

This method of obtaining relative transmissivities seems to have some potential, considering the probability of errors in the specific capacity measurements and differences in bore construction. However, the information in Figure 5.53 cannot be used quantitatively to compare the N-D and S-SP catchments as there were almost no data from the S-SP catchment.

(C) Pump test:

A 10-hour pump test was carried out using a bore located about 65 metres north of the absorption basin in the N-D catchment. The bore was flow-tested before the pump test was carried out and found to produce 20.6 m$^3$/h under open flow conditions (this decreasing to 10.2 m$^3$/h at 400 kPa).

The plan and elevation of the pumping and observation bores are shown in Figure 5.54. The pumping bore was screened over about 40 per cent of the lower part of the aquifer. It was intended to
FIGURE 5.53: Specific capacities of private bores in the Dalkeith peninsula.
(Source: Drillers logs on file at the GSMA).
FIGURE 5.54: Plan and elevation (inset) of the pumping and observation bores used in the pump test.
locate an observation bore (D1) about one aquifer thickness from the pumping bore and slotted over the same interval. However at a depth of 10 metres into the aquifer (14 m from the surface), hard limestone was encountered and the observation bore could not be extended below this depth using the method adopted for its insertion (i.e. jetting). A shallow bore was also located at this site (S1) and two moderately deep bores (D2 and D3) were located about two and three aquifer thicknesses away from the pumping bore respectively. The N-D absorption basin was about four aquifer thicknesses from the pumping bore. Water from the pumping bore was discharged into a cement-lined pond. Overflow from this pond flowed into the absorption basin after the pump test had been going for 240 minutes. The absorption basin showed almost no response to the pumping until overflow from the cement-lined discharge pond commenced. After this time, the basin water level gradually increased until by the end of the test, it was 17 mm higher than at the start.

As no observation bore was located at the same depth (or in the same lithology) as the screen in the pumping bore, the data from the pumping test could not be used to obtain a measure of transmissivity (and therefore, hydraulic conductivity) and storage coefficient of the aquifer. However, the test did show the hydraulic response of the aquifer to pumping, as measured at varying depths and distances from a pumping bore.

Figure 5.55 shows the time-drawdown curves obtained from the four observation bores. The very low drawdowns in the observation bores indicate the transmissivity of the aquifer is very high and/or the aquifer is very anisotropic. In a pump test in a sand exhibiting a semi-confined response to pumping, Webb and Watson (1979) recorded drawdowns of about two metres in piezometers situated at a similar radius and location as D1, when pumping at about 100 m³/h. The drawdown at D1 at the same time was only 0.08 metres, for a pumping rate of 20.6 m³/h. The shallow piezometer (S1) was much slower in responding to pumping than was site D1, but the difference in drawdowns decreased with pumping time. Both sites
FIGURE 5.55: Time-drawdown curves for the four observation bores used in the pump test.
show evidence of delayed yield between 10 and 100 minutes. There was only two metres between the top of the screen in D1 and the bottom of the screen in S1. The difference in response indicates some degree of anisotropy in the aquifer. Pollett (1981) reported an absence of response in some observation bores screened over the top of the aquifer during pump tests at Jandakot, south of Perth.

The slightly greater drawdown at site D3 relative to site D2 may be due to the greater depth of the screen at D3, despite its greater distance from the pumping bore.

The discharge rate of the pumping bore was about four times that of an average private bore (Section 5.6). Individual private bores are therefore likely to have only minor, short-term effects on the water table. Interference between private bores is therefore unlikely to be a problem.

5.10.6.3 Flow calculations and computed recharge:

Groundwater flow into and out of the study areas was calculated on a fortnightly basis using the hydraulic gradients that were shown in Section 5.10.6.1, aquifer thicknesses that were also calculated on a fortnightly basis and hydraulic conductivities of 30 m/d for the S-SP catchment and 50 m/d for the N-D catchment. The hydraulic conductivity for the S-SP catchment was decided from considerations of lithology and the pump test results from the sandy, Gwelup wellfield (MWA, unpubl. data). The hydraulic conductivity for the N-D catchment was estimated relative to that in the S-SP catchment using the flow net results.

Figure 5.56 shows the flows into and out of the two urban areas for the water balance period. The figure also shows estimates of net recharge/discharge computed from:

\[ \text{Recharge} = (\text{outflow} - \text{inflow}) + \text{change in storage}. \]
FIGURE 5.56: Groundwater inflow and outflow for the N-D (top) and S-SP (middle) catchments and calculated net recharge/discharge for the two catchments (bottom).
Groundwater inflows and outflows show the same relationship as was shown by hydraulic gradients (Fig. 5.51) as transmissivities changed little with time. As the difference between groundwater inflow and outflow remains fairly constant throughout the year, calculated recharge/discharge shows a similar relationship to change in storage (Fig. 5.51).

On an annual basis, recharge and discharge took place for similar periods in both catchments. The S-SP catchment had a higher recharge and/or lower discharge rate than the N-D catchment for about 95 per cent of the 20 month study period. The difference between the catchments was most marked at high recharge periods. This difference in short term response is probably due to the greater percentage of roads and greater amount of imported drainage waters in the S-SP catchment.
CHAPTER 6

WATER BALANCES - INTEGRATION AND DISCUSSION

6.1 INTEGRATION

The integration of measurements that were detailed in Chapter 5 needs to take account of the different time scales involved. Some measurements were of very short term events (e.g. road runoff, interception losses by roofs and trees), while others were long-term events (e.g. recharge which was calculated on a six monthly basis). The shortest integration possible was that of the longest event, recharge.

In deciding where to divide the water balance periods, consideration had to be given to lags in the processes. Thus, groundwater storages began declining in early October while garden irrigation commenced in late October and recharge of winter rainfall continued until late November.

The periods over which the water balances were integrated were a compromise between the various cycles. Three six-month periods were integrated (Table 6.1), resulting in data from the first and last four-week periods of measurement being omitted from the calculations.

Table 6.1

Water balance periods

<table>
<thead>
<tr>
<th>Period</th>
<th>Interval</th>
<th>Duration (days)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Summer 1981/82</td>
<td>2nd November 1981 to 2nd May 1982</td>
<td>182</td>
</tr>
<tr>
<td>Winter 1982</td>
<td>3rd May 1982 to 29th October 1982</td>
<td>182</td>
</tr>
</tbody>
</table>
A check on the overall precision of the water balances was made by comparing the estimates of net recharge of the groundwater by two independent methods. The first method involved adding all the 'vertical fluxes' to/from the groundwater (i.e. all except groundwater inflows and outflows) while the second involved calculating net recharge from groundwater flows and changes in storage ('horizontal flux' method), as was carried out in Section 5.10. Table 6.2 gives the results of these calculations.

Table 6.2

Net recharge calculations for different water balance periods for the two urban catchments (all figures in mm)

<table>
<thead>
<tr>
<th></th>
<th>Vertical flux method</th>
<th>Horizontal flux method</th>
</tr>
</thead>
<tbody>
<tr>
<td>N-D Catchment</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Winter 1981</td>
<td>+ 197</td>
<td>+ 219</td>
</tr>
<tr>
<td>Summer 1981/82</td>
<td>- 178</td>
<td>- 120</td>
</tr>
<tr>
<td>Winter 1982</td>
<td>+ 160</td>
<td>+ 111</td>
</tr>
<tr>
<td>Summer 1981/82 + Winter 1982</td>
<td>- 18</td>
<td>- 9</td>
</tr>
<tr>
<td>S-SP Catchment</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Winter 1981</td>
<td>+ 251</td>
<td>+ 309</td>
</tr>
<tr>
<td>Summer 1981/82</td>
<td>- 15</td>
<td>- 42</td>
</tr>
<tr>
<td>Winter 1982</td>
<td>+ 211</td>
<td>+ 210</td>
</tr>
<tr>
<td>Winter 1981 + Summer 1981/82</td>
<td>+ 236</td>
<td>+ 267</td>
</tr>
<tr>
<td>Summer 1981/82 + Winter 1982</td>
<td>+ 196</td>
<td>+ 168</td>
</tr>
</tbody>
</table>

Table 6.2 shows that there was good agreement between the methods in the S-SP catchment but some discrepancy in the N-D catchment for the summer 1981/82 and winter 1982 periods. When grouped into 12-month periods, some of these discrepancies cancelled
out. Good agreement between the methods does not necessarily mean the results are accurate, as there are likely to be compensatory errors in the measurements (e.g. soil recharge and phreatophyte water use may both be under- or over-estimated). Groundwater flows in the N-D catchment were at a maximum during the summer of 1981/82, as a result of the high water levels following the wetter 1981 winter. It is possible that this lag in groundwater flow accounts for some of the discrepancy that exists between the vertical and horizontal methods of calculating net recharge, particularly as the discrepancy diminishes when annual volumes are considered.

Both catchments were strong recharge areas during the winter periods. During the summer period, the N-D catchment was a strong discharge area while the S-SP catchment was only a slight discharge area. On an annual basis, the period Nov 1981 to Oct 1982 was a slight discharge period in the N-D catchment but a strong recharge period in the S-SP catchment. The discharge in the N-D catchment is greater when imported drainage waters (19 mm) are taken from the vertical recharge.

Rainfall in the winter 1981 + summer 1981/82 period was 10 per cent above the Perth average in the N-D catchment and 3 per cent above average in the S-SP catchment. Rainfall in the summer 1981/82 + winter 1982 period was 3 per cent below the Perth average in the N-D catchment and 10 per cent below average in the S-SP catchment. However, the distribution of rainfall in 1982 was atypical, winter rainfall being 10 per cent below average in the N-D catchment and 18 per cent below average in the S-SP catchment.

The errors in estimating the fluxes in the water balance are variable. Over a season the error in estimating rainfall inputs is likely to be ±10 per cent, although the error will be greater for individual storms. The error in estimating metered water use is likely to be ±15 per cent, once the error in meter recording is added to the 10 per cent error arising from the sample size. It is much more difficult to estimate errors for the other fluxes,
although it may be assumed that phreatophyte water use has a large percentage error given its small flux and difficulties in its measurement.

Given the errors involved, it can be said with confidence only that the N-D catchment is close to balance under average rainfall conditions and that the S-SP catchment is a net recharge site under similar conditions.

6.2 DETAILED BALANCES OF THE URBAN AREAS

Figure 6.1 (a,b,c) shows the detailed water balance for the N-D catchment for each six-month period while Figure 6.2 (a,b) shows the two annual water balances for the same catchment. The inputs and outputs to the groundwater storage box do not balance, as the fluxes were calculated separately. The major sources of recharge to the groundwater on an annual basis are roof runoff (45%), soil recharge (30%) and road (absorption basin) recharge (24%). The shedding area surfaces therefore contribute almost 70 per cent of recharge while occupying only 31 per cent of surface area. The major discharges from the groundwater are private- (78%) and public- (6%) bores and phreatophytes (15%). Imported MWA water volumes were 25 to 30 per cent of rainfall inputs to the urban water cycle, but almost half of this imported water was lost to an ocean outfall, due to the almost complete sewerage of the area. The amount of water intercepted by tree canopies exceeds the amount withdrawn by phreatophytes. The effect of canopy interception is even greater when the effect on direct recharge is taken into account (Section 5.7).

Figures 6.3 (a,b,c,) and 6.4 (a,b) show the detailed seasonal and annual water balances for the S-SP catchment respectively. The major recharge to the groundwater on an annual basis was from roof runoff (43%), road runoff (36%) and soil (direct) recharge (20%). The contribution of recharge from shedding areas was almost 80 per cent, from 43 per cent of the surface area of catchment. From an
FIGURE 6.1 (c) : Water balance of the N-D catchment, May - Oct 1982.
FIGURE 6.3 (a) : Water balance of the S-SP catchment, May - Nov 1981.

- **Water Source/Sink**: Water flux (mm/yr)
- **Temporary Water Storage**: Volume (mm)

**Legend**:
- **WATER SOURCE/SINK**: Water flux (mm/yr)
- **TEMPORARY WATER STORAGE**: Volume (mm)

**Streams and Flows**:
- **Atmosphere**: Rainfall (150 mm), Runoff (619 mm), Evaporation (50 mm)
- **Soil**: Rainfall (150 mm), Runoff (619 mm), Evaporation (50 mm)
- **Septic Tank**: Inflow (1 mm), Outflow (74 mm)
- **Public Bore**: Inflow (2 mm), Outflow (5 mm)
- **Private Bore**: Inflow (2 mm), Outflow (5 mm)
- **Fountain**: Inflow (4 mm), Outflow (5 mm)
- **Phreatophytes**: Inflow (2 mm), Outflow (5 mm)
- **Ocean Outfall**: Inflow (74 mm), Outflow (0 mm)
- **Groundwater**: Inflow (349 mm), Outflow (286 mm)

**Flow Rates**:
- **Atmosphere**: Rainfall (150 mm), Runoff (619 mm), Evaporation (50 mm)
- **Soil**: Rainfall (150 mm), Runoff (619 mm), Evaporation (50 mm)
- **Septic Tank**: Inflow (1 mm), Outflow (74 mm)
- **Public Bore**: Inflow (2 mm), Outflow (5 mm)
- **Private Bore**: Inflow (2 mm), Outflow (5 mm)
- **Fountain**: Inflow (4 mm), Outflow (5 mm)
- **Phreatophytes**: Inflow (2 mm), Outflow (5 mm)
- **Ocean Outfall**: Inflow (74 mm), Outflow (0 mm)
- **Groundwater**: Inflow (349 mm), Outflow (286 mm)

**Other Flows**:
- **Reticulation System**: Inflow (101 mm), Outflow (82 mm)
- **Residences**: Inflow (1 mm), Outflow (17 mm)
- **Roads and Paths**: Inflow (88 mm), Outflow (77 mm)
- **Soakwells**: Inflow (27 mm), Outflow (101 mm)
- **Flood Mitigation Drain**: Inflow (101 mm), Outflow (77 mm)
- **Imported Drainage**: Inflow (92 mm), Outflow (0 mm)
- **Spoon Drains Path Edges**: Inflow (38 mm), Outflow (50 mm)

**Water Balance**:
- **Total Inflow**: 101 mm + 82 mm + 1 mm + 17 mm + 88 mm + 27 mm + 101 mm + 92 mm + 38 mm = 349 mm
- **Total Outflow**: 82 mm + 17 mm + 77 mm + 101 mm + 0 mm + 50 mm = 286 mm
- **Net Inflow**: 349 mm - 286 mm = 63 mm

**Note**: The diagram is a representation of the water balance for the S-SP catchment, showing the inflows, outflows, and storage for various water bodies and flows.
extra 12 per cent of shedding area relative to the N-D catchment, the proportion of recharge from shedding areas in the S-SP catchment was only 10 per cent higher. Unlike the N-D catchment, however, the imported drainage waters to the absorption basin, plus some extra water, are lost to a flood mitigation drain. Direct recharge during summer is less in the S-SP catchment due to lower rates of watering. The effect of the smaller percentage of soak wells in the S-SP catchment also results in recharge from roof outlets being proportionally lower. Imported MWA water in the S-SP catchment was 30 to 35 per cent of rainfall additions, 5 per cent higher than in the N-D catchment, due to greater housing density and lower bore density.

The major discharges from the groundwater in the S-SP catchment were phreatophytes (37%), private- (21%) and public- (15%) bores and the fountain in Shenton Park Lake (19%). Bore extraction in the S-SP catchment during the 1981/82 summer was only 16 per cent that of the N-D catchment. While draining Shenton Park Lake, the flood mitigation drain indirectly discharges the groundwater and is actually the major discharge process in the S-SP catchment.

6.3 DISCUSSION

The major difference between the urban and woodland areas (Section 5.7.7.1) appears to be the production of indirect recharge by shedding surfaces in urban areas. Indirect recharge occurs after all, except very light, rainfalls. The January 1982 rainfall resulted in major recharge in the urban areas but only wet the top two metres of soil in the woodland sites. Interception losses were also greater in woodland areas and the soil water deficits at the start of winter were much greater in the woodland soil profiles. There was also about 30 per cent more water in the urban water cycle as a result of mains water additions.

In the N-D catchment, evaporation from all sources was greater than rainfall (113 and 126% in the two annual periods respectively)
while in the S-SP catchment, evaporation was slightly less than rainfall (87% and 97% respectively). Evaporation in the woodland sites was at least 100 per cent of rainfall and in the Kings Park area, probably exceeded rainfall as a result of phreatophyte extraction. The difference in evaporation rates between the two urban areas results from evaporation and transpiration of soil water in the N-D catchment, due to the 55 per cent higher irrigation rate in this catchment.

Thus, availability of water, rather than evaporative demand, limits the amount of evaporation in the urban and woodland catchments. Any effect of urbanization on potential evaporation rates is therefore likely to have little consequence in Perth.

The fact that the N-D catchment appears to be approximately in balance during average rainfall years indicates that groundwater resources are probably fully committed in this area. There is only a small potential for an increase in groundwater yield in the catchment through a lowering of water levels. Most of this induced yield would come from an increase in groundwater inflow and a decrease in groundwater outflow. There is only a small potential yield from phreatophyte use and almost no yield from evaporative losses from basin waters. However the salt water intrusion problem around the Dalkeith peninsula limits any lowering of the water table in this catchment.

A comparison of groundwater levels in the basin in the N-D catchment during 1981/82 with some spot levels from historical records indicates that groundwater levels have not declined very much (ca. 40 to 50 cm) during the last 30 years (Fig. 6.5). As groundwater demand has almost certainly increased appreciably during this time, extra recharge, decreased groundwater outflow and/or increased groundwater inflow must have occurred to prevent a significant decline in levels. Although the decline is only small in magnitude, it represents a significant percentage decrease in hydraulic gradient, given that groundwater levels are only about one metre above the Australian Height Datum. Increased groundwater
FIGURE 6.5: Comparison of current and historic water levels in Masons Gardens basin, N-D catchment.
inflow from net recharge areas up-gradient (such as the S-SP catchment area) may have played an important part. This inflow would be assisted by the high transmissivity of the aquifer. Also, the increased use of bores will result in greater direct recharge by rainfall through a decrease in soil water deficits at the commencement of winter. Some increase in percentage shedding area would also have taken place over the 30-year period as a result of additional building and paving.

The amount of recharge in the S-SP catchment appears limited by the presence of the flood mitigation drain in Shenton Park Lake. Extra recharge is likely when groundwater levels are low around the Lake at the start of winter. However, for down-gradient areas, the length of time that water levels are maintained at flood levels each year will be important in determining the amount of groundwater flow which takes place away from the catchment.

Figure 6.4 indicates that the volume of imported waters to Shenton Park Lake is equal to the volume of water exported by the flood mitigation drain. The drain sometimes operates when groundwater levels around the basin are low, if a sudden influx of water occurs (e.g. May 1981) and there is insufficient time for the runoff waters to enter the aquifer. Diverting the imported waters to another absorption basin (e.g. in the Nedlands-Dalkeith area) would result in much less water being lost to the drain. However it would also result in lower lake levels at certain times of the year. It will be seen in the next chapter that recharging the groundwater system at absorption basins, which are also wetlands, has some effect on the quality of the down-gradient groundwater.

Previous suggestions have been made (e.g. P.G. Harris, *The West Australian*, 15/3/1978) that there is a critical percentage of bore ownership, above which groundwater overdevelopment occurs in Perth. Estimates of critical ownership have ranged from about one
house in ten to one in four. The present study has highlighted other factors which need to be considered in determining a 'safe yield' of the aquifer under Perth. Most important of these factors are the percentage of shedding area, whether runoff from these areas is added to the groundwater or lost to drainage, and also whether an area is sewered. A worst possible case for an urban area would be a high percentage of bore ownership, a low percentage of shedding area, sewerage, and road runoff being added to an ocean/estuarine outfall rather than to an absorption basin. Such conditions exist in estuarine and oceanic suburbs around Perth and it is perhaps surprising that saline intrusion is not a more common occurrence. The existence of all these factors, except loss of road runoff waters, in the N-D catchment shows that quite high percentages (e.g. 45 per cent) of bore usage are possible without overdraw of groundwater resource, at least during years of average rainfall.
CHAPTER 7

URBAN EFFECTS ON GROUNDWATER QUALITY

7.1 BACKGROUND

Urbanization may affect groundwater quality through the alteration of the quantities and distribution of recharge- (and discharge-) waters, as well as through the addition of substances which are not found in undeveloped areas. This chapter examines the effect of urbanization on groundwater chemistry in the two urban areas as an aid to understanding the physical changes that were identified in the last two chapters. The chapter also examines the likelihood that any changes in groundwater quality are serious enough to diminish the usefulness of the groundwater.

Chapter 2 identified salinity and iron content as the quality parameters most likely to affect groundwater use in the study areas. There is also the possibility that both groundwater and surface water qualities in the urban catchments will be affected by pollutants in the storm waters that are added to absorption basins and by leachates from the sanitary landfill materials that are buried near each basin. The main effects that these two activities may have on water quality are due to nutrients, heavy metals, BOD and oils (Chapter 2).

Section 7.2 examines the source of ions in groundwaters in the Perth area, as indicated by other workers. The methods used to collect and analyse the water samples are then outlined in Section 7.3. Section 7.4 examines inputs from stormwater additions, as determined by analysing stormwaters from two dissimilar storms in the N-D catchment. Section 7.5 outlines spatial, depth and temporal variations in groundwater chemistry in the two urban areas. Efforts are made to relate these variations to the recharge-discharge
mechanisms known to exist in the catchments. A regional survey of groundwater quality in the top of the aquifer, aimed at determining whether any regional variations in groundwater quality can be attributed to urbanization, is detailed in Section 7.6. Conclusions are drawn from the whole chapter in Section 7.7.

7.2 SOURCE OF IONS IN THE GROUNDWATER

7.2.1 Atmospheric and irrigation inputs

Hingston and Gailitis (1976), in an investigation of the atmospheric inputs over the whole of Western Australia, found the mean chloride concentration (i.e. mg/L) in rainwater to be less variable than chloride input (i.e. kg/ha/y) and suggested concentrations be used in hydrologic studies in preference to inputs. The Hingston and Gailitis study investigated the distribution of seven ions (Na⁺, K⁺, Ca²⁺, Mg²⁺, Cl⁻, SO₄²⁻, and HCO₃⁻) in rainfall and dryfall during 1973 and 1974. Two of their sampling sites (Floreat and Perth) straddle the S-SP urban catchment. Table 7.1 gives the volume-weighted concentrations of the seven ions and the pH of rainwater from these two sites.

Table 7.1
Concentration of ions in rain waters at Floreat and Perth during 1973

<table>
<thead>
<tr>
<th>Site</th>
<th>pH</th>
<th>Na⁺</th>
<th>K⁺</th>
<th>Ca²⁺</th>
<th>Mg²⁺</th>
<th>Cl⁻</th>
<th>SO₄²⁻</th>
<th>HCO₃⁻</th>
</tr>
</thead>
<tbody>
<tr>
<td>Floreat</td>
<td>5.5</td>
<td>7.2</td>
<td>0.4</td>
<td>1.1</td>
<td>0.9</td>
<td>13.4</td>
<td>3.1</td>
<td>0.8</td>
</tr>
<tr>
<td>Perth</td>
<td>5.6</td>
<td>6.7</td>
<td>0.4</td>
<td>1.7</td>
<td>0.9</td>
<td>12.3</td>
<td>4.2</td>
<td>1.4</td>
</tr>
</tbody>
</table>

(Source: Hingston and Gailitis 1976).
In comparing the composition of salts in rainwater and sea water, Hingston and Gailitis found excess quantities of Ca\(^{2+}\) and SO\(_{4}\(^{2-}\) in the rainwater. Calcium was measured as being about 11 kg/ha higher than expected and the excess was attributed to finely-dispersed dust from calcareous dunes. The excess sulphate (about 18 kg/ha) was attributed to sulphur dioxide or sulphuric acid produced by the combustion of fossil fuels.

Table 7.2 shows the monthly chloride and TDS concentration of mains water to the two urban areas. Chloride and TDS levels are higher during summer, because borewater is the major source of water. In winter, dam water is the main source. By comparing Tables 7.1 and 7.2 it can be seen that chloride concentrations in mains water are more than 10 times those in rainwater.

7.2.2 Soil inputs

McArthur and Bettenay (1974) presented a partial chemical analysis of a Karrakatta sand* profile to a depth of 2.8 m. The results of these analyses are given in Figure 7.1. The analyses showed a very low soluble salt and NaCl content in the top metre. A relatively high concentration of organic matter (as indicated by ignition loss) in the top 50 cm of the profile coincided with a zone of lower iron content. Iron contents were at a maximum at about 2 m in the Karrakatta sand profile, which corresponded with the brightest (yellow) coloration of the soil. Sharma et al. (1983) found accumulations of cyclic salts in sands under densely-planted pine plantations north of Perth as a result of negligible recharge rates and reduced flushing of the sands.

In a series of infiltration column experiments, Martin and Harris (1982) found that iron dissolution from Karrakatta sand was considerably increased if humus was present. This increase occurred under both reducing (waterlogged) and oxidising (intermittent infiltration) conditions. When the sand was sterilized to preclude microbial activity, iron dissolution was markedly reduced.

* One of three soil series within the Spearwood dunes
Table 7.2
Monthly chloride and TDS levels in mains water supplied to the urban areas for the period March 1981 to November 1982
(Source: Metropolitan Water Authority)

<table>
<thead>
<tr>
<th>Month</th>
<th>Chloride (mg/L)</th>
<th>TDS (by evaporation) (mg/L)</th>
</tr>
</thead>
<tbody>
<tr>
<td>1981</td>
<td></td>
<td></td>
</tr>
<tr>
<td>March</td>
<td>201</td>
<td>440</td>
</tr>
<tr>
<td>April</td>
<td>219</td>
<td>460</td>
</tr>
<tr>
<td>May</td>
<td>189</td>
<td>400</td>
</tr>
<tr>
<td>June</td>
<td>133</td>
<td>290</td>
</tr>
<tr>
<td>July</td>
<td>84</td>
<td>180</td>
</tr>
<tr>
<td>August</td>
<td>65</td>
<td>140</td>
</tr>
<tr>
<td>September</td>
<td>57</td>
<td>130</td>
</tr>
<tr>
<td>October</td>
<td>74</td>
<td>160</td>
</tr>
<tr>
<td>November</td>
<td>150</td>
<td>300</td>
</tr>
<tr>
<td>December</td>
<td>206</td>
<td>410</td>
</tr>
<tr>
<td>1982</td>
<td></td>
<td></td>
</tr>
<tr>
<td>January</td>
<td>193</td>
<td>380</td>
</tr>
<tr>
<td>February</td>
<td>205</td>
<td>410</td>
</tr>
<tr>
<td>March</td>
<td>165</td>
<td>330</td>
</tr>
<tr>
<td>April</td>
<td>144</td>
<td>250</td>
</tr>
<tr>
<td>May</td>
<td>182</td>
<td>360</td>
</tr>
<tr>
<td>June</td>
<td>123</td>
<td>300</td>
</tr>
<tr>
<td>July</td>
<td>80</td>
<td>200</td>
</tr>
<tr>
<td>August</td>
<td>67</td>
<td>140</td>
</tr>
<tr>
<td>September</td>
<td>126</td>
<td>230</td>
</tr>
<tr>
<td>October</td>
<td>166</td>
<td>300</td>
</tr>
<tr>
<td>November</td>
<td>219</td>
<td>430</td>
</tr>
<tr>
<td>Mean Values</td>
<td>145</td>
<td>297</td>
</tr>
<tr>
<td>DEPTH (cm)</td>
<td>COLOUR</td>
<td>pH</td>
</tr>
<tr>
<td>-----------</td>
<td>---------------------------------</td>
<td>------</td>
</tr>
<tr>
<td>0</td>
<td>DARK BROWNISH GREY</td>
<td>6.5</td>
</tr>
<tr>
<td>50</td>
<td>LIGHT BROWNISH GREY</td>
<td></td>
</tr>
<tr>
<td>100</td>
<td>PALE YELLOW</td>
<td></td>
</tr>
<tr>
<td>150</td>
<td>PALE BROWNISH YELLOW</td>
<td></td>
</tr>
<tr>
<td>200</td>
<td>BROWNISH YELLOW</td>
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<tr>
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<td>BRIGHT YELLOW</td>
<td></td>
</tr>
<tr>
<td>300</td>
<td>YELLOW</td>
<td></td>
</tr>
</tbody>
</table>

**FIGURE 7.1**: Partial chemical analysis of a Karrakatta Sand profile.
(Source: McArthur and Bettenay 1974).
Glassford and Killigrew (1976) observed that the goethite-pigmented clay coating on the grains of Karrakatta sand were removed in aqueous environments.

7.2.3 Aquifer inputs

Bestow (1970) noted that the 'superficial formations' in the Gnangara area had been continually leached throughout the Quaternary and therefore all original NaCl should have been removed. Balleau (1973) considered that chemical equilibrium with soluble salts would have been reached in aquifers in the Gnangara area due to this continual leaching.

Comparisons of ion concentrations in rainfall with those in groundwater have been made for the Safety Bay Sand aquifer by Passmore (1970) and for the Tamala Limestone aquifer by Martin and Harris (1982). In both cases, Schoeller plots showed the groundwaters were enriched in calcium and bicarbonate due to the dissolution of calcite.

7.3 METHODS OF SAMPLING AND ANALYSIS

Stormwater samples were collected during runoff events in litre PVC bottles when flow rates in the drains showed major changes. Measurements of rainfall and depth of flow in the drains were taken at regular (2 or 5 minute) intervals. Flow depths were converted to flow volumes using the Manning equation, as explained in Section 5.3. The stormwater samples were refrigerated following collection, to prevent changes in nutrient and TOC contents.

All groundwater samples were taken from pumping bores. Sites were selected so as to achieve three objectives:

(i) Uniform spatial distribution,

(ii) Paired sites were chosen where it was apparent that
neighbouring bores produced water of different quality (i.e. depth
distribution),

(iii) Bores were sampled if previous analyses were available
(i.e. temporal distribution).

Full chemical analyses were made on samples from 41 bores from
each of the two urban catchments during April–July 1981. Partial
chemical analyses were also made on samples from bores for which
partial analyses had been made by other workers in previous years.
A small follow-up survey of bores sampled in April and May 1981 was
made in April 1983.

The method of collecting groundwater samples involved the
connection of a hose to a tap or sprinkler outlet on the
reticulation system. The water sample was pumped through a perspex
box in which measurements of pH and Eh were made using an Orion 407A
specific ion meter. Once readings of pH and Eh had stabilized
(usually after 20 minutes), samples were collected for analysis in
the laboratory. Three separate samples were taken,

(i) A litre sample for cation and anion analysis;

(ii) A 50-ml sample which was complexed with 2 ml of TPTZ
(2,4,6-tripyridyl-s-triazine) solution to form a blue complex for
ferrous iron determination (Dougan and Wilson 1973).

(iii) A 50-ml sample for determination of total organic carbon
(TOC). A similar sized sample was taken when nutrients were
analysed at μg/L levels of accuracy (as opposed to mg/L levels of
accuracy).

Care had to be taken to ensure that automatic fertilizing
devices were not connected to the reticulation system and also that
rust from pipes was not present. The use of the perspex box allowed
rust and air bubbles to be detected in the sample.

Table 7.3 summarises the methods used in the chemical analysis
of the samples. Full particulars of these methods have been
detailed in Martin (1980).
Table 7.3

Methods of chemical analysis

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Method</th>
</tr>
</thead>
<tbody>
<tr>
<td>Bicarbonate</td>
<td>Titration with HCl - methyl orange indicator.</td>
</tr>
<tr>
<td>Carbonate</td>
<td>Titration with HCl - phenolphthalein indicator.</td>
</tr>
<tr>
<td>Chloride</td>
<td>Mohr method - titration with silver nitrate - potassium chromate indicator.</td>
</tr>
<tr>
<td>Cations (incl. heavy metals)</td>
<td>Atomic absorption spectroscopy - SrCl$_2$ ion-exchange agent.</td>
</tr>
<tr>
<td>Conductivity</td>
<td>Philips PR 9501 conductivity meter.</td>
</tr>
<tr>
<td>Ferrous iron</td>
<td>TPTZ complexing - read on a Pye Unicam SP6-200 spectrophotometer.</td>
</tr>
<tr>
<td>Fluoride</td>
<td>Orion specific ion analyser.</td>
</tr>
<tr>
<td>Kjeldahl nitrogen</td>
<td>Auto analyser.</td>
</tr>
<tr>
<td>Nitrate + nitrite (NO$_3^-$ + NO$_2^-$)</td>
<td>(i) Nitrite reduction (Cd column), diazolisation, complexing (mg/L level).</td>
</tr>
<tr>
<td></td>
<td>(ii) Auto analyser (µg/L level).</td>
</tr>
<tr>
<td>Phosphorus</td>
<td>Auto analyser.</td>
</tr>
<tr>
<td>Silica</td>
<td>Colorimetry - molybdate blue complex.</td>
</tr>
<tr>
<td>Sulphate</td>
<td>Nephelometry - barium sulphate suspension.</td>
</tr>
<tr>
<td>Suspended solids (SS)</td>
<td>Weight gain of millipore filter paper</td>
</tr>
<tr>
<td></td>
<td>(Stormwater samples only).</td>
</tr>
<tr>
<td>Total dissolved solids (TDS)</td>
<td>(i) Evaporation in crucibles.</td>
</tr>
<tr>
<td></td>
<td>(ii) Summation of anions and cations.</td>
</tr>
<tr>
<td>Total organic carbon (TOC)</td>
<td>TOC analyser.</td>
</tr>
</tbody>
</table>

The general procedures adopted once the samples were brought to the laboratory were:

(i) Refrigeration of TOC and nutrient samples until analysed.

(ii) Analysis of bicarbonate, TDS and ferrous iron immediately on returning from the field.

(iii) Batch analysing of samples for major cations and anions.

The sum of cation and anion equivalences was compared as a check on the analyses. In almost all cases, the balance was within 5 per cent.
7.4 STORMWATER QUALITY

It was noted in Chapter 2 that the concentration of pollutants in stormwater varied by factors of 10 or more within a single storm, from area to area and also from storm to storm. To gain an accurate estimation of stormwater quality requires an extensive and long-term study, such as was carried out by Gutteridge et al. (1981).

Despite this problem of variability of stormwater quality, it was decided to obtain some measurements of the quality of storm waters entering the N-D catchment, as the physical measurements had highlighted their importance for groundwater recharge. Accordingly, two dissimilar storms were chosen for investigation - a short, intense storm at the beginning of the winter period (May) and part of a series of low intensity storms at the end of the winter period (September). By choosing dissimilar storms, it was hoped to sample waters carrying extremes of water quality to provide some measure of the range of values in stormwaters. The May storm (including a 6-minute period at 33 mm/h) fell on a catchment that had only 30 mm rainfall during the previous seven weeks. The September storm was typical of many low-intensity storms (long periods of 2 to 3 mm/h) in which individual showers are interspersed by dry periods of 20 minutes or more. Such storms may continue for many hours during winter associated with the passage of a cold front over the Perth area. Sometimes runoff from previous rainfall continued between showers, so there was a mixture of water running off the catchment.

All stormwater samples were analysed for suspended solids (SS), TDS, Cl\(^-\), NO\(_3\)\(^-\) + NO\(_2\)\(^-\), Kjeldahl N, total P, Cu\(^{2+}\), Zn\(^{2+}\), Pb\(^{2+}\), Ni\(^{2+}\) and Cr\(^{3+}\). Some samples were also tasted and smelt for hydrocarbons but none were detected. All analyses (except SS) were made on filtered samples because:

(i) sampling the SS material accurately was considered impossible because of the floating and bed-load components,

(ii) dissolved materials are more representative of the quality of water entering the groundwater. SS materials will become
incorporated into the basin sediments and probably not released unless mineralized.

Figures 7.2 and 7.3 show the hydrographs and chemical concentrations for the two storms that were measured. The drain which was monitored conducted water from the south-eastern part of Masons Gardens catchment and the adjoining catchment (see Fig. 5.8). In the May storm, 4.3 mm of rainfall fell in a 15-minute period. There was a lag of about 10 minutes between the maximum rainfall intensity and maximum runoff from the catchment. Figure 7.2 also shows that there was a marked 'first-flush' of SS, TDS, Cl\textsuperscript{−}, Kjeldahl N and NO\textsubscript{3}\textsuperscript{−} + NO\textsubscript{2}\textsuperscript{−} in the May storm, followed by a stepped fall-off in concentration. It is probable that this stepping is due to the later addition of first-flush waters from the adjoining catchment. The hydrograph is also slightly stepped at this point.

In the September storm, the hydrograph was falling when the first sample was collected as a result of a previous shower. Despite a 20-minute break in the rainfall, runoff in the drain continued at low levels until a later, 100-minute shower took place. There was a 10-15 minute lag between the maximum intensity of rainfall and maximum runoff. This appears to be the time of concentration for the catchment. There was also a step in the decline of this hydrograph which may correspond to waters arriving from the adjoining catchment. The concentration of SS in the September storm was only about 5 per cent that of the May storm. SS values show a close correspondence with flow while most other parameters show a decline with time.

Table 7.4 shows the flow-weighted mean concentrations for the two storms and the significance of any differences (2 tailed t-test). The May storm was significantly higher in SS, NO\textsubscript{3}\textsuperscript{−} + NO\textsubscript{2}\textsuperscript{−} and Zn\textsuperscript{2+} but significantly lower in TDS, Cl\textsuperscript{−}, Kjeldahl N and total P. The higher SS would be expected after a dry period. Bishaw (1980) noted a decrease in NO\textsubscript{3}\textsuperscript{−} + NO\textsubscript{2}\textsuperscript{−} in rainwaters and stormwaters in the Ballajura area as winter progressed. TDS and Cl\textsuperscript{−} were only
FIGURE 7.2: Hydrograph and chemical concentrations of stormwater from a high-intensity storm in May.
FIGURE 7.3: Hydrograph and chemical concentrations of stormwater from a low-intensity storm in September.
Table 7.4

Flow-weighted mean concentrations of storm waters (mg/L)

<table>
<thead>
<tr>
<th></th>
<th>SS</th>
<th>TDS</th>
<th>Cl⁻</th>
<th>NO₃⁻ + NO₂⁻</th>
<th>Kjeldahl N</th>
<th>Total P</th>
<th>TOC</th>
<th>Cu²⁺</th>
<th>Zn²⁺</th>
<th>Pb²⁺</th>
<th>Ni²⁺</th>
<th>Cr³⁺</th>
</tr>
</thead>
<tbody>
<tr>
<td>May</td>
<td>227</td>
<td>39</td>
<td>15</td>
<td>0.211</td>
<td>0.416</td>
<td>0.102</td>
<td>5.7</td>
<td>0.01</td>
<td>0.03</td>
<td>0.01</td>
<td>&lt;0.01</td>
<td>&lt;0.01</td>
</tr>
<tr>
<td>September</td>
<td>16</td>
<td>49</td>
<td>20</td>
<td>0.100</td>
<td>1.352</td>
<td>0.257</td>
<td>5.8</td>
<td>&lt;0.01</td>
<td>&lt;0.01</td>
<td>&lt;0.01</td>
<td>&lt;0.01</td>
<td>&lt;0.01</td>
</tr>
</tbody>
</table>

Significance of difference

|       | *** | *   | *   | *** | *** | *** | n.s. | n.s. | *   | n.s. | n.s. | n.s. |

*** = P < 0.001  ** = P < 0.01  * = P < 0.05  n.s. = non-significant
slightly higher in the September storm but Kjeldahl N and total P were substantially higher. The reasons for the higher values during the lower intensity and late winter storm are unknown.

With the exception of the SS values in the May storm, all values are markedly lower than those reported for the Sydney and Melbourne areas (Table 2.4). Possible reasons for the lower values include the small amounts of soil runoff in the N-D catchment and the lower fallout of atmospheric pollution in the less-industrialised Perth area. Roads in the N-D catchment are also swept every three months. In the S-SP catchment, road sweeping is carried out much more frequently (weekly during leaf fall). As no samples of stormwaters were taken from this catchment, the effect of sweeping on water quality cannot be assessed.

The values shown in Tables 2.4 and 7.4 indicate that stormwaters in the Perth area had an SS value similar to that of raw sewage for high intensity storms. A large part of the SS fraction consisted of dead leaves and lawn clippings. Nutrient concentrations in the stormwaters were only about 5 per cent those of secondary treated sewage while TOC contents were about 20 per cent.

The mean phosphate levels in the stormwaters were about 80 per cent of the mean values noted by Congdon (1979) for Lake Joondalup (20 km north of the study area), while total nitrogen levels are about half those of the lake. This lake has been classified as mildly eutrophic. There was some evidence of excessive algal growth in Shenton Park Lake during the summer of 1981/82. At this time there was also excessive growth of the benthotic plant, Hydrilla verticillata. Jack (1977a) has noted that stormwaters are generally an insignificant source of phosphate in the Swan River.

While BOD was not measured in the stormwaters, TOC values of filtered samples were very low. However, the SS fraction appeared to be composed of significant amounts of large organic detritus as well as inorganic clastics (sand, silt and clay). Heavy metal
values are similar to those noted by Taylor (1980) for drains entering the Swan River (i.e. highest for Cu$^{2+}$ and Zn$^{2+}$ with occasional samples above the toxic threshold).

7.5 GROUNDWATER QUALITY

7.5.1 Spatial distribution

The spatial distribution of chemical parameters in the groundwaters was determined from carrying out full analyses of 82 bores during mid-1981. Each parameter was contoured and overlain with groundwater flow maps. The contours for pH, Eh, TOC and total Fe are shown for both catchments as Figures 7.4 to 7.7 respectively. The contours for the other parameters are given in Appendix 8.

The absorption basins in both catchments are associated with low pH-Eh conditions and high TOC-Fe. The high TOC-Fe values in the S-SP catchment are displaced down-gradient because samples were collected after recharge in the basin had commenced (see later). The TOC in the basins probably arises from biological growth in the surface waters (and also, to a lesser extent, from road runoff). Low pH and Eh conditions would be expected under such conditions, due to the presence of organic acids and the microbial breakdown of organic matter respectively. Stability diagrams for iron (e.g. Hem 1963) show that under acid-reducing conditions, iron is in the soluble, ferrous form. The dissolution of iron from the goethite-bearing yellow sands around the basins would therefore be expected, as was reported by McArthur and Bettenay (1974) for waterlogged areas in the Bassendean Sands, and was shown by Martin and Harris (1982) for Spearwood Sands. The extra water that is directed into these basins following urbanization would be expected to increase the dissolution of iron, through increasing the waterlogging of the yellow sands which usually occur about a metre above the normal water table (Section 5.7.3.1). Absorption basins also exist up-gradient of both the N-D and S-SP catchments (see Fig. 4.1). It is likely that these basins are responsible for some of the other low
FIGURE 7.4 (a): Groundwater pH in the N-D catchment

FIGURE 7.4 (b): Groundwater pH in the S-SP catchment
FIGURE 7.5 (a): Groundwater Eh (mV) in the N-D catchment

FIGURE 7.5 (b): Groundwater Eh (mV) in the S-SP catchment.
FIGURE 7.6 (a): TOC concentrations (mg/L) in the groundwater in the N-D catchment

FIGURE 7.6 (b): TOC concentrations (mg/L) in the groundwater in the S-SP catchment
FIGURE 7.7 (a): Iron concentrations (mg/L) in the groundwater in the N-D catchment

FIGURE 7.7 (b): Iron concentrations (mg/L) in the groundwater in the S-SP catchment
pH–Eh and high TOC-Fe values seen in (or just outside) the eastern and western boundaries of the catchments.

Ninety per cent of the samples collected in the 1981 survey were analysed for NO$_3^-$ (mg/L level of sensitivity) to establish background levels for this parameter and also to see if road runoff or the sanitary landfill materials in the absorption basins could be producing high levels of this nutrient as outlined in Chapter 2. The contours for NO$_3^-$ are shown in Figure 7.8.

In the N–D catchment, the highest values were obtained in the NE of the catchment and there was no correlation with the location of absorption basins or known sanitary landfill sites. The levels of NO$_3^-$ in the groundwater were generally low in this catchment (mean value = 2.0 mg/L, maximum value = 7.8 mg/L).

In the S–SP catchment, spot highs of NO$_3^-$ occurred both above and below the absorption basin. However, levels in the groundwater were generally low (mean value = 1.8 mg/L, maximum value = 10.0 mg/L). Drinking water limits for nitrate-nitrogen are 45 mg/L (World Health Organisation standards).

Ten further samples were collected for analysis of NO$_3^-$ + NO$_2^-$, Kjeldahl N and total P at μg/L levels of sensitivity in April 1983. Sample sites in both catchments included bores up- and down-gradient of the absorption basins.

The mean and range of values for these samples were: NO$_3^-$ + NO$_2^-$ = 2.5 mg/L (0.003 to 14.7), Kjeldahl N = 0.4 mg/L (0.116 to 1.17) and total P = 0.035 mg/L (0.016 to 0.059). There was no obvious relationship between nutrient levels and proximity to the absorption basins. The analyses show that phosphorus is generally low in the groundwater. This would be expected in limestone areas due to precipitation of apatite. Fertilizer phosphate would be expected to be adsorbed onto goethite in the vadose zone sands.
FIGURE 7.8 (a) : Nitrate concentration (mg/L) in the groundwater in the N-D catchment

FIGURE 7.8 (b) : Nitrate concentration (mg/L) in the groundwater in the S-SP catchment
To test the associations between chemical parameters in the groundwaters, correlation coefficients were calculated. These coefficients are shown in Table 7.5.

There was a very high degree of correlation between parameters in both catchments (46% of all possible correlations were significantly correlated at $P < 0.05$ in the N-D catchment and 40% were correlated in the S-SP catchment). Examination of Table 7.5 shows three ion associations in the N-D catchment, i.e.

(i) Low lying area (shallow water table or low DWT) — low pH and Eh, high TOC, Fe$^{2+}$, K$^+$, Mg$^{2+}$ and SO$_4^{2-}$. This is probably the association already shown in Figures 7.4 to 7.7. The K$^+$ may be produced by kaolinisation of feldspars in the sands around the basin.

(ii) Elevated areas — high pH, Ca$^{2+}$, HCO$_3^-$ and Si. These waters are probably older and there has been time for dissolution of limestone to take place.

(iii) High TDS, Na$^+$, Cl$^-$, Mg$^{2+}$, SO$_4^{2-}$ waters. These waters probably arise from concentration-by-evaporation of waters which still are dominated by ions of atmospheric origin, rather than those of aquifer dissolution.

There are also two associations which can be seen in the S-SP catchment waters, i.e.

(i) Low Eh, high Fe, deep below the water table and at high elevations. This association appears similar to (i) in the N-D catchment, except for its location deep within the aquifer and its association with high Ca$^{2+}$ and HCO$_3^-$.

(ii) Low elevation waters lower in TDS, Ca$^{2+}$, Fe$^{2+}$, Si, HCO$_3^-$, and SO$_4^{2-}$. This association probably reflects recently-recharged waters which have been added to the absorption basin from road runoff.

The absorption basin in the N-D catchment had high TDS waters associated with it, as most sampling in this catchment took place in April and May 1981 when there had been only slight or moderate re-
Table 7.5

Correlation coefficients of chemical and physical parameters of groundwaters from the N-D and S-SP catchments

(upper right, N-D catchment; lower left, S-SP catchment)

<table>
<thead>
<tr>
<th></th>
<th>pH</th>
<th>Eh</th>
<th>COND</th>
<th>TDS</th>
<th>Ca^{2+}</th>
<th>Mg^{2+}</th>
<th>Na^{+}</th>
<th>K^{+}</th>
<th>Fe(total)</th>
<th>Fe^{2+}</th>
<th>Si</th>
<th>HCO_{3}^{-}</th>
<th>Cl^{-}</th>
<th>SO_{4}^{2-}</th>
<th>NO_{3}^{-}</th>
<th>F^{-}</th>
<th>TOC</th>
<th>DWT</th>
<th>DBWT</th>
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<tbody>
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<td>pH</td>
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</tr>
<tr>
<td>COND</td>
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<td>-30</td>
<td>-43</td>
<td>-45</td>
<td>-79</td>
<td>-84</td>
<td>0.64</td>
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<td>TDS</td>
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<td>0.63</td>
<td>-0.27</td>
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<td>0.38</td>
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<td>-0.59</td>
<td>0.55</td>
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<td>K^{+}</td>
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<td>-0.59</td>
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<td>Fe(total)</td>
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<td>0.96</td>
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<tr>
<td>Fe^{2+}</td>
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<td>-36</td>
<td>-27</td>
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<td>0.96</td>
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<td>0.81</td>
<td>0.31</td>
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<td>HCO_{3}^{-}</td>
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<td>-38</td>
<td>0.44</td>
<td>0.44</td>
<td>0.62</td>
<td>0.45</td>
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<tr>
<td>Cl^{-}</td>
<td>0.44</td>
<td>-44</td>
<td>0.68</td>
<td>0.62</td>
<td>0.45</td>
<td>0.34</td>
<td>0.59</td>
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<tr>
<td>SO_{4}^{2-}</td>
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<td>0.74</td>
<td>0.55</td>
<td>0.36</td>
<td>-0.36</td>
<td>0.34</td>
<td>-0.29</td>
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<tr>
<td>NO_{3}^{-}</td>
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<td>-34</td>
<td>-0.34</td>
<td>-0.36</td>
<td>-0.34</td>
<td>-0.29</td>
<td>-0.30</td>
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<tr>
<td>F^{-}</td>
<td>0.80</td>
<td>-38</td>
<td>-26</td>
<td>-35</td>
<td>-0.34</td>
<td>-0.34</td>
<td>-0.29</td>
<td>-0.30</td>
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<tr>
<td>TOC</td>
<td></td>
<td>-37</td>
<td>-37</td>
<td>-37</td>
<td>0.31</td>
<td>0.35</td>
<td>0.30</td>
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<td>0.47</td>
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<td>0.31</td>
<td>0.35</td>
<td>0.58</td>
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</tbody>
</table>

Note: DWT = depth of water table below ground surface (m)
DBWT = depth of sample below water table (m)
Only significant (P < 0.05) correlations are shown.
charge. However, the absorption basin in the S-SP catchment had low TDS waters associated with it as sampling in this catchment extended into July when recharge through the basin was extensive. Before the survey was carried out it was thought that the time of sampling would not be very critical, as the amount of recharge was considered to be small in comparison to the volume of groundwater in storage. However, most private bores are screened in the top third of the aquifer and there was a considerable seasonal variation in groundwater quality in bores located near the absorption basins.

The detection of ion associations from Table 7.5 indicated that it may be possible to identify different hydrochemical facies in the groundwaters. Chebotarev (1955) proposed categorizing waters into bicarbonate, sulphate and chloride groups with various genetic types within these groups, as he considered anions to be independent ingredients of groundwaters. Back (1961) and Morgan and Winner (1962, cited in Freeze and Cherry 1979) subdivided the commonly-used trilinear diagram of Piper (1944) for the purposes of defining hydrochemical facies. Piper's diagrams are restricted to major ions, however, and are restricted to showing proportions rather than actual concentrations. The use of trilinear plots also reduced the number of independent variables from six to four (Dalton and Upchurch 1978). Factor analysis has been preferred by Dalton and Upchurch and also by Ashley and Lloyd (1978) on the basis that actual concentrations are used, and because minor ions, neutral chemical species and non-chemical data can be included.

Factor analysis was carried out independently on the N-D and S-SP data sets. Table 7.6 shows the factors that were obtained from each catchment while Figures 7.9 and 7.10 show the spatial distribution of each factor. The type of factor analysis carried out involved Varimax rotation of axes (method PA2 in 'Statistical Package for the Social Sciences', Nie et al. 1970). Depth of the water table, depth of sampling, NO$_3^-$ and F$^-$ were left out of the factor analyses as complete data sets were not available for these parameters.
Table 7.6

Factor loadings for chemical analyses of groundwater from the two urban catchments

<table>
<thead>
<tr>
<th>Variable</th>
<th>N-D catchment</th>
<th>S-SP catchment</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Factor 1</td>
<td>Factor 2</td>
</tr>
<tr>
<td>pH</td>
<td>-.31</td>
<td>.40</td>
</tr>
<tr>
<td>Eh</td>
<td>-.80</td>
<td>.45</td>
</tr>
<tr>
<td>COND</td>
<td>.91</td>
<td>.65</td>
</tr>
<tr>
<td>TDS</td>
<td>.95</td>
<td>.57</td>
</tr>
<tr>
<td>Ca(^{2+})</td>
<td></td>
<td>.74</td>
</tr>
<tr>
<td>Mg(^{2+})</td>
<td>.83</td>
<td>.53</td>
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<tr>
<td>Na(^+)</td>
<td>.94</td>
<td>.39</td>
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<td>K(^+)</td>
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<td>.67</td>
</tr>
<tr>
<td>Fe (total)</td>
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<td>.94</td>
</tr>
<tr>
<td>Fe(^{2+})</td>
<td>.95</td>
<td>.97</td>
</tr>
<tr>
<td>Si</td>
<td>-.61</td>
<td></td>
</tr>
<tr>
<td>HCO(_3^-)</td>
<td></td>
<td>1.00</td>
</tr>
<tr>
<td>Cl(^-)</td>
<td>.92</td>
<td></td>
</tr>
<tr>
<td>SO(_4^{2-})</td>
<td>.69</td>
<td>.31</td>
</tr>
<tr>
<td>TOC</td>
<td>.89</td>
<td></td>
</tr>
<tr>
<td>Eigen value</td>
<td>5.23</td>
<td>3.95</td>
</tr>
<tr>
<td>% of explained variation</td>
<td>46.8</td>
<td>35.3</td>
</tr>
</tbody>
</table>

NB: Only loadings greater than 0.3 are shown.
FIGURE 7.9: Contours of factor scores in the N-D catchment
FIGURE 7.10: Contours of factor scores in the S-SP catchment
Factor 1 in both catchments is a salinity factor with high, positive loadings for TDS, conductivity, Na\(^+\), Mg\(^{2+}\) and Cl\(^-\). In the N-D catchment this factor is highest around the absorption basin area and there are long extensions to the SW and SE of the basin. The SW extension of this factor coincides with a thick sequence of sand in the aquifer (see Section CC in Fig. 5.43). The SE extension of this factor does not, however, appear to correspond to any aquifer feature. The high loading in the east of the catchment is directly down-gradient of another absorption basin (see Fig. 4.1).

Factor 1 in the S-SP catchment is lowest around the basin, probably due to the recently recharged waters at the time of sampling. The previous section showed that the salinity of road runoff waters is only about 35 per cent higher than the salinity of rainfall. The high area immediately west of the basin does not correspond to any up-gradient absorption basin or known aquifer feature. It is possible that higher salinity waters from the basin have been displaced 100 m to the west during the rapid recharging of the basin which took place in 1981. Figure 5.48 indicated that groundwater flow is to the NW at times of high recharge. These waters may have remained behind while those down-gradient were transported away more rapidly. Such large rates of movement are unusual in natural groundwater systems. However in Section 7.7 a plume of higher salinity water is reported being transported 100 to 150 m downstream in about eight weeks from a similar absorption basin.

Factor 2 in the N-D catchment and factor 3 in the S-SP catchment are high iron factors, associated with reduced waters and higher TOC contents. In the N-D catchment the close correspondence of this factor with the absorption basin is very evident. The high iron area to the west of the catchment is also down-gradient from an absorption basin. In the S-SP catchment, the iron factor is highest down-gradient from the basin and in the west, north and east of the catchment. Only the first occurrence can be clearly identified as originating from an absorption basin.
In the N-D catchment it is clear that the salinity and iron factors have a common point of origin; the absorption basin. However, these associations are separate enough for two factors to be identified. In the N-D catchment (and to a lesser extent in the S-SP catchment), the iron factor is closest to the basin while the salinity factor is displaced. It is possible that the initial recharging of the basin, at the commencement of winter, rapidly removes the higher salinity waters but it is only when the water table rises around the basin that iron is released into the groundwater. The January 1982 rainfall (100 mm in two days) resulted in a rapid rise in the water table around Shenton Park Lake, as the rainfall was heavy and the ocean outfall was blocked. It was very noticeable that soon after this event the bore water used around the basin left a marked stain. Iron dissolution requires both a high water level and reducing waters. It is probable that the waters in the absorption basin are reducing at the beginning of winter but become more oxidising as more road runoff takes place.

Factor 3 in the N-D catchment and factor 2 in the S-SP catchment are limestone factors, with high Ca\(^{2+}\) and HCO\(_3^-\). In the N-D catchment it is highest in areas where factor 1 was lowest. The fact that high factor scores exist for waters from a limestone area immediately down-gradient from the basin indicates that these waters need not be very old to have been influenced by the limestone. In the questionnaire that was conducted in the N-D catchment, 44 per cent of bore owners acknowledged a white lime deposit was left behind when their bore water evaporated.

In the S-SP catchment, the limestone factor is highest in the central-east of the catchment. Figure 5.45 indicated that there are discontinuous areas of limestone throughout the S-SP catchment, including the central-east area. The high factor score in the central-east area may indicate only that the limestone is present at the depth at which the samples were taken, rather than a greater
proportion of limestone.

Factor 4 in the S-SP catchment is a high Si, SO\textsubscript{4}^{2-}, Eh (low TOC) association which is located up-gradient from the absorption basin. These waters may be older waters which have not experienced a concentration-by-evaporation or reduced episode in an absorption basin, or encountered limestone in the aquifer. The silica probably results from kaolinization of microcline which is very prevalent in the Spearwood Sands (see Appendix 4). The origin of the sulphate is unknown. Possible sources are oxidation of pyrite by the high Eh waters, fertilizer additions, decomposition of organic matter, and atmospheric fallout.

The difference in groundwater chemistry between the two urban catchments was tested for significance by carrying out two-tailed t-tests on the data obtained from the 82 samples. Table 7.7 shows the mean values for each catchment and the significance of any differences.

Table 7.7 shows that the N-D catchment waters are significantly higher in pH, TDS, Ca\textsuperscript{2+}, Na\textsuperscript{+}, Si, HCO\textsubscript{3}\textsuperscript{-} and Cl\textsuperscript{-}, and significantly lower in K\textsuperscript{+}, Fe (total), Fe\textsuperscript{2+} and TOC. In general terms, then, the S-SP catchment waters have a lower salinity but contain greater iron contents. Part of the difference in salinities may be due to the different times of sampling in the catchments. However the fact that the S-SP catchment is both a large net recharge area and also contains proportionally less limestone, probably explains most of the difference. The lack of limestone in the S-SP catchment may also partially explain the higher iron contents in this catchment, as iron will remain soluble under low Eh-pH conditions. The higher K\textsuperscript{+} values in the S-SP catchment may be due to the greater prevalence of microcline in aquifer materials and/or increased rate of kaolinization due to the lower pH of the waters.
### Table 7.7

**Difference in mean values of chemical parameters for groundwater from the two urban catchments**

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Mean value</th>
<th>Significance of difference</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>N-D catchment</td>
<td>S-SP catchment</td>
</tr>
<tr>
<td>pH</td>
<td>7.07</td>
<td>6.82</td>
</tr>
<tr>
<td>Eh (mV)</td>
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<td>188</td>
</tr>
<tr>
<td>Conductivity (mS/m)</td>
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<tr>
<td>TDS mg/L</td>
<td>667</td>
<td>482</td>
</tr>
<tr>
<td>Ca$^{2+}$ mg/L</td>
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<td>Mg$^{2+}$ mg/L</td>
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<td>Na$^+$ mg/L</td>
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<td>K$^+$ mg/L</td>
<td>6.5</td>
<td>7.7</td>
</tr>
<tr>
<td>Fe (total) mg/L</td>
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<td>1.6</td>
</tr>
<tr>
<td>Fe$^{2+}$ mg/L</td>
<td>0.5</td>
<td>1.6</td>
</tr>
<tr>
<td>Si mg/L</td>
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<td>11.7</td>
</tr>
<tr>
<td>HCO$_3^-$ mg/L</td>
<td>267</td>
<td>140</td>
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<tr>
<td>Cl$^-$ mg/L</td>
<td>191</td>
<td>149</td>
</tr>
<tr>
<td>SO$_4^{2-}$ mg/L</td>
<td>55</td>
<td>59</td>
</tr>
<tr>
<td>NO$_3^-$ mg/L</td>
<td>2.8</td>
<td>1.8</td>
</tr>
<tr>
<td>F$^-$ mg/L</td>
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<td>0.02</td>
</tr>
<tr>
<td>TOC mg/L</td>
<td>0.58</td>
<td>0.90</td>
</tr>
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</table>

| Depth of water table (m) | 15.6 | 12.9 | n.s. |
| Depth of sampling (m) below the water table | 9.9 | 8.4 | n.s. |

*** P < 0.001  
** P < 0.01  
* P < 0.05  
n.s. non significant
7.5.2 Depth distribution

In the N-D catchment, there were two bore owners in the vicinity of the basin who substantially decreased the amount of iron in their bore water by deepening their bores. A similar case was reported for an absorption basin to the south of the N-D catchment. This evidence would appear to confirm that the absorption basin in the N-D catchment is a source of some of the iron in the groundwater. Iron content was highly correlated ($r = 0.53$, $P < 0.001$) with shallow depths of the water table in this catchment (see Table 7.5). An examination of two neighbouring bores located 250 m up-gradient of the basin in the N-D catchment indicated that the shallow bore had the highest iron content (1.3 vs <0.01 mg/L). The location of this iron up-gradient of the basin may be the result of the gradient reversals that take place following the addition of road runoff waters to the N-D basin, although some iron may be leached directly from the soil if the recharging waters are sufficiently reducing.

In the S-SP catchment, iron was positively correlated with both depth of the water table and depth of sampling. The largest areas of high iron contents in this catchment are in the north and east of the catchment. There are no absorption basins immediately up-gradient from these areas, but the presence of the iron at depth (e.g. 30 m) in the aquifer would indicate that these waters are old and may have travelled some distance. There is a large wetland located four kilometres up-gradient from this area. One bore owner in the S-SP catchment reported an increase in iron content following a deepening of his bore. In this catchment it is more difficult to predict the iron content of groundwater.

Table 7.5 indicated that pH, Si, $\text{HCO}_3^-$ and $\text{NO}_3^-$ were positively correlated with depth of the water table in the N-D catchment but only pH and Si were positively correlated with depth below the water
table. In the S–SP catchment, TDS, Ca$^{2+}$, Fe$^{2+}$, Si, HCO$_3^-$ and SO$_4^{2-}$ were positively correlated with depth of the water table but only Fe$^{2+}$ and SO$_4^{2-}$ were positively correlated with depth below the water table. The correlation of Ca$^{2+}$ and HCO$_3^-$ with depth of the water table is probably due to limestone in high elevation areas, as was indicated in Figures 5.43 and 5.45. The fact that these ions are not correlated with depth below the water table probably indicated that all the CO$_2$ (aq) and H$_2$CO$_3$ in infiltrating soil waters have been depleted by encountering limestone in the vadose zone and upper aquifer. Drillers have reported cavernous zones above the water table in limestone areas, indicative of limestone dissolution by CO$_2$-bearing waters and removal of Ca$^{2+}$ and HCO$_3^-$ in solution. Below the water table, closed-system conditions operate with no replenishment of CO$_2$.

7.5.3 Temporal distribution

An estimation of the variations in chemical parameters with time was obtained from the questionnaire, completed by about 150 bore owners. Very few bore owners had noticed any change in water quality. However, three had noted an increase in iron content. Only one of these bore owners had noted the change after a long period of no staining. In the other two cases it is possible the staining was minor, but cumulative. The one owner who noted an increase after a 10 to 15 year period of no staining was located immediately down-gradient from the N–D catchment. One other bore owner reported an increase in the lime deposit that was left behind after the bore water evaporated. No quantified increase in iron or lime was reported in the questionnaire.

To test changes in water quality with time, bores for which water analyses had been carried out by the Government Chemical Laboratories (GCL) were resampled. Very few such bores were available from within the two study areas, so bores from outside the catchments and also some bores which had been sampled in the 1981 survey were analysed. The GCL usually analysed only for Cl$^-$ and
pH, so these parameters were the ones which were analysed for in the resampled bores. The GCL analyses had been made in the laboratory, some time after sampling. The pH of groundwater samples can alter with time if CO₂ degasses, or if iron precipitates ('ferrolysis'). Thus the pH values obtained in the field may not be directly comparable with the laboratory analyses. Therefore only Cl⁻ remains as an accurate parameter for comparison.

Figure 7.11 shows the comparison of Cl⁻ values obtained at different times from bores in the vicinity of the N-D and S-SP catchments. Generally the Cl⁻ values are very similar, although more have risen than have fallen. All that can be said from the data is that no significant (P < 0.05) change in Cl⁻ concentration has taken place.

Figure 7.11 also shows the change in pH values that has taken place in the groundwaters. The scatter in these values is much greater than occurred for Cl⁻, probably as a result of analytical errors. Using a paired t-test analysis as for Cl⁻, there was no significant change in pH detected.

To determine the effect on the salinity of down-gradient groundwater of adding stormwaters to an urban wetland, the conductivity of the groundwater in a bore located 100 to 150 metres down-gradient of a large urban wetland was monitored over a 10-month period. The wetland received significant amounts of road runoff waters after April and May 1980 and the salinity of the water increased in the down-gradient bore at the end of June (Fig. 7.12). The higher salinity water probably resulted from concentration-by-evaporation in the wetland during the summer months. The salinity of the water decreased in October-November as the fresher, road-runoff waters reached the down-gradient bore. This wetland was considerably larger than Shenton Park Lake (see Fig. 7.14(b)), but the processes are likely to be similar.
FIGURE 7.11: Comparison of groundwater - chloride (above) and pH (below) values over time.
FIGURE 7.12 : Average salinity in the top 12 metres of the groundwater in a bore located immediately down-gradient of an urban wetland/absorption basin.
7.6 REGIONAL SURVEY OF GROUNDWATER IN THE UPPER AQUIFER

7.6.1 Methods

A survey of selected physical and chemical parameters was carried out in the northern Perth area to determine if urbanization had a regional effect on the overall quality of the groundwater in the Perth area. Four parameters were measured in situ - pH, Eh, conductivity and temperature using a Martek Mark VII water quality analyser. Eh and pH readings were made separately and one week apart as there was provision for only one of these probes at a time. The need to log holes twice allowed checks to be made on the temperature and conductivity profiles. Forty-three observation bores, slotted over the top 12 metres of the aquifer and a few deeper bores were available for use in the survey. Readings of the four parameters were made at one-metre intervals over the slotted interval.

It was considered that areas of high recharge would have lower conductivity readings and be oxidizing, unless the waters had passed through a reducing zone in the soil. The pH of rainfall (5.5 in Perth, Hingston and Gailitis 1976) is also likely to be altered in the soil and aquifer by organic matter and limestone and is unlikely to be a good indicator of high or low recharge areas. Wood (1976) considered that temperature measurements were critical in identifying recharge from nearby surface water bodies.

The profiles obtained from the logging were visually examined, and the average values over the 12 metre interval were statistically correlated with controlling (independent) variables. For statistical purposes, the three independent variables (formation, flow stage, and urban stage) were assigned ordinal values for each hole by ranking them according to the following criteria. Formation was ranked from young to old, flow stage was ranked from early to late, and urbanization stage was ranked from woodland to old urban, as shown in Table 7.8.
<table>
<thead>
<tr>
<th>Ordinal value</th>
<th>Variable</th>
<th>Formation</th>
<th>Flow stage*</th>
<th>Urbanization stage</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>2</td>
<td>Tamala Limestone (Cottesloe)</td>
<td>0-5 km</td>
<td>Woodland</td>
</tr>
<tr>
<td>2</td>
<td>3</td>
<td>Tamala Limestone (Karrakatta)</td>
<td>5-10 km</td>
<td>Rural</td>
</tr>
<tr>
<td>3</td>
<td>4</td>
<td>Bassendean Sand</td>
<td>10-15 km</td>
<td>New urban (post 1942)</td>
</tr>
<tr>
<td>4</td>
<td>5</td>
<td>Guildford</td>
<td>&gt; 20 km</td>
<td>Old urban (pre 1942)</td>
</tr>
</tbody>
</table>

* distance measured from the axis of the Gnangara Mound.
7.6.2 Results

7.6.2.1 Profiles and contour plots:

Most (90%) profiles showed a slight decrease (ca. 1°C) in temperature with depth, the decrease being smaller in areas where the water table was deepest. The average decrease over the 12 metres was 0.9°C. Most (80%) profiles also showed an increase in conductivity with increasing depth. In half of these cases, the increase took the form of a definite, stepped increase over a few metres. In two cases, the increase was about 100 mS/m. The average increase in conductivity over the 12 metres was 20 mS/m. A slight decrease in pH with increasing depth was noted in 76 per cent of cases (average over 12 metres of 0.2 pH units). Three of the 43 bores showed a stepped increase of 0.5 pH units or more, two of these increases being correlated with increases in conductivity. However most pH profiles changed little with depth. Most profiles showed a decrease in Eh with increasing depth, the average decrease being 60 mV. About a quarter of the profiles decreased by more than 150 mV, the most common decrease being a stepped drop over the top few metres.

Figure 7.13 shows examples of stepped changes in parameters within profiles. Stepped increases in conductivity about two metres below the water table were encountered in about 35 per cent of bores. The zones of lower conductivity were taken to represent recently recharged waters. The fact that many of the zones were oxidizing tends to confirm this assessment. The survey was carried out during April-May 1980 and it was apparent in some multi-logged holes that the recharge-affected zone increased with time and rainfall.

Figure 7.13 also shows examples of stepped increases in conductivity at depth (e.g. GD3, GE4, M30). These increases were sometimes accompanied by increases in pH and temperature and
FIGURE 7.13: Chemical and physical properties of the groundwater in five bores
decreases in Eh. While appearing similar in nature to those attributed to recharge zones, the depth of the changes and the presence of recharge zones in the same profile suggests some form of aquifer stratification may be occurring in these holes. Martin (1980) reported stratification in a survey of nine holes in the same area. In some of these cases the stratification could be correlated with clay bands within the aquifer. Martin reported an increase in temperature with increasing depth in seven of the nine profiles, which was different from the trend shown in the present survey. As the surveys were six months apart, it is possible that seasonal temperatures are being transmitted to the groundwater, perhaps via the bore casing.

Figure 7.14 shows the contours of temperature, conductivity, pH and Eh for the top metre of the water table in the surveyed area. High recharge areas would be expected to be low conductivity-high Eh areas.

Figure 7.14(a) shows that waters with the highest temperatures occur along a central corridor, peaking in the south. A bias in the location of sampling sites may mean that the contours shown do not truly reflect the regional variation in groundwater temperature. Depth of the water table was found to influence temperatures and several of the bores in this corridor are situated in areas where the water table was close to the surface.

Figure 7.14(b) shows that low conductivity areas occur in the north-east and south, with a small area in the central west. The north-eastern area corresponds to an area of known recharge on the west of the Gaangara Mound. The southern area corresponds to the older urban area but, as the area is defined by only five bores, it is not certain that it represents a high recharge area. Significant spatial variability in salinity has already been reported from the urban catchments that were studied. The effect of evaporation from a large wetland area is evidenced by the high conductivity in the central-west area. This is the wetland for which salinity cycles
FIGURE 7.14 (a): Temperature of the top metre of groundwater.

FIGURE 7.14 (b): Conductivity of the top metre of groundwater.
FIGURE 7.14 (c): pH of the top metre of groundwater

FIGURE 7.14 (d): Eh of the top metre of groundwater
were reported in the previous section (Fig. 7.12).

Figure 5.14(c) shows a general increase in pH westwards. The contact of the Bassendean Sand and Tamala Limestone corresponds approximately to the pH 6 contour in the west.

Figure 5.14(d) shows a general increase in Eh westwards, with an isolated high in the east. The recharge situation with regard to Eh is complicated due to the presence of more organic matter in the older, Bassendean Sands (McArthur and Bettenay 1974) which tends to lower the Eh of recharging waters in this formation. Mean Eh values (not shown) indicate a much clearer increase in values to the west than do surface Eh values.

The value of conductivity and Eh values in identifying recharge areas was limited in the present study by both the small number of measuring sites and the influence of factors other than recharge. However, some urban areas do show evidence of lower conductivities and higher Eh.

7.6.2.2 Statistical tests:

Because of the problem of confounding between the independent variables (e.g. urban stage and flow stage had a correlation of 0.58), partial correlations were carried out between the independent and dependent variables mentioned in Section 7.8.1. Using this method, only three correlations proved to be significant, i.e.

\[
\begin{align*}
\text{flow stage - mean conductivity} & \quad r = 0.46 \ (**) \\
\text{formation - mean pH} & \quad r = -0.75 \ (***) \\
\text{urban stage - mean temperature} & \quad r = 0.55 \ (***)
\end{align*}
\]

*** $P < 0.001$, ** $P < 0.01$.

The first two correlations were expected (i.e. increasing salinity with distance from a major recharge zone, increased pH in the Tamala Limestone), but the correlation between urban stage and
mean temperature was not expected. The correlation is possibly due to leakage of warm artesian waters from the Yarragadee Formation (Allen 1978) or to the fact that several urban monitoring bores were located adjacent to absorption basins. It is also possible that indirect recharge waters pick up some heat from the surfaces that they run off. As 70 to 80 per cent of all recharge is indirect recharge, this energy input may be significant. In a multiple regression equation for mean temperature, both urban stage and depth of the water table had significant coefficients and the correlation coefficient of the regression increased from 0.55 to 0.62.

7.7 DISCUSSION

The most significant chemical inputs from atmospheric sources appear to be Na\(^+\) and Cl\(^-\) while Ca\(^{2+}\) and SO\(_4^{2-}\) values are higher than expected. The concentration of chloride in mains water during the summer months was about 200 mg/L which is almost 15 times higher than the concentration in rainfall and slightly higher than groundwater concentrations. The salinity of mains water additions will increase markedly in the soil, as little of these additions become recharge. Salts left behind in the soil will be picked up by winter rainfall. Chloride concentrations in road runoff waters are about 35 per cent higher than in rainfall and therefore will lower groundwater salinities around absorption basins.

The main soil inputs appear to be K\(^+\) and Si from feldspars and Fe\(^{2+}\) from goethite. The latter ion is most soluble under reducing conditions and therefore TOC and Eh values are important. The virtual absence of goethite in the sands below the water table in areas adjacent to the absorption basins means that flooding of the vadose zone by reducing waters is required for iron dissolution. Once flooding has removed most of the iron, the quality of groundwater in these areas would be expected to improve. The levels of CO\(_2\)(aq) and H\(_2\)CO\(_3\) in soil water will be important in limestone dissolution. The effect of urbanization on limestone dissolution is unknown.
The major contribution from aquifer materials comes from dissolution of limestone. Dissolution appears to be greatest at the top of the aquifer, possibly because CO$_3^{(aq)}$ and H$_2$CO$_3$ levels are high there and transport of Ca$^{2+}$ and HCO$_3^-$ can take place.

The quality of road runoff in the N-D catchment appears to be better than that reported for many other urban areas, probably because of the lack of soil runoff, low levels of atmospheric fallout by pollutants and the regular sweeping of roads. Gully pits are also cleaned before each winter.

The main effect of adding road runoff to absorption basins that are also wetlands seems to be an indirect effect (increased iron) which results when reduced waters encounter goethite in the yellow, Spearwood sands, following the onset of winter rains. By the end of winter, waters in the absorption basins are less reduced and less saline. The levels of nutrients in the road runoff may contribute to increased rates of eutrophication in urban wetlands.

Iron concentrations in the groundwaters appear to vary with depth. In the high-limestone, N-D catchment, the iron is confined to shallow, sandy areas while in the sandy, S-SP catchment, it is also found in old waters at depth. The salinity and pH of groundwaters in the areas do not appear to have undergone any large changes with time.

The effect of urbanization on salinity is likely to be very uneven in both a spatial and temporal sense. It is possible to calculate the Cl$^-$ mass balances for the annual water balances that were shown in Figures 6.2 and 6.4, using data on concentrations of Cl$^-$ in rainfall (Hingston and Gailitis 1976), mains water (Table 7.2) and aquifer water (Table 7.7).

During the November 1981 to October 1982 period, the concentration of Cl$^-$ in the groundwater leaving the N-D catchment
would have to be about 30 per cent (60 mg/L) higher than that entering it, if Cl⁻ concentrations in the groundwater were to remain constant within the catchment. The high concentration of Cl⁻ (calculated to be 950 mg/L) in direct recharge waters is probably the main reason for this increase in groundwater salinity. The 1982 winter rainfall was 10 per cent below average for this catchment, so the effect of urbanization on salinity is likely to be less in an average year.

Cl⁻ concentrations of groundwater entering and leaving the S-SP catchment would have been similar during the November 1981 to October 1982 period if Cl⁻ concentrations were to remain constant within the catchment. The lower rate of irrigation in this catchment means that the calculated chlorinity of direct recharge waters is only about 500 mg/L. As the 1982 winter rainfall was 18 per cent below average in this catchment, a decrease in groundwater salinity would be expected in an average year. The low salinity zones in Figure 7.14(b) probably represent such areas of enhanced winter recharge.
CHAPTER 8

CONCLUSIONS AND RECOMMENDATIONS

8.1 CONCLUSIONS

The first specific aim of the thesis (i.e. the identification and measurement of the components of the urban water balance) have been substantially met by the water balance diagrams shown in Chapter 6. Despite the errors involved in the estimations of the water balance, the effects of difference degrees of urbanization have been delineated. In arriving at the estimates, a clearer understanding of urban hydrologic processes has been obtained (the second specific aim). Thus the importance of indirect recharge in the urban hydrologic cycle has been indicated, as has the relative unimportance of phreatophyte extraction. The impact of urbanization on groundwater quality has also been assessed (the third specific aim) with the indirect effects due to changing recharge amounts and sites proving to be more important than the direct addition of pollutants.

In general, urbanization has a beneficial effect on groundwater quantity in Perth. In areas with a high housing density, urbanization may also benefit groundwater quality through lowering salinities. The benefits of urbanization come from increased indirect recharge of low-salinity roof and road runoff waters. In most cities these waters are lost to the groundwater system and a consequent decline in groundwater levels occurs. In Perth, the higher the proportion of shedding areas, the greater the recharge so that in areas with a high housing density, excess water has to be drained to an ocean or estuarine outfall. The use of runoff waters for recharge in Perth is made possible by a very porous and permeable aquifer and vadose zone and a separate wastewater system.

The beneficial effect of urbanization on groundwater quantity
can be substantially negated by overpumping of private bores. In
managing groundwater levels, the main input to the groundwater
system capable of being managed has been shown to be indirect
recharge (and septic tank input in unsewered areas) and the main
output to be bore extraction (and drains in high density urban
areas). Converting gully pits in road drains into soak wells would
both increase indirect recharge and decrease iron dissolution from
the vadose zone in areas where flood mitigation drains are necessary
in absorption basins.

More specific conclusions from the study, in the order in which
they were treated in the thesis (i.e. atmospheric, surface water and
groundwater effects), are:

1. There is no direct evidence of an increase in rainfall in the
Perth area as a result of urbanization. Indirect evidence (low
pollution levels, unimportance of convectional rainfall) suggests
urbanization is unlikely to have an appreciable effect on the amount
and distribution of rainfall.

2. During winter, higher temperatures and lower relative (and
actual) humidities in the central business district of Perth
relative to surrounding areas are likely to result in slightly
enhanced potential rates of evaporation for this district. However,
availability of water during summer is the main limit to actual
evaporation rates in Perth. In a 69 per cent vegetated urban area
receiving moderate to high rates of irrigation, actual evaporation
was about 20 per cent higher than rainfall. In a 57 per cent
vegetated urban area receiving low to moderate rates of irrigation,
actual evaporation was about 10 per cent below rainfall. In an
adjoining woodland area, evaporation rates were about equal to
rainfall. Thus urbanization in Perth can either increase or
decrease actual evaporation rates, depending upon housing density
and irrigation factors.

3. Urbanization in Perth results in a much greater change in
runoff rates than reported from most other cities due to the virtual
absence of runoff before urbanization on the sandy soils of the Swan Coastal Plain.

4. The disposal of public stormwaters in Perth is not a major problem (e.g. causing stream flooding) because:
   (i) they are small in volume through containing no roof runoff (except in the central business district) and sewage, and virtually no soil runoff,
   (ii) there are readily available disposal sites which are hard to flood (i.e. absorption basins, estuary, ocean).

5. Water balance studies carried out in two urban areas over an 18-month period have shown that indirect recharge of runoff waters contributes 70 to 80 per cent of all groundwater recharge, despite impermeable surfaces only occupying 30 to 40 per cent of the ground surface area. Indirect recharge occurs in urban areas after all, except very light (< 1 mm), rainfall events. Indirect recharge also occurs from rain falling after long dry periods, when direct recharge is small due to the presence of large soil water deficits.

6. Direct recharge of rainwater is greater in urban areas than in native woodland areas. This is due both to lower canopy interception losses and to lower soil water deficits in the urban areas.

7. Garden irrigation results in lower soil water deficits at the end of the summer period. In addition to adding water to the soil profile, irrigation encourages surface rooting by plants which results in less extraction of soil water at depth.

8. Direct recharge of summer irrigation waters in the urban areas are low, except in cases of high application rates and shallow depths to the water table. When irrigation rates are less than 60 per cent of potential evaporation rates, soil profiles dry out as the summer progresses.
9. Interception losses in tree canopies result in large (e.g. 70%) decreases in direct recharge in both urban and native woodland areas. As urban areas have only about 30 per cent the canopy area of woodland areas, interception losses are probably lower by a commensurate amount in urban areas.

10. Phreatophyte water use in urban areas is minor in comparison with other losses from the groundwater. With the exception of Salix babylonica, only inadequately watered trees within 18 metres of the water table extract significant amounts of groundwater. These trees often lose contact with the water table when it falls at the end of summer and become stressed.

11. In an urban area with large blocks (and therefore relatively low shedding area) and 45 per cent bore use, the groundwater resources were found to be fully committed during periods of average to slightly below-average winter rainfall. As groundwater levels appear to have fallen by only 40 to 50 cm over the last 30 years in the area, it is possible that groundwater inflow from adjoining high recharge areas or decreased groundwater outflow is sufficient to offset any withdrawals. Such a situation is possible due to very high aquifer transmissivities found in the area.

12. In an urban area with small blocks (and therefore high shedding area) and 4 per cent bore use, large amounts of recharge occurred even in years of below-average winter rainfall. Excess water (100 mm) in this area is drained to an ocean outfall and annual recharge was limited by the height of the water table around the absorption basin at the start of winter.

13. The quality of road runoff waters in Perth appears to be significantly better than that reported from many other urban areas. This probably results from a lack of soil runoff, low levels of atmospheric fallout of pollutants and frequent sweeping of roads in the Perth area.
14. The addition of road runoff to absorption basins which are also wetlands results in a seasonal salinity cycle in the groundwater in down-gradient areas. It may also result in a seasonal variation in the amount of iron which is dissolved from flooded goethite-bearing sands by reducing waters when absorption basins are sited within Spearwood Sands.

15. In most parts of the urban area, groundwater salinities would be expected to be increased as a result of evaporation of irrigated waters during summer. However in areas with a high housing density the salinity of the groundwater may be unchanged or even decreased due to the large amounts of indirect recharge of low salinity waters.
8.2 RECOMMENDATIONS

1. The amount of direct and indirect recharge taking place in urban areas should be confirmed using chloride as a conservative tracer. The advantage of soak wells over splash blocks/spoon drains in producing recharge could thereby be determined. If the chloride stored in the soil profiles can be ascertained for different times of the year, and if the seasonal concentration of chloride in the groundwaters can be accurately determined, chloride mass balances would provide a check on the water balance fluxes that have been determined by physical methods.

2. A more comprehensive investigation of stormwater quality is required to determine accurately the range of pollutants in stormwaters and the effects of street sweeping and gully pit clearing on stormwater quality.

3. Road drainage schemes should be examined for modifications which would result in increased recharge of stormwaters and decreased losses to estuarine- and oceanic- outfalls. The threat of saline intrusion in estuarine areas (which are characterised by a high bore ownership and loss of stormwater) makes this examination a high priority.

4. The quality of groundwater downstream of an urban wetland which receives stormwater runoff could be monitored throughout the year to determine accurately the cycles in iron concentrations and salinity. Other parameters of interest (e.g. nitrogen, phosphorus and sulphate) could also be examined. The decay of groundwater mounds around absorption basins and the movement of iron and salinity cycles could be investigated as a means of obtaining information of aquifer transmissivities and stratification.
CHAPTER 9

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LEGEND

1.4  SURFACE CONTOUR (mAHBD)

APPENDIX 1.1: Bathymetry of Masons Gardens basin in the N-D catchment
APPENDIX 1.3: Cores from Shenton Park Lake (above) and Masons Gardens (below)
## APPENDIX 2 (A)

**WEEEKLY WATER BALANCE OF MASONS GARDENS ABSORPTION BASIN**

_All units in m$^3$_.

<table>
<thead>
<tr>
<th>Week</th>
<th>Total runoff from N-D catchment</th>
<th>Runoff from adjoining catchment</th>
<th>Rainfall</th>
<th>Evaporation</th>
<th>Net groundwater addition</th>
<th>Change in storage</th>
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**WEEKLY WATER BALANCE OF MASON'S GARDENS ABSORPTION BASIN**

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### APPENDIX 2.2

**WEEKLY WATER BALANCE OF SHENTON PARK LAKE ABSORPTION BASIN**

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### WEEKLY WATER BALANCE OF SHENTOH PARK LAKE ABSORPTION BASIN

(All units in m³)

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<th>Rainfall</th>
<th>Pumping</th>
<th>Fountain addition</th>
<th>Evaporation</th>
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The University of Western Australia

Department of Geology
Nedlands, Western Australia 6009
Telegrams Uniwest Perth, Telex AL92992
Telephone (09) 380

February, 1981

Dear Householder,

Mr Don McFarlane, a postgraduate student at the University, is carrying out research on the groundwater in the Subiaco-Shenton Park area, and would greatly appreciate your help in completing the attached questionnaire. The aim of the research is to better understand how the growth of the city, and changes in water use, have affected the underground water in the area. To ascertain this, it is necessary to know how much groundwater is extracted by bores and wells, and how much water is returned to the groundwater system by garden watering. As both bore- and mains- (Metropolitan Water Board) water can recharge the groundwater, it is necessary to understand general watering practices. The Subiaco-Shenton Park area has been specially selected for this intensive study and the results will help understand the situation for the whole city.

The study is being carried out independently of any Government agency, and the results will be used in such a way that the information from individuals cannot be identified. For the study to be successful, it is very important that everyone in the area complete the questionnaire (it has been found to take about 15 minutes to answer all questions). I would be grateful if you could complete the questionnaire before the weekend, and either give it to Mr McFarlane, who will call on Saturday or Sunday to collect it, or leave it in an obvious place (e.g. on the door) so that he may pick it up. Once the study has been completed, a summary of the questionnaire results will be forwarded to you, so that you may compare your answers with the averages for the Subiaco-Shenton Park area.

If you have any difficulties completing the questionnaire, Mr McFarlane will be happy to help you when he calls.

I would be most grateful for your cooperation.

Yours sincerely,

[Signature]

P.G. Harris
Professor of Geology
QUESTIONNAIRE

Please tick the circular boxes (☐), or enter a number in the rectangular boxes ([]), as required.

SECTION I: ALL RESPONDENTS

1. How many people live in your house on average?

2. How many of these people fall into the following age groups?

   Age last birthday, in years
   0-2
   3-4
   5-14
   15-24
   25-64
   65 or more

3. Is this house connected to the sewerage system, or do you use septic tanks?

   Sewerage
   Septic tanks
   Other
   Don’t know

4. Which of the following methods are used to water lawns and gardens on your property? (Please tick one or more boxes).

   Fixed reticulation, usually manually operated
   Fixed reticulation, usually operated by time clock
   Movable sprinkler, usually manually operated
   Movable sprinkler, usually with a water minder
   Trickle system
   Hand-held hose
   Watering can or bucket
   Some other method (please specify)

5. In summer, how many times per week do you water your lawns and gardens on average. (If you water your lawns and gardens at the same time, only complete the lawn section).

   (Don’t water, Once a week, 1 time a week, 2 times a week, 3 times a week, 4 times a week, 5 times a week, 6 times a week, 7 times a week, More than 7 (Please state)

   Lawns
   Gardens
   Roadside verge
   Vegetable garden
6. In summer, how long on average does it take to completely water your lawns and gardens, each time you water? (If you water your lawns and gardens at the same time, only complete the lawn section).

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<th>45</th>
<th>60</th>
<th>90</th>
<th>2</th>
<th>3</th>
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<td>Hours (hrs)</td>
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<td></td>
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<td></td>
</tr>
</tbody>
</table>

(Fields are: 0 mins; 15 mins; 30 mins; 45 mins; 60 mins; 90 mins; 2 hrs; 3 hrs; More than 3 hours)

Lawns
Gardens
Roadside verge
Vegetable garden

7. On average how many water outlets (sprinklers, hoses) do you operate at any one time while watering? (If you water your lawns and gardens at the same time, only complete the lawn section).

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<th>2</th>
<th>3</th>
<th>4</th>
<th>5</th>
<th>6</th>
<th>7</th>
<th>8</th>
<th>More than 8</th>
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8. At what time of the day do you generally water?

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<th>Morning (8am-noon)</th>
<th>Afternoon (noon-5pm)</th>
<th>Evening (5pm-9pm)</th>
<th>Night (9pm-5am)</th>
<th>Don't water no lawn/gardens</th>
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9. Do you also water your lawns and gardens during the winter period? (May to September)

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<td>Sometimes</td>
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<tr>
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<td>Only for bore/well maintenance</td>
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10. Are there any rainwater tanks on your property? Yes | No
11. Is there a swimming pool in use on your property?
   Yes, fixed below ground ........................................... 1
   Yes, fixed above ground .......................................... 2
   Yes, movable above ground ....................................... 3
   No pool ............................................................. 4

12. Do you have a bore or well on your property which you use?
   Yes, bore (go to Question 15) .................................... 1
   Yes, well (go to Question 15) ..................................... 2
   No bore or well (Go to Question 13) ............................. 3

13. Do you use water from a neighbour's bore or well?
   Yes (go to Question 15) ............................................ 1
   No (go to Question 14) ............................................. 2

14. Do you plan to install a bore or well, or start to use water from a neighbour's bore or well?
   Yes, plan to install a bore/well .................................. 1
   Yes, plan to use a neighbour's bore/well ....................... 2
   No, inhibited by financial considerations ..................... 3
   No, don't plan to ever install a bore/well .................... 4

Please go to the GENERAL COMMENTS section at the end of the Questionnaire.

SECTION II: BORE AND WELL USERS ONLY

15. Do you also use scheme (Met. Water Board) water on your lawns and gardens?
   Never ........................................................................... 1
   Rarely (less than 10 times per year) .................................. 2
   Sometimes (10 to 50 times per year) ................................. 3
   Often (more than 50 times per year) ................................... 4

If you use water from a neighbour's bore or well, please go to the GENERAL COMMENTS section at the end of the questionnaire.

If you have a bore or well on your property, please try and answer questions 16 to 27 as accurately as possible.

SECTION III: BORE AND WELL OWNERS ONLY

16. When was your bore/well constructed? Before 1940 ............. 1
    1940 - 1950 ............................................................ 2
    1950 - 1960 ............................................................ 3
    1960 - 1970 ............................................................ 4
    1970 - 1981 ............................................................ 5
    Don't know ............................................................. 6

If constructed after 1970, please state what year ............

17. What drilling/reticulation company installed your bore/well?
   Name ........................................................................... 1
   Don't know .................................................................... 2
18. What make and model pump do you have on your bore/well?

Make ........................................
Model ........................................
Don't know .................................

19. What horsepower is your pump?  
   3/4 H.P.  ...............  
   1 H.P.  ...............  
   1½ H.P.  .............  
   2 H.P.  ...............  
   3 H.P.  ...............  
   5 H.P.  ...............  
Other (please state) .....................
Don't know ...............................  

20. In summer, how long do you operate your bore/well each week?  
   ............... minutes  
   or ............... hours  

21. Is your bore/well operated by a time clock?  
   Yes...  ...............  
   No...  ...............  

22. Do you have an estimate of how much bore water you use each week in summer?  
   Yes...  ...............  
   No...  ...............  

If yes, please state how much ...........................

23. Have you ever had to:  
   a) Deepen your bore/well  
      Yes...  ...............  
      No...  ...............  
      Don't know. .............
   
   b) Clean or replace your screen?  
      Yes...  ...............  
      No...  ...............  
      Don't know. .............

24. How deep is your bore/well now?  
   ............... feet  
   or ............... metres  
   Don't know ......................  

25. Does your bore/well water leave a brown iron stain on fences and paving?  
   No stain ...............  
   Slight stain ............  
   Moderate stain ........  
   Considerable stain ....

25 a) Does your bore/well water leave a white lime deposit on windows and leaves?  
   No deposit .............  
   Slight deposit ........  
   Moderate deposit .......
26. Have you noticed any change in the quality of water you receive from your bore/well?

Yes .... ○ | 1
No .... ○ | 2

If yes, please give details ...........................................
........................................................................

27. What was the main reason you installed your bore/well?

Convenience of watering lawns and gardens .... ○ | 1
Water restrictions .............................. ○ | 2
Increasing cost of Met. Water Board water .... ○ | 3
Better garden ...................................
Other (please state) ................................. ○ | 5
Don't know, didn't install the bore/well .... ○ | 6

SECTION IV: GENERAL COMMENTS, ALL RESPONDENTS

(If you wish to make any comments about any of your above answers, or about water use practices in general, please do so in this section. If you know the water level in your bore/well on construction (or at any time since), or the nature of the material through which it was constructed, this information would be very welcome).

As part of the study it is necessary to estimate the amount of mains-water used each month in the Subiaco-Shenton Park area. Would you be agreeable to readings being made of your meter every 4 weeks over the 18 month study period?

Yes .... ○ | 1
No .... ○ | 2

Thank you for your co-operation. A summary of the results of the questionnaire will be forwarded to you as soon as possible.
APPENDIX 3.2

QUESTIONNAIRE RESULTS ON BORE WATER USE

Only bore owners answered the following questions. There were only 12 bore owners in the S-SP sample and therefore the percentages shown are only rough guides for this catchment.

All figures are per cent unless otherwise stated.

1. Use of mains water in addition to groundwater use

<table>
<thead>
<tr>
<th>Frequency of use</th>
<th>N-D</th>
<th>S-SP</th>
</tr>
</thead>
<tbody>
<tr>
<td>Never</td>
<td>23</td>
<td>0</td>
</tr>
<tr>
<td>Rarely (&lt; 10 times/year)</td>
<td>26</td>
<td>0</td>
</tr>
<tr>
<td>Sometimes (10-50 times/year)</td>
<td>36</td>
<td>27</td>
</tr>
<tr>
<td>Often (&gt; 50 times/year)</td>
<td>15</td>
<td>73</td>
</tr>
</tbody>
</table>

2. Period when bore was installed

<table>
<thead>
<tr>
<th>Period</th>
<th>N-D</th>
<th>S-SP</th>
</tr>
</thead>
<tbody>
<tr>
<td>1940-50</td>
<td>4</td>
<td>12</td>
</tr>
<tr>
<td>1950-60</td>
<td>13</td>
<td>0</td>
</tr>
<tr>
<td>1960-70</td>
<td>7</td>
<td>13</td>
</tr>
<tr>
<td>1970-81</td>
<td>70 (see below)</td>
<td>63</td>
</tr>
<tr>
<td>Don't know</td>
<td>6</td>
<td>12</td>
</tr>
</tbody>
</table>

N-D: 1970-81

<table>
<thead>
<tr>
<th></th>
<th>'70</th>
<th>'71</th>
<th>'72</th>
<th>'73</th>
<th>'74</th>
<th>'75</th>
<th>'76</th>
<th>'77</th>
<th>'78</th>
<th>'79</th>
<th>'80</th>
<th>'81</th>
<th>Total</th>
</tr>
</thead>
<tbody>
<tr>
<td>%</td>
<td>0</td>
<td>0</td>
<td>1</td>
<td>0</td>
<td>1</td>
<td>2</td>
<td>0</td>
<td>5</td>
<td>27</td>
<td>22</td>
<td>10</td>
<td>2</td>
<td>70</td>
</tr>
</tbody>
</table>
3. Horsepower of pump

<table>
<thead>
<tr>
<th>Horsepower</th>
<th>N-D</th>
<th>S-SP</th>
</tr>
</thead>
<tbody>
<tr>
<td>3/4</td>
<td>3</td>
<td>12</td>
</tr>
<tr>
<td>1</td>
<td>4</td>
<td>38</td>
</tr>
<tr>
<td>1 1/2</td>
<td>14</td>
<td>12</td>
</tr>
<tr>
<td>2</td>
<td>19</td>
<td>0</td>
</tr>
<tr>
<td>3</td>
<td>18</td>
<td>25</td>
</tr>
<tr>
<td>5</td>
<td>8</td>
<td>0</td>
</tr>
<tr>
<td>Other</td>
<td>4</td>
<td>0</td>
</tr>
<tr>
<td>Don't know</td>
<td>30</td>
<td>13</td>
</tr>
</tbody>
</table>

4. Duration of pumping operation during summer (minutes/week)

    N-D  333   S-SP  210

5. Time clock used?  Yes.  N-D 44%;  S-SP 13%

6. Estimate of bore water used?  Yes.  N-D 15%;  S-SP 44%

    Estimate (m³/week)  N-D 34;  S-SP 25

7. Need to deepen bore?  Yes.  N-D 4%;  S-SP 12%

    Need to clean screen?  Yes.  N-D 6%;  S-SP 12%

8. Depth of bore known?  Yes.  N-D 80%;  S-SP 100%

    Average depth (m)  N-D 25;  S-SP 15

9. Extent of iron staining

<table>
<thead>
<tr>
<th>Extent</th>
<th>N-D</th>
<th>S-SP</th>
</tr>
</thead>
<tbody>
<tr>
<td>No stain</td>
<td>81</td>
<td>12</td>
</tr>
<tr>
<td>Slight</td>
<td>6</td>
<td>25</td>
</tr>
<tr>
<td>Moderate</td>
<td>6</td>
<td>38</td>
</tr>
<tr>
<td>Considerable</td>
<td>7</td>
<td>25</td>
</tr>
</tbody>
</table>

10. Extent of lime deposits

<table>
<thead>
<tr>
<th>Extent</th>
<th>N-D</th>
<th>S-SP</th>
</tr>
</thead>
<tbody>
<tr>
<td>No deposit</td>
<td>56</td>
<td>63</td>
</tr>
<tr>
<td>Slight deposit</td>
<td>31</td>
<td>25</td>
</tr>
<tr>
<td>Moderate deposit</td>
<td>13</td>
<td>12</td>
</tr>
</tbody>
</table>
11. **Any change in water quality noticed?**

Yes. N-D 4%; S-SP 25%

**Changes noted:**
- N-D: 1 increased iron, 1 increased odour,
  1 slightly improved quality,
  2 eliminated iron after deepening.
- S-SP: 1 increased iron, 1 increased iron
  after deepening.

12. **Main reason for installing bore/well**

<table>
<thead>
<tr>
<th>Reason</th>
<th>N-D</th>
<th>S-SP</th>
</tr>
</thead>
<tbody>
<tr>
<td>Convenience</td>
<td>32</td>
<td>25</td>
</tr>
<tr>
<td>Water restrictions</td>
<td>26</td>
<td>13</td>
</tr>
<tr>
<td>Cost of water</td>
<td>12</td>
<td>25</td>
</tr>
<tr>
<td>Better garden</td>
<td>8</td>
<td>25</td>
</tr>
<tr>
<td>Don't know</td>
<td>22</td>
<td>12</td>
</tr>
</tbody>
</table>
APPENDIX 4

CHEMICAL AND MINERALOGICAL STUDY OF THE MA AND KC SOIL PROFILES

4.1 CHEMICAL

Figure A4.1 shows the trend in eight oxides as determined by XRF analysis of four samples from the MA profile. The water table in this profile was at a depth of six metres. Almost all oxides show a progressive increase to five metres, followed by a decrease below the water table. Na$_2$O and Fe$_2$O$_3$ show the largest percentage decrease while SiO$_2$ is the only oxide which increases below the water table.

Figure A4.1 also shows the analyses of two samples from the KC profile. The sample at 4.5 metres is at the peak in clay content as determined by textural analysis. Almost all oxides show an increase at 4.5 metres with the largest percentage increases occurring for Fe$_2$O$_3$ and Al$_2$O$_3$. SiO$_2$ is the only oxide which shows a significant decrease at 4.5 metres.

4.2 MINERALOGY

XRD analysis was carried out on the fine grained sand and the silt + clay fractions for eight samples from the MA and KC profiles. In both profiles, the fine sand fraction was dominated by quartz and microcline, with minor amounts of kaolinite (spherites?) occurring in the upper parts of the KC profile. The microcline peak on the XRD charts decreased in size with increasing depth within the KC profile and also below the water table in the MA profile. The silt + clay fraction in both profiles was dominated by kaolinite and goethite with lesser amounts of gibbsite. Both goethite and gibbsite were absent below the water table in the MA profile. Microscopic examination of the coarse and fine sand fractions indicated a relative enrichment of microcline over quartz in the fine sand fraction.
FIGURE A 4.1: Oxide distribution in the MR (above) and KC (below) profiles
4.3 SYNTHESIS

The **MA profile** consists of quartz-microcline-kaolinite-goethite-gibbsite with a gradual increase in all minerals except quartz above the water table, this trend being reversed below the water table. Trace amounts of plagioclase and ilmenite (as evidenced by $\text{Na}_2\text{O}$ and $\text{TiO}_2$) probably also follow this trend.

The **KC profile** consists of the same minerals as in the MA profile with all minerals except quartz and microcline increasing towards the clay layer. The decline in microcline in the fine grained sand fraction at the clay layer is accompanied by an increase in clay, perhaps indicating **in situ** kaolinisation is taking place.
APPENDIX 5

SOIL WATER PROFILES FOR URBAN AND NATIVE WOODLAND AREAS

About one third of the soil water profiles that were obtained for each soil profile are shown, along with brief explanatory notes on relevant site conditions and soil characteristics. The profiles exhibit all the variability encountered in urban areas. In particular, irrigation regimes are variable with time, especially following changes in home ownership and during holiday periods. Occasional heavy watering results from car- and boat-washing and from leaving sprinklers on overnight.

Details of site conditions include estimates of canopy cover to the west and north of the access hole. The cover area was taken as that between 45° and the vertical. The western sector is an indicator of rainfall interception during winter while the northern sector is an indicator of shading (which affects evaporation).

Two soil water profiles (KA and DA) were previously shown as Figures 5.30 and 5.31.
SITE NAME: KB

STUDY AREA: Kings Park

SETTING: Native woodland. High interception site (Banksia-Casuarina)

PERCENTAGE CANOPY

Northern sector: 60

SATURATED HYDRAULIC CONDUCTIVITY:

18 m/d at 6.9 m depth.

GRAIN SIZE DISTRIBUTION OF PROFILE:

Medium-grained sand throughout. Progressive increase in fineness and decrease in sorting with depth, with a band with higher clay content at 3.6-3.9 m. Charcoal layer at 1.8 m. Limestone and tree roots at 7.2 m.

IRRIGATION REGIME:

None.

REMARKS:

Evidence of increased water holding capacities at 1.8 and 3.6 m. Low water contents at 7.2 m correspond to the area with a higher concentration of tree roots and the presence of limestone. The wetting front was encountered at a depth of 6.9 m when the hole was drilled in October 1980. The site was disturbed in summer 1981 when branches were cut away from surrounding trees by the Kings Park Board. The profile thereafter responded similarly to the two low interception sites in the Kings Park area (i.e. KA and KC). Deep drainage during 1981 was only 45 per cent (50 mm) that of KA (Fig. 5.30) and KC while in 1982 about 20 mm deep drainage occurred at KB while none occurred at KA and KC. Understorey species were disturbed when the branches were cut down which may explain this extra deep drainage and also why the January 1982 rainfall penetrated to a greater depth at KB than at KA and KC. The higher clay content zone at 3.6 m is less well developed at KB than at KC and KA. The charcoal layer at 1.8 m identifies a previous soil profile.
SITE NAME: KC

STUDY AREA: Kings Park

SETTING: Native woodland. Low interception site.

PERCENTAGE CANOPY:

Western sector: 30
Northern sector: 50

SATURATED HYDRAULIC CONDUCTIVITY:

14 m/d at 720 cm.

GRAIN SIZE DISTRIBUTION OF PROFILE:

Discussed in some detail in Section 5.7.3.1 and Appendix 4. Medium sand throughout with a band of sandy loam (to 16% clay) at 420 cm.

IRRIGATION REGIME:

None.

REMARKS:

Despite the layer with higher clay content at 420 cm, the advance of the wetting front appears similar to profile KA (Fig. 5.30) which has a less well developed clay layer. The difference between the maximum and residual water contents in the clay-enriched layer appears to be slightly less (4-5%) than in the rest of the profile (6-7%). Deep drainage of KC profile was about 75 per cent that at KA during 1981 (95 mm vs 125 mm) which may have been due to reasons other than the finer textured zone. The wetting fronts stalled at 660 and 630 cm in the KA and KC profiles respectively during 1982. There was almost no carryover of soil water from the January 1982 rainfall into the 1982 winter period.
SITE NAME: SA

STUDY AREA: Selby Street

SETTING: Native woodland. Low interception site.

PERCENTAGE CANOPY:

Western sector : 15
Northern sector : 60

SATURATED HYDRAULIC CONDUCTIVITY:

18 m/d at 720 cm.

GRAIN SIZE DISTRIBUTION OF PROFILE:

Medium-grained sand throughout. Decrease in sorting and increase in fine skew with increasing depth. Higher clay zone at 390 cm.

IRRIGATION REGIME:

None.

REMARKS:

The form of the moisture characteristic curve is shown behind the wetting fronts in this hole. Deep drainage was 128 mm (15% of rainfall) in 1981 and 65 mm (8%) in 1982. The advance of the wetting front was not as rapid, or as uniform, in 1982.
SITE NAME: SB
STUDY AREA: Selby Street
SETTING: Native woodland. High interception site (Casuarina sp).
PERCENTAGE CANOPY:
   Western sector: 90
   Northern sector: 75
SATURATED HYDRAULIC CONDUCTIVITY:
   33 m/d at 600 cm.
GRAIN SIZE DISTRIBUTION OF PROFILE:
   Medium-grained sand throughout. Better sorted than SA (low, fine skew). Decrease in grain size and sorting with increasing depth.
IRRIGATION REGIME:
   None
REMARKS:
The retardation of the wetting front advance is clearly shown in a comparison with holes SA and SC. Only 24 mm (3% of rainfall) deep drainage occurred during 1981 and no deep drainage occurred during 1982. The wetting front was encountered at 630 cm when the hole was drilled during October 1980, indicating the limited wetting of the soil profile during this year. The January 1982 rainfall penetrated to 150 cm which was similar to the SA (180 cm) and SC (150 cm) holes, probably due to the continuous nature of the rainfall limiting interception losses and also the influence of shading.
SITE NAME: SC
STUDY AREA: Selby Street
SETTING: Native woodland. Low interception site.
PERCENTAGE CANOPY:

Western sector : 25
Northern sector : 20
SATURATED HYDRAULIC CONDUCTIVITY:

40 m/d at 660 cm.
GRAIN SIZE DISTRIBUTION OF PROFILE:

Medium sand throughout. Coarser grained than either the SA or SB profiles. Finest textured soil appears in the centre of the profile with coarser textured, lime-rich sand at the base of the hole.
IRRIGATION REGIME: None
REMARKS:

Very similar soil moisture profile to hole SA, with the exception that the clay-enriched layer at 390 cm is less pronounced. The two holes are about 300 m apart. Deep drainage was 151 mm (19% of rainfall) during 1981 and 81 mm (10%) during 1982.
SITE NAME: SS

STUDY AREA: N-D catchment

SETTING: Lawn within a private garden.

PERCENTAGE CANOPY:

Western sector: 10
Northern sector: 15

SATURATED HYDRAULIC CONDUCTIVITY:

20 m/d at 660 cm.

GRAIN SIZE DISTRIBUTION OF PROFILE:

Medium-grained sand throughout. Clay-enriched layer at 480 cm. Progressive decrease in grain size with increasing depth, apart from the finer-textured layer at 480 cm.

IRRIGATION REGIME:

Mains water. 25-30 mm/week added during summer. Occasional extra watering of lawn during boat washing in summer.

REMARKS:

The profiles show a progressive drying out of the soil during summer, except when subject to occasional extra water additions (i.e. 9.81, 11.81, 4.82). The winter rainfall becomes deep drainage by June in this profile, compared with September (or later) in the native woodland sites. The rapid fall in water content (and organic matter) between 30 and 60 cm, as evidenced by the decreased neutron count, is typical of many urban lawns.
SITE NAME: DB

STUDY AREA: N-D catchment

SETTING: Lawn within a private garden

PERCENTAGE CANOPY:

Western sector: 5
Northern sector: 60

SATURATED HYDRAULIC CONDUCTIVITY:

28 m/d at 660 cm

GRAIN SIZE DISTRIBUTION OF PROFILE:

Medium-grained, moderately-well sorted sand throughout. Progressive decrease in grain size and sorting with increasing depth. Finer-textured layer at 390 cm.

IRRIGATION REGIME:

Bore water. 45-90 mm/week during summer (the larger amounts being applied during the hottest months).

REMARKS:

An example of a heavily watered urban lawn, under which the soil water profile alters little throughout the year. The profile is only 10 metres from the DA profile (Fig. 5.31) which received 20 to 50 mm per week during summer. Wetting fronts produced by winter rainfall fully penetrate the profile fully by May or June.
SITE NAME: AR

STUDY AREA: N-D catchment

SETTING: Lawn within a private garden. Moderate interception site.

PERCENTAGE CANOPY:

Western sector: 65
Northern sector: 80

SATURATED HYDRAULIC CONDUCTIVITY:

27 m/d at 390 cm.

GRAIN SIZE DISTRIBUTION OF PROFILE:

Medium-grained yellow and brown sand throughout. Progressive decrease in grain size and sorting with increasing depth (including that part of the profile beneath the water table).

IRRIGATION REGIME:

Bore water. 80 mm/week throughout the summer (timeclock-operated).

REMARKS:

The high rate of irrigation and amount of shading of this profile results in the significant recharge of irrigated water during summer. There is some indication that tree roots exist at 480-540 cm, the higher readings at 510 cm probably resulting from a nearby root. There are indications that some soil water is being extracted from the 480-540 cm interval, despite the heavy irrigation regime. The bore at this site was installed during the 1980/81 summer. The moisture characteristics for this hole were shown in Figure 5.28, indicating appreciable deep drainage/recharge during summer.
SITE NAME: MJ

STUDY AREA: N-D catchment

SETTING: Lawn within an urban garden. Low interception site.

PERCENTAGE CANOPY:

Western sector: 5
Northern sector: 5

SATURATED HYDRAULIC CONDUCTIVITY:

17 m/d at 660 cm.

GRAIN SIZE DISTRIBUTION OF PROFILE:

Medium-grained sand throughout but becoming progressively finer with depth and becoming almost a fine sand at 690 cm. Sorting is best at the top and bottom of the profile, being least in the vicinity of a higher clay content zone at 390-420 cm.

IRRIGATION REGIME:

Bore water. Heavy watering (80 mm/week) during the 1980/81 summer, being less (30 to 45 mm/week) during the 1981/82 and 1982/83 summers following a change of ownership.

REMARKS:

The lower watering regime during 1981/82 (compared with 1980/81) resulted in a delay in the advance of the winter wetting front by about a month. Watering during the 1982/83 summer did not commence until late, resulting in a marked drying out of the soil profile in comparison with previous summers. This drying of the sandy soils is often accompanied by the onset of non-wetting characteristics which can inhibit uniform water penetration of the soil profile.
SITE NAME: BS

STUDY AREA: N-D catchment

SETTING: Garden bed containing low bushes in private garden.

PERCENTAGE CANOPY:

Western sector: 80%
Northern sector: 95%
variable due to pruning

SATURATED HYDRAULIC CONDUCTIVITY:

20 m/d at 570 cm.

GRAIN SIZE DISTRIBUTION OF PROFILE:

Medium-grained, moderately well sorted sand throughout, becoming progressively finer grained and less well sorted with depth. Limestone encountered at 600 cm.

IRRIGATION REGIME:

Hand watered with bore water. 40 mm/week applied during the summer (approximate due to variability inherent in the watering method).

REMARKS:

The very high 'water contents' at the surface are due to the presence of loam added to the garden bed and to roots. The winter wetting fronts became deep drainage by June, signifying a low soil water deficit in the soil profile. The slight decrease in soil water contents at the bottom of the hole probably signify a lower water holding capacity of the lime-rich sand encountered immediately above the solid limestone at 600 cm.
SITE NAME: RA

STUDY AREA: N-D catchment

SETTING: Lawn in private garden. Low interception site adjacent to the high interception, RB site.

PERCENTAGE CANOPY:

Western sector: 5
Northern sector: 15

SATURATED HYDRAULIC ACTIVITY:

30 m/d at 510 cm.

GRAIN SIZE DISTRIBUTION OF PROFILE:

Medium-grained, moderately well sorted sand throughout, becoming progressively finer grained and less well sorted with depth (including below the water table).

IRRIGATION REGIME:

Bore water. Poorly watered area (5 to 10 mm/week) due to poor sprinkler coverage. The topsoil at this site is also slightly compacted due to vehicles which, in addition to non-wetting due to soil dryness, may result in some runoff during heavy rainfall.

REMARKS:

Very dry soil profile throughout the summer months. Winter wetting fronts reach the water table in July, while the water table begins to rise in May. Wetting fronts are not as sharply defined as in other holes. Evidence of water extraction by tree roots at 510 cm.
SITE NAME: RB
STUDY AREA: N-D catchment
SETTING: Lawn in private garden. High interception site.
PERCENTAGE CANOPY:
Western sector: 95
Northern sector: 90
SATURATED HYDRAULIC CONDUCTIVITY:
Not measured.
GRAIN SIZE DISTRIBUTION OF PROFILE:
Not measured (assumed to be similar to RA, only five metres away). Tree root encountered at 120 cm while drilling the hole.
IRRIGATION REGIME:
Bore water. 7 to 11 mm/week. Extra water added in November 1981 and March 1982 (car washing?).
REMARKS:
Despite the higher watering regime than at RA, winter wetting fronts did not reach the water table until September and October due to interception losses. As for RA, there is evidence of extraction by tree roots at 510 cm. Recharge at this site was only about 33 per cent that at RA during 1981 and 1982.
SITE NAME: GA

STUDY AREA: N-D catchment

SETTING: Lawn in private garden.

PERCENTAGE CANOPY:

Western sector: 10
Northern sector: 50

SATURATED HYDRAULIC CONDUCTIVITY:

30 m/d at 600 cm.

GRAIN SIZE DISTRIBUTION OF PROFILE:

Medium-grained throughout, although the sand at 1 m is almost a coarse sand (Mz = 1.083). A progressive decrease in mean grain size and sorting with increasing depth.

IRRIGATION REGIME:

Mains water. Low rate of irrigation (10 to 15 mm/week) during late summer 1981 and the summer of 1981/82. Extra watering (boat washing) in January 1982 and during November 1982 following a change of ownership.

REMARKS:

The low water contents (3%) at 90 cm during late summer indicate the failure of irrigation at < 60 per cent potential evaporation rates to provide deep drainage waters. Apart from a tree root at 120 cm, this profile shows a monotonic increase in 'water content' with increasing depth in late summer. This increase is due to an increase in bulk density and water holding capacity and a decrease in root densities.
SITE NAME: GB

STUDY AREA: N-D catchment

SETTING: Lawn in private garden, beneath the canopy of a deciduous tree (Melia azedarach)

PERCENTAGE CANOPY:
Western sector: 25) winter value decreases to about 30 in the northern sector.
Northern sector: 80

SATURATED HYDRAULIC CONDUCTIVITY:
24 m/d at 660 cm.

GRAIN SIZE DISTRIBUTION OF PROFILE:
Very similar to GA, which is located on another property about 10 m away. Distribution shown in Figure 5.22.

IRRIGATION REGIME:
Bore water. Generally low (10-20 mm/week) except for two heavy waterings during February and March 1982 (sprinkler left on overnight) and following a change of ownership in October 1982.

REMARKS:
The very slow advance of the wetting front during winter 1981 was due to enhanced interception resulting from a vehicle being parked to the west of the hole during several heavy storms. Generally, irrigation rates were insufficient to prevent the soil profile from drying out during the summer, as was the case for site GA. The January 1982 rain failed to penetrate the soil profile much beyond one metre as a result of a high soil water deficit and interception by the overhead deciduous canopy.
SITE NAME: GC

STUDY AREA: N-D catchment

SETTING: Lawn in private garden (back yard of same block as GB)

PERCENTAGE CANOPY:

Western sector: 75
Northern sector: 10

SATURATED HYDRAULIC CONDUCTIVITY:

23 m/d at 630 cm.

GRAIN SIZE DISTRIBUTION OF PROFILE:

Medium-grained, moderately well sorted sand throughout, as for GA and GB. Progressive decrease in grain size but little change in sorting with increasing depth.

IRRIGATION REGIME:

Bore water. Slightly better watered than GB. Extra heavy watering during November and December 1981 and March 1982 due to an uneven watering regime. Very heavy watering following the change of ownership in October 1982.

REMARKS:

The progressive drying out of the soil profile during summer in this profile was disrupted by the erratic heavy watering periods. This type of watering is shown to be wasteful of water with the whole profile (> 720 cm) being wet once a month with only the water in the top 150 cm being available to the lawn. Watering during October and November 1982 was in excess of 100 mm/week which also resulted in excessive leaching of the soil profile and wasted pumping.
SITE NAME: CC

STUDY AREA: S-SP catchment

SETTING: Lawn in Local Government Authority's public gardens

PERCENTAGE CANOPY:

Western sector: 5
Northern sector: 5

SATURATED HYDRAULIC CONDUCTIVITY:

6.5 m/d at 690 cm.

GRAIN SIZE DISTRIBUTION OF PROFILE:

Medium-grained to two metres, fine-grained below this depth. Bimodal distribution with a fine skew at the top of the hole becoming a coarse skew at the bottom. A marked increase in clay content below 390 cm.

IRRIGATION REGIME:

Bore water. Very low watering regime (10-15 mm/week) applied regularly (time clock).

REMARKS:

A large soil water deficit in the top 360 cm of this profile at the end of each summer delayed drainage beneath 720 cm until August each year. As for profile KC, the existence of a higher clay content zone appeared to have little effect on the rate of advance of the wetting front. Wetting fronts in the coarser-textured part of the profile appear sharper than in the finer-textured part. Little recharge is likely to take part at this site due to the low watering regime and the extreme depth (30 metres) of the water table.
<table>
<thead>
<tr>
<th>SITE NAME:</th>
<th>GS</th>
</tr>
</thead>
<tbody>
<tr>
<td>STUDY AREA:</td>
<td>S-SP catchment</td>
</tr>
<tr>
<td>SETTING:</td>
<td>Non-watered verge</td>
</tr>
<tr>
<td>PERCENTAGE CANOPY:</td>
<td></td>
</tr>
<tr>
<td>Western sector: 5</td>
<td></td>
</tr>
<tr>
<td>Northern sector: 0</td>
<td></td>
</tr>
<tr>
<td>SATURATED HYDRAULIC CONDUCTIVITY:</td>
<td>6 m/d at 720 cm.</td>
</tr>
<tr>
<td>GRAIN SIZE DISTRIBUTION OF PROFILE:</td>
<td>Fine-skewed, medium-grained sand grading to a coarse-skewed, fine-grain sand at the bottom of the profile. Strongly bimodal (Fig. 5.22).</td>
</tr>
<tr>
<td>IRRIGATION REGIME:</td>
<td>None</td>
</tr>
<tr>
<td>REMARKS:</td>
<td>The very slow advance of the wetting front during the 1981 and 1982 winters resulted from a vehicle being parked over this hole during some heavy storms. The profile shows evidence of lateral dispersion of water with the rapid wetting of the profile at depth during August 1981 and September 1982. A progressive increase in water-holding capacity with increasing depth, in addition to some increase in bulk density, was probably the cause of the increase in residual water content from 3 per cent at the surface to 7-8 per cent at 720 cm.</td>
</tr>
</tbody>
</table>

* not correlated with increased rainfall.
SITE NAME: SL

STUDY AREA: S-SP catchment

SETTING: Lawn in gardens surrounding Shenton Park Lake, watered by Local Government Authority

PERCENTAGE CANOPY:
Western sector: 15
Northern sector: 0

SATURATED HYDRAULIC CONDUCTIVITY:
49 m/d at 390 cm

GRAIN SIZE DISTRIBUTION OF PROFILE:
Moderately well sorted, medium-grained sand throughout. Mean grain size decreases below the water table. Sand immediately above the water table is particularly well sorted which may explain the very high saturated hydraulic conductivity.

IRRIGATION REGIME:
Bore water. Watered every night, 20–30 mm/week

REMARKS:
Wetting fronts reached the water table by June or July due to its shallow depth. The profile is about 120 m up-gradient of Shenton Park Lake. The water table rose before the wetting fronts reached it. The water table dried out to about 3 per cent water content by the end of summer, indicating little deep drainage was occurring despite a nightly watering at about 40 per cent of potential evaporation rates.
SITE NAME: DR

STUDY AREA: S-SP catchment

SETTING: Unwatered street verge

PERCENTAGE CANOPY:

Western sector: 0
Northern sector: 0

SATURATED HYDRAULIC CONDUCTIVITY:

32 m/d at 390 cm

GRAIN SIZE DISTRIBUTION OF PROFILE:

Moderately well sorted, medium-grained sand throughout. Decrease in mean grain size and sorting until 500 cm followed by little change between 500 and 725 cm.

IRRIGATION REGIME:

None

REMARKS:

This profile is located about 300 m east of Shenton Park Lake yet the water table rose well before the wetting front reached it. This profile developed a substantial soil water deficit during summer, the surface drying out to less than 2 per cent moisture content. Consequently wetting fronts only reached the water table in August, despite its shallow depth.
SITE NAME: HR

STUDY AREA: S-SP catchment

SETTING: Lawn in private garden

PERCENTAGE CANOPY:

Western sector: 20
Northern sector: 70

SATURATED HYDRAULIC CONDUCTIVITY:

16 m/d at 690 cm

GRAIN SIZE DISTRIBUTION OF PROFILE:

Moderately sorted, medium-grained sand throughout. Pronounced bimodal distribution with an increase in the fine mode with increasing depth. Finer textured horizon at 390 cm.

IRRIGATION REGIME:

Mains water. Light-moderate watering regime 25-30 mm/week during the 1981/82 summer, heavier watering (not measured) during the 1980/81 summer.

REMARKS:

The heavier watering during the 1980/81 summer resulted in deep drainage by July. During the 1981/82 summer, the soil profile dried out to only 2 per cent water content and deep drainage did not commence until September. The difference between minimum and maximum water contents was about 5 per cent for all parts of the profile during 1982.
SITE NAME: MA

STUDY AREA: S-SP catchment

SETTING: Lawn on verge (low interception site for comparison with site MB)

PERCENTAGE CANOPY:
- Western sector: 5
- Northern sector: 5

SATURATED HYDRAULIC CONDUCTIVITY:
12 m/d at 390 cm

GRAIN SIZE DISTRIBUTION OF PROFILE:
Moderately well sorted, medium-grained sand throughout. Increase in grain size and sorting below the water table. Profiles shown in Figures 5.22 and 5.24(b). Mineralogy and chemistry discussed in Appendix 4.

IRRIGATION REGIME:
Bore water. Low irrigation rate, 5-10 mm/week

REMARKS:
Located about 250 m west of Shenton Park Lake. Irrigation rates were too low to prevent the soil profile drying out each summer. Wetting front reached the water table by June each year, about one month after the water table began to rise. Some evidence of root extraction at 450 cm, probably by a T. conferta tree located about four metres away.
SITE NAME: MB

STUDY AREA: S-SP catchment

SETTING: Lawn on verge. High interception site - beneath *Tristaniopsis conferta* canopy.

PERCENTAGE CANOPY:

Western sector : 95  
Northern sector : 95

SATURATED HYDRAULIC CONDUCTIVITY: Assumed to be the same as for GRAIN SIZE DISTRIBUTION OF PROFILE: profile MA, located 10 m away

IRRIGATION REGIME:

Bore water. Slightly better watered than MA (10-20 mm/week)

REMARKS:

The high interception equivalent of the MA profile. Recharge during the 1982 winter was only 26 mm, being about a quarter that of the MA site (110 mm). The advance of the wetting front was about 10 weeks behind the MA profile during winter. However the January 1982 rainfall penetrated to similar depths, probably as a result of the continuous form of the rainfall and decreased evaporation due to shading at the MB site. No evidence of major water extraction by tree roots from the capillary fringe, despite the proximity (1 m) of the tree.
SITE NAME: ER

STUDY AREA: S-SP catchment

SETTING: Unwatered verge, adjacent to T. conferta tree

PERCENTAGE CANOPY:

<table>
<thead>
<tr>
<th>Sector</th>
<th>Percentage</th>
</tr>
</thead>
<tbody>
<tr>
<td>Western sector</td>
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<tr>
<td>Northern sector</td>
<td>65</td>
</tr>
</tbody>
</table>

SATURATED HYDRAULIC CONDUCTIVITY:

16 m/d at 660 cm.

GRAIN SIZE DISTRIBUTION OF PROFILE:

Moderately sorted, medium-grained sand throughout. Increase in mean grain size and sorting below the water table.

IRRIGATION REGIME:

None

REMARKS:

The best profile for showing water extraction by T. conferta roots in the capillary fringe zone. The considerable soil water deficit at the end of summer at this site resulted in the wetting front reaching the water table only in September. As for all soil profiles located in low topographic positions, the finer-textured zone at 390-450 cm was absent.
SITE NAME: OR

STUDY AREA: S-SP catchment

SETTING: Lawn in private garden. Moderate interception site, becoming a high interception site as the study progressed.

PERCENTAGE CANOPY:

Western sector : 75
Northern sector : 90

SATURATED HYDRAULIC CONDUCTIVITY:

28 m/d at 240 cm

GRAIN SIZE DISTRIBUTION OF PROFILE:

Moderately sorted, medium-grained sand throughout. Progressive decrease in grain size and sorting with increasing depth.

IRRIGATION REGIME:

Mains water. Moderate, 25-30 mm/week.

REMARKS:

A profile with a very shallow water table located about 200 m down-gradient of Shenton Park Lake. The soil profile dried back to 3 per cent water content each summer, despite the irrigation and a high degree of shading. The water table rose before the wetting front reached it, despite the very shallow depth to water.
SITE NAME: HA

STUDY AREA: S-SP catchment

SETTING: Lawn in private garden. Beneath deciduous tree canopy (*Melia azederach*).

PERCENTAGE CANOPY:

Western sector : 35
Northern sector : 60

SATURATED HYDRAULIC CONDUCTIVITY:

8 m/d at 660 cm

GRAIN SIZE DISTRIBUTION OF PROFILE:

Moderately well sorted, medium-grained sand throughout, becoming almost a fine-grained sand at 700 cm depth.

IRRIGATION REGIME:

Mains water. Moderate, 35-44 mm/week during the 1981/82 summer, being less during the 1980/81 summer and greater during the 1982/83 summer.

REMARKS:

A soil profile which was irrigated at less than critical rates for deep drainage during 1980/81 but at greater than critical rates during 1981/82 and 1982/83.
SITE NAME: HB
STUDY AREA: S-SP catchment
SETTING: Lawn in private garden
PERCENTAGE CANOPY:

Western sector : 20
Northern sector : 5

SATURATED HYDRAULIC CONDUCTIVITY:

6 m/d at 690 cm

GRAIN SIZE DISTRIBUTION OF PROFILE:

Moderately-sorted, medium-grained sand at the top of the profile grading to a moderately well sorted, fine sand at 700 cm.

IRRIGATION REGIME:

Mains water. 15-20 mm/week. Extra heavy watering during February 1982.

REMARKS:

A profile only 15 m from HA but receiving appreciably less irrigation which resulted in a progressive drying out of the soil profile as the summer progressed (except when uncharacteristic watering regimes were introduced (e.g. 2.82) when the owners were on holiday).
APPENDIX 6

DAWN XYLEM PRESSURE POTENTIALS (BARS) OF TRISTANIA CONFERTA TREES IN THE N-D CATCHMENT

<table>
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<th>Feb</th>
<th>Mar</th>
<th>Apr</th>
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<td>-7.5</td>
<td>-19.0</td>
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</tr>
</tbody>
</table>

(A) UNWATERED TREES

Elevation above water table (m)

(B) WATERED TREES

Elevation above water table (m)

<table>
<thead>
<tr>
<th></th>
<th>Nov</th>
<th>Dec</th>
<th>Jan</th>
<th>Feb</th>
<th>Mar</th>
<th>Apr</th>
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<tr>
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<td>-7.0</td>
</tr>
</tbody>
</table>

* average of two trees.
APPENDIX 7

METHODS OF INSTALLING WATER LEVEL PROBES IN PRIVATE BORES AND WELLS

1. GENERAL

In this appendix, bores refer to systems which have been installed by drilling from the ground surface. Usually 100 or 150 mm diameter PVC casing is run in the drilled hole and a submersible, turbine or jet pump installed. Wells refer to systems in which well liners (about 120 cm diameter) are sunk to within a metre or so of the water table and a spear emplaced below the water table from the bottom of the well. A centrifugal pump is installed on a platform at the bottom of the well. Wells are now installed only when the water table is within 10 or 12 metres of the ground surface, whereas most old systems are wells.

2. BORES

Experience with the installation of water level probes has been gained only with systems using submersible and jet pumps. Turbine pumps pose greater difficulties for access. Fortunately, such pumps are not common among private bore owners (although Local Government Authorities and institutions often use them when watering large areas).

Submersible pumps are usually placed well below the water level within the bore casing and just above the screen. As the water level probe does not extend past the water surface, there is little likelihood of the probe jamming between the pump and the casing. Most private bores use 100 mm PVC casing. Within the casing there is usually a 50 mm polyethylene water return pipe, an electrical cable going to the pump and a safety wire or rope, also attached to the pump. It is necessary to get the water level probe passed all
these obstacles to reach the water surface.

Figure A7.1 shows a section through a typical private bore containing a submersible pump. There are three possible access sites for the probe shown in the figure.

**Site 1:**

It is usually possible to drill a 5/8" (16 mm) hole in the top plate of the bore, far enough away from the electrical and safety cables to allow free access for the probe. The low speed setting on a 2-speed electrical drill is recommended. It is also advisable to drill the hole in two stages (e.g. 3/8" before 5/8"). A magnet attached to the bore plate will prevent drill cuttings from falling down the hole. In some bores, the water return pipe is not central in the casing and this improves access. The drilled hole should be as close to the PVC casing and as far from the water return pipe and cables as possible. Occasionally the electrical cable enters through a large notch in the top plate. The probe should be tried through the notch to see if an extra hole is necessary.

There are different types of fittings used to join the PVC and polyethylene pipes. Some are very large and it is not possible to by-pass them with a hole drilled in the top plate. In such bores, sites 2 and 3 should be investigated.

**Site 2:**

In some bores the casing rises 10 or more centimetres into the asbestos-lined cavity. It is then possible to drill a horizontal hole through the casing below the top plate. Care must be taken not to breach the polyethylene pipe which is only 25 mm from the inside of the PVC. A small, battery-driven drill containing a circular saw-bit is best for such holes, to overcome the problems of limited access.
FIGURE A 7.1. Access sites in a bore for a water level probe
Site 3:

As a last resort it is possible to enter the casing below the asbestos-lined cavity. This requires a hole to be dug beside the asbestos well to expose the casing and a hole drilled through the casing as explained for Site 2. A 45° elbow then needs to be glued to the casing and a PVC pipe taken to the surface to allow the hole to be backfilled. While less convenient than the other two sites, this method will always give access for a water level probe.

3. WELLS

It is not possible to gain access to the spear installed in the bottom of wells because of the presence of a non-return valve. A separate hole must be drilled from the bottom of the well, as far from the spear as possible. Depending on the depth of the water table, a three or four metre length of SWV grade PVC (50 mm) should be slotted and hammered into position using a slip hammer. Once below the water table, the PVC should be bailed with a sludge pump. Jetting in the pipe is quicker if the electrical parts of the centrifugal pump can be covered. If a probe is not left in the well, a funnel should be placed in the top of the PVC to help entry of a probe dangled from the top of the well.

4. CONSTRUCTION OF A SIMPLE WATER LEVEL PROBE

The water level probes consist of a length of flat 2-core cable (CMA 24/0.20 cable has been found suitable) with 2 or 3 barrel lead weights (10 mm diameter x 56 mm length) attached to one of the cores. By leaving a length of cable between the lead weights, the probe remains flexible enough to negotiate corners. Such a probe is inexpensive enough to allow one to be left at each bore and well to enhance accuracy and speed of reading.
APPENDIX 8: Contours of chemical constituents in the groundwater.

APPENDIX 8.1: TDS concentrations in the N-D (above) and S-SP (below) catchments.
APPENDIX 8.2: Sodium concentrations in the N-D (above) and S-SP (below) catchments
APPENDIX 8.3: Potassium concentrations in the N-D (above) and S-SP (below) catchments
APPENDIX 8.4: Magnesium concentrations in the N-D (above) and S-SP (below) catchments
APPENDIX 8.5: Calcium concentrations in the N-D (above) and S-SP (below) catchments
APPENDIX 8.6: Chloride concentrations in the N-D (above) and S-SP (below) catchments.
APPENDIX 8.7: Bicarbonate concentrations in the N-D (above) and S-SP (below) catchments
APPENDIX 8.8: Sulphate concentrations in the N-D (above) and S-SP (below) catchments
APPENDIX 8.9: Silicon concentrations in the N-D (above) and S-SP (below) catchments