



**The influence of drains and perennial pastures on
soil-water-nutrient dynamics of texture contrast soils down a
toposequence at Keyneton, South Australia.**

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Abstract

Land and water degradation issues such as salinity, acidification and eutrophication are all linked to the movement of water through the landscape. All are indications of a system that is no longer in equilibrium, with excess water draining beyond the root system in most agricultural settings, and in many instances leaching significant amounts of nitrate and other nutrients along with it. This study looked at quantifying some of these water and nutrient fluxes in a sloping duplex catchment, and examined the role of drains and perennial pastures in their potential to mitigate degrading processes.

In low and average rainfall years, lucerne was found to produce significantly more dry matter than phalaris and cocksfoot down a typical Adelaide Hills' toposequence. Lucerne roots also penetrated deeper into the subsoil utilising water from a greater region of the soil profile including underlying sandstone. Recharge under lucerne was low (<2% of rainfall) compared to the other two perennials (>10% of rainfall).

The performance of lucerne, in terms of both herbage productivity and recharge reduction, was unexpected on these soil types. The use of lucerne in these environments should be promoted. A perennial legume better adapted to salinity and waterlogging is needed for toe-slope soils.

Drain flows from these pastures were dominated by throughflow, highlighting the importance of lateral flows. Drains removed up to 16% of rainfall in an average rainfall year. Minimal flows were recorded in below average years. Drainage in the soils varied according to toposequence position, with vertical drainage in the upper (reddish) soils, and lateral flow more prevalent in the mid and toe-slope (yellow/grey/white) soils. Drainage from cocksfoot and phalaris was up to twice that from lucerne.

Significant concentrations of nutrients were recorded in drainage water, with DOC and nitrate concentrations higher in throughflow, and phosphorus only slightly higher in overland flow. Nitrate concentrations were highest under lucerne (up to 67 mg/L) and exceeded recommended standards for drinking water. Phosphorus mainly moved in the dissolved state and is therefore likely to enter local streams.

The results have implications for the management of water in these agricultural landscapes. Replacing annual, or shallow-rooted perennial, pastures with perennials like lucerne can reduce recharge to levels more approaching that under native vegetation and at the same time provide graziers with increased pasture productivity. The use of drains can be effective in intercepting surface and through flow on these duplex soils. However, the drainage water contains high levels of nitrate and DOC, and its careful disposal is required if drains are to be a sustainable option in this landscape.

This work contains no material which has been accepted for the award of any other degree or diploma in any university or other tertiary institution and, to the best of my knowledge and belief, contains no material previously published or written by another person, except where due reference has been made in the text.

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Chapter 1

Introduction

1.1 Background

Land degradation has become an increasingly important issue in rural Australia, from a social, economic and environmental perspective. Dryland salinity, acidification and waterlogging are all processes operating in our agricultural environment, reducing production, and incurring opportunity costs and the expense of remediation. In the Keynes catchment, Mt Lofty Ranges, South Australia, waterlogging and increasing salinity were identified as issues of concern to the property owners in the early 1990s and Landcare activity commenced there with a trial of salt tolerant plants in the worst affected, low productive soils. The Landcare group then sought advice on best management options for the most productive parts of the farm. These options included alternative pastures that use more soil water, and therefore reduce deep drainage to shallow saline watertables



Plate 1.1 Landcare project in Keynes catchment, Keyneton, SA.

1.2 Aims and Objectives

This study commenced early in 1995, and data collection ceased in early January 1998. Specifically this research project aimed to;

1. compare the variability in some of the soil physical and chemical properties that may affect pasture development and the water balance of the texture contrast soils which occur in the catchment,
2. measure runoff, throughflow and changes in stored soil water as components of the water balance of the texture contrast soils, at different positions in the landscape under various pastures,
3. model all components of the water balance of the texture contrast soils at different positions in the landscape under various pastures, and
4. determine if the sowing of new pasture species changes the chemical concentrations and loads in runoff and drainage waters.

1.3 Thesis layout

The thesis presents a review of the recent literature relevant to each research area, then describes the methodology of the research and discusses the subsequent results. Due to the breadth of the study, the thesis deals with each research issue separately, then summarises all the results in the final chapter.

Chapter 2 is the primary literature review, dealing with the background of land degradation in southern Australian farming landscapes. It includes soil hydraulic features which can influence plant development and water use, texture contrast soils and their variable chemical and physical properties, through to a discourse on drainage, the ecology of different perennial pastures, and the methodology of root measurement.

Field trial layout and management are in Chapter 3. This chapter deals with the management of an established pasture commonly sown by farmers in the area, and the establishment of two alternative pastures. It also describes the hydrological isolation of the three pasture types using a series of instrumented drains for measuring specific components of the water balance, and the monitoring of changes in soil water content below the pastures using a neutron moisture meter.

In Chapter 4 the results of the above research are presented and discussed. This section describes the soils along the toposequence and looks at their variability. It also presents some of the soil physical parameters and shows the heterogeneity present on-site. Drain flows are compared spatially and temporally, and their performance discussed.

Chapter 5. This chapter reports on the rooting depth and productivity of the three pasture types and considers these as influences on components of the water balance. Neutron probe data over an 18-month period is presented graphically to support assertions on water use, in conjunction with root length density data from root washing work. Two methodologies that estimate evapotranspiration are described and compared, and deep drainage is calculated and compared with other studies for each pasture type.

Chapter 6 includes a literature review of recent material on water and associated nutrient movement from agricultural landscapes. It then discusses the concentrations and loads of chemicals in runoff (overland flow and throughflow) from the pastures, and discusses the off-site implications of introducing new pasture species to a landscape that has had a cocksfoot (*Dactylis glomerata* L.) and subterranean clover (*Trifolium subterranean* L.) based pasture for 120 years.

Chapter 7 firstly reviews some of the recent literature on water balance modelling and then describes the particular model (TOPOG) used to simulate the water balance in this study. It also presents the results of simulations to validate the model and discusses the outcomes of this in terms of model complexity and suitability of input data.

The final chapter, Chapter 8 is a summary that shows how extreme spatial variability and “harshness” in soil physical and chemical properties can affect both pasture growth and development and the water balance. Despite this variability, some clear trends emerged which could help landholders in this environment in their choice of options for land management that could provide both an economic and environmental benefit.

Chapter 2

Literature Review

2.1 Introduction

This chapter considers the background to the relevant degradation processes operating within the study catchment. It also reviews the current knowledge of the peculiarities of farming duplex soils and the impact of their hydrology on plant growth and development. As well, the processes (both mechanical and physiological) involved in hydrological management, particularly as they have been applied in an Australian context are discussed.

Given the breadth of this research project, this review could become unwieldy if it covered all aspects of the study. For this reason, specific issues such as the impacts of perennial pastures on runoff water quality, as well as modelling perennial water use, have only been discussed briefly in this chapter. These issues will be dealt with in self-contained chapters, and the literature relating to them will be reviewed at that time.

2.2 Background

Land degradation is widespread in the agricultural regions of Australia (Johnston 1992), affecting vast areas of potentially productive land. In WA 1,800,000 ha is salt-affected and it is predicted that this will rise over the next two decades to 6,000,000 ha (Berry 1997; Bennett 1998). Bennett (1998) reports that there are >400,000 ha of land affected by dryland salinity in SA, predicted to rise to 600,000 ha in the next few decades (see Table 2.1). Recent figures for the Murray-Darling Basin (MDBC 1999), suggest an increase in salt-affected land from 300,000 ha in 1996, to 9,000,000 ha (previous estimate 6,000,000 ha) before a new hydrologic equilibrium is reached.

Although evident in many forms (eg. dryland salinity, waterlogging, erosion), most of this degradation is the result of past widespread clearance of the native vegetation (Hatton *et al.* 2002) leading to changes in local hydrological regimes as deep rooted evergreen perennial vegetation was replaced with mainly annual crops and pastures (Williamson 1990; Gardner *et al.* 1992; Greenwood *et al.* 1992; Saunders and Hobbs

1993). As well as the environmental consequences, these changes in the water balance have significant economic impacts including losses in yield and opportunity costs in land lost to production. Berry (1997) suggests that the capital value of WA's current land loss is c. \$1.5 billion. The estimated annual cost of dryland salinity in Australia is \$270 million (Bennett 1998), with \$130 million in lost agricultural production, \$100 million in infrastructure damage and at least \$40 million in lost environmental assets.

Table 2.1 Area of land affected by dryland salinity in Australia. Adapted from Bennett (1998).

State	Area salt affected in 1996 (ha)	Potential area affected at equilibrium* (ha)
Western Australia	1,804,000	6,109,000
South Australia	402,000	600,000
Victoria	120,000	unknown
New South Wales	120,000	5,000,000
Tasmania	20,000	unknown
Queensland	10,000	74,000
Northern Territory	minor	unknown
Total	2,476,000	>11,783,000

* The potential area affected at equilibrium is the likely area to be affected if no remedial action is taken.

On shallow texture contrast or duplex (Northcote 1979) soils which cover approximately 20% of the Australian continent (Chittleborough 1992), these problems are exacerbated by the development of rapidly fluctuating perched water tables on relatively impermeable clay B horizons (Gardner *et al.* 1992; MacEwan *et al.* 1992; Fritsch and Fitzpatrick 1994; Cox and McFarlane 1995). Any salt stores in deeper horizons, the regolith or groundwater that have been mobilised to the extent that they discharge above this horizon, may be redistributed downslope by these perched water tables (R. MacEwan, pers.comm.).

In other studies (Weyman 1973; Greacen and Williams 1983; Cox 1988; Hatton *et al.* 2002) on sloping texture contrast soils, it has been shown that significant quantities of water travel as interflow (or throughflow) downslope along the A/B horizon interface. This leads to the increased probability of waterlogging on the lower slopes (Prebble and

Stirk 1988) or salinity in the gully bottoms as saline groundwater rises with the increase in recharge (Saunders and Hobbs 1993).

Additionally, it has been suggested that throughflow water may be a significant transport mechanism for dissolved organic matter and metal ions (Fitzpatrick *et al.* 1992; Naidu *et al.* 1993), contributing to water quality problems in the catchment. Smettem *et al.* (1991) note the pollution of water storages in southern Australia, by suspended clays from non-point sources in many catchments. However, even though duplex soils are widespread throughout these catchments, “information on current hydrogeological processes that affect colloid and solute transport is scarce” (Smettem *et al.* 1991). All these matters raise concerns over the sustainability of current farming systems and practices in this environment (see Chapter 6).

2.3 Soil hydrology

Texture contrast soils have a variety of problems/limitations (eg. erosion, low fertility, waterlogging) that are a consequence of their hydrological properties (Hook 1990), and influenced by management practices carried out on them. To enable recommendations to be made to improve on current management practices, there is a need for a greater understanding of the hydrological functioning within these soils, a knowledge of the vertical and horizontal movement of water and the changes in storage of the different soil horizons.

The quality of water in the catchment is inexorably linked to the hydrologic processes operating within it. Movement of naturally occurring or anthropogenically derived chemicals is possible through the transporting function of water on and through the soil, and their distribution and ultimate condition is dependent on bio-geochemical processes operating within the soil (Kutilek and Nielsen 1994).

To better understand and describe soil hydrological processes, there is a primary need to analyse and quantify the fundamental processes of water movement (infiltration, evapotranspiration, redistribution, and drainage) and storage. These operate under field conditions imposed by the local meteorological situation, landscape position, vegetation and soil hydraulic properties (Kutilek and Nielsen 1994). From pedon studies

(macroscopic scale), it is becoming common to extrapolate to the catchment (megascopic) scale through computer simulation modelling (Kutilek and Nielsen 1994).

However, due to the degree of spatial variability of soils and their hydraulic properties, and hence the number of replicate measurements required, Jaynes and Tyler (1984) report that there is widespread research into characterising those soil properties which may act as surrogates for determining hydraulic conductivity.

2.3.1 Soil permeability and hydraulic conductivity

The permeability of a soil describes its capacity to conduct gases and fluids, especially water (Charman and Murphy 2000). Given the close relationship between the movement of water through soils and a range of land degradation issues (eg. salinisation, soil acidification), permeability is an important soil property.

Hydraulic conductivity (K) measures a soil's permeability to water. It is a measure of the rate of water movement in response to a potential gradient, either gravimetric, matric, or potential (Phil Ward, pers.comm.). The rate of viscous flow of water in isotropic porous media is proportional to, and in the direction of, the hydraulic gradient (Darcy's Law, named for Henry Darcy of Dijon, who formulated it in 1856 from extensive work on the flow of water through sand filter beds). As formulated by Darcy the law is:

$$Q = k \cdot S \cdot (H + e) / e$$

where

Q is the volume of water passed in unit time,

S is the area of the bed,

e is the thickness of the bed,

H is the height of the water on top of the bed, and

k is a coefficient (depending on the nature of the sand) and for cases where the pressure under the filter is equal to the weight of the atmosphere (Kutilek & Nielsen 1994).

K is dependent on soil structure and in particular on the geometry and continuity of water-filled pores (Jaynes and Tyler 1984; Charman and Murphy 2000). As the conductivity of a pore is related to the 4th power of its radius (Poiseuille's Law), K is particularly dependent on the frequency and distribution of larger pores.

K can be expressed as a function of soil water content, $K(\theta)$ or the associated soil water potential $K(\psi)$ (Jaynes and Tyler 1984). In a saturated soil all pores are filled allowing the maximum flux of water (K_s). However as the soil dries (decreasing water potential) the larger pores begin to empty of water and fill with air, and K declines as soil water follows a longer and more indirect path.

There are a number of methods used to measure K, and in the field K_s can be determined using permeameters (Perroux and White 1988; Talsma and Hallam 1980), infiltrometers and piezometers among other techniques (McMurray 1994). Laboratory measurement of K_s is usually conducted on intact soil cores and determined by measuring the water flux across a saturated core. In their evaluation of DRAINMOD in south-west Australia, Cox *et al.* (1994) demonstrated the importance of K_s to allow adequate modelling simulations. However they state that no method "may adequately combine the effects of soil heterogeneity and anisotropy for predicting a true field K_s " for their soils.

As with many other soil properties, K is highly variable over space and time (Jaynes and Tyler 1984; Charman and Murphy 2000). Spatially this variability occurs laterally (eg. the result of differences in compaction or vegetative cover) and generally with depth (usually a decline with depth, although compaction may impact near the surface). Temporally, K may vary over a growing season due to soil swelling (Phil Ward, pers. comm.) or as plant growth and the activity of soil fauna generates macropores (Murphy *et al.* 1993). Whilst it is apparent that significant differences in K_s values may result from some of the above methods (McMurray 1994), Bouwer and Jackson (1974) state that (the variability of) the soil is a much greater source of inaccuracy than errors associated with the methods themselves.

The high degree of spatial variability in K has led to studies that attempt to predict it by correlation with other soil properties (Jaynes and Tyler 1984; McKenzie and Jacquier 1997). In the Australian context where there is no database of matching hydraulic and laboratory data (McKenzie and Jacquier 1997) a need has been recognised that prediction of hydraulic properties is necessary. This could be achieved by developing correlations between hydraulic properties and the morphological variables commonly used in mapping.

These studies, whilst of mixed success in the past, have often highlighted the difficulty in transferring results from one soil (or range of soils) to another (MacEwan *et al.* 1992; McKenzie and Jacquier 1997). Charman and Murphy (2000) suggest that these predictive methods will have difficulties “encompassing the wide variability of K_s in the field” and support Gray (1973) in advocating direct measurement for any critical application.

McKenzie *et al.* (1991) evaluated a number of schemes for predicting air and water properties of soils, using soils from the mid-north of South Australia. They found that none of the systems provided statistically significant predictions of available water capacity and results for air-filled porosity were moderate ($R^2 = 0.58-0.70$). They were however encouraged by the Hollis and Woods (1989) prediction of field saturated hydraulic conductivity ($R^2 = 0.77$), particularly as that system was developed on a limited number of UK soils. Based on UK criteria, it included estimates of pore size and distribution whereas in the Australian system, they note that structural description focuses on the solid component with only an incomplete description of the pore space that has more influence on the water-flow properties of the soil (McKenzie *et al.* 1991).

More recently, McKenzie and Jacquier (1997) have identified that statistically significant relationships exist between soil morphological properties observable in the field and K_s , however individually these relationships are noisy and do not provide a good prediction of K_s . They found that the complex interactions they identified between field texture, grade of structure, bulk density, aggregate stability and macroporosity were better presented in a tree-based model using simple morphological predictors, and showed a reduction in variance by about 70%.

Particle size

A number of studies have attempted to correlate soil texture and particle size distribution with K_s (eg. Gupta and Larson 1979; Burch *et al.* 1987; McKenzie and MacLeod 1989; Espeby 1990; McKenzie and Jacquier 1997). Schuh and Bauder (1986) found that texture data alone accounted for c.50% of the variability in K_s (as a function of soil water suction S) in multiple regression equations based on physical data. When using only particle size data, best fit predictive models varied from measured K_s by a factor of 2 to 3 at the 60% confidence interval and 7 to 12 at the 95% level, indicating a lack of precision in any modelling based on this data.

In this same study (ie. Schuh and Bauder 1986), analysis of regression correlation coefficients showed that clay percentage “provided the single most significant and most consistent correlation with K_s of all physical variables”. For sand, higher content was significantly related to higher conducting porosity at 1.5 kPa suction, but to lower conducting porosity at 5.0 kPa suction. Near the crossover point around 3.0 kPa, there was no significant relationship with K_s . Conversely, Jaynes and Tyler (1984) developed an equation based on only the soil sand content that described 83% of the hydraulic conductivity variation, and Pershinger and Yahner (1970) also found a positive correlation between sand content and percolation rate.

In Schuh and Bauder (1986), silt percentage was also variable at different suctions (as with the sand example above) and this supports the observations of Nielsen *et al.* (1973) “that silt is not related to steady-state infiltration”. In contrast to this, Jaynes and Tyler (1984) report regression coefficients showing significant correlation at the 99% confidence level, as did Derr *et al.* (1969).

McKenzie and Jacquier (1997) found that the main feature in their study was the amount of overlap in K between all texture classes. Whilst the sand and clay classes at either end of the spectrum were distinct, there were still some clays that were as conductive as the highest value sand. McKenzie and MacLeod (1989) also concluded that field textural classes could only provide a general guide to the proportions of each particle size grade.

Bulk density

Bulk density is inversely related to soil porosity (Marshall, Holmes & Rose 1996). Schuh and Bauder (1986) found that bulk density was a poor predictor of K_s on soils with varying textures, minerals and organic matter, and suggested that bulk density was primarily an indicator of grain size. Jaynes and Tyler (1984) however, developed regression coefficients for bulk density that were significant at the 99% confidence level, although this was in coarse-textured C horizon soils and may not, in their opinion, be adequate for finer-textured soils.

Whilst Schuh and Bauder (1986) and Jaynes and Tyler (1984) found varying degrees of correlation with porosity, the uncertainty evident in these and other earlier contradictory studies (eg. Bouma and Hole 1971), limit the use of these physical properties as estimators of hydraulic conductivity. Given the extreme heterogeneity evident in soils, Nielsen *et al.* (1973) suggest that the degree to which these physical properties influence hydraulic properties is itself highly variable and inadequate as a reliable predictor for modelling flow processes.

McKeague *et al.* (1986) proposed that reliable estimates of bulk density could be made from simple descriptive guidelines, although McKenzie *et al.* (1991) could not support this. They did however find a reasonable relationship with their 'calliper' method ($r^2 = 0.89$) involving the weighing of a shaped clod on an electronic balance and measuring its geometric mean diameter with callipers. McKenzie and Jacquier (1997) demonstrated a trend of lower K_s at higher bulk densities but the relationship was noisy and generally of low value for any prediction. Hugh Brown (pers. comm.; unpublished data from the Keyneton catchment) also reported that regression of K on bulk density showed a non-significant trend of lower K with increasing bulk density.

2.3.2 Soil porosity

Soil porosity is an expression of the relative volume of pores, that part of the soil space "not filled by the soil solid phase" (Kutilek and Nielsen 1994)

$$P = V_P / V_T \quad \text{Eq. 2.1}$$

where V_P is the volume of pores and V_T the total bulk volume of soil. Porosity is expressed as a percentage (see Table 2.2) by multiplying P by 100.

Table 2.2 Range of porosity and bulk density in mineral soils (from Kutilek and Nielsen 1994, p.19)

Type of soil	Porosity (P) %	Bulk Density (ρ_T) g/cm ³
sandy soils	56 - 36	1.16 - 1.70
silty soils	56 - 39	1.26 - 1.61
loamy soils	55 - 30	1.20 - 1.85
clayey soils	70 - 35	0.88 - 1.72

Porosity (P) generally lies within the range of 0.4 and 0.6, however the wider range is from < 0.3 to > 0.9 . The theoretical minimum $P = 0.26$ represents the porosity of the most dense packing of identically sized spherical particles (Kutilek and Nielsen 1994), the theoretical minimum porosity of sands. As clay content increases, so too does soil porosity, with a theoretical maximum P in loam soils due to optimum aggregation potential. Agricultural practices however have generally led to soil compaction, and porosity may not be determined solely by texture (Kutilek and Nielsen 1994).

Pores may be classified as (a) *sub microscopic*, ($< 0.05 \mu\text{m}$) in which pores are so minute that water molecules cannot cluster sufficiently to produce a continuous flow path, (b) *capillary*, in which capillary menisci form and the dominant flow of soil water occurs, and (c) *macropores* ($> 5 \mu\text{m}$), which are large enough to preclude the formation of capillary menisci, and in which flow may be as a film or at times turbulent. Although their relative volume may be small, their capacity to transmit water and dissolved substances may be far greater (Kutilek and Nielsen 1994).

2.3.3 Soil water content

The soil water content is an expression of the liquid phase of a soil (Kutilek and Nielsen 1994) and is defined either volumetrically (Equation 2.2)

$$\theta = V_W / V_T \quad \text{Eq. 2.2}$$

or gravimetrically (ie. by mass) (Equation 2.3)

$$\omega = m_W / m_S \quad \text{Eq. 2.3}$$

where θ = volumetric soil water content, V_W is the volume of water, V_T the volume of the intact soil sample, ω = gravimetric soil water content, m_W the mass of water and m_S the oven dry mass of soil.

Table 2.3 Symbols related to soil water content (from Kutilek and Nielsen 1994 p. 28)

	solid phase	soil water	soil air	whole soil
volume	V_S	V_W	V_A	V_T
mass	m_S	m_W	m_A	m_T
particle density			$\rho_S = m_S/V_S$	
bulk density			$\rho_T = m_S/V_T$	
porosity			$P = (V_W + V_A)/V_T$	
volumetric soil water content			$\theta = V_W/V_T$	
gravimetric soil water content			$\omega = m_W/m_S$	
soil water ratio			$v = V_W/V_S$	
volume of pores			$V_p = V_W + V_A$	

Volumetric water content is the volume of water held at a particular time by a certain volume of soil. It can be expressed as a fraction or a percentage of the volume or as a depth of water per depth of soil (eg. mm/m) (see Table 2.3 for some of the relationships between soil hydrological properties). Soil water content can be measured destructively by taking soil samples for drying and gravimetric analysis, or non-destructively using gypsum blocks, capacitance probes, or a neutron moisture meter.

2.4 Duplex soils and Australian agriculture

2.4.1 Definition

Chittleborough (1992) reviewed the formation and pedology of duplex soils in Australia, examining a range of hypotheses on their genesis. Northcote (1960), first introduced the term 'duplex' as a primary diagnostic of soil profiles, with his three mineral soil Primary Profile Forms being Duplex, Gradational and Uniform. Although a uniquely Australian term, 'duplex' soils occur in other countries, mainly in those areas exhibiting strong cyclic wetting/drying climates (Chittleborough 1992). 'Duplex' and 'texture contrast' are used interchangeably throughout this work. However it should be

noted that soil scientists are tending to favour 'texture contrast' to reduce overseas confusion with the Australian usage of 'duplex'.

A duplex soil has a contrast of one and a half (or more) texture groups between the surface and subsurface horizons, and a sharply defined horizon boundary (Northcote 1960). Under the classification system based on Northcote's (1960) 'A Factual Key for the Recognition of Australian Soils', these two textural properties alone determine the classification of 'Duplex,' despite a diversity of other physical, mineralogical and chemical properties. Chittleborough (1992) suggests that most of Australia's duplex soils fall within the "Red-Brown Earths, Podsolis, Solonetz and Soloth Great Soil Groups" (Stace *et al.* 1968), and given their diversity there are various hypotheses relating to their genesis and development, from geological to biological (eg termite bioturbation - Lee and Wood 1971; Lee 1983 in Chittleborough 1992).

Firman (1969) cited in Chittleborough (1992), suggests that sedimentary layering was the origin of the A and B horizons of the widespread red and yellow duplex soils of South Australia. Firman's conclusion is based on Quaternary geological mapping of the southern-central region of South Australia, much of which is covered with yellow and red duplex soils. Given the relationships shown to exist between soil type and the underlying parent material, Chittleborough (1992) notes that "it is more plausible" that the textural contrast is due to soil forming processes occurring *in situ*, rather than a geological origin.

Earlier work by Chittleborough *et al.* (1984) on a number of chronosequences demonstrated that sedimentary layering could not explain the texture contrast in the oldest sequences. Clay translocation is of primary import in texture contrast soil development but *in situ* weathering "becomes progressively more important with time" (Chittleborough *et al.* 1984). Fritsch and Fitzpatrick (1994), introducing their conceptual soil-water-landscape model of a Typic Palexeralf-Natraqualf soil sequence (Soil Survey Staff 1992) in the Mt Lofty Ranges, indicate the importance of ancient and modern hydrological processes and conditions in the formation of soil domains (eg. lateritic, hydromorphic) within the overall soil sequence.

Chittleborough (1992) concludes that there is no single criterion (or set of criteria) that will distinguish between the various hypotheses put forward to explain the complex genesis of duplex soils. He states that greater confidence can be placed on studies where the hypotheses are tested against a range of diverse criteria.

2.4.2 Distribution of duplex soils

Duplex soils are widespread throughout Australia and have developed over a range of metamorphic, sedimentary and igneous parent materials (Chittleborough 1992). They occur on flat terrain to relatively steep sloping landscapes and cover about 20% of the continent (Chittleborough 1992), underlying much of Australia's settled area. Greacen and Williams (1983) suggest that the widespread development of these hydrologically complex soils in the Australian landscape is the result of a combination of our geologically ancient soils and long periods of semi-aridity.

In south-eastern Australia, duplex soils dominate, with two main exceptions; the uniform and gradational clays (Wimmera soils) in a wide band north of Horsham (Vic), and the gradational sands and loams of the Mallee soils in north-western Victoria and the east of South Australia (Gardner *et al.* 1992). South Australian duplex soils include the red-brown earths, and those with a bleached sub-surface such as podsoles and solonetz where dryland salinity is increasingly evident (Gardner *et al.* 1992). In the Mt Lofty Ranges (where this study is located), duplex soils formed from the Kanmantoo metasediments (see Fig. 2.1) are prone to both waterlogging and salinity (Fitzpatrick *et al.* 1997).

2.5 Aiming for a sustainable water balance on duplex soils.

It is widely acknowledged that recharge under most introduced crops and pastures is significantly greater than that which occurs under natural vegetation (Kennett-Smith *et al.* 1993; Cox *et al.* 1994; Richardson and Narayan 1995; Ridley *et al.* 1997). This inability of annuals to use available water as well as perennials is a function of the seasonality of the growth phase, less rooting depth (eg annual weeds 25-60 cm, subclovers 40-50 cm, perennials >120 cm) and environmental limitations to transpiration and growth (Crawford and Macfarlane 1995). As a consequence, agricultural soils have increased quantities of water available for recharge.

It is apparent then, that catchment water balances need to be manipulated to return the system to a more sustainable basis (Gregory *et al.* 1992; O'Loughlin 1992). The simplest water balance is:

$$\text{Inflow} = \text{Outflow} \pm \text{Changes in Storage}$$

where Inflow includes precipitation, the upward capillary flow, runoff (surface and lateral flow). Outflow includes canopy interception, surface runoff, lateral flow (throughflow or interflow), soil surface evaporation, transpiration and vertical drainage. There is a need to reduce the components of runoff (surface and lateral flow) and deep drainage whilst increasing evapotranspiration, and the options for achieving this may include artificial drainage or changed agronomy. Engineering works (drains) physically modify the pathways and storages of water in the landscape whilst agronomic approaches intercept and evaporate/transpire rainfall and soil water, before it drains to the water table (O'Loughlin 1992).

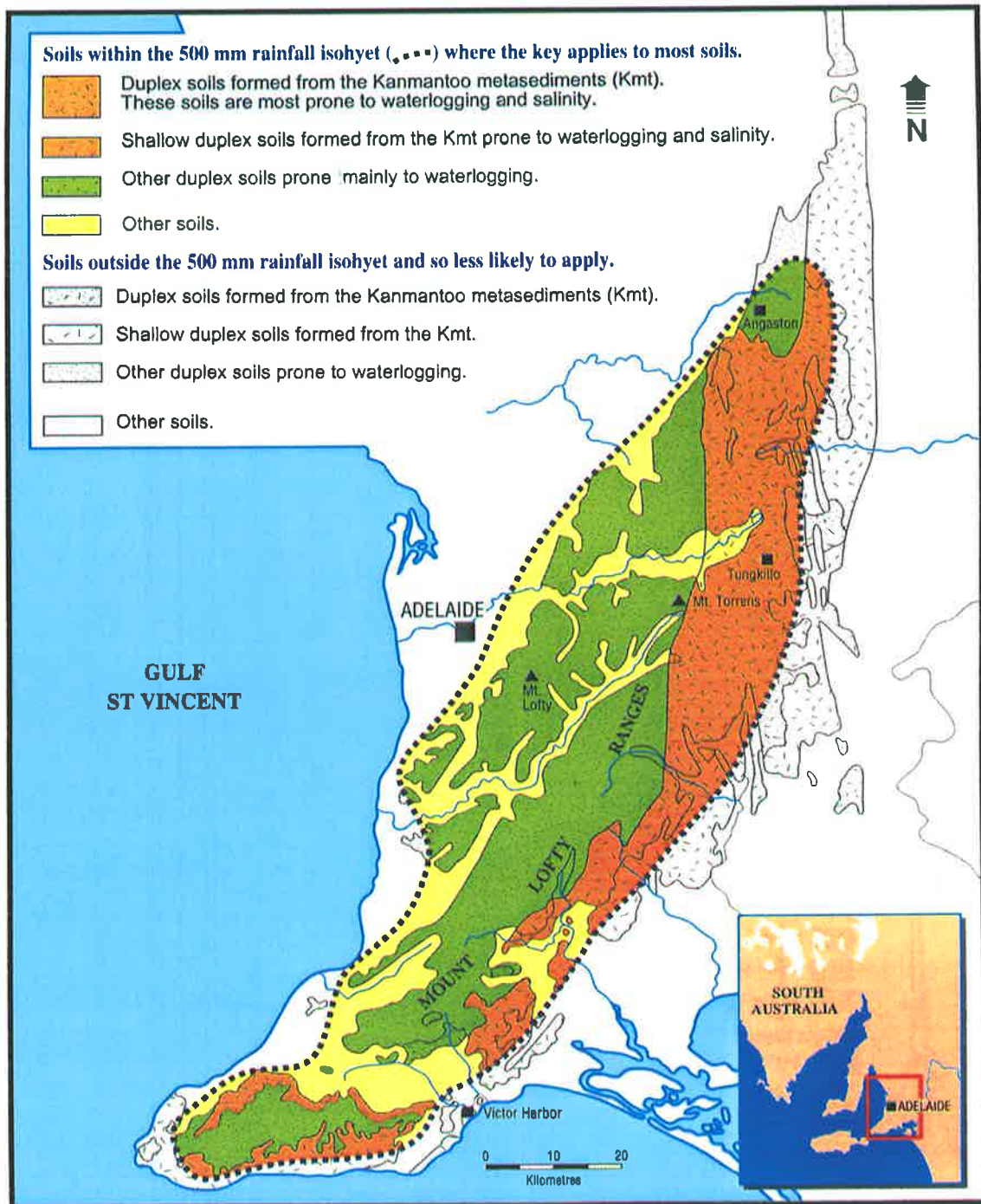


Fig. 2.1 Duplex soil distribution in the Mt Lofty Ranges, South Australia (adapted from Fitzpatrick *et al.* 1997). Note: soils “within the 500 mm rainfall isohyet” receive >500 mm rainfall, those “outside” receive <500 mm rainfall.

2.6 Hydrology of duplex soils and implications for agriculture

The hydrological behaviour of duplex soils is influenced by their morphology. Both texture and structure affect the basic hydraulic properties of the profile (Greacen and Williams 1983), with a dominant influence being the sudden textural change of the clay B horizon (Prebble and Stirk 1988). The extent of this influence is then a function of the hydraulic differences (absolute and relative) across the A/B horizon interface (Greacen and Williams 1983).

Passioura (1992) noted the serious physico-chemical problems associated with duplex soils, in particular those contributing to the soil water balance. With the low hydraulic conductivities associated with the B horizon, and an excess of winter rainfall over evaporation, perched water tables and waterlogging develop leading to reduced plant growth from crops/pastures with depauperate root systems (Setter and Belford 1990). However, even without root damage associated with waterlogging, the high bulk densities of the B horizon, combined with possible sodicity and/or other chemical problems, reduce the likelihood of root colonisation other than within old root channels or other macropores (Dracup *et al.* 1992).

Edwards (1992) discusses the “irony” with duplex soils, where perched water tables cause early waterlogging of the A horizon leading to shallow rooted plants that are then unable to reach potential plant available water at the end of the season to fully realise the crop’s potential. This phenomenon of an excess of water followed by a shortage within a relatively brief time span (Turner 1992) resulting in plant water deficits developing rapidly as spring rains decrease, is mainly the result of the low permeability of the B horizon caused by massive, prismatic or columnar structure and low drainable pore space.

Tennant *et al.* (1992) argue that permeability is the soil property most impacting on duplex soil behaviour, severely restricting water entry and root penetration (Belford *et al.* 1992; Tennant *et al.* 1992). Tennant *et al.* (1992) feel that the effect (of permeability) on soil function is such that problems are more those of ‘permeability contrast’ than ‘texture contrast’.

Prolonged waterlogging at critical growth phases has been shown to reduce Victorian grain yields substantially (Gardner 1995). Wheat generally showed the greatest sensitivity to waterlogging early in the season when small plants were less able to supply sufficient oxygen to the plant's root system (Belford *et al.* 1990). In WA, wheat yields on duplex soils were shown to be partially correlated with the extent (height) and duration of waterlogging (Belford *et al.* 1992) and Belford *et al.* (1990) found that waterlogging accounted for one half of the wheat yield variation at a Beverley (WA) site.

McFarlane and Wheaton (1990) showed a similar influence of waterlogging on pasture growth in the Upper Great Southern (WA), where growth was affected by the timing and extent of the waterlogging event. In mid-winter they found no difference in growth between waterlogged and non-waterlogged areas, probably influenced by slow growth responses to low temperatures. Generally, severe waterlogging decreased growth throughout the season, and moderate waterlogging whilst decreasing winter growth tended to compensate with increased spring growth. However, early winter pasture has a greater marginal value (McFarlane and Wheaton 1990) therefore this early loss was significant. Only in normally dry areas of the landscape was *mild* waterlogging shown to increase pasture growth.

They also found that waterlogged areas showed compositional changes, with waterlogged areas having a lower clover to grass ratio than non-waterlogged areas. Depending on the actual composition of the pastures, this could have nutritional and/or economic impacts. If the pasture is there to build up soil fertility then obviously the higher the clover composition the better, and for animal production, Rossiter (1978) feels that clover levels <30% may start to impact through effects on pasture yield. Taylor and Clifton (1993) recommend a balance of 50% perennial grasses and 50% clover (legume) for optimum stock performance.

Waterlogging is often associated with salinity, and the interaction of these factors increases the risk of adverse effects on plant growth more than either factor individually (Barrett-Lennard *et al.* 1990; Belford *et al.* 1992). Waterlogging greatly lowers available oxygen in the plant root zone, leading to anaerobic respiration of the reserves of

carbohydrates stored within the roots. This action leads to a 95% reduction in energy production and a compensatory reduction in growth and water uptake. In saline conditions it also leads to an increased salt uptake and a diminishing likelihood of survival (Barrett-Lennard *et al.* 1990) as plants “lose their ability to exclude sodium and chloride at the surface of the roots” (Belford *et al.* 1992).

Passioura (1992), in reviewing the difficulties inherent in farming duplex soils, spoke of the need to quantify the issues and impacts of nutrient leaching, throughflow, runoff, and drainage to groundwater, and assess the impact of various management impacts on them. “Until we can readily quantify these losses, we will remain ignorant of the long term consequences of our management practices” (Passioura 1992).

2.7 Hydrological management of duplex soils

2.7.1 Introduction

Hydrological management is based on the concept of manipulating the water balance of the catchment (O’Loughlin 1992). Physical modification of storage areas and flow paths can be achieved with deep ripping, which has variable results (Brouwer and van de Graaff 1988; Belford *et al.* 1992; McFarlane and Cox 1992; Brouwer 1989; Gardner 1995), as does the application of ameliorants such as lime or gypsum (Belford *et al.* 1992). Drain construction is probably the most ‘engineered’ physical modification of flow paths, whilst biological methods are usually aimed at changing the fluxes to evapotranspiration, and drainage to the water table.

2.7.2 Drains

Reverse interceptor

In temperate Australia, surface runoff from most duplex soils (see Talsma 1969, cited in Greacen and Williams 1983, p. 523) is unlikely under ‘normal’ rainfall intensities. However, once the A horizon is saturated, or near saturation, runoff can be expected (Greacen and Williams 1983; Prebble and Stirk 1988). Gregory *et al.* (1992) comment on the variability on and between sites, of run-off and lateral flow and whilst recognising that substantial lateral flows may occur, state that the dominant direction of flow is vertical. In other studies most flow is assumed to move laterally as throughflow

along the 'throttle' imposed by the texture contrast of the B horizon (Greacen and Williams 1983; Prebble and Stirk 1988; Gardner *et al.* 1992; McFarlane and Cox 1992; Fleming and Cox 1998) often coming to the surface at saturated areas on the lower or foot slopes, or elsewhere through preferential flow paths (Greacen and Williams 1983).

Shallow drainage (eg. seepage interceptor drains, Cox and McFarlane 1995) is now recognised as an important management strategy that can be adopted at an early stage in the remediation of waterlogged soils. Drainage can form part of an overall strategy for sustainable farm management in catchments with texture contrast soils (McFarlane and Cox 1992), encouraging deeper plant root growth and greater production which in turn increases transpiration and reduces potential recharge to deeper saline groundwater.

McFarlane and Cox (1990, 1992) reviewed the effectiveness of three interceptor drains (reverse, conventional, WISALTS) (Fig. 2.2). on duplex soils in WA. They found that the percentage of annual rainfall intercepted by the drains varied from 1.1% to 7.3% in areas of 370-470 mm annual rainfall, and up to 18.8% in a higher rainfall area (535-710 mm). The reverse interceptor appeared to have least ongoing management problems (eg. silting, rilling of batters) and could be established at a higher gradient as erosive storm waters are not carried in the grader-cut channel (McFarlane and Cox 1992). Using a financial spreadsheet program, Bathgate and Evans (1990) showed that for a property in the Denbarker region, west of Albany (WA), an increase in the stocking rate of 0.4 DSE/Ha was needed for reverse interceptors to be profitable, and that such an increase (<4%) in production was not an unreasonable expectation.

Seepage interceptor drains will not solve all the problems associated with excess soil water and may not be suitable for all texture contrast soils. There was also some concern expressed by the landowner, about the amount of land used by reverse interceptors (see for instance Fig. 3.3) and the difficulty imposed by them in managing stock movements using motor cycles. MacEwan *et al.* (1992) commented on the potential maintenance of 1 km/ha of surface drains and other management issues as militating against the exclusive use of surface drains. Buried tube drains (for throughflow) with contour banks for overland flow were a combination that resolved the landholder's perceived difficulties.

Contours and drains

In rural South Australia, contour banks are the most common earthworks designed to collect rainfall runoff and direct it to a suitable position in the landscape. Drains *per se*, are less common and usually on relatively flat country. Such drains are relief drains whereas those situated on slopes are termed interceptors (van Hoorn and van der Molen 1974). A relief drain is generally a means of lowering a high watertable on a floodplain or similar flat ground. Interceptor drains however, capture and transport water from upslope areas with the aim of protecting lower areas in the landscape. Drains may be designed to collect both overland flow and subsurface flow (ie. open or surface drains) or subsurface flow only (ie. buried or subsurface drains).

Buried agricultural drain

In a Victorian study (Gardner 1995), subsurface drainage was shown to have a greater positive effect on wheat and triticale yields than deep ripping on shallow duplex soils. However due to the significant costs involved this is unlikely to be feasible on non-cropping (or less productive) land. In contrast, Brouwer and van de Graaff (1988), again in Victoria, compared conventional tillage and deep ripping on the yields of phalaris and lucerne pastures, and on water movement in the duplex soil beneath them and showed significant yield increases for both pasture types as well as reduced waterlogging. Deep ripping in this case however, increased flows to preferential downward flow channels, leading to greater recharge of the groundwater, exacerbating the salinity potential.

As deep ripping responses seem to be site specific (McFarlane and Cox 1992), subsurface drains may be a better environmental option as they decrease the magnitude of recharge whilst removing large volumes of excess winter rainfall (Gardner *et al.* 1990).

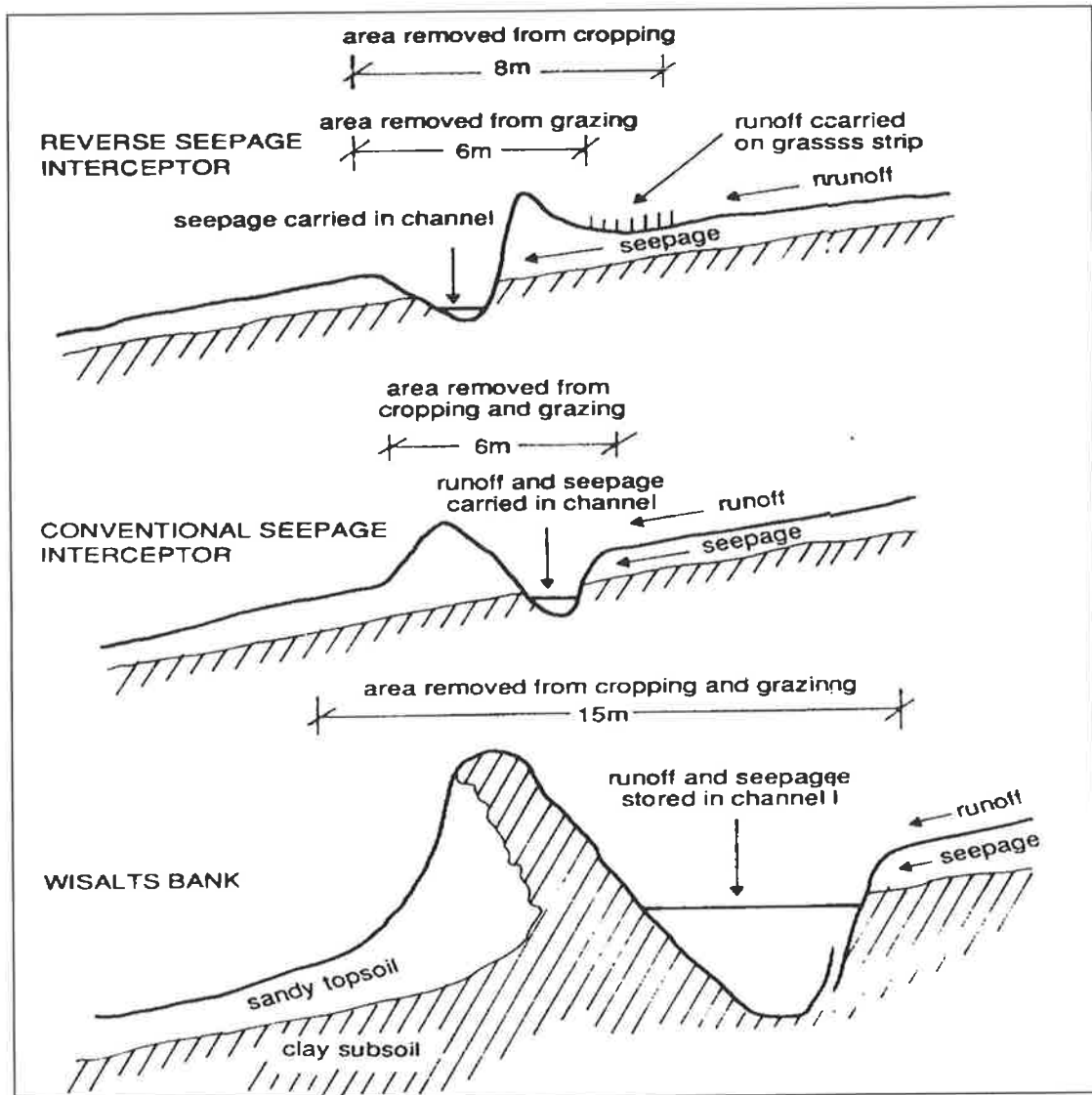


Fig 2.2 Comparison of three interceptor drains (Source: McFarlane and Cox 1990).

2.7.3 Agronomic solutions

The plant physiological approach to manipulation of the soil water balance is based on the belief that perennial plants will use more water than annual crops and/or pastures (Nulsen and Baxter 1982; O'Loughlin 1992). A deeper root system, and longer growing season than annuals underlies the above assumptions (Scott and Sudmeyer 1993), with many researchers showing the decreased water use of our annual based agricultural systems compared to the previous perennial based native vegetation (Carbon *et al.* 1982; Greenwood *et al.* 1985; Kennett-Smith *et al.* 1993). Allison and Hughes (1983) estimated groundwater recharge under mallee (< 0.3 mm/year) had increased by up to 100 times following clearing, and Kennett-Smith *et al.* (1993) found that recharge in

this environment (mallee), on duplex soils, increased by approximately an order of magnitude.

With the increased awareness of the consequences of the above changes, in terms of land lost to production through salinisation, waterlogging and erosion, there has been a drive to rehabilitate the landscape by establishing vegetation systems where water use corresponds more with that of the original vegetation of the area (Anon 1992).

In the cropping/grazing areas of south-eastern Australia, the establishment of perennial pastures has been advocated to reduce deep drainage and recharge (Anon 1992; Dainton 1992; Garrett 1992; Taylor and Clifton 1993). Phalaris and lucerne are generally recommended as having the greatest potential to dry the soil profile (Crawford and Ransom 1993; Whitfield *et al.* 1993) although cocksfoot may have potential in areas unsuited to phalaris (Anon 1992). Ridley *et al.* (1997) indicate the productivity and persistence of both phalaris and cocksfoot in the higher rainfall areas (>600 mm/yr) where lucerne is restricted by poor drainage, soil acidity or high aluminium levels, however Anon (1992) also highlights the sensitivity of phalaris to acid soils (pH < 5.2) and aluminium, and the consequent importance of the CSIRO breeding program for acid tolerant cultivars (Oram *et al.* 1992; Oram and Culvenor 1994).

Despite the above advocacy, Ridley *et al.* (1997) report only two studies (Burke's Flat, Victoria and Marnoo, Victoria) that have clearly shown a reduction in recharge by the use of perennials in lower rainfall areas (400-450 mm/yr), although a number of studies showed higher water use than annual pastures. In >600 mm/yr rainfall areas they feel the role of perennial pastures in reducing recharge is in doubt "because of incomplete or inadequate research and a lack of predictive modelling", however, even if unlikely to stop 100% of recharge in >600 mm rainfall areas, perennial pastures provide profit and reduce degradation (Ridley *et al.* 1997).

In the Burke's Flat study (trees, lucerne, phalaris), trees were planted on high recharge areas, with lucerne and phalaris on the yellow texture contrast soils of the lower slopes (Day *et al.* 1993). Monitoring of groundwater pressures indicated a drop in the water

table under trees of 5 m in 5 years (Nulsen 1993) and under lucerne of 4 m in 10 years (Taylor and Clifton 1993), with a slight rise in adjacent untreated areas.

A similar drop in groundwater pressure was not evident under phalaris, however this was thought to be attributable to a shorter monitoring period. Direct evapotranspiration measurements (in 1989) had shown that a phalaris sward transpired more than a lucerne sward (470 mm vs 395 mm) and should therefore be similarly effective in recharge control (Clifton *et al.* MS, cited in Taylor and Clifton 1993). However, whether this would be the case given the potentially greater rooting depth of lucerne is debatable. Annual pasture on the other hand used c. 100 mm/yr less than the perennials (Crawford and Ransom 1993).

At Marnoo, in the south-eastern Wimmera, 40 ha was sown to lucerne and lowered the moderately saline watertable (12000 $\mu\text{S}/\text{cm}$) by 2-3 metres in comparison with that under conventional rotations on adjacent land (Young *et al.* 1992). In yet another study near Wagga Wagga (NSW), Beale (1993) found that phalaris water use was > 100 mm less than lucerne and more equal to annual pasture in that environment. In contrast, Lolicato (1993^b) at Tatura (Vic) reported SWC under phalaris as lower than lucerne in the first year and equal at about 18 months, after which lucerne became dominant at depth. Despite these inconsistencies, and the natural variation inherent in these sorts of studies (see Table 2.4), there is widespread acknowledgment of a role for perennial pastures in recharge control (Beale 1993; Day *et al.* 1993; Lolicato 1993^{ab}; Taylor and Clifton 1993) as part of an integrated catchment management approach (Nulsen 1993). The difficulty in measuring water movement at depth is reflected in the paucity of measured effects of different agronomic options in the reduction of recharge (Nulsen 1993; Lolicato 2000).

Table 2.4 Average, maximum and minimum annual evapotranspiration (AET) for range of crops/pastures in SW Australia (from Scott and Sudmeyer 1993).

Species	Annual Evapotranspiration (mm)		
	Average	Maximum	Minimum
Lupins	471	580	230
Oats	458	583	264
Lucerne	444	608	281
Phalaris	349	553	261
Annual pasture	339	501	159

To be an attractive option for farmers however, perennial pastures must also show an economic advantage to offset the cost of their establishment and maintenance. Berry (1997) compares various physical and agronomic options (Table 2.5) showing the net return on investment as well as an environmental benefit (reduction in recharge). This approach allows land managers to choose those approaches that best suit their specific requirements, both economic and environmental.

In the Southern Avoca catchment (Vic), the economic benefit of shifting to perennial pastures for salinity management (Anon 1992) is shown by the potential increase in stocking rate (Table 2.6) based on French's formula (Equation 2.4);

$$\text{DSE / hectare} = [\text{Annual rainfall (mm)} / 25] - 10 \quad \text{Eq. 2.4}$$

Similarly, in the Marnoo study (Young *et al.* 1992), a regional survey showed that the higher, more even dry matter production over the year, allowed lucerne to support 60% higher stock numbers than the usual medic-based pastures.

Table 2.5 Economic / environmental returns for recharge reduction options. Adapted from Berry (1997).

	Net return for \$ investment in improved production	% reduction in net accessions to groundwater
Trees	0.05	100
Earthworks	0.675	18
Liming	0.77	10
Lucerne	1.50	68
Tagasaste	2.15	90

Table 2.6 Current and potential stocking rates (DSE/Ha). Adapted from Anon (1992).

Rainfall	Potential Stocking Rates (per French formula)	Current Average Stocking Rates	Range of Average Stocking Rates	Realistic Target with Improved Pastures	Achievable Increases
(mm)	DSE/Ha	DSE/Ha	DSE/Ha	DSE/Ha	DSE/Ha
400	6	2.50	0.5-4.5	4.5	2.0
500	10	3.75	1.0-6.0	6.0	2.25
600	14	5.00	1.0-8.0	8.0	3.0

2.8 Pasture ecology/Root growth and measurement

2.8.1 *Lucerne*

Lucerne is reputedly the “most widely grown perennial pasture legume in Australia” (Lodge 1991), and is grown from the temperate regions through to the subtropics. However, survival data shows the rapidly increasing death rate with progression away from the temperate zone to subtropical areas (Leach 1978). The two factors of most importance to the performance and persistence of grazed lucerne in temperate environs are timing of grazing (ie. in relation to plant growth stage/physiology) and grazing management.

As climates vary, so does the growth of lucerne varieties. The effect can be a restriction in the active growing season and the potential growth during the cool-cold season. Lucerne varieties are grouped into 4 categories as follows (from Heritage Seeds 2002) .

Winter dormant (3) 0-5% winter growth: A distinct dormancy period triggered by temperature making it able to withstand very low temperatures for long periods. Sometimes called summer active, these varieties will produce the equivalent dry matter of the more active varieties in spring, summer and autumn.

Semi dormant (5) 10% winter growth: Characterised by a slowing of growth in the winter, however autumn growth is quicker than the winter dormant varieties. These plants are generally more leafy and have broader crowns than winter varieties. They also tend to persist for longer periods.

Winter active (6-8) 10-20% winter growth: Although growth slows in winter, it is never fully dormant. Provided temperatures are suitable, these varieties are not affected by day length. Recovery after cutting/grazing is much quicker than dormant varieties. Plants are characterised by narrow crowns and have a lower leaf density, fewer tillers and bigger leaves.

Highly winter active (9-10) 20-30% winter growth: These varieties show the fastest regrowth after cutting or grazing. Although these varieties do slow in winter if temperatures are suitable, reasonable growth is achievable.

Traditionally, the timing of grazing is based on 10% flowering to achieve good quality and yield whilst limiting the depletion of root carbohydrate levels (Leach 1978; Lodge 1991). Although this recommendation has been in vogue since Roman times, Lodge (1991) in a review of management practices, indicates a range of factors from temperature, to stubble length and residual leaf area that may be at least as important, if not more so, than carbohydrate root reserves. Lodge (1991) suggests that for persistence, a grazing regime that ensures “adequate numbers of developed basal buds depends mainly on defoliation at a late stage of maturity and lenient harvesting”, rather than the 10% stage of flowering. There is however a tradeoff as forage value decreases with increased maturity of the stand.

Grazing systems must be rotational or intermittent (McKinney 1974; Leach 1978; Lodge 1991) with a spell of 5-7 weeks between successive grazing (McKinney 1974; Lodge 1991), so that in drier regions a 6-10 paddock rotation system is needed (Lodge 1991). Wolfe and Southwood (1980) suggest that fewer paddocks are needed if there is a substantial contribution to the overall feed supply by annuals and other perennials in winter and early spring, and an increased grazing period (ie. > 7-10 days) would similarly reduce the required paddock number. Leach (1978) reports that longer grazing periods may not be as detrimental as previously supposed, provided there is an adequate recovery period, although any benefit from prolonged grazing is reduced in older herbage.

In practice, lucerne stands often tend to be used opportunistically, partially due to the additional cost in dollar terms as well as in time and labour in creating and managing a rotational system. The need for smaller paddocks, and the expense involved in fencing is a major obstacle to overcome in recommending a change in the current set stocking of much of our temperate grazing country.

For the greatest productivity, lucerne should be grown on fertile, well drained, alluvial soils. However, in dryland farming, most is grown where conditions are less favourable, with a restricted moisture supply and unfavourable soil conditions. This generally leads to lower productivity and other problems associated with its adaptation to the physical environment (Leach 1978).

In ideal situations (eg. some American and European soils) lucerne will develop root systems extending beyond 8 m in 4 years (Heichel 1983). Under normal Australian conditions, this is unlikely because of the many constraints to plant root growth, but Holford and Doyle (1978) note root growth >2 m within 18 months of sowing on red-brown, and black, earths in northern NSW.

Rooting depth increases with the age of the lucerne stand, with the root system dominated by a deep tap root, and proliferating fibrous roots in the upper 15-20 cm of the soil profile (Heichel 1983). Edaphic factors can however influence the growth of pastures (Davidson 1978) and the high bulk densities found in duplex subsoils may limit root penetration (Cresswell and Kirkegaard 1995). Gregory (1998), reporting on wheat and lupin experiments, found shallower root systems on duplex soils than on deep yellow earths, with only 1.3-3.8% of total root length below 40 cm. Dracup *et al.* (1992) similarly reported shallow constrained root systems on duplex soils with only a small proportion of the roots in the B horizon. Heichel (1983) notes that lucerne root mass decreases logarithmically with depth, with c.70% of the total mass in the top 15-20 cm.

There is general agreement though on the superior ability of tap-rooted plants to penetrate (or 'biologically drill') the high bulk density B horizons of duplex soils (Blackwell *et al.* 1990; Materechera 1991, 1992; Cresswell and Kirkegaard 1995; Gregory 1998). Although difficult to generalise on (Cresswell and Kirkegaard 1995), the penetration capability of a plant root seems strongly influenced by its size (diameter) (Materechera 1992), and lucerne roots some millimetres in diameter have been found at 2-3 m depths (Cresswell and Kirkegaard 1995).

Lucerne taproots are reported to use biopores (eg. earthworm channels) to improve their access to depth (Blackwell *et al.* 1990) as well as create stable biopores themselves for subsequent root growth (Blackwell *et al.* 1990; Materechera 1991, 1992; Cresswell and Kirkegaard 1995; Gregory 1998). In duplex soils, the importance of macropores to root growth, and as a pathway for the vertical movement of water and gases is well recognised (Blackwell *et al.* 1990; Materechera *et al.* 1991; Dracup *et al.* 1992). In many duplex soils there are cracks and other macroscopic planes of weakness which

function as preferential flow paths for water and nutrients (Smettem *et al.* 1991; Cox *et al.* 1996^b), and Whiteley and Dexter (1983) found that under these circumstances, preferential root growth on ped surfaces was more likely than growth within the adjacent structural units.

Whilst these fissures allow greater root penetration to depth in the profile, it also introduces other considerations such as the degree of root-soil contact in these voids, and the efficiency of exploration of the soil matrix for the uptake of water and non-mobile nutrients (Whiteley and Dexter 1983; Cresswell and Kirkegaard 1995). Heichel (1983) notes that lucerne taproots follow the cracks and faults in heavy subsoils and may extend laterally (up to 5 m) if impeded by an impenetrable layer. Under these circumstances, patterns of root development in the field are unlikely to be well understood given the sampling difficulties, and root proliferation is generally inferred from soil water use patterns (Heichel 1983).

2.8.2 *Phalaris*

Different adaptive mechanisms have evolved in grasses from the Mediterranean region to ensure their survival over the characteristically arid summer months (McWilliam and Kramer 1968). Whereas annual grasses rely on early flowering, and heavy seed production as survival mechanisms, perennial grasses produce seed but also have an adaptive capacity for summer dormancy. It is their persistence as individual plants over summer, rather than their less significant reseeding, that is crucial for stability in perennial stands of pasture (McWilliam and Kramer 1968), and in any case, seedling phalaris is usually too weak to compete in grazed, established swards (Oram and Culvenor 1994).

In phalaris, summer dormancy commences following seed maturity, with a slow senescence of the leaves and stems (culms), until often only a residual amount of green tissue is found in the basal internodes (McWilliam and Kramer 1968). This reduction in surface area conserves moisture, and the living tissue remaining, a storage for carbohydrates and water, is essential for regrowth in autumn.

Normally, the plant's water requirements are met largely by the surface roots, which comprise c.90% of the total root weight of the plant (Cashmore 1934, cited in McWilliam and Kramer 1968). However, as summer drought progresses, these surface roots gradually die off as the soil dries, and the plant relies on those few roots which penetrate into the subsoil to extract the small quantity of water required for bud survival in the dormant state (Oram and Culvenor 1994). McWilliam and Kramer (1968) found that although very few roots penetrated to c.2 m, those that did were critical for summer survival, as if they were severed (at a depth of 90 cm), plants invariably died.

At the height of dormancy, the buds are the only remaining meristematic tissue in the plant (McWilliam 1968), and survival depends on the continual, though small, supply of water to the basal culms through the limited deep root system. Oram and Culvenor (1994) noted that given this dependence on subsoil moisture, those soil conditions which could constrain root growth (eg. acidity, rock, waterlogged subsoil) would all be likely to diminish the plant's survival.

The dormancy, which is greatest at the start of summer, is in general terms an induced dormancy in that it can be broken by removing the moisture stress and it declines as autumn approaches with a reduction in temperatures and increased likelihood of rainfall (McWilliam and Kramer 1968). Levels of bud dormancy have however been shown to differ between genotypes (Oram and Culvenor 1994), and are a factor to be considered in the survival of plants in southern Australia (Oram and Freebairn 1984; Oram and Culvenor 1994) where summer thunderstorms may break dormancy.

Oram and Freebairn (1984) reported that some cultivars with a 'low' dormancy (eg 'Australian', 'Seedmaster'), although capable of surviving extended summer drought, were likely to die if bud development was promoted by brief periods of cool, wet weather. Under these conditions the developing buds would require a greater amount of water, and unless this could be supplied by new surface roots, any new growth would be killed by desiccation.

Some ecotypes developed from Moroccan strains have buds with a higher dormancy level (Oram and Freebairn 1984) and only a small proportion of buds will start to grow

under similar influences, giving these cultivars a better chance of survival. The cultivar, Sirosa, is considered to have 'intermediate' dormancy (Oram and Freebairn 1984). It is a winter-active cultivar "developed from 26 diverse Mediterranean introductions onto 'Australian' and a Turkish accession" with similarities to 'Australian' (Oram and Culvenor 1994).

Summer dormancy though limits a plant's annual water use no matter how well it grows during the winter/spring period (Johnston 1993). Those species that continue to grow during the warmer months will use water at a time when evaporative demand is high (Cook 1992). From the point of view of recharge, and the importance of establishing an autumn soil water deficit, summer dormancy is not a desirable attribute. Even before the onset of dormancy, summer-active lucerne has been shown to dry the profile more rapidly than phalaris (Dear and Cocks 1997).

Taylor and Clifton (1993) report that phalaris persistence is reliant on management factors and that a lack of persistence of the newer varieties (eg. Sirosa) seems related to grazing management, particularly set stocking. They feel that "a short respite from grazing in late spring is necessary to promote the development of dormant buds on the crown of the plant". Oram and Culvenor (1994) acknowledge that the persistence of winter-active cultivars of phalaris is under review, and that they may be more sensitive to spring management operations than the Australian type.

Although there is a range of interacting factors (edaphic, drought, grazing management) that will influence the survival and regeneration of individual or groups of cultivars, phalaris shows better drought survival than cocksfoot (Oram and Freebairn 1984) and generally persists "better than other introduced temperate grasses" (Oram and Culvenor 1994). However, Taylor and Clifton (1993) state that cocksfoot is more persistent than phalaris on acid soils.

2.8.3 Cocksfoot

Cocksfoot (*Dactylis glomerata* L.) is acknowledged as a useful pasture species throughout southern Australia. Although its nutritive value is generally considered less

than that of phalaris, it is often more persistent in “open-textured soils” (Lamp *et al.* 1990), and unlike phalaris is free from anti-nutritional factors.

Ridley (1995), found in field experiments that cocksfoot “had a reasonably dense root system from 10-50 cm depth” and that this was consistent with the view of Weaver (1926) that cocksfoot developed a substantial root system to 75 cm depth. These studies (Ridley 1995) also showed the rapid development of tillers in both cocksfoot and phalaris following autumn rains, however cocksfoot root growth did not commence until secondary tillering whereas phalaris commenced earlier.

Cocksfoot may also be less persistent than phalaris in <600 mm rainfall environments, with plants subject to uprooting by stock in summer/early autumn (see Ridley 1995, p.151). The lower proportion of roots surviving over summer compared with phalaris, as well as the poor early root growth may have some bearing on this. Ridley (1995) suggests that the exclusion of stock from cocksfoot paddocks for a month after opening rains, may improve its persistence by allowing the plants time to develop new root systems. Whilst this may not be very practical from an animal nutrition perspective, it may reduce reliance on cocksfoot seedling regeneration.

The cultivar ‘Currie’ is intermediate in dormancy characteristics between the non-dormant European cultivars and the North African ecotypes which are strongly dormant (Oram 1990). It is able to survive hot, dry summers reasonably well and in a low to medium rainfall environment (450-635 mm) Oram (1990) suggests this cultivar may be more productive in summer and autumn than phalaris due to its quicker growth response to rainfall. More recent publications (Duncan 2001; Kondinin Group 1995) report that Currie is an early maturing variety, maturing at least a week earlier than Porto, the variety grown in the Ridley (1995) study above. Currie, will provide useful summer growth in higher altitudes if moisture is available but the bulk of its growth is in autumn, winter and spring. Porto requires a higher rainfall (very productive with rainfall >700 mm) and has better summer growth in suitable areas.

2.8.4 Root measurement

Perennials have been reported to reduce the deep infiltration of water in comparison with annual species (Joffre *et al.* 1988; Nulsen 1993; Taylor and Clifton 1993) and it is assumed that this is as a result of their deep rooting capability. However as noted previously, root proliferation is generally inferred from soil water use rather than from sampling (Hiechel 1983).

Mackie-Dawson and Atkinson (1991) review a variety of methods of determining root distribution (quantitative and non-quantitative) and analysing the outcomes. They suggest that whilst visual comparisons of root systems are not quantitative, they may still have value as a means of developing conceptual models and giving insights into planning further studies. In the 'profile wall method' root distribution can be mapped directly onto plastic overlay, or plotted indirectly as root count versus depth.

Soil coring is the most common methodology employed for root analysis (Mackie-Dawson and Atkinson 1991) and the sectioned cores are usually washed on sieves to separate root material from the soil. Smucker *et al.* (1982) reported on a less laborious method using pressurised water jets and air flotation (hydro-pneumatic elutriation), and in comparisons with careful handwashing, for a range of species and using primary and secondary sieves (930 and 500 μm respectively), there was between 15 - 25% additional root dry weight recovery by the elutriation method (Mackie-Dawson and Atkinson 1991).

2.8.5 Root length

Whilst small samples may be measured directly (eg. onto graph paper), larger samples or longer root lengths are normally measured using variants of Newman's (1966) line-intersect principle (see Tennant 1975), based on counting the number of intersections between root and a pattern of random or regular lines. In the past this was performed visually, however with the development of new technologies, there is an expanded use of desktop scanners for accurate and precise estimates of root length, area and other morphological parameters (Pan and Bolton 1991).

2.9 Summary and rationale for study

Land degradation related to the hydrological imbalance caused by widespread clearing of native vegetation across Australia is well documented. Whether it is increasing dryland salinity, development of acid soils or waterlogging, all are contributing to a decline in agricultural sustainability in vast areas of this country. CSIRO (1999), reporting for the Murray Darling Basin Commission, outlined the causes and extent of salinity in the Basin and concluded that for much of this area, current best practice farming systems could not control the problem.

“The large mismatch between the leakage below current farming systems and the capacity for groundwater systems to accept this leakage is the fundamental cause of our expanding dryland salinity problem” (CSIRO 1999).

Texture contrast soils tend to exacerbate farm management difficulties in relation to these soils' hydrological peculiarities (Passioura 1992). With the low hydraulic conductivity of their B horizons, the potential for waterlogging is increased, leading to lateral flows along 'throbbles' such as the A/B (McFarlane and Cox 1992) or B/C horizon (Smettem *et al.* 1991) interfaces. Along with increasing lower slope salinity and sodicity (Fitzpatrick *et al.* 1997), this throughflow drains to the generally saline groundwater of the foot and toe slopes (ie. the 'break of slope' and the 'valley floor'). We need to investigate and quantify these movements of water through the landscape, and investigate the capability of various options to deal with it, if we are to understand the nature of the problem and recommend management options to mitigate it.

Reverse interceptor drains are known to perform best in situations of a strong hydraulic throttle (eg. at the A/B soil horizon boundary) where they intercept throughflow moving downslope on top of the B horizon (Cox 1988; McFarlane and Cox 1992). The same drains may not perform well on texture contrast soils with a high level of macroporosity (and hence vertical preferential flow) in the B horizon clay and these 'leaky' soils have been reported in parts of the Mt Lofty Ranges (Smettem *et al.* 1991). Under these conditions the buried tube (pipe) drains should function well although MacEwan *et al.* (1992) reported on a study that showed that slope position may influence the efficiency of pipes in alleviating waterlogging. They found that whilst mid-slope soils were

drained adequately by pipes alone, the soils of the lower slopes were drained better with a combination of pipe and mole drains.

CSIRO (2000) recognise that there needs to be a range of options in place if we are to control leakage from our farming systems. Whilst annual cropping would be the preferred economic option, it has no potential to reduce leakage by the magnitude required. However, the potential to reduce salinity by recharge reduction is physically feasible (Peck 1977), particularly in lower rainfall areas.

Additional leakage under current farming systems is generally small (15-130 mm/year) relative to annual rates of potential evapotranspiration (>1500 mm/year), leaving scope for agronomic research to find ways to reduce this leakage to more closely resemble that under native vegetation (Peck and Hurle 1973; CSIRO 2000). Lucerne, phalaris and cocksfoot are perennials commonly recommended for recharge control in Australia with little supporting evidence (Ridley *et al.* 1997). The determination of the soil water dynamics and growth of these perennials in the study environment is important in relation to manipulating the hydrology of current farming practices to limit the potential for degrading processes within the catchment.

The problem with recharge is not its occurrence, it is a natural process. Now it occurs more frequently and at a greater magnitude than pre-clearing and agricultural development (Kennett-Smith *et al.* 1993; Mitchell *et al.* 1997), and that is the problem. Salinity management therefore should not aim to prevent recharge, but to reduce it to acceptable limits. Increasing water use by the more widespread planting of deep-rooted perennials may be all that is needed in some areas. The challenge, as seen by Mitchell *et al.* (1997), "is to achieve this over the whole of the landscape where current recharge rates are too high".

Other studies (Sharpley *et al.* 1994; Fleming and Cox 1998, 2001; Cox and Pitman 2001) have identified the potential for phosphorus, sulfur and other chemicals to move in quantity in surface runoff (see also Chapter 6). This suggests that surface water transported in drains will need careful handling to prevent high concentrations of pollutants entering local waterways. Further studies (Cox and Kirkby 1995; Kirkby *et al.*

1996, 1997; Cox *et al.* 1996^b, 2000) have shown that the issue of macropore flow needs to be better understood to deal with potential leaching of nutrients to groundwater supplies. By studying a range of agronomic and engineering options this study addresses some of the above challenges, and also identifies potential problems for the future.

Chapter 3

Field trial layout and management

3.1 Introduction

Land and water degradation problems are common to many catchments throughout the Mt Lofty Ranges. Erosion, waterlogging and salinity have developed since the land was cleared of its perennial vegetation cover. This research was undertaken in a catchment that is representative of the eastern Mt Lofty Ranges “where an old lateritic regolith has formed on Cambrian Kanmantoo Group metasediments during the Mesozoic” (Moore 1980).

3.2 Site description

The experimental site is located in Keynes' catchment (c.200 ha) near Keyneton in South Australia (Fig. 3.1), c.100 km northeast of Adelaide in the Mt Lofty Ranges (139°09'E, 34°31'S). The topography of the area is undulating with elevations ranging from 390 to 430 m above mean sea level, and water leaving the site drains towards the Murray River via the North Rhine and Marne Rivers. The land is mostly cleared, clearance having commenced in 1850 (McMurray 1994), and creek lines are badly eroded and salt encrusted (Plate 3.1).

Highly saline areas in the lower slopes are indicated by the presence of scalds or by the dominance of sea barley grass (*Hordeum marinum* Huds.) which began to establish in the area in the 1930s (McMurray 1994). There has been increasing evidence of surface scalding since then (G.Keynes, pers. comm.) and over the period of this study a gradual spread of sea barley grass onto the toe-slopes was noticeable.

3.2.1 Land use

The land is predominantly used for grazing (sheep and cattle) carrying c. 4 DSE/Ha (G. Keynes, pers. comm.), and some small areas sown to oats. Pastures are based on a cocksfoot (*Dactylis glomerata* L.) and sub-clover mix, and are declining through overgrazing.

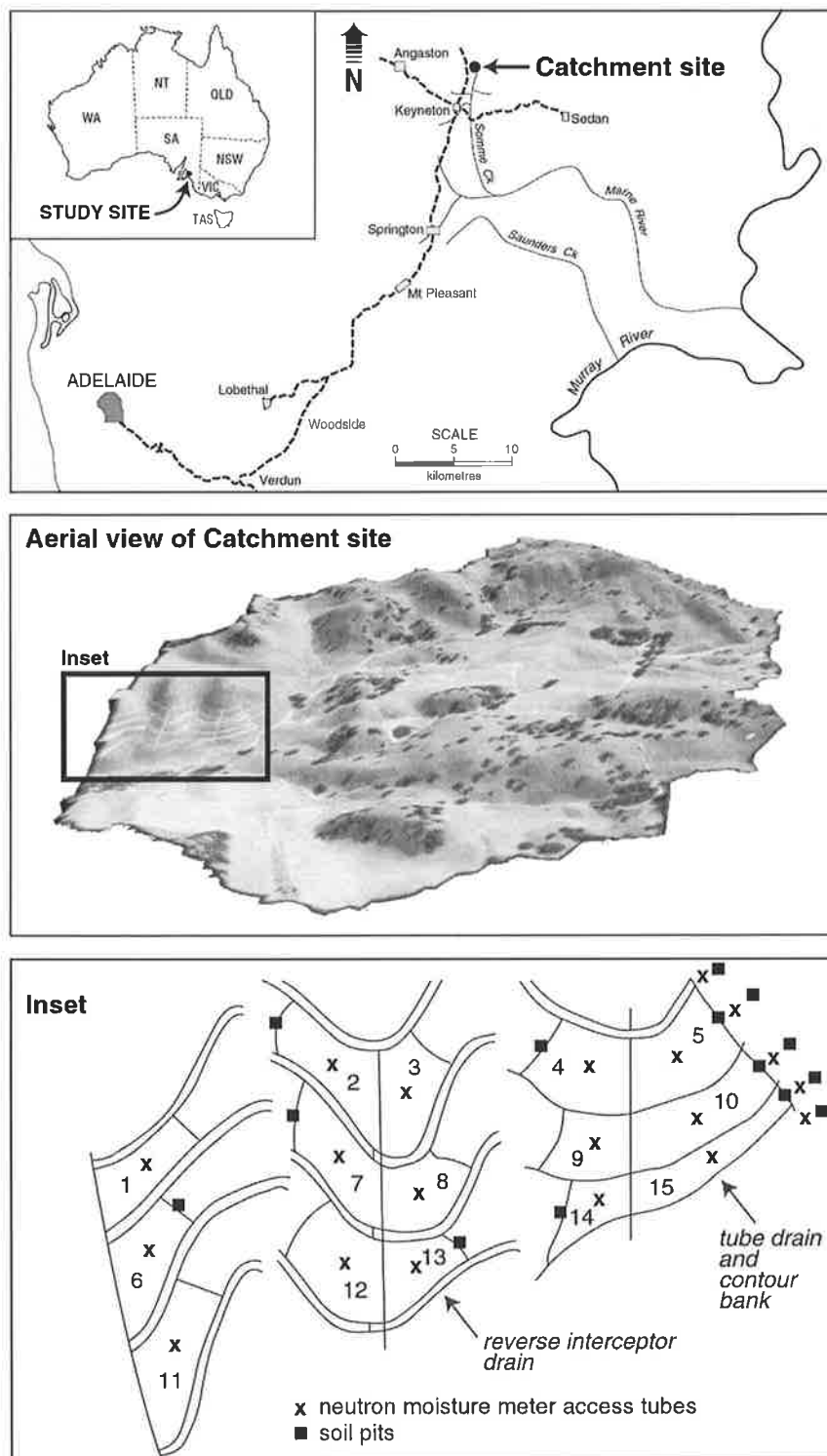


Fig. 3.1 Location of catchment, drains, soil pits and neutron access tubes (modified from Cox and Pitman 2001).

3.2.2 Climate

The climate is Mediterranean, with hot summers (mean maximum c.26.5°C) and cool winters (mean minimum c.5.5°C). The mean annual rainfall is 540 mm (30 year average) with >75% of the fall in the period April to October. Rainfall and other climate data allowing potential evapotranspiration (PET) to be calculated by the Penman-Monteith equation (Penman 1948; Monteith and Unsworth 1990) were recorded daily at an automatic weather station (Monitor Sensors) located 200 m from the experimental site. For those occasions when on-site data was unobtainable through equipment malfunctions, records were available from Keyneton Post Office, c. 5 km south of the site.



Plate 3.1 Badly eroded creekline with sodic soils and saline scalds.

3.2.3 Soil physical and chemical properties

Saturated hydraulic conductivity (K_s) was measured in duplicate for A and B horizons at six sites (ie. two sites representing the upper, mid and lowerslopes) using the constant-head well permeameter technique of Talsma and Hallam (1980) with the analysis of Elrick *et al.* (1989). A 50 mm diameter auger was used to drill a hole to the required

depth (0.02-0.1 m in the A horizon, and up to 0.95 m in the B horizon), and a wire brush used to reopen smeared pores as recommended by Reynolds and Elrick (1989).

Surface saturated hydraulic conductivities were measured at the same sites (large disc permeameter) using the design and analysis of Perroux and White (1988). A tension of 10 mm, equivalent to a pore diameter of 3 mm was used to eliminate outflow through pores greater than 3 mm in diameter. Where required, all vegetation was removed with scissors without soil disturbance.

Samples from soil cores were collected for profile description and particle analysis by dispersion, sedimentation and gravimetric analysis methods (C.Grant, CSIRO Soils, pers. comm.). Bulk densities were measured by the intact core method (McKenzie, Coughlan & Cresswell 2002, p.39) from soil cores taken at the time of installation of the NMM access tubes (see 3.4.4), and detail of chemical analyses is in Chapter 6.

3.3 Experimental design and statistical analysis

Research on the hydrology of sloping land with texture-contrast soils is far more complex than most of the published research which is on flat land. The ability to carry out this research depends firstly on the availability of suitable landscapes deemed to be 'hydrologically similar' on which treatments can be imposed, and secondly on financial resources required to install instruments for monitoring all water and nutrient fluxes. Both landscape and financial resources place limitations on the experimental design that can be imposed. Small replicated plot studies which are used on flat land are unacceptable when trying to quantify hillslope processes.

The research site chosen for this study made use of an existing site established by CSIRO at Keyneton. The advantage of utilising this existing site was that certain site characterisation of climate, soil, and water had already been conducted and a certain amount of instrumentation, for example, a network of piezometers, had already been installed.

Selecting this site imposed limits on the area available for new research. In order to isolate treatment plots from each other, expensive earth works were required (the total

cost of installing and instrumenting the site was around \$120,000). For hydrological reasons each plot was relatively large, measuring 40 m by 50 m – the minimum size considered appropriate for these studies. As a consequence of the available landscape and the need for large hydrologically isolated treatment plots it was not possible to impose a completely balanced experimental design. As a compromise it was decided to only have one replication of each treatment and no replication of the control plots at each of three positions along each toposequence.

At the commencement of the research these practical constraints on experimental design were acknowledged and it was assumed that detailed point-based measurement of water and nutrient dynamics combined with simulation modelling would overcome any limitations in statistical design. In hindsight this strategy was overly optimistic, but at the time it was considered a reasonable strategy by many researchers faced with similar constraints around Australia (e.g. Ward *et al.* 2001; Hodgson *et al.* 2002).

In the course of data collection the true nature of heterogeneity in soil and preferential water flow at this site became apparent. The movement of water and nutrients through sloping landscapes is now understood to be much more complex than previously thought. Preferential flow pathways related to physical fractured rock patterns complicated by differential weathering rates result in great complexity. This complexity is increased even further when biological pathways, old root channels, macro fauna, etc, are included in the soil matrix. This greater awareness of spatial heterogeneity at both the micro and macro scale is a major outcome of this research.

Great time and effort was invested in parameterising the TOPOG model for this site. TOPOG is a very parameter intensive model and previous authors have found that model performance is strongly dependent on the quality of parameters and the effort invested in measuring attributes that can be used to derive parameters at the scale needed to be relevant to the catchment in question. Many previous users of the model have fallen back on using average parameters in the hope that less intensive effort on deriving parameters would still result in acceptable model performance.

Unfortunately, in this catchment against this data set, performance of the TOPOG model was poor and unable to reproduce observed patterns of water movement. A major reason for this poor performance was the inability of parameters derived for ‘average’ soil conditions to adequately simulate water dynamics in this highly heterogeneous catchment. This is another major finding from this research with significant implications for further research and the use and merit of simulation models on heterogeneous sloping land.

Recent publications (Silberstein *et al.* 2002; Ali *et al.* 2004) have reached similar conclusions on the applicability of hydrological models on sloping fractured rock catchments. This knowledge was not available at the time of the commencement of this research.

Two major findings from this work were:

1. Spatial heterogeneity in this sloping fractured rock catchment was far greater than previously expected.
2. Use of ‘average’ parameters to simulate such a catchment result in poor model performance but attempting to derive spatially variable parameters for the model soon becomes prohibitively expensive.

Both these findings have important implications for further research, further model development and application, and practical on-ground works (drains, revegetation, etc).

3.3.1 Site layout

The standard (established) pasture (plots 1, 6, and 11; Fig. 3.1) was 10% cocksfoot (*Dactylis glomerata* L.), 15% subterranean clover (*Trifolium subterranean* L.) and annual grasses and weeds, including Salvation Jane (*Echium plantagineum* L.), storksbill (*Erodium moschatum* L. Ait) and smaller amounts of soursob (*Oxalis pes-caprae* L.), capeweed (*Arctotheca calendula* (L.) Levyns) and silver grass (*Vulpia bromoides* (L.) Gray).

The improved (newly sown) pastures were Aquarius lucerne (*Medicago sativa* L.) (plots 3, 4, 7, 10, 13, and 15; Fig 3.1) and also Sirosa phalaris (*Phalaris aquatica* L.) with a mix of Dalkeith, Trikkala, Junnee and Balansa subterranean clovers (plots 2, 5, 8, 9, 12, and 14; Fig 3.1). A legume fertiliser (Hifert 6:16:0:10) was applied to all plots at the start of the trial. Plots were approx. 0.2 ha and replicated at three levels in the landscape; upper, mid, and lower slope. (See also Fig. 3.2 for pasture layout).

Overland flow and throughflow were excluded from entering the plots from upslope by physical barriers (see Plate 3.2). Overland flow and A/B horizon throughflow (to about 0.5 m depth) from the three pastures (nine plots) was measured and sampled as it passed through v-notch weirs in reverse interceptor drains installed across the lowest part of the plot (plots 1, 2, 3, 6, 7, 8, 11, 12, and 13; Fig. 3.1). In six other plots (plots 4, 5, 9, 10, 14, and 15; Fig. 3.1), overland flow was measured as it passed through a v-notch weir in a contour bank whereas throughflow was collected in a tube drain and measured in a tipping bucket at the lowest point.

Given the possible increased salinity and sodicity in the lower slope B horizon soils (McMaster 1992), it was expected that there would be differences in pasture growth that were related more to the position in the landscape than the pasture type itself. The experimental design should quantify any differences attributable to slope position.

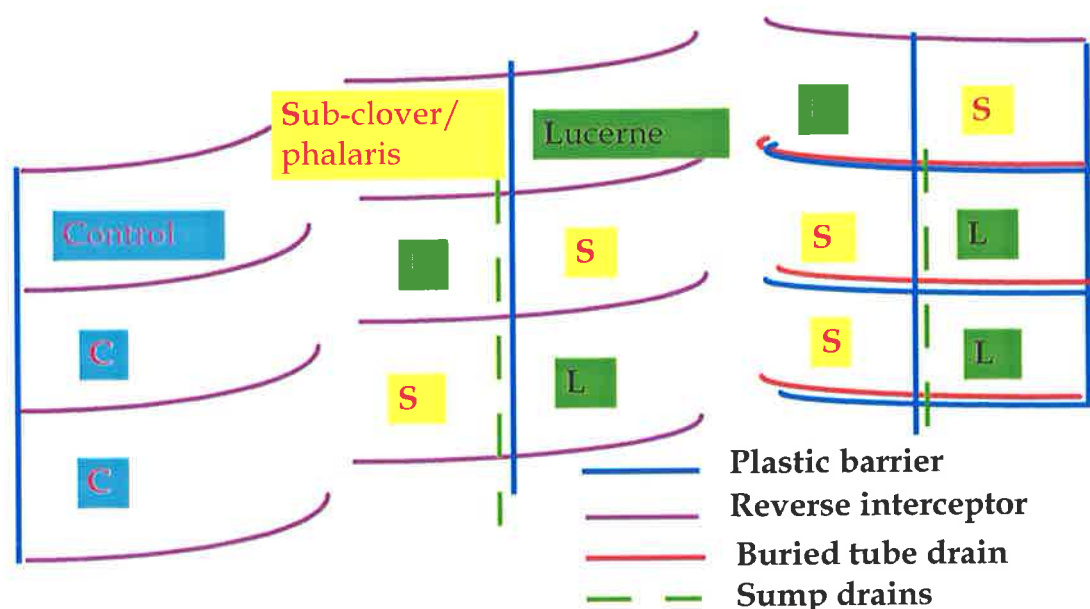


Fig. 3.2 A simplified illustration of plot layout and experimental design at Keyneton.

3.3.2 Surveying and installing drains

There was an area of hillslope suited to this experiment, delineated in part by natural local conditions. A sudden gradient increase and a significant quantity of sizeable surface rock outcroppings defined the upslope limit. The lower limit was set at the footslope, above the obviously saline area denoted by sea barley grass. Laterally, grassed waterways and/or changes in drainage direction imposed by cross-slope conditions defined the natural limits.

Deep reverse interceptor drains were graded in above the three slope segments chosen, to prevent any surface or throughflow water ingress from upslope. The drains were surveyed in at a slope of 0.6 % starting at the discharge point in the grassed waterway and subsequently marking a rise of 6 cm every 10 m across the slope. At approximately 40 m intervals down the slope, 3 further drain lines (0.6 % fall) were surveyed in on each slope segment to divide the areas into zones of upper slope, mid and lower slope (see Figs. 3.1 & 3.2).

Two sets of reverse interceptor drains (Fig. 3.3) were constructed across two slope segments whilst on the third, slotted agricultural drain (80 mm diameter) was installed in grader cuts along the surveyed lines (Fig. 3.2). (There was a need to study buried drains as well as interceptors because of farmer concerns noted in Chapter 2). A plastic barrier (Plate 3.2) down the face of the cut and partially enclosing the pipe, was placed to prevent downslope movement of water through the profile above the position of the drain pipe. It should be noted that the pipes were installed c. 5 to 10 cm further into the B horizon than the floors of the interceptor drains. After backfilling, a contour bank was formed above each pipe drain to capture surface flow.

To provide adjacent plots for phalaris and lucerne in two locations at each slope position (see Figs. 3.1 & 3.2), a grader cut (c. 1 metre in depth) was excavated down the slope (Plate 3.3), bisecting the drain lines. Sumps were installed to collect and remove water flow from those treatment areas without direct drainage to the grassed waterways (see Fig. 3.2), and drainage pipes from each sump were enclosed in a V of plastic within the grader excavation (Plate 3.4). This V of plastic that extended above the soil surface after

backfilling, prevented lateral flow between treatment areas. At either end of the drain arrangement, another grader cut was excavated down the slope and a plastic barrier installed, again to prevent lateral flow intrusion.



Plate 3.2 Plastic barrier preventing throughflow to lower site.

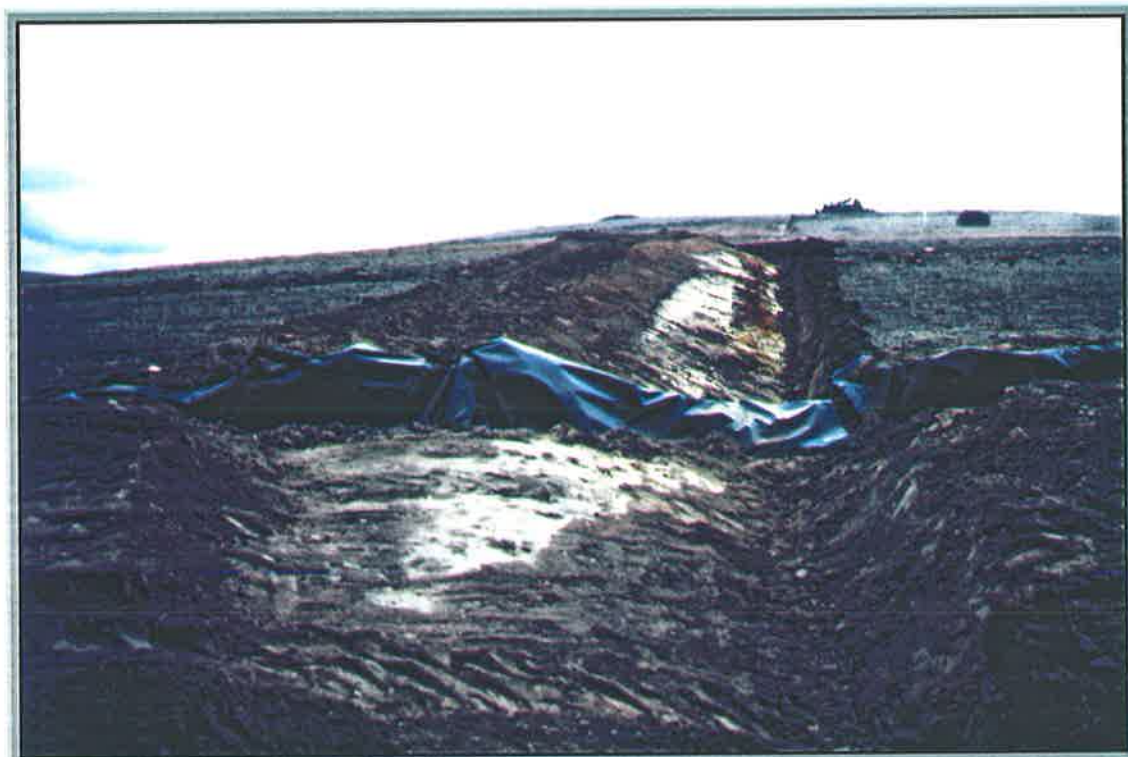


Plate 3.3 Excavation to carry drainage from sumps installed upslope.

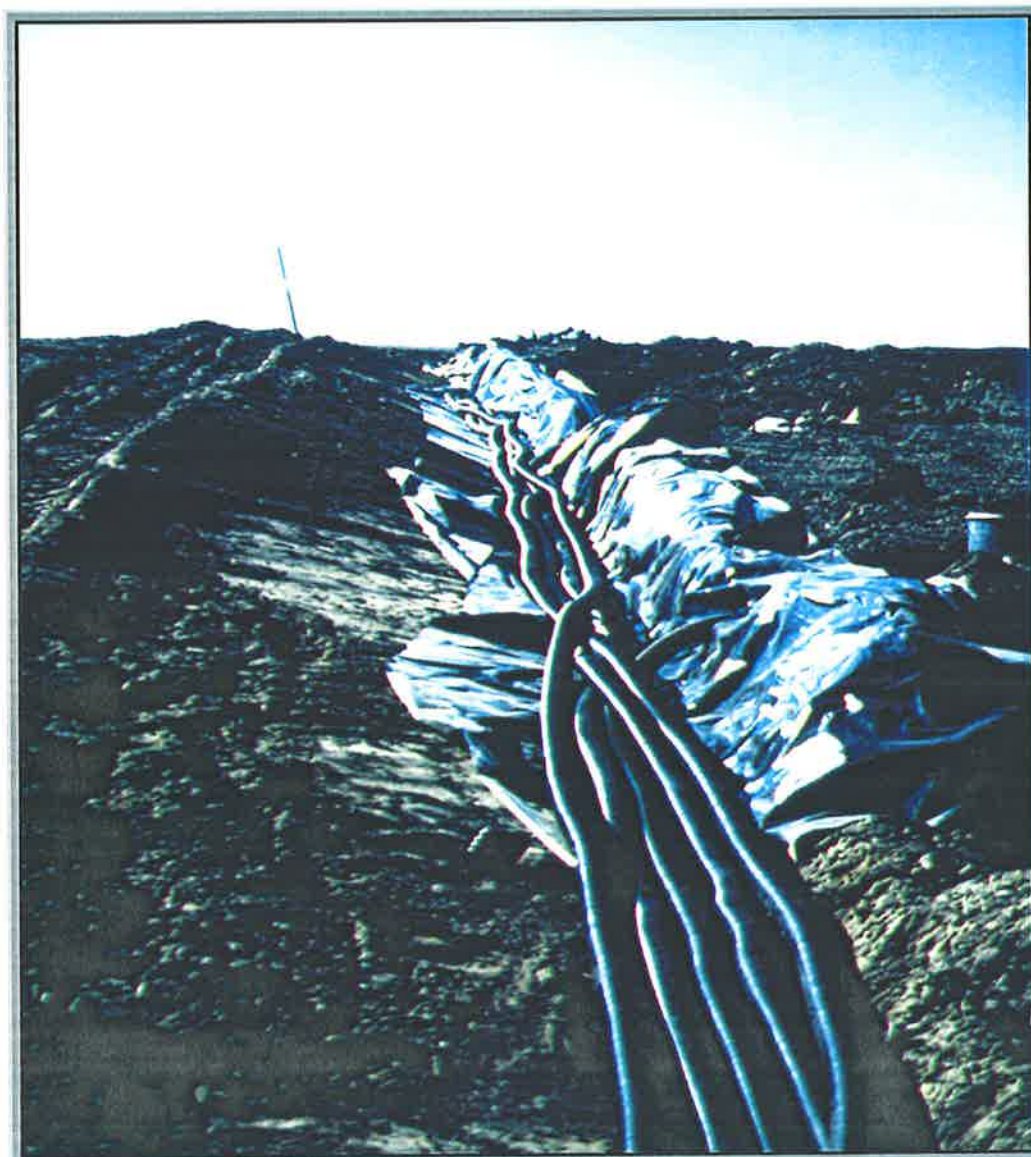
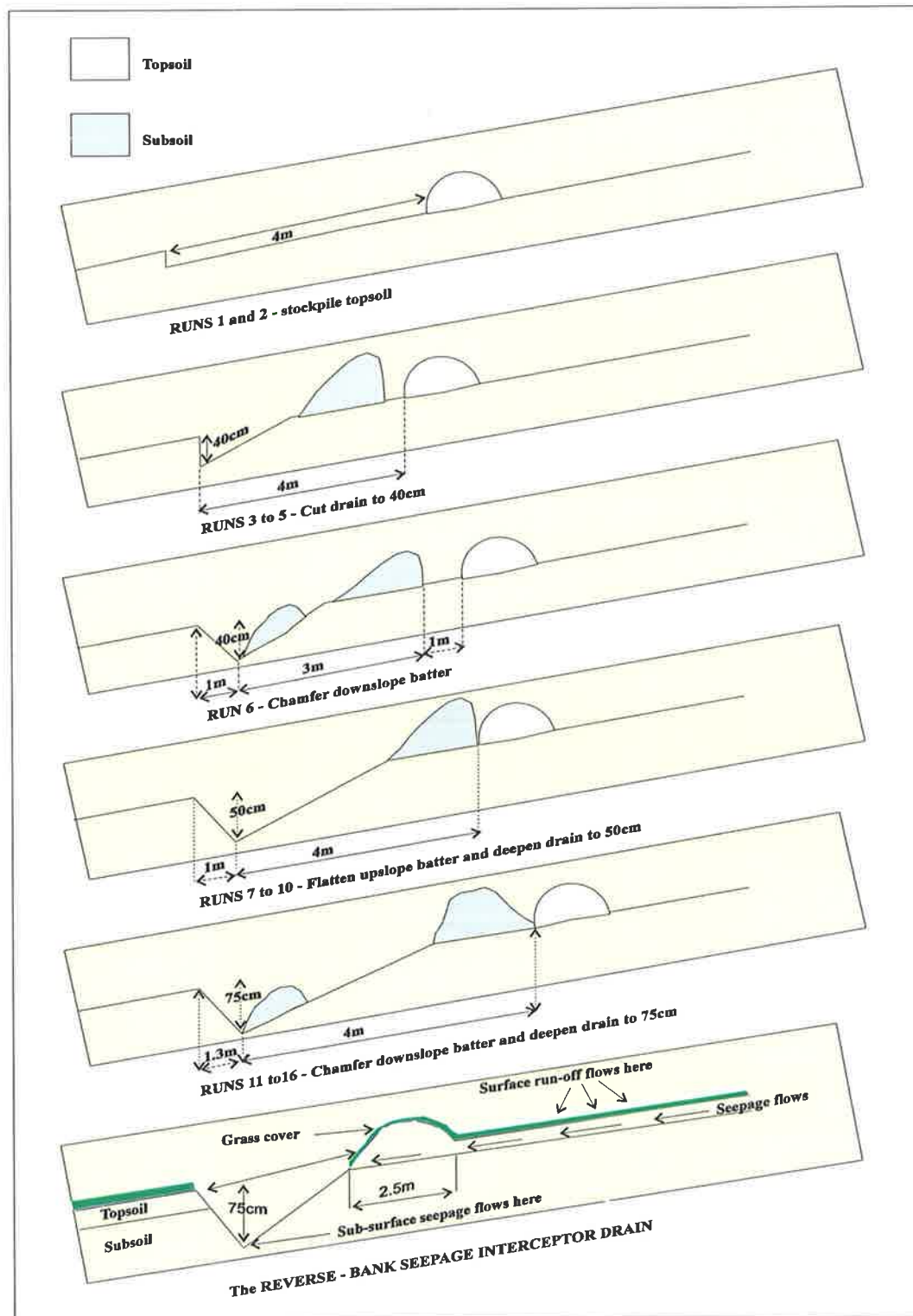


Plate 3.4 Drains from sumps, enclosed in plastic sheeting, and directed off-site.

6 STAGES IN THE CONSTRUCTION OF A REVERSE INTERCEPTOR DRAIN

Fig. 3.3 Construction of reverse interceptor drains (from Reynolds *et al.* 1996).

3.4 Instrumentation / monitoring / measuring

3.4.1 Drains

Reverse interceptors and contour banks

To quantify the movement of water, as surface flow and/or throughflow, V-notch weirs were installed in the reverse interceptor and contour drains. Given the likelihood of short duration, high volume surface runoff, and prolonged lower volumes from throughflow, the V-notches were cut to 120° and 30° respectively (Fig 3.4).

The weirs were cut from 2.5 mm mild steel and were 2.5 m in width and 0.5 m in height. At each location a trench was dug such that the weir was positioned across the drain with the V-notch located in the optimum flow path. The likelihood of any lateral diversion of water flow around the weir was remote considering the distance they were set into each side bank, and any probable undercutting was reduced by setting the base of each weir c. 100 mm below the level of the drain floor. A further seal was gained by the application of bentonite within the locating trenches before backfilling and tamping the soil in place.

The weirs on the surface flow drains were additionally supported on their upslope ends by earthen 'wings', placed both for this purpose and to prevent high, short duration, flood flows from escaping unrecorded around the weir. On the downstream side of each weir, a flat stone was placed under the V-notch to prevent erosional scouring at this discharge point.

Capacitance probes connected to data loggers (DATAFLOW Pty Ltd) were installed upstream of each weir (beyond the stilling pool) to measure stage heights for the subsequent calculation of flows. The probes were housed within PVC tubing that was securely attached to star pickets (see Plate 3.5). Probes were calibrated from their lower support position, a bolt through the PVC tubing set c.100 mm below the base of the V-notch, or the cease to flow (c.t.f.) point.



Plate 3.5 V-notch weirs installed on reverse interceptor drain

The flow rate through the V-notch was calculated with the discharge equation in the form;

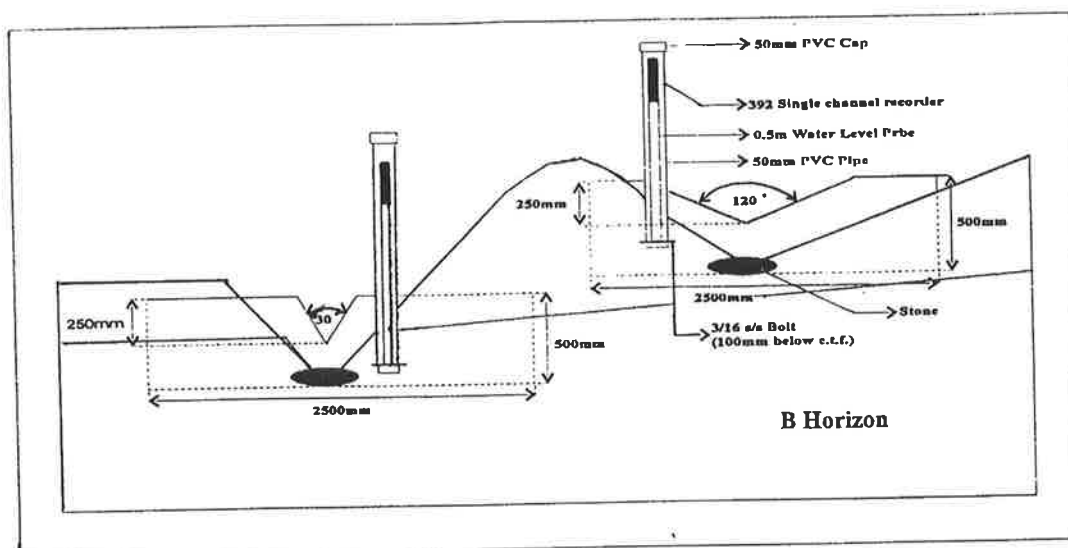
$$Q = CH^{2.5} \quad \text{Eq. 3.1}$$

where Q = flow rate, H = head on the weir, and C = a constant, dependent on the notch angle and measurement unit (see Table 3.1).

Table 3.1 Discharge equations for V-notch weirs with Head in Metres (from: Grant & Dawson 1995)

V-notch angle	l/s	m ³ /hr
22.5°	$Q = 274.4H^{2.5}$	$Q = 987.8H^{2.5}$
30°	$Q = 373.2H^{2.5}$	$Q = 1344H^{2.5}$
45°	$Q = 571.4H^{2.5}$	$Q = 2057H^{2.5}$
60°	$Q = 796.7H^{2.5}$	$Q = 2868H^{2.5}$
90°	$Q = 1380H^{2.5}$	$Q = 4969H^{2.5}$
120°	$Q = 2391H^{2.5}$	$Q = 8606H^{2.5}$

**CROSS SECTION OF REVERSE INTERCEPTOR DRAIN
WITH OVERLAND FLOW V'NOTCH WEIR AND INTERFLOW V'NOTCHWEIR
ALSO SHOWING SINGLE CHANNEL DATA LOGGER AND WATERLEVEL PROBE**



**CROSS SECTION OF INTERCEPTOR DRAIN
WITH OVERLAND FLOW V'NOTCH WEIR AND BURIED TUBE DRAIN**

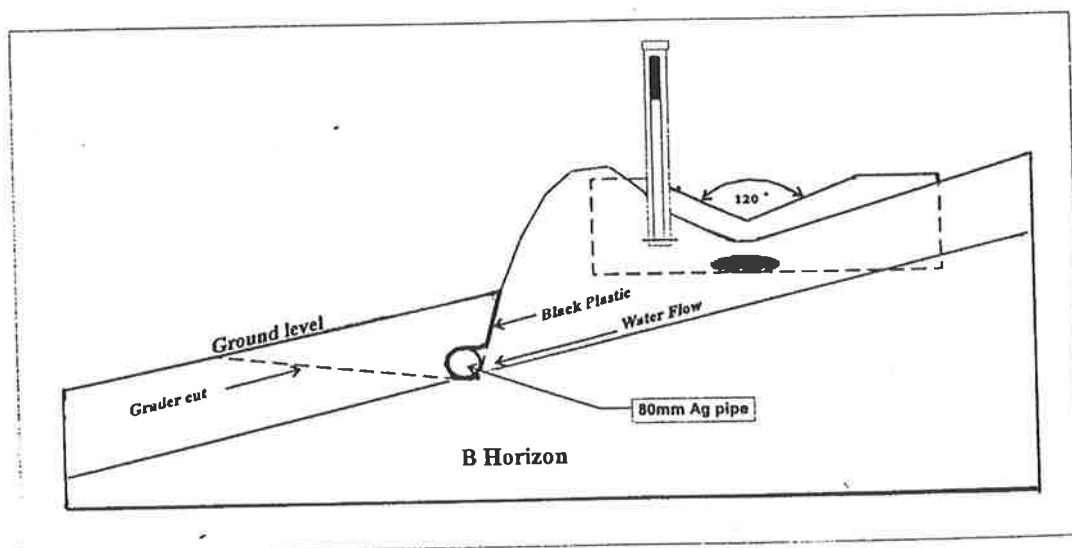


Fig 3.4 Design and placement of V-notch weirs.

Tipping buckets

Throughflow collected by the buried agricultural drains was directed to tipping buckets, housed in covered sumps, where a recorder attached to a data logger recorded each tip (2 L) (Plate 3.6).

3.4.2 Calculating site areas

For layout details see Figure 3.1. Drain flow on plots 4/5, 9/10 and 14/15 is from right to left of the diagram. On all other plots drain flow is from left to right.

A digital elevation model (DEM) at 1 m x 1 m node spacings was created from aerial photogrammetric data collected by the University of South Australia (David Bruce, pers. comm.). Drains, and the barriers to lateral flow and end flow (perpendicular to the drains), were super-imposed on the DEM as arcs, with 1 m node spacings. Using an aerial photograph underlay, the weir positions were marked on each arc (drain). These positions were field checked for accuracy.

The shape of the land surface determines the flow of water across it. In the Arcinfo® module 'GRID', hydraulic analysis tools are employed to describe the physical characteristics of a surface. Using the above DEM as input, a drainage system was delineated.

When the ARC coverage of drains and dividers etc., was overlain on this drainage network in ARCEDIT, polygons were completed which represented the catchment area within each pasture treatment plot that contributed flow to each V-notch weir. The characteristics (e.g. area, perimeter) of these polygons were then quantified. This methodology allowed all rainfall quantities and water movements to be standardised to that per unit area for comparison across sites.

3.4.3 Water quality

Water samples were collected during or following rainfall events, from tipping buckets and weirs. Samples were analysed for pH, electrical conductivity (EC), dissolved organic carbon (DOC), N, P, and a range of metals. Further details are in Chapter 6.

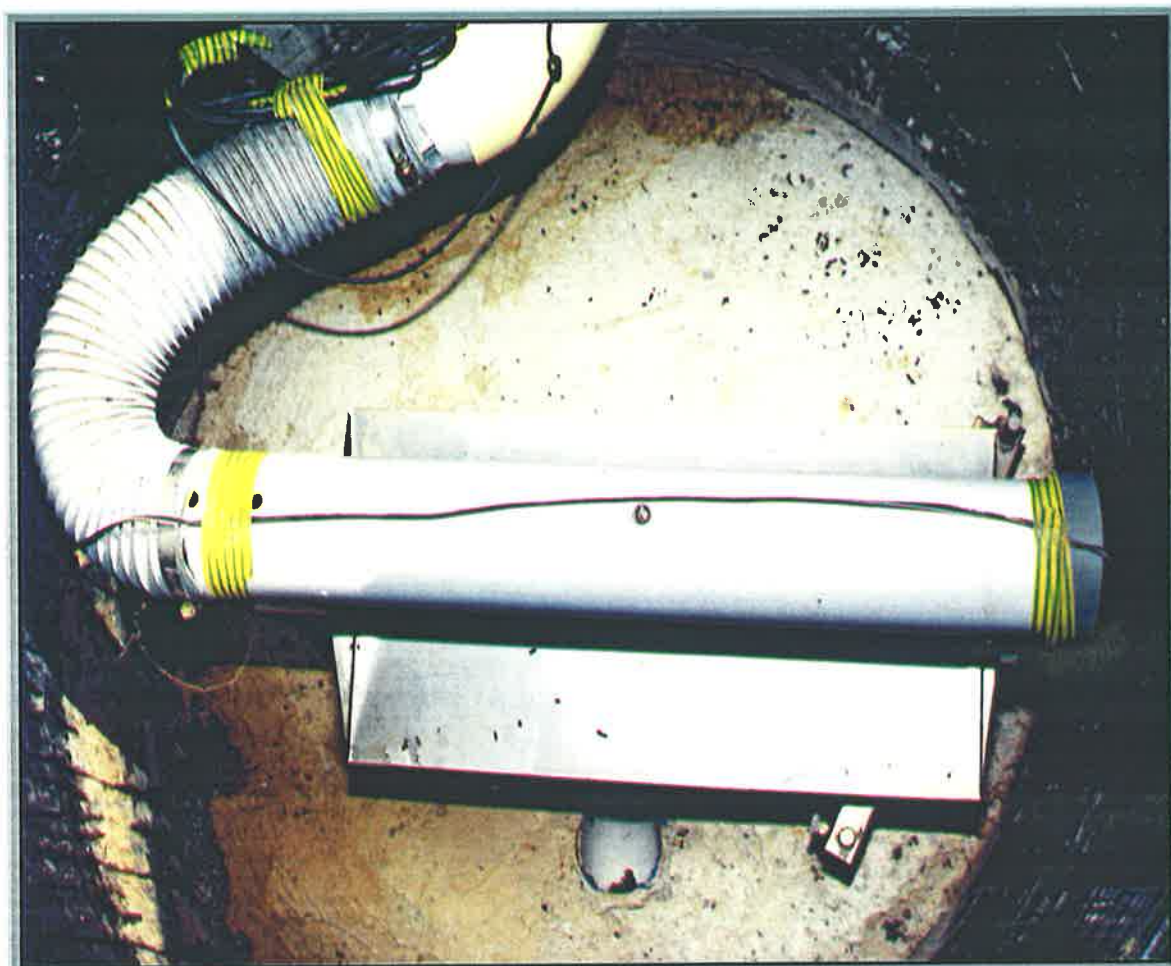


Plate 3.6 Tipping bucket recorder in 'Ribloc' sump, with concrete base and drain hole leading to sump drain.

3.4.4 Soil water content

Regular monitoring (2 weekly in summer, weekly in winter) of the soil water storage in the profile was carried out from March 1996, using a neutron moisture meter (CPN Corporation, California, USA; Greacen 1981). Readings were taken at 9 depths to 170 cm (10, 30, 50, 70, 90, 110, 130, 150, 170 cm).

Installation of access tubes

A single aluminium access tube (5 cm OD) sealed at the base, was installed in the centre of each plot to a depth of 170 cm (this provided five tubes at each slope position). Holes for the access tubes were drilled with a specially prepared auger which provided a tight fit between soil and access tube. Using the method of Prebble *et al.* (1981), a slurry of kaolin, cement and water was poured down each hole and the access tube then pushed to

its limit in the soil. Excess length of access tube remaining above ground was cut to a standard 10 cm.

Calibration of the neutron probe

Soil cores were taken from each plot and sectioned to correspond to probe reading depths (core size: 200 mm length and 31.5 mm radius). Additional measurements were taken at later core sampling events eg. root length cores (note that root core samples were 200 mm in length and 33.5 mm radius). Core sites were along a transect from upper to lower drain on each plot, although the dimensions of the drilling rig precluded sampling within c. 2.5 m of any drain.

After being oven-dried at 105° C for 24 hours, gravimetric and then volumetric soil water contents (SWC) were calculated. Volumetric SWC measurements were paired with corresponding probe readings and a linear relationship established that linked volumetric SWC to probe reading values (see Lolicato 2000). Separate calibration curves were determined for each soil horizon ($R^2 = >0.8$). The bulk density of each sample was also calculated at this time from dry soil mass and known soil volume.

Volumetric soil moisture for the 0-10 cm depth could not be measured with great certainty due to the nature of the recording device (ie. the attenuated neutrons are recorded from a sphere of influence which in this instance would include an area above the soil surface). As in other studies (eg. Ridley *et al.* 1997) the volumetric soil moisture of this surface layer was assumed to be the same as that for the 10-30 cm depth. The two displayed a similar texture although organic matter was greater in the 0-10 cm sample, a difference reflected in their respective bulk densities (0-10 cm, 1.36; 10-30 cm, 1.78 g/cm³). Subsequent sampling from 10 cm and 30 cm, over a range of soil moisture conditions, led to the establishment of a common calibration curve for the A and E horizons ($R^2 = 0.9132$).

From soil moisture plots established using these criteria (supported by field notes made at recording times), the surface soil moisture contents seem to broadly match expected patterns. During the wettest and driest periods, this relationship is strongest, however there are likely to be varying degrees of divergence in those periods just prior to and

following this 'uniform' wetness/dryness when the estimates will be less accurate. They do however indicate trends and are sufficient to show the changes over time within the surface soil layer.

Volumetric soil moisture for each layer (incremental depth) was converted to mm of stored soil water and summed for totals showing amounts above and below 35 cm (ie. an approximate level of the A/B horizon interface). A horizon SWC (mm) therefore equals volumetric SWC x 350 (mm) and profile SWC is equal to the sum of soil water (mm) held in each depth layer (Lolicato 2000).

3.5 Pasture establishment and maintenance

3.5.1 Sowing

The paired experimental sites were sprayed with glyphosate at 900g a.i/ha prior to direct drilling of seed in mid-May, 1995. A legume fertiliser (Hifert 6:16:0:10) was applied at the rate of 50 kg/ha at seeding.

Lucerne (*Medicago sativa* L.) cv. Aquarius was sown at 10 kg/ha following inoculation with a blend of acid tolerant rhizobia (WSM 922, WSM 879, MSUR52A). Aquarius was selected as a highly winter active cultivar, showing a rapid recovery after cutting/grazing (Lowe 1994), a rapid growth rate (Table 3.3), and importantly, was claimed to have the highest resistance to phytophthora root rot (R.Oram, pers. comm.), a disease of poorly drained soils. The graphs in Figure 3.5 show the growth patterns of one cultivar from each dormancy category, demonstrating that the main differences occur during the winter months. Aquarius is similar to CUF101 in this figure (Lowe 1994).

The second treatment comprised a mix of Sirosa phalaris (3 kg/ha), Dalkeith sub-clover (3.3 kg/ha), Trikkala sub-clover (3.3 kg/ha), Junnee sub-clover (3.4 kg/ha) and Balansa (2 kg/ha), for a total rate of 15 kg/ha. Balansa was included to extend the growing season of the sub-clovers, and the Sirosa phalaris as a winter active perennial grass (Oram & Culvenor 1994). Sirosa is a cultivar selected for reduced levels of the dimethyltryptamine alkaloids responsible for staggers and sudden death syndrome in

cattle and sheep (Lamp *et al.* 1990; Oram 1990), a problem experienced by the landowners in the past.

Seed was 'scratched' into the soil on May 15th 1995, with seed depth set at c. 25 mm. Germination was first noticed on May 28th, however by mid June it was apparent that much of the lucerne had failed to emerge through too deep seed placement, and growth was very patchy (Note: at the time of seeding, some patches along the contour banks were seeded by hand broadcasting onto the soil surface, with excellent emergence results and this indicated that non emergence was due to depth of seed placement). The lucerne sites were reseeded on July 5th with seed dropped directly onto the moist soil surface, and the subsequent establishment was successful.

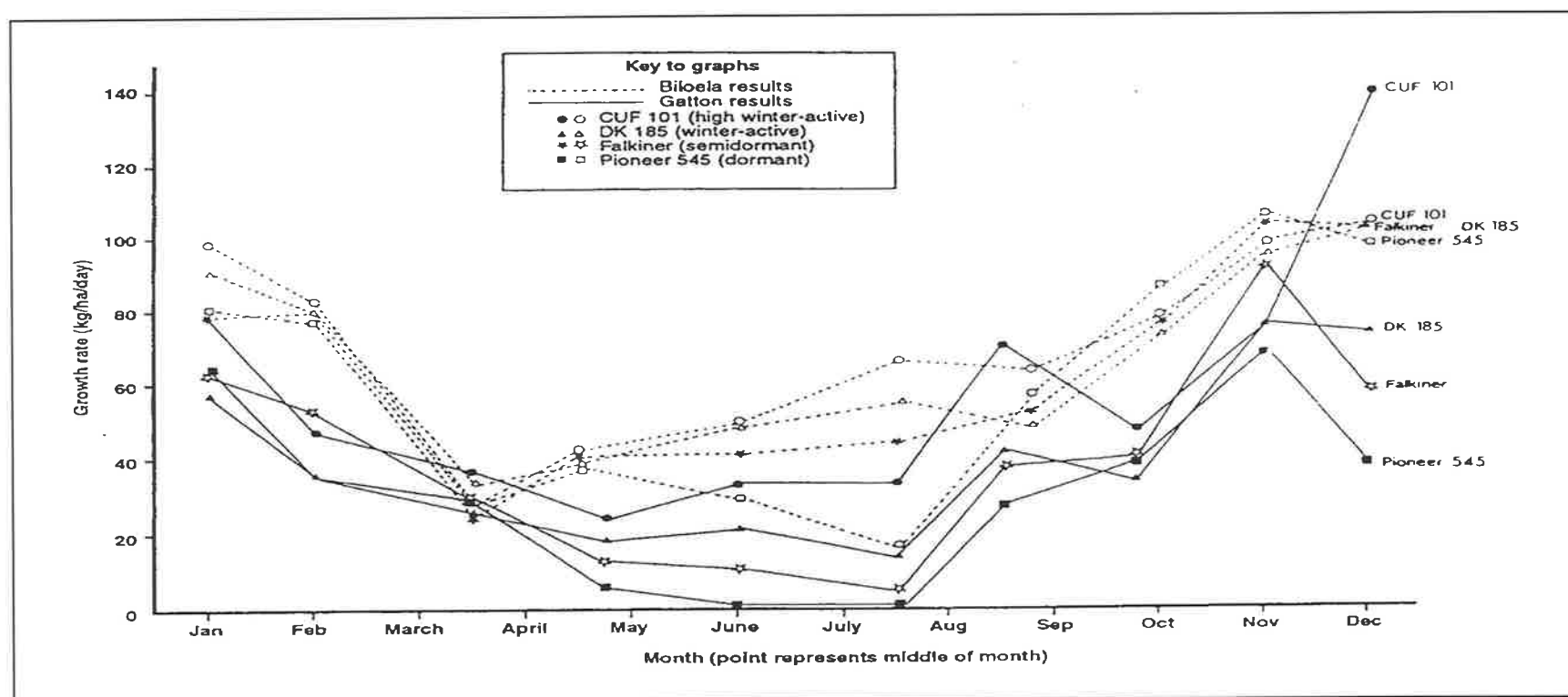
In mid August, the seedling lucerne was observed to have a reddish tinge to the foliage, possibly an indication of developing root problems due to poor drainage (Lowe 1994). In this stressed condition it was susceptible to insect damage, with some patches badly affected. However, by late September the stands had recovered and were growing well.

3.5.2 Pest control

Shortly after germination, red-legged earth mites (*Halotydeus destructor*) were noted on the site and treated on two occasions with omethoate at the recommended rate.

In the early establishment period, the seedling lucerne plants were badly damaged by a combination of brown pasture looper caterpillars (*Ciampa arietaria*), bluegreen aphid (*Acyrtosiphon kondoi*) and red-legged earth mite (RLEM). Alpha-cypermethrin at 10g a.i/ha successfully controlled these pests and was used for a subsequent outbreak of RLEM (again at 10g a.i/ha), this time in conjunction with Flumetsulam (20g a.i/ha) to control Salvation Jane.

Fig. 3.5 Graph indicating growth stages of lucerne with varying degrees of dormancy. CUF 101 in this illustration, most closely resembles Aquarius, the cultivar used in this study. Source: (Lowe 1994)



Following the above treatments, there were minor infestations of RLEM and blue-green aphids, but the damage to plants did not indicate any necessity for control measures. This was probably a result of cultivar selection for insect and disease resistance, and cv. *Aquarius* compares favourably with cv. *Hunter River*, one of the most common 'old' varieties that is still grown in South Australia (see Table 3.2).

Table 3.2 Properties of two lucerne cultivars (pest and disease resistance).

Cultivar	SAA	BGA	PA	CCR	PRR	St & Lep
Hunter River	S	S	S	MR	S	LR
Aquarius	HR	MR	LR	LR	HR	S

Key to abbreviations in headings

SAA	spotted alfalfa aphid
BGA	bluegreen aphid
PA	pea aphid
CCR	colletotrichum crown rot
PRR	phytophthora root rot
St & Lep	a leaf damage rating for stemphylium and leptosphaerulina leaf diseases

Key to pest disease ratings

HR	highly resistant
MR	moderately resistant
LR	low resistance
S	susceptible

3.5.3 Fertiliser application

In early April 1997, 1 tonne of 'Pasture Plus' (P 17%; S 9.7%) was applied to the experiment site, primarily to assess the movement of chemicals in subsequent rainfall events. This application rate translates to 28.3 kg/ha (2.83 g/m²) of P and 16.2 kg/ha (1.62 g/m²) of S.

3.5.4 Grazing management

The experimental sites were fenced to protect them from the relatively heavy grazing routine normally practised on this property. Following dry matter cuts, a high density of animals were introduced and kept on site until pastures had been grazed to an acceptable level, usually between 5 and 10 days. Control plots were grazed more often at a lower animal density, as part of the normal farm routine.

3.6 Production / density counts

3.6.1 Dry matter cuts

Pasture was harvested on four occasions (August & October 1996, April & October 1997) for the calculation of dry matter production. On the unfenced 'control' plots, herbage was cut from three pasture cages (600 mm x 600 mm) at each landscape position (ie. upper, mid and lower slope). Within the fenced area, a measured strip (10

m) was cut diagonally across each phalaris and lucerne plot with a mower, and the total fresh weight of herbage was recorded. Adjacent cocksfoot pasture within the fenced area was similarly collected with the mower. Subsamples were then taken from each bulk sample, placed in plastic bags and sealed. Subsequently these subsamples were removed from their bags and weighed, then oven dried at 70° C for 24 hours before re-weighing to determine moisture content. Within 48 hours of each pasture cut, the land-owner introduced stock to the site to graze the remaining herbage.

3.6.2 Plant density and pasture composition

At intervals throughout the study, plant density and groundcover of the pasture components was measured. A 0.36 m² quadrat was thrown 15 times per plot and at each throw, the number of the principal pasture plants (ie. lucerne, phalaris, cocksfoot) within the quadrat was recorded. At the same time, an estimate was made of the percentage of the various components of the pasture (including bare ground).

3.7 Root growth / measurement

3.7.1 Root distribution

In October 1996, ten soil cores were taken by push tube (33.5 mm radius) from each of the control plots and from a lucerne and a phalaris plot at each landscape position. Cores were taken on a diagonal transect across each plot between upper and lower drains. One core from each site was sectioned to correspond with neutron probe reading depths, sealed in plastic bags and used for additional NMM calibration. A second core was placed on a core tray, which was sealed and labelled for later detailed profile description. The remaining eight cores were cut into sections (0-10 cm, and then in 20 cm intervals) and placed into plastic bags which were sealed and labelled, and stored for root washing. All samples were stored at < 4°C prior to subsequent analysis.

Root washing was carried out using the hydro-pneumatic elutriation method described by Smucker *et al.* (1982) in equipment shown in Fig. 3.7. Two sieve sizes, 0.5 mm and 1.0 mm, are employed to retain roots using this technique. Soil pits excavated in April 1997 and December 1997 allowed a further analysis of root growth and distribution

including a qualitative description of root growth and penetration down the profile, and a relative count of roots per depth interval down the profile.

3.7.2 Analysis

Morphological measurements of length, area, and root length density etc were determined from scanned samples in the WinRHIZO™ package (Regent Instruments Inc., Quebec, Canada), using the 'line intersection method' (Tennant 1975).

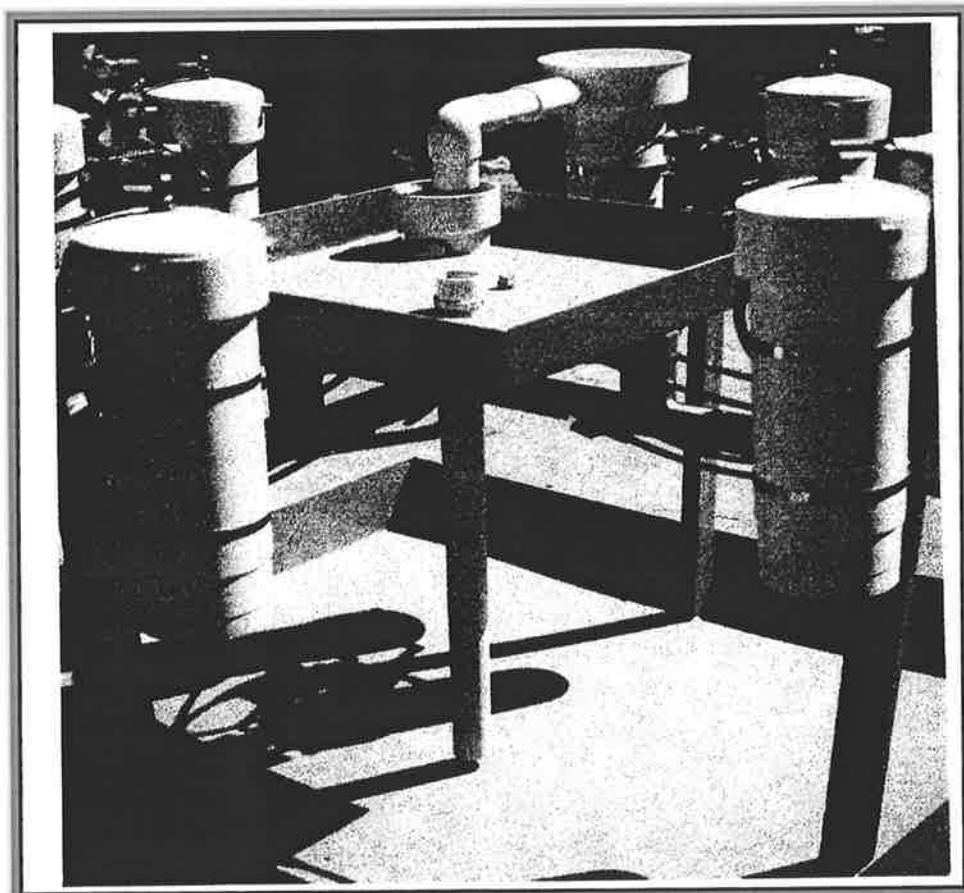


Fig. 3.7 Hydro-pneumatic elutriation equipment used for root washing.

Chapter 4

Results and Discussion – (General)

4.1 Climate

4.1.1 Temperature

The mean maximum and minimum monthly temperatures throughout the three-year study period were generally around the average (Table 4.1).

Table 4.1 Mean monthly temperatures (maximum/minimum °C) for the Keyneton site.

	1995		1996		1997		30 year av.	
	Max	Min	Max	Min	Max	Min	Max	Min
January	26.70	11.48	26.38	9.48	27.56	12.33	27.3	12.1
February	26.88	12.72	27.29	11.99	29.72	14.56	27.1	12.6
March	21.84	9.84	24.53	9.81	21.16	7.72	24.2	10.8
April	17.45	7.34	17.30	6.46	20.68	7.66	19.9	8.0
May	13.80	6.12	16.99	5.67	13.68	6.17	15.5	5.7
June	12.19	6.09	13.00	6.04	12.21	4.86	12.7	4.1
July	10.88	4.44	12.25	6.12	11.02	1.18	11.7	3.4
August	14.86	5.12	12.62	5.92	13.03	4.16	12.8	3.8
September	14.96	4.17	14.04	5.90	13.94	5.46	15.3	4.8
October	17.59	5.92	18.53	6.02	18.34	5.61	18.7	7.0
November	21.80	7.43	20.38	6.36	23.23	8.81	22.3	8.4
December	23.86	8.49	24.50	8.67	24.93	9.71	24.8	10.3

4.1.2 Rainfall

The annual rainfall was 15% below average in 1995, close to the average in 1996 and 27% below average in 1997 (Table 4.2). April to October rainfall followed a similar pattern, with a 14% reduction from the long-term average in 1995, a slight increase (2%) in 1996 and a substantial 40% decrease in 1997 (Table 4.2). The variable nature of the rainfall distribution during the study period (particularly in the years 1995 and 1997) is illustrated graphically in Fig. 4.1, along with Penman-Monteith evaporation data (Monteith 1965; FAO 1992).

In 1995, what could potentially have been a drought period from April to October, was saved by the fact that at least 5-10 mm of rain fell at almost weekly intervals during this time. Given the pattern of rainfall distribution and the limited potential for soil saturation as a result, drain flows were generally minimal.

In the following year (1996), April – May rainfall was well below average, with a late break to the season in June. This was followed by average rainfalls through to October, with sufficient excess over ET to contribute to the development of perched water tables and subsequent drain flows. In several instances there were short duration rainfall events >20 mm which, in association with a previously saturated A horizon, led to the generation of overland flow on a number of plots.

1997 was essentially a drought year. Following a promising break of season in May, the June – July rainfall (38 mm) was well below average (145 mm). August and September were around the average whilst October (1 mm) barely registered any rainfall.

Table 4.2 Monthly rainfall and potential evaporation (1995, 1996 and 1997) for the Keynes' catchment, and long-term average monthly rainfall and Class A pan evaporation for the town of Keyneton (all data in mm). Shaded areas show months where rainfall > PET.

Month	1995		1996		1997		Average	Average
	Rain	PET*	Rain	PET	Rain	PET	Rain (78 years)	Class A pan evaporation (30 years)**
January	48.4	119	47.2	123	13	101	18.3	139
February	15.4	102	7.8	77	58	84	22.7	108
March	13.2	75	29.8	63	3	62	19.5	97
April	37.2	40	10.2	29	2	40	36.3	53
May	37	16	10	23	61	16	63.4	30
June	67.4	13	132	31	26	12	64.4	23
July	121.0	7	84.4	13	12	18	79.6	26
August	9.2	17	79.6	20	91	25	76.4	32
September	48	37	73.4	37	69	33	64.1	49
October	58.2	53	53.4	61	1	70	50.5	73
November	9	81	14.8	77	27	73	28.5	89
December	4.6	89	6.6	95	39	99	25.1	119
April- October	378	183	443	214	262	214	434	286
Total	468.3	649	548.8	649	402	633	548.8	839

* PET is Potential Evaporation by the Penman-Monteith Equation.

** Source: Bureau of Meteorology data in Cox and Ashley (2000)

[Note: PET (Priestley & Taylor 1972) was 812 mm (1995), 781 mm (1996) and 778 mm (1997), calculated from on-site weather station data.]

4.1.3 Evaporation

Penman-Monteith annual potential evaporation (PET) for the catchment is > 600 mm (see Table 4.2) and generally there is only likely to be an excess of rainfall over evaporation in the winter/early spring period of the year. The winter surplus in 1995 was 205 mm, and in 1996 and 1997, 268 mm and 161 mm respectively. For the same periods, the annual deficits (ie. PET - Rainfall) were 181 mm (1995), 100 mm (1996) and 231 mm in 1997.

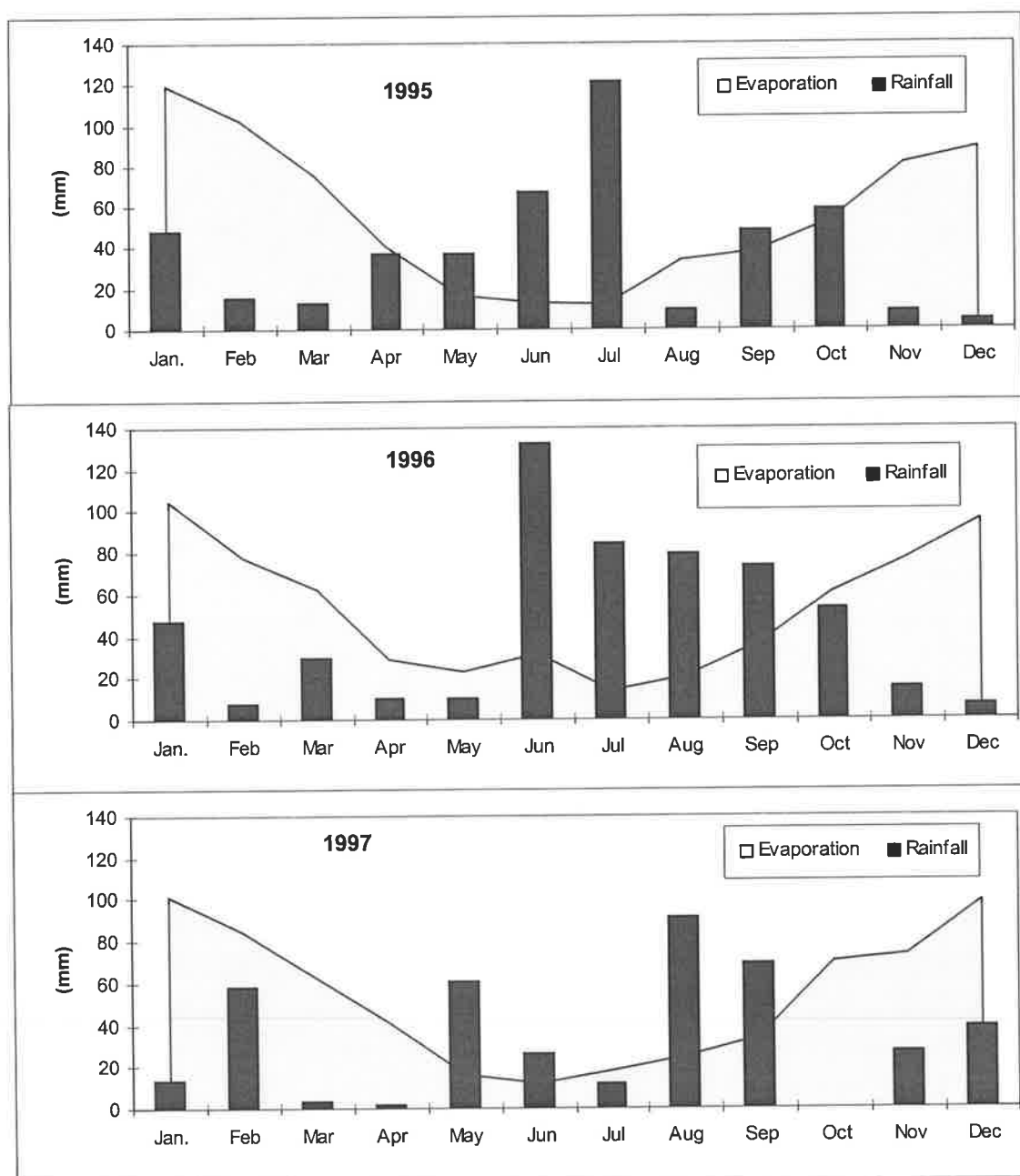


Fig. 4.1. Monthly rainfall (and evaporation) for the three years of the study in Keynes' catchment.

Maps published on the Bureau of Meteorology website indicate that annual point PET for Keyneton should be about 1600 mm, and average areal actual evapotranspiration around 400 mm (see: [<http://www.bom.gov.au/climate/averages/climatology/evapotrans/et.shtml>]). The climate of Keynes' catchment is moderated by its location in the Adelaide Hills. Average temperatures are substantially lower than those of the plains country with fog and cold winds common on site well into summer. Weather station data supporting a lower PET than predicted by the Bureau is attached as an appendix.

4.2 Soil and catchment characteristics

The catchment of approximately 260 ha is located in the most western part of the Murray-Darling Basin and drainage flows south-west out of the catchment then east to the Murray River (a major water supply source for Adelaide). Climate, soils and landscape features are typical of those throughout the Adelaide Hills. The native vegetation of the area (predominantly *Eucalyptus* spp.) was heavily cleared in the period 1850-1870 and is currently about 90% cleared. Regeneration is poor and most remnants are to be found on roadsides, drainage lines and parts of the catchment unsuitable for cropping. Topography of the area is undulating with maximum slope of 18%.

4.2.1 Soil types/profiles

A general soil survey showed the dominant soil types were Typic Palexeralfs (Soil Survey Staff 1996) on the upper slope positions grading to Aquic Palexeralfs on the mid to lower slopes and Natraqualfs on the lower slopes to flats. Xerorthents (shallow soils over rock) have formed at various positions down the toposequence (see Appendices for correlation between Australian Soil Classifications and the US Soil Taxonomy). Soils in the landscape are part of a catenary sequence of red through yellow to grey texture contrast soils (Northcote 1979).

A number of cores were taken across the study site (supported by the visual inspection of soil pits dug in some of the sites) for morphological description (Tables 4.3-4.5), and these showed the variations in soil colour and texture, as well as horizon depth and extent down the toposequence. Equally apparent was the same variability across any single slope position (eg. upper, mid or lower slope).

The upper slope (generally brown) very fine loamy sand A horizon ranged from 14-21 cm in depth and in the mid-slope where the texture was predominantly sandy loam, the variation was similar (13-22 cm), although the soil colour was tending to show the greyish indications of waterlogging. In the greyish sandy loams or very fine loamy sands of the lower slopes the range was from a shallow 10 cm, up to 28 cm, probably indicative of the variable deposition/erosion potential of this slope position. This variation in A horizon depth, is characteristic of duplex soils and Tennant *et al.* (1992) state that (in Western Australia) “it is thought to represent remnants of the drainage surface existing on the truncated laterite profile during early stages of duplex soil formation before mantling with sedimentary materials”.

In the E horizon, texture varied from sandy loam in the upper slopes, clayey sand or sandy loam at mid-slope and very fine loamy sand or sandy clay loam in the lower slopes. The lighter red and brown soil colours were common at all slope positions although there were examples of greyer soils and nodules of iron and/or manganese indicative of periodic saturation. This horizon generally extended to a depth between 30-45 cm, although in Site L4 (see above; 10 cm A horizon) it only went to 21 cm, and in Site M2 was completely absent (see Fig. 4.6 for Site locations).

The B horizon soils extending to depths between 0.5 and 1 metre were all heavy clay with a strong angular blocky structure. Reddish brown was the predominant colour of these soils with mottling and increasing yellow occurring in the mid and lower slopes, again indicating periodic waterlogging. The heterogeneity of these soils however, restricts conceptual modelling to generalisations eg. soil indicators suggest the potential for waterlogging of the B horizon increases as one moves downslope. In at least one toposequence this appears reversed, as mottling occurred in the upper slope (U4) whilst the B2 horizon of the lower slope (L4) was a uniform red.

In the C/B intergrade, the general reddish brown of the heavy clay B horizon gave way to the brownish yellow of the C horizon, with texture changing to silt, silty loam, sand, clayey sand, or fine sandy loam. The yellow/grey/white C horizon soils' upper boundary was from 60-100 cm depth, with the lower boundary as shallow as 82 cm or as deep as 143 cm, below which was massive white sandstone

Table 4.3 Morphological descriptions of three upper-slope soil cores from Keyneton.

Upper control (Uc) [cocksfoot]		Upper 2 (U2) [lucerne]		Upper 4 (U4) [phalaris]	
Soil horizon and colour	Soil texture & structure	Soil horizon and colour	Soil texture & structure	Soil horizon and colour	Soil texture & structure
A: Brown (7.5YR5/2) & dark brown (7.5YR3/2) moist	A: 0-14 cm, very fine loamy sand; massive structure	A: Light brown (7.5YR6/3) dry & dark brown (5YR4/2) moist	A: 0-15 cm, very fine loamy sand; massive structure	A: Brown (7.5YR5/2) dry & dark brown (7.5YR3/2) moist	A: 0-21 cm, very fine loamy sand; massive structure
E: Light reddish brown (5YR6/3) dry & reddish brown (5YR5/3) moist	E: 14-33 cm, gravelly sandy loam; massive structure	E: Light brown (7.5YR6/3) dry & reddish brown (5YR5/3) moist	E: 15-42 cm, sandy loam; massive structure	E: Pinkish grey (7.5YR7/2) dry & brown (7.5YR5/2) moist; weak red (10R4/4) dry nodule (3%)	E: 21-37cm, sandy loam; weak angular blocky structure
B21: Red (2.5YR5/6) dry & red (2.5YR4/6) moist	B21: 33-63 cm, heavy clay; strong angular blocky structure				
B22: Reddish brown (2.5YR5/4) Dry & reddish brown (2.5YR4/4) moist	B22: 63-83 cm, heavy clay; strong angular blocky structure	B22: Reddish brown (2.5YR4/4) dry & dark reddish brown (2.5YR3/4) moist	B22: 42-65 cm, heavy clay; moderate angular blocky structure	B22: Mixture of red (10R4/6, 50%) and brown (7.5YR5/4, 30%) dry & dark red (10R3/6) and dark brown (7.5YR4/4) moist	B22: 37-59 cm, heavy clay; strong angular blocky structure
C/B: Brownish yellow (10YR6/6) dry and moist for C horizon, and light weak red (5R4/3) dry and dusky red (5R3/3) moist for the B patchy	C/B: 83-101 cm, silt texture for the C and light clay for the B patchy; moderate angular blocky structure for the B patchy			C/B: Mixture of red (10R4/6, 50%) and brown (7.5YR5/4, 10%) dry & dark red (10R3/6) and dark brown (7.5YR4/4) moist for the B horizon, massive for C and mixture of strong brown (7.YR5/6, 30%) and light grey (7.5YR7/0, 10%) dry for the C horizon	C/B: 59-90 cm, medium heavy clay for the B and clayey sand for the C; strong angular blocky structure for the B and for the B horizon, massive for C
C: Mixture of yellow (2.5Y7/6, 60%) and white (2.5Y8/0, 40%) dry	C: 101-112 cm, silty loam; massive structure	C: Mixture of brownish yellow (10YR6/6, 70%) and white (10YR8/2, 30%) dry	C: 65-83 cm, clayey sand; massive structure	C: Mixture of strong brown (7.5YR5/6, 80%) and light grey (7.5YR7/0, 20%) dry	C: 90-100 cm, clayey sand; massive structure



Fig. 4.3 Soil profile of pit dug in the upper-slope position

Table 4.4 Morphological descriptions of three mid-slope soil cores from Keyneton.

Mid slope control (Mc) [cocksfoot]		Mid slope 1 (M1) [lucerne]		Mid slope 2 (M2) [phalaris]	
Soil horizon and colour	Soil texture & structure	Soil horizon and colour	Soil texture & structure	Soil horizon and colour	Soil texture & structure
A: Greyish brown (10YR5/2) dry & dark greyish brown (10YR4/2) moist	A: 0-15 cm, very fine loamy sand; massive structure	A: Dark greyish brown (10YR4/2) dry & dark greyish brown (10YR3/2) moist	A: 0-13 cm, sandy loam; weak subangular blocky structure	A: Brown (7.5YR5/3) dry & dark brown (7.5YR3/2) moist	A: 0-22 cm, sandy loam; massive structure
E: Light reddish brown (6.5YR6/3) dry & reddish brown (7.5YR5/3) moist; yellowish red (5YR5/6) dry iron nodule (20%)	E: 15-33 cm, clayey sand; massive structure	E: Light reddish brown (5YR6/3) dry & reddish brown (5YR4/3) moist	E: 13-31 cm, sandy loam; weak subangular blocky structure		
B22: Reddish brown (2.5YR4/4) dry & dark reddish brown (2.5YR3/4) moist	B22: 33-62 cm, heavy clay; strong angular blocky structure	B21: Mixture of weak red (10R4/4, 80%) and reddish yellow (7.5YR6/6, 20%) dry and dusky red (10R3/4) and strong brown (7.5YR5/6) moist B22: Mixture of reddish brown (7.5YR6/6, 60%) and reddish brown (2.5YR4/3, 40%) dry & strong brown (7.5YR5/6) and dark reddish brown (2.5YR3/3) moist	B21: 31-60 cm, heavy clay; strong angular blocky structure B22: 60-79 cm, heavy clay; strong angular blocky structure	B22: Mixture of light yellowish brown (10YR6/4, 55%) and weak red (10YR4/3, 45%) dry & yellowish brown (10YR5/4) and dusky red (10YR3/2) moist	B22: 22-56 cm, medium heavy clay; moderate angular blocky structure
C: Mixture of yellowish brown (10Y5/6, 90%) and white (2.5Y8/0, 10%) dry	C: 62-120 cm, sand; massive structure	C: Mixture of reddish brown (7.5YR6/6, 70%), reddish brown (2.5YR4/3, 20%) and light grey (2.5YR7/0, 10%) dry	C: 79-140 cm, clayey sand; massive structure	C/B: Weak red (10R4/3, 60%) and dusky red (10YR3/2) moist for the B horizon, and brownish yellow (10YR6/6, 40%) dry for the C horizon C: Mixture of reddish grey (5YR5/2, 50%), light grey (2.5Y7/0, 35%) and olive yellow (2.5Y6/6, 5%) dry	C/B: 56-68 cm, fine sandy loam; moderate angular blocky structure for the B and massive for C C: 68-130 cm, fine sandy loam; massive structure

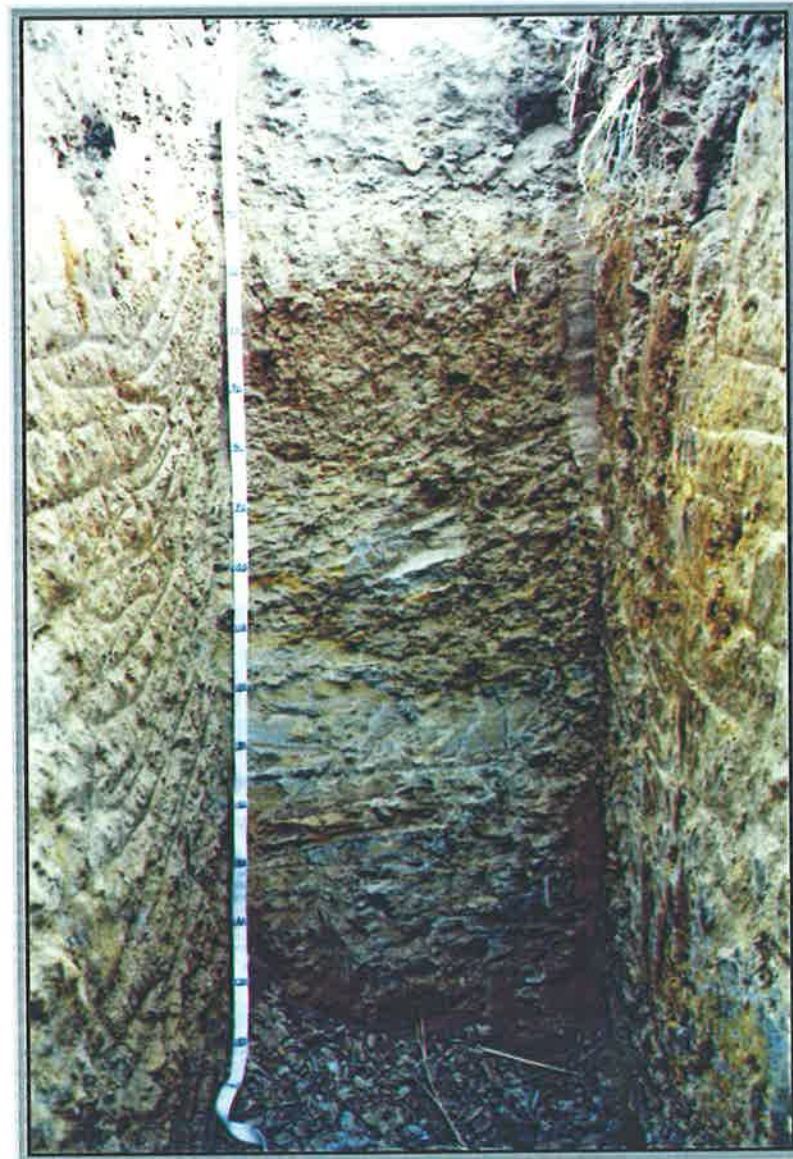


Fig. 4.4 Soil profile of pit dug in mid-slope position

Table 4.5 Morphological descriptions of three lower slope soil cores from Keyneton.

Lower slope control (Lc) [cocksfoot]		Lower slope 3 (L3) [phalaris]		Lower slope 4 (L4) [lucerne]	
Soil horizon and colour	Soil texture & structure	Soil horizon and colour	Soil texture & structure	Soil horizon and colour	Soil texture & structure
A: Brown (10YR5/3) dry & very dark brown (10YR3/3) moist	A: 0-19 cm, very fine loamy sand; massive structure	A: Greyish brown (10YR5/2) dry & very dark greyish brown (10YR3/2) moist	A: 0-28 cm, very fine loamy sand; massive structure	A: Greyish brown (10YR5/2) dry & very dark greyish brown (10YR3/2) moist	A: 0-10 cm, sandy loam; massive structure
E: Very pale brown (10YR7/3) dry & pale brown (10YR6/3) moist; dark reddish brown (5YR3/2) dry; nodule of manganese and iron (20%)	E: 19-45 cm, gravelly very fine loamy sand; massive structure	E: Light grey (10YR7/2) dry & greyish brown (10YR5/2) moist	E: 28-42 cm, very fine loamy sand; massive structure	E: Light reddish brown (5YR6/3) dry and reddish brown (5YR5/3) moist	E: 10-21 cm, sandy clay loam; massive structure
B21: Yellowish brown (10YR5/4) dry and dark yellowish brown (10YR4/4) moist; weak red (10R4/4) dry mottle (35%) B22: Yellowish brown (10YR5/4) dry and dark yellowish brown (10YR4/4) moist; weak red (10R4/4) dry mottle (10%)	B21: 45-75 cm, heavy clay; strong angular blocky structure B22: 75-106 cm, heavy clay; strong angular blocky structure			B22: Red (2.5YR4/6) dry and dark red (2.5YR3/6) moist	B22: 21-51 cm, heavy clay; strong angular blocky structure
C/B: Yellowish brown (10YR5/8) dry and moist for the C horizon, light reddish brown (5YR6/3) dry and reddish brown (5YR5/3) moist for the B horizon	C/B: 106-146 cm, light clay for the B horizon and silty loam C; moderate angular blocky structure for B and massive for C horizon	C/B: Brownish yellow (10YR6/6) dry and yellowish brown (10YR5/6) moist for the B horizon; grey (10YR6/1) dry and moist for the C horizon; dark reddish brown (2.5YR3/4) dry mottle C: Mixture of grey (10YR6/1) and light red (10YR6/6) moist R: White (2.5Y8/0)	C/B: 42-67 cm, medium clay for B horizon and sand for the C; moderate angular blocky structure for B and massive for C horizon C: 67-82 cm, sand R: 82-115 cm, sandstone	C/B: Yellowish brown (10YR5/8) dry and dark yellowish brown (10YR4/6) moist for the C horizon; red (2.5YR4/6) dry and dark red (2.5YR3/6) moist for the B horizon C: Mixture of yellowish brown (10YR5/8) and white (10YR8/1) dry and dark yellowish brown (10YR4/6) and white (10YR8/1) moist	C/B: 51-70 cm, medium heavy clay for the B horizon and sandy loam C; moderate platy structure for the B and massive for C horizon C: 70-143 cm, sandy loam; massive structure



Fig. 4.5 Soil profile of pit dug in lower slope position.

4.2.2 Soil properties

The following table (Table 4.6) summarises the results of a number of soil properties at the different slope positions.

Table 4.6. Typical soil chemical properties on upper, mid and lower slope

Horizon	Depth (cm)	EC (mS/cm) 1:5 soil:water ratio	pH	Particle size			Exchangeable cations				Sum cations	CEC (NH ₄)	ESP (%)
				Clay	Silt (%)	Sand	Ca	Mg	Na	K	cmol(+)/kg		
<i>Crest to upper slope</i>													
A1	0-14	0.059	6.2	10	15	74	5.7	1.0	0.1	0.5	7.3	8.0	1.3
E	14-33	0.014	6.7	14	18	68	2.4	0.9	0.2	0.2	3.7	3.8	5.3
B21	33-63	0.023	7.1	35	26	39	4.3	2.9	0.3	0.3	7.8	8.0	3.8
B22	63-83	0.036	7.2	29	31	40	4.3	3.0	0.4	0.3	8.0	7.6	5.3
B patchy	83-103B	0.044	7.2	16	40	44	2.0	1.9	0.3	0.2	4.3	4.0	7.5
C patchy	83-103C	0.042	7.3	21	24	56	2.5	2.1	0.3	0.2	5.1	5.0	6.0
C	101-112	0.045	7.6	8	13	79	1.0	1.2	0.3	0.1	2.6	2.9	10.3
<i>Mid slope</i>													
A	0-15	0.054	6.4	9	11	80	4.8	0.9	0.1	0.5	6.3	7.5	1.3
E	15-33	0.013	6.5	9	14	77	1.9	0.4	0.1	0.2	2.7	3.6	2.8
B2	33-62	0.021	6.6	47	15	39	3.6	3.6	0.3	0.3	7.7	9.6	3.1
C	62-120	0.027	6.4	7	19	74	0.6	0.9	0.1	0.1	1.8	1.9	5.2
<i>Lower slope</i>													
A1	0-19	0.028	6.0	10	12	79	3.3	0.5	0.2	0.3	4.3	6.3	3.2
E	19-45	0.045	6.9	15	12	73	1.2	1.1	0.5	0.2	3.1	3.3	15.2
B21	45-75	0.190	6.6	62	11	27	3.9	6.0	2.5	0.5	12.9	13.8	18.1
B22	75-106	0.215	7.3	37	31	31	3.3	4.7	2.0	0.4	10.3	9.1	22.0
C/B	106-146	0.126	7.8	19	36	45	1.7	2.2	1.0	0.2	5.1	5.4	18.5

Particle size distribution

The mean particle size distribution trend shown (Table 4.6) is consistent with other reports on texture contrast soils in the literature, with a marked increase in clay content of the B21/B22 horizon in comparison to that of A/E horizon soils. However in both Hook's (1990) Canberra study and that of Abdur Rab *et al.* (1987) in Bundoora, Victoria, clay content was more evident in the C horizon (19-50% and c.70% respectively), whilst in Western Australian soils the A horizon clay content was usually <5% (Tennant *et al.* 1992).

Keyneton soils generally have a fine sandy loam A horizon with about 12% clay content, over a leached E horizon that varies from gravelly sandy loam through clayey sand to very fine loamy sand. Clay content is higher in both A and E horizon soils in the lower slopes than in the mid and upper slopes and this could be due to material washed down-slope or less eluviation in the lower slopes. The E horizon overlies a B21 horizon that contains a highly variable amount of illuviated clay. The B22, which is more likely to be evident in most profiles, has the highest average clay content and is the most variable. In the C horizon, the highest clay contents were in the upper slopes, whilst the lower slopes were in the low area of the range. In general terms though, the variability shown, presents substantial difficulties in trying to model any horizon.

Bulk density

Generally the bulk densities of the Keyneton soils (Table 4.7) are higher than those of a range of other texture contrast soils in the literature (eg. Talsma 1983; Abdur Rab *et al.* 1987; Hook 1990; Tennant *et al.* 1992; Kutilek and Nielsen 1994). This is particularly the case in relation to the B horizon soils with a range from around 1.3 Mg/m³ to a maximum of 2.51 Mg/m³ (at around 1 metre depth in the lower slope position) (see Table 4.7).

Brouwer and van der Graaff (1988) reported two samples from the lower BC/upper C horizon in their Victorian study, that had bulk densities of 2.5 Mg/m³. They attributed to this to "very high iron cementation", which may also apply in Keyneton soils where there is noticeable iron/manganese nodulation. Hook's (1990) weathering parent

material C horizon also approached the upper limits of C horizon bulk density at Keyneton. At all profile depths other than within the A horizon and the deeper part of the C horizon, bulk density increased from upper to lower slope

Tennant *et al.* (1992) reported that A horizon bulk densities in WA often exceeded 1.8 Mg/m^3 which could be expected in apedal sandy soils but not in pedal clays. Even so, all the figures from Keyneton put the soils in the range of densities ($>1.4 \text{ Mg/m}^3$) considered to restrict root elongation (Tennant *et al.* 1992). The relative ranges of the bulk densities in Table 4.8 again illustrate the extreme variability inherent in these soils throughout the profile when compared to other studies.

Bulk density measurements from a number of studies in Keynes catchment all show measurements $>2.0 \text{ Mg/m}^3$ (see Appendix). Although the coring procedure used to collect the samples recorded in Table 4.7 may have been responsible for additional compaction and some anomalous measurements (eg. 2.51 Mg/m^3), Keyneton soils do show exceptionally high bulk densities. In other studies showing similar high measurements, bulk density samples were taken by rings (4.7 cm diameter and 5 cm height; Hugh Brown, pers. comm.) or steel cores (7.2 cm diameter and 5 cm height; McMurray 1994), and these samples were less likely to be compacted.

Table 4.7 Mean bulk densities (and their respective ranges) of soils, in 20 cm increments, for three positions down a toposequence.

(cm)	Mean bulk density (Mg/m^3)			Range		
	upper slope	mid slope	lower slope	upper slope	mid slope	lower slope
0-20	1.50	1.57	1.49	1.42-1.59	1.40-1.74	1.44-1.68
20-40	1.74	1.72	1.63	1.71-1.79	1.43-1.89	1.49-1.77
40-60	1.66	1.80	1.82	1.41-1.84	1.71-1.95	1.69-2.02
60-80	1.63	1.74	1.97	1.31-1.82	1.66-1.87	1.76-2.24
80-100	1.67	1.83	2.03	1.44-1.89	1.77-1.98	1.80-2.51
100-120	1.82	1.93	2.01	1.66-1.92	1.81-2.01	1.78-2.25
120-140	1.86	1.82	1.98	1.68-1.95	1.81-1.82	1.76-2.13
140-160	1.89	1.89	1.84	1.72-2.01	1.88-1.90	1.70-1.98

The A horizons had lower bulk densities (1.5 to 1.6 Mg/m^3) than the subsurface horizons (Fig. 4.6) due to the presence of soil organic matter and grass roots. On the upper slopes, bulk density increased to 1.74 Mg/m^3 by 0.3 m depth, decreased to 1.63

Mg/m³ by 0.7 m then gradually increased with depth to 1.89 Mg/m³ by 1.6 m. Mid-slope bulk density changes with depth followed a similar pattern to those of the upper slopes. Bulk density increased to 1.80 Mg/m³ by 0.5 m, decreased to 1.74 Mg/m³ by 0.7 m then generally increased with depth to 1.89 Mg/m³ by 1.6 m. On the lower slopes, bulk density increased with depth to 2.03 Mg/m³ by 0.9 m then decreased to 1.84 Mg/m³ by 1.6 m.

Table 4.8 Bulk densities of a range of texture contrast soils (Mg/m³)

Horizon	Study				
	Keyneton	Tennant <i>et al.</i> (1992) WA	Hook (1990) ACT	Talsma (1983) ACT	Abdur Rab <i>et al.</i> (1987) Victoria
A	1.39 – 1.89	1.30 – 1.56	1.33 – 1.61	1.18 – 1.71	1.23 – 1.53
B	1.31 – 2.51*	1.54 – 1.72	1.29 – 1.80	1.47 – 1.75	1.47 – 1.64
C	1.43 – 2.25		1.43 – 2.01	1.57 – 1.69	

* this figure is highly questionable. The next highest measurement in this horizon was 2.25 Mg/m³.

Given the inverse relationship between bulk density and porosity, the above values indicate generally reduced porosities of the mid and lower slopes in comparison to those of the upper slopes. This in turn directly affects the water holding capacity of these soils, particularly below the A horizon. Results indicated that in the top 35 cm of the profile, potential soil water storage at saturation of 120-140 mm, was likely regardless of toposequence position. From 35 cm to 170 cm, the profile in the upper slope would hold between 400-420 mm of soil water, in the mid-slope from 340-360 mm and in the lower slope from 250-285 mm.

Hydraulic conductivity

As in Williams (1983) there was a significant effect of horizon on K_s, with A horizons (range = 0.2 - 0.9 m/day) having values an order of magnitude greater than B horizons (0.007 - 0.09 m/day) and C horizons (0.02 - 0.2 m/day) (Table 4.9). The A and E horizons had sandy loam textures and both were expected to have relatively high K_s. However, the massive structure and high bulk densities in the E horizons compared to the A horizons caused much lower K_s values in the E horizons. K_s of the E horizons was similar on the lower and mid slopes but much lower on the upper slope. Neither between-site nor site-by-horizon differences in K_s were statistically significant (at P = 0.05). K_s of the B horizon was high for a clay soil (with the exception of the lower

slope) and indicative of a soil with many fine to medium macropores (Kirkby *et al.* 1997). Variability in K_s was greatest in mid slope plots (standard deviation was almost as high as the mean) and least on the lower slope plots. Lowest K_s was measured in sodic B horizons. K_s of the C horizon was lower than the B horizon on the upper and mid slopes but higher on the lower slope. K_s in C horizons increased down the toposequence from the upper slope with hard saprolite in the C horizon, to the lower slope, which had softer saprolite.

Table 4.9. Means and standard deviations of hydraulic conductivity (m/day)

Horizon	Lower slope	Mid slope	Upper slope	Mean
A	0.406 ± 0.137	0.204 ± 0.094	0.896 ± 0.820	0.532 ± 0.511
E	0.042 ± 0.015	0.171 ± 0.153	0.059 ± 0.017	0.111 ± 0.097
B	0.007 ± 0.000	0.092 ± 0.046	0.084 ± 0.021	0.069 ± 0.040
C	0.075 ± 0.077	0.062 ± 0.018	0.017 ± 0.001	0.078 ± 0.068

4.3 Calculation of Site areas

From the drainage system defined in ARCEDIT, the following areas (Table 4.10) were calculated for the various polygons representing treatment plots. To assess the accuracy of the ARC representation of the field situation, a number of distances between fixed points (eg. bank intersects) were compared on the ground and from the computed model. Differences were only minimal (c. 0.1 m over a distance of > 40 m), and are unimportant at this scale.

Given the subtle changes in micro-topography within each plot, the digital elevation model and subsequent delineation of drainage, were invaluable tools. This allowed for the defining of a cut-off line running from each V-notch weir to the upper drain (in effect a sub-catchment boundary), beyond which runoff was excluded, and a similar boundary (where needed) to define the alternate end of some plots (see Fig. 4.6).

Table 4.10. Drain slope position, type and area (m²) and soil classification

Plot #	Plot	Drain	Slope position	Drain type	Drained area (m ²)	Soil classification
1	Uc	7i/o	Upper	Reverse interceptor	1281	Chromosol
2	U1	4i/o	Upper	Reverse interceptor	1769	Chromosol
3	U2	1i/o	Upper	Reverse interceptor	1536	Chromosol
4	U3	ot1	Upper	Tube drain and contour bank	1856	Chromosol ¹
5	U4	ot4	Upper	Tube drain and contour bank	1964	Chromosol
6	Mc	8i/o	Mid	Reverse interceptor	1536	Chromosol
7	M1	5i/o	Mid	Reverse interceptor	1678	Dermosol
8	M2	2i/o	Mid	Reverse interceptor	1295	Chromosol
9	M3	ot2	Mid	Tube drain and contour bank	1242	Sodosol
10	M4	ot5	Mid	Tube drain and contour bank	2103	Chromosol
11	Lc	9i/o	Lower	Reverse interceptor	2000	Sodosol
12	L1	6i/o	Lower	Reverse interceptor	1981	Sodosol
13	L2	3i/o	Lower	Reverse interceptor	1109	Sodosol
14	L3	ot3	Lower	Tube drain and contour bank	877	Sodosol
15	L4	ot6	Lower	Tube drain and contour bank	1360	Sodosol

¹ the drain also intersected Sodosols

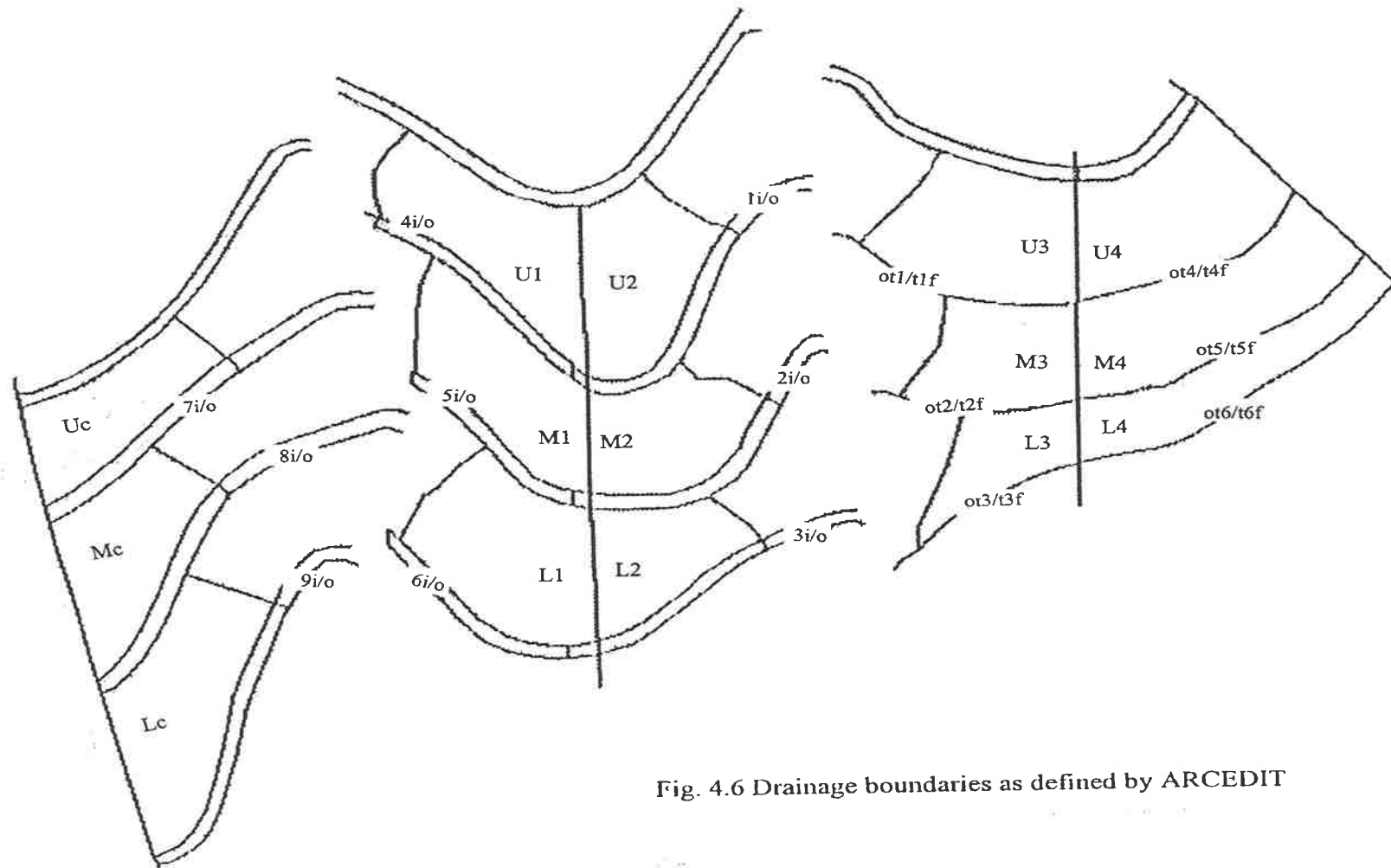


Fig. 4.6 Drainage boundaries as defined by ARCEDIT

4.4 Drain function

4.4.1 Drainflow

Over the period of this study, drainflows were extremely variable in both the volume of runoff transported and the relative percentages moving as overland flow and/or throughflow (Table 4.11). In the two dry years (1995, 15% below average rainfall, and 1997, 27% below) when mean drainflow was 1.1 and 1.2% of period rainfall, almost no overland flow was recorded. In 1996, a year of average rainfall, the mean drainflow was 5.6% of annual rainfall. Overland flow was recorded from eleven (of fifteen) drains and was often a substantial component of total runoff (see Tables 4.12-4.16 for more detail).

Figure 4.7 shows the extreme variability in flow between drains from 1995 to 1997. Flows were highest in 1996, the wettest year, with up to 137 mm (25% of annual rainfall) measured in drain 11 (a reverse interceptor drain on the lower slope; Table 4.10). About 80% of this flow was throughflow. High flows were also measured in 1996 in drain 12 (132 mm in a reverse interceptor drain on the lower slope), and 7 (104 mm in a reverse interceptor drain on the mid slope) but throughflow was only a small component of the total flow in these two drains. In some drains (4, 5, 6, 8, and 13) nearly all flow each year was throughflow.

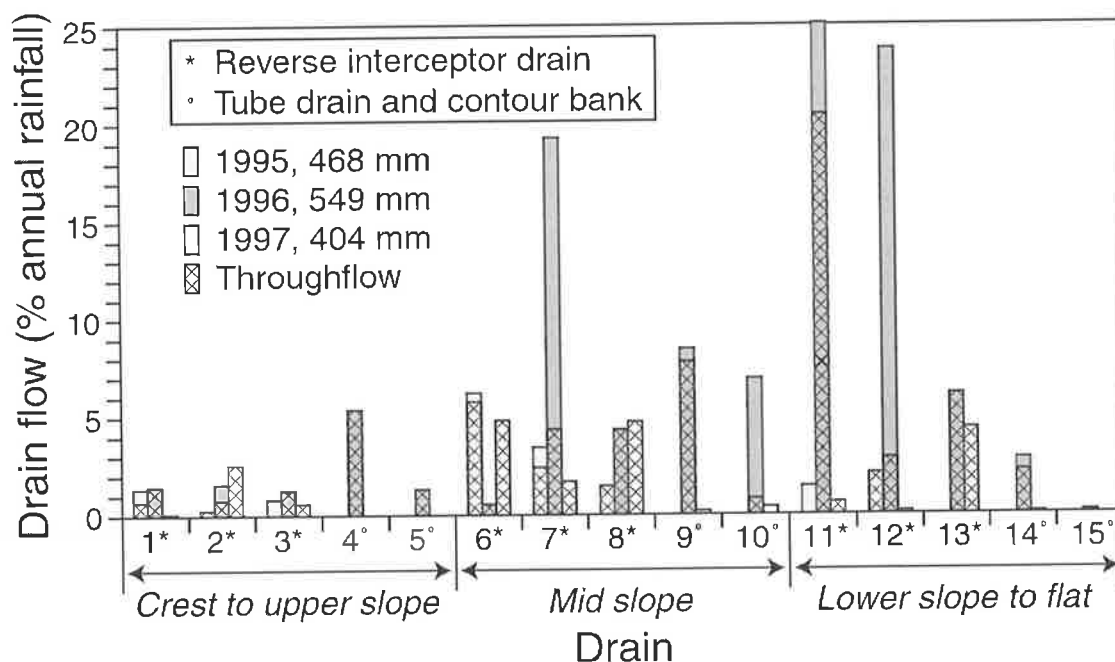


Fig. 4.7. Flow (total flow and throughflow) in each of the 15 drains from 1995 to 1997.

Table 4.11 A summary of drain flows at Keyneton, 1995-97

Drain ID * ¹	Drain No* ²	Combined drain flow(mm)			Combined drain flow (% of annual rainfall)			Percentage total flow as overland flow		
		1995* ³	1996	1997	1995* ³	1996	1997	1995* ³	1996	1997
1i/o	3	2.9	5.6	1.8	0.8	1.0	0.4	93.5	9.9	0
2i/o	8	6.6	23.3	19.1	1.9	4.2	4.8	0	0	0
3i/o	13	0	38.9	20.5	0	7.1	5.1	0	2.0	0
4i/o	2	0.8	6.1	7.3	0.2	1.1	1.8	0	38.9	0
5i/o	7	12.2	81.1	5.0	3.4	14.8	1.2	0	78.2	0
6i/o	12	6.4	84.7	0.3	1.8	15.4	0.07	0	87.4	0
7i/o	1	6.3	7.8	0	1.8	1.4	0	75.9	0	0
8i/o	6	19.2	2.2	16.2	5.4	0.4	4.0	0	0	0
9i/o	11	4.2	90.1	1.6	1.2	16.4	0.3	0	20.2	0
ot1	4	0	21.4	0	0	3.9	0	0	2.2	0
ot2	9	0	49.4	0.4	0	9.0	0.09	0	7.2	0
ot3	14	0	24.0	0.1	0	4.4	0.02	0	24.3	0
ot4	5	0	4.7	0	0	0.9	0	0	0	0
ot5	10	0	23.	0.9	0	4.2	0.2	0	87.7	100
ot6	15	0	0.5	0	0	0.09	0	0	100	0

*¹ Refer to Fig. 4.6*² Refer to Fig. 3.1*³ Based on rainfall for the period May-DecemberPeriod: 1995

In the period May - December, rainfall was 354 mm and distribution can be seen in Fig. 4.8. Variable amounts of overland flow and/or throughflow were recorded after rainfall events, with no flow at all recorded for the entire period from some sites (Table 4.11). This was particularly noticeable on those sites with buried drains/contour banks, with none of these six sites showing any form of runoff.

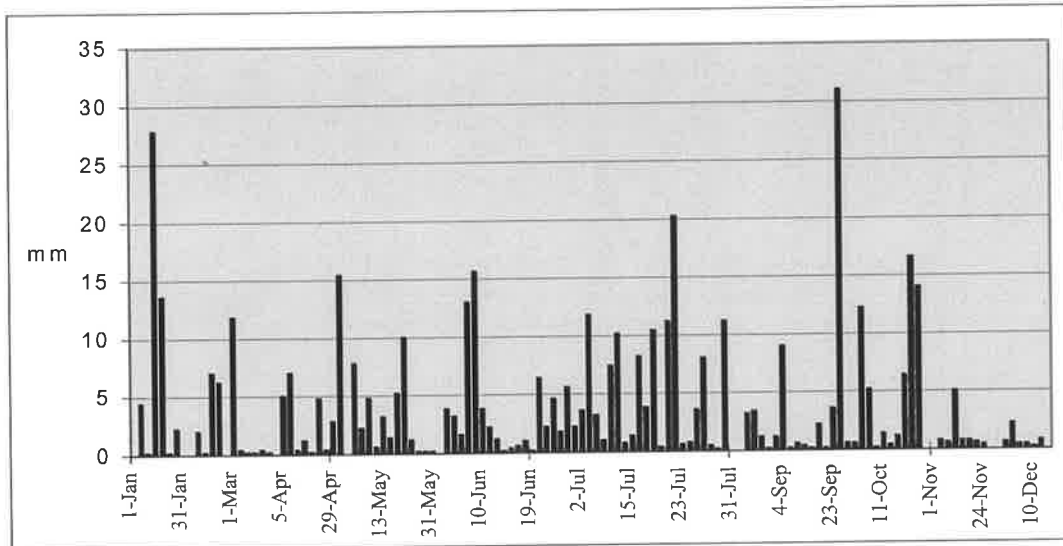


Fig. 4.8 Daily rainfall distribution in Keynes' catchment during 1995

On the few events that overland flow did occur (and this in only 2 of a possible 15 drains), it ranged from 3.8% to 11.5% of the amount of rain falling on site during these events. In terms of the total period rainfall, it represented <1.5% (Table 4.12) and only occurred following rain events >30 mm. Throughflow, whilst occurring more often and in greater volume, was still variable both between and within sites (0.1 – 42.9% of a single rainfall event). Generally it only took place after successive days of rainfall, or after large rain events. Up to 5.4% of a site's rainfall over this period drained as throughflow (Table 4.13).

Period: 1996

Again there was considerable variation in drain flow throughout the year (Tables 4.11, 4.14 and 4.15). Given the increase in rainfall from that of the previous year (Fig. 4.9), drains were found to flow more often, however there was a level of unpredictability in these flows. Even large rainfall events were no guarantee that drainflow would commence (eg. following 33 mm of rain on 27/28 June the range of throughflow recorded across the sites was between 0 and 5.4 L m⁻²). Between 0.7 and 13% of annual rainfall was removed as throughflow.

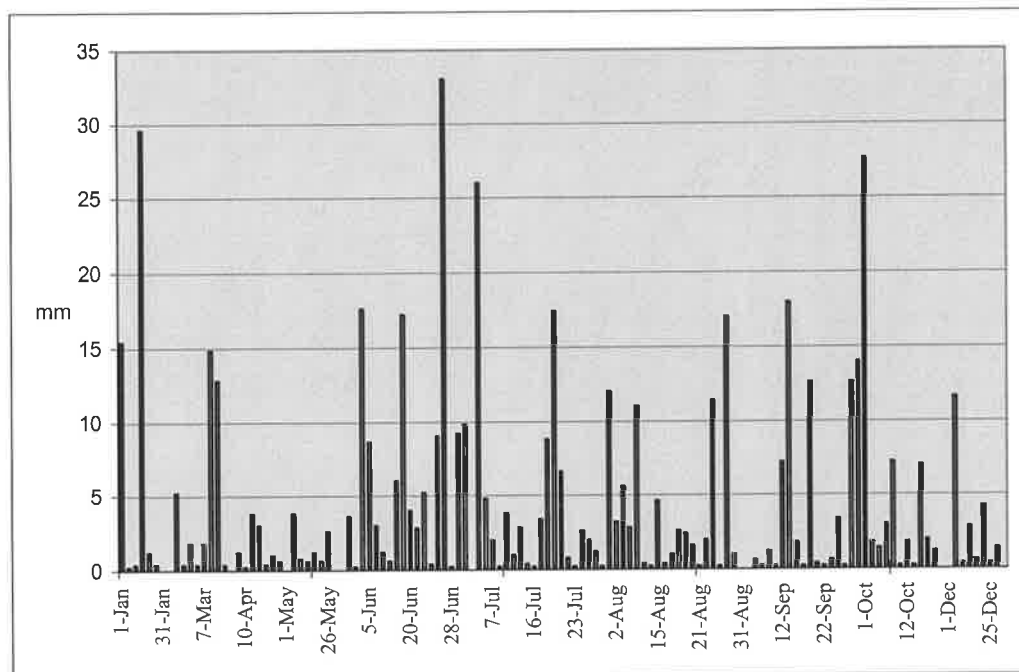


Fig. 4.9 Daily rainfall distribution in Keynes' catchment, 1996.

Overland flow occurred more frequently throughout the year. Conditions that led to this seemed to be a rain event >25 mm, and a soil that had previously been wetted, although even given these factors, all rainfall may still have infiltrated the soil. However, in a three-day period (29 Sept - 1 Oct) a total of 54 mm of rainfall was recorded and the overland flow that followed was substantial (in some cases up to 90% of rainfall). Although occurring less frequently than throughflow, overland flow accounted for up to 13.5% of annual rainfall removed as drain flow from a particular site (Table 4.14).

Period: 1997

Drain flow was limited (Tables 4.11, 4.16 and 4.17) in this low rainfall year (Fig 4.10), and overland flow was only recorded in one instance. The variability in flows between plots however was still marked, with for instance, a range of 1.9 - 32% of rainfall for the rain event of 9 - 10 February. On an annual basis, drainflow accounted for between 0.4 and 5.1% of total rainfall.

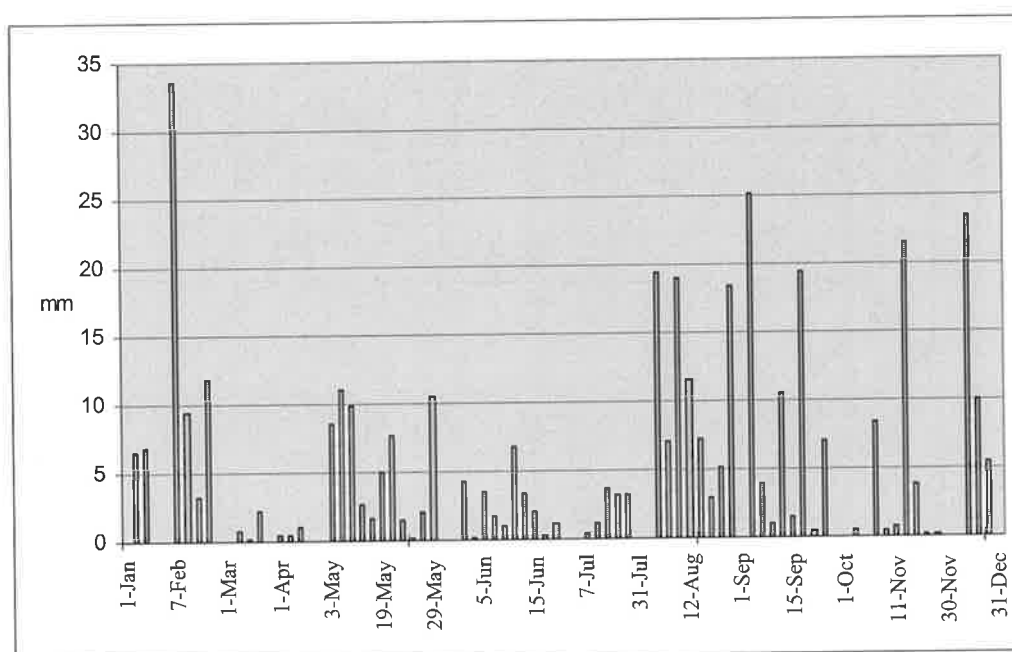


Fig. 4.10 Daily rainfall distribution in Keynes' catchment during 1997.

Drain flow in relation to treatments

Over the three years of the study, drain flow from cocksfoot pasture was higher than phalaris, and this in turn was greater than that from lucerne (Table 4.12). In the two drier years, overland flow was minimal in real terms (0-1.2 mm). For 1996, overland flow was a significant component in both real terms (7.5-17.5 mm) and particularly in the case of lucerne and phalaris, as a percentage of total drain flow. These flows were from the more poorly draining mid and lower slopes, generally from a single rain event on already wet soils. Although data are limited, this probably resulted from a rise in the level of perched water in the already water-filled, low permeability mid and lower slopes. Profile descriptions show historic evidence of waterlogging in these slope positions.

Table 4.12 Overland flow in relation to total drain flow from three perennial pastures at Keyneton.

	1995			1996			1997		
	Drain flow (mm)	Overland flow (mm)	Overland as % of drain flow	Drain flow (mm)	Overland flow (mm)	Overland as % of drain flow	Drain flow (mm)	Overland flow (mm)	Overland as % of drain flow
Cocksfoot	9.5	1.2	13.3	40.2	7.5	18.8	5.8	0	0
Phalaris	2.4	0	0	32.9	17.5	53.4	4.2	0	0
Lucerne	2.5	0.4	16.7	21.5	15.8	73.6	3.7	0.2	5.1

Table 4.13 Drain flow recorded during the period May-December 1995 in Keynes' catchment

Table 4.13 Drain flow recorded during the period May-December 1995 in Keynes catchment										
Drain	May-Dec Rainfall (litres)	Drain flow [litres and (%)]						May-Dec Total flow (litres)	May-Dec % total rainfall	
		14 July	17 July	18 July	19 July	21 July	22 July	24 Sept.		
1i	543744						167 (0.3)	53 (0.1)	286	0.05
1o							4182 (8.6)		4182	0.8
2i	458430				1521		2178 (5.4)	524 (1.3)	8590	1.9
2o										
3i	392940									
3o										
4i	626226							1424 (2.6)	1424	0.2
4o										
5i	594012	586	1603		3764	540	12052 (23.0)		20527	3.5
5o										
6i	701274						12278 (19.7)	405 (0.65)	12683	1.8
6o										
7i	453474	76	22	10			643 (1.6)	1180 (3.0)	1931	0.43
7o							1548 (3.8)	4561 (11.5)	6109	1.3
8i	543744	2930	457		1659	166	3915 (8.1)	20409 (42.9)	29536	5.4
8o										
9i	708000	4505	711		1143		1666 (2.7)		8431	1.2
9o										

Table 4.14 Flow recorded in the reverse interceptor drains throughout 1996 in Keynes' catchment.

Drain	Jan-Dec Rainfall (litres)	Drain flow [litres and (%)]								Jan-Dec Total flow (litres)	Jan-Dec % total rainfall
		28/29 June	3/4 July	17/18 July	19/20 July	4-10 August	24-30 Aug.	11/12 Sept.	28/29 Sept.		
1i	841447	147	69 (0.17)	2016 (4.0)		1118 (6.6)	100		4065 (4.7)	7762	0.92
1o			852 (2.0)							852	0.10
2i	709371	3762 (8.8)	5895 (13.9)	3977 (9.4)		3069	1037	5483 (15.7)	6500 (9.0)	30170	4.25
3i	607908	536	32848 (90.0)	6834 (23.5)	2053 (28.0)					42271	6.95
3o			881 (3.0)							881	0.14
4i	969092		5458 10.4	769 (4.9)						6619	0.68
4o			4209 (9.2)							4209	0.43
5i	919131	824	1258 (2.3)	11	4627 (8.2)	1399	2903	4086	13134 (14.0)	29700	3.23
5o					9765 (31.4)	642	10156 (33.6)		85039 (90.5)	106332	11.57
6i	1085170			195	1454 (2.8)	2135 (9.8)	173		12268 (11.1)	21142	1.95
6o					32259 (67.8)	7800 (35.6)	14087 (39.5)		87416 (78.8)	146655	13.51
7i	701946				1694 (5.5)	794	519	2985 (8.6)	4059 (7.6)	10051	1.43
7o											
8i	841340		233 (0.6)		86 (0.1)		360		2582 (4.0)	3399	0.40
8o											
9i	1095827	10867	15846 (24.2)		15022	7429	8760	22881 (42.4)	59050 (55.8)	143940	13.14
9o							3457 (9.6)		32895 (58.7)	36352	3.32

Table 4.15 Flow recorded in the contour bank / buried tube drains throughout 1996 in Keynes' catchment.

Drain	Jan-Dec Rainfall (litres)	Drain flow [litres and (%)]								Jan-Dec Total flow (litres)	Jan-Dec % total rainfall
		28/29 June	3/4 July	17/18 July	19/20 July	6 – 10 August	24 – 30 August	11/12 Sept.	28/29 Sept.		
ot1	1016905		494		368					862	0.08
t1f			(1.0)		(1.1)				19024	38838	3.81
			2736		17070						
ot2	680402	80	2054		2287					4421	0.65
		(0.65)	(6.4)		(10.6)						
t2f		6588	1114		21112				27564	56918	8.4
ot3	480510	62	62							5116	1.06
		(0.72)	(0.27)								
t3f		116	3470		6070				5896	15902	3.3
ot4	1075806										
t4f			488		2470				5616	9172	0.9
ot5	1152071	4856	8157				7221		10851	42982	3.73
		(23.6)	(14.9)				(30.1)		(12.4)		
t5f		1100	126		2006				2804	6036	0.5
ot6	745012		654							654	0.09
			(1.8)								
t6f											

Table 4.16 Flow recorded in the reverse interceptor drains throughout 1997 in Keynes' catchment.

Drain	Jan-Dec Rainfall (litres)	Drain flow [litres and (%)]							Jan-Dec Total flow (litres)	Jan-Dec % total rainfall	
		7/8 Feb.	9/10 Feb.	29/30 May	7/8 August	11/12 Aug.	30 August	2 – 4 Sept.	18 Sept.		
1i	617491	334 (0.5)	27		105 (0.25)	1825 (3.9)	294	10	85	2792	0.45
1o											
2i	520568	509 (0.9)	365 (1.9)	162		17326 (43.9)		4825 (12.3)	689 (2.7)	24752	4.75
2o											
3i	446110	11465 (24.0)	5410 (32.0)				689 (3.4)	3268 (9.8)	703 (3.3)	22734	5.1
3o											
4i	711163	634 (0.8)			112	3046 (5.6)	7042 (21.6)	42		12902	1.8
4o											
5i	674499	812 (1.1)	8		16	5188 (10.1)			866 (2.7)	8444	1.25
5o											
6i	796346	614 (0.7)								614	0.08
6o											
7i	515119										
7o											
8i	617413	2569 (3.9)	643 (2.8)		2117 (5.2)	13389 (28.6)			2140 (7.2)	24908	4.03
8o											
9i	804166	950 (1.1)	1386 (4.6)	520 (2.5)						3181	0.40
9o											

Table 4.17 Flow recorded in the contour bank / buried tube drains throughout 1997 in Keynes' catchment

Drain	Jan-Dec Rainfall (litres)	Drain flow [litres and (%)]							Jan-Dec Total flow (litres)	Jan-Dec % total rainfall
		7/8 Feb	9/10 Feb	29/30 May	7/8 August	11/12 Aug.	30 Aug.	2 – 4 Sept.	18 Sept.	
ot1	746250									
t1f										
ot2	499309									
t2f							140		416	0.1
ot3	352620									
t3f							29		75	0.03
ot4	789475									
t4f										
ot5	845441								1834 (4.5)	0.22
t5f										
ot6	546723									
t6f										

4.4.2 Example hydrographs of rainfall events

Below are 'typical' hydrographs for rainfall events that generated overland flow. Overland flow only occurred when soils were already wet, and rainfall exceeded infiltration capacity, generally in a storm event.

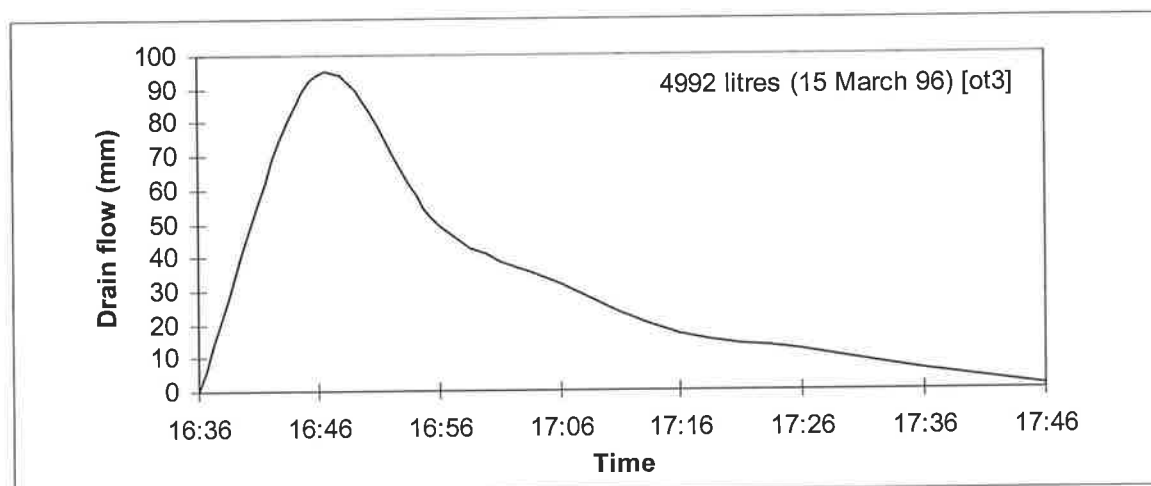


Fig. 4.11(a) Hydrograph of overland flow from contour drain ot3, March 1996.

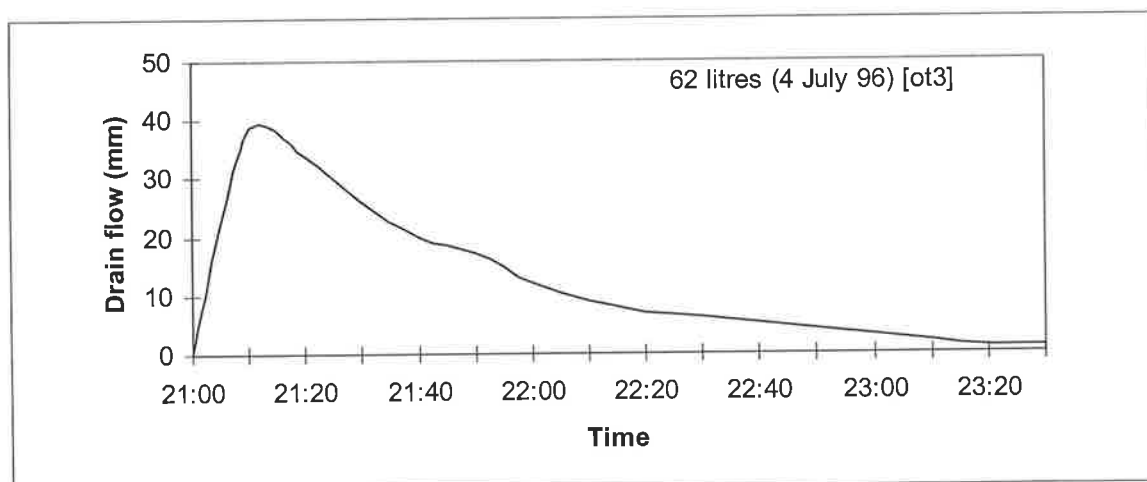


Fig. 4.11(b) Hydrograph of overland flow from contour drain ot3, July 1996.

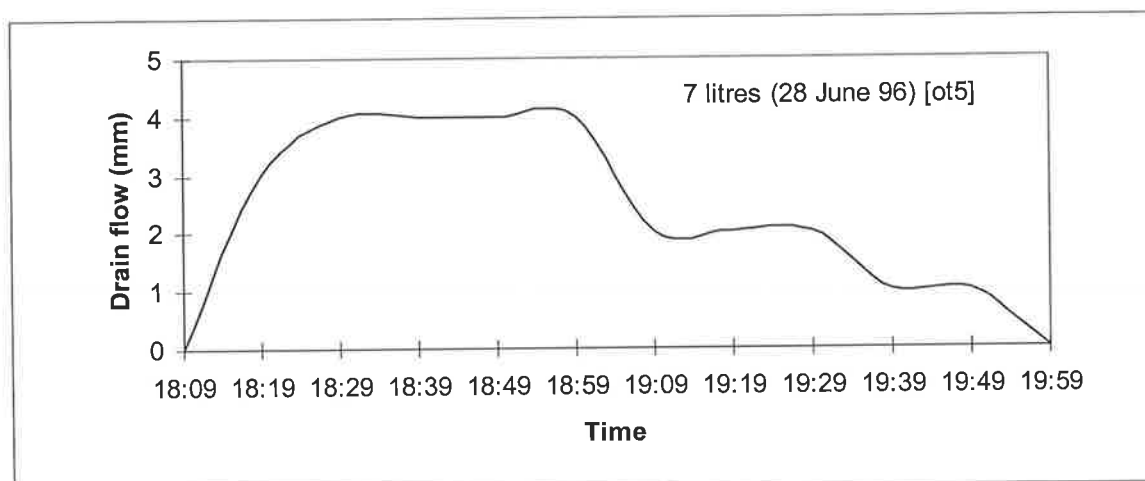


Fig. 4.11(c) Hydrograph of overland flow from contour drain ot5, June 1996.

In Figs. 4.11 and 4.12, the drain flow (mm) on the Y axes relate to mm above the V-notch of the weir per 10 minute recording interval. The duration of overland flow was usually quite brief in comparison to throughflow, with peak flows occurring rapidly but often lasting only a matter of minutes (Figs. 4.11a-c).

In the preceding hydrographs, (a) and (b) are from the same plot (phalaris in a lower slope position), the former on 15th March 1996 and the latter July 4th 1996. In both cases the time from the commencement of flow to the peak of the hydrograph is around 10 minutes. The pattern in both of these cases is one of rapid flood peak, then a tapering off of flow over a relatively brief period (1-2 hours). The significant difference between these two hydrographs is the discharge volume (see Equation 3.1), and the exponential function of the height over the weir is clearly illustrated (Flow rate = $2391H^{2.5}$, where H is head on the weir) (Grant and Dawson 1995). These are more 'typical' hydrographs than the lower example (mid-slope lucerne) where a small discharge (7 litres or 0.003 L m^{-2}) occurs with the peak flow extended over approximately 50 minutes.

The hydrograph examples below (Figs. 4.12a-b), are a result of the one rainfall event. The upper example, from drain 6i, is throughflow discharge from a phalaris plot in one of the lower slope positions. Discharge of 12,278 litres (6.2 L m^{-2}) was recorded over an 18 hour period, and the hydrograph pattern suggests that sufficient rain fell to initiate throughflow, and then 6 or so hours later another fall occurred which prolonged and substantially increased the flow volume. The discharge from this plot was around 20% of the rainfall deposited on-site.

In the second example (from drain 7i), discharge of 643 litres (0.5 L m^{-2}) was recorded from a cocksfoot site on the upper-slopes. Although the pattern of this hydrograph was similar to the first, discharge was substantially less and was c. 1.6% of rainfall. Again the recorded discharge is a function of the height of flow at the weir, in this instance with a 30 degree V-notch, discharge = $373.2H^{2.5}$ (Grant and Dawson 1995).

The trend of higher discharge from the lower slope positions in comparison to the upper-slopes was generally consistent once the soil profile had 'wet up', and was influenced by inherent drainage potential and soil water. On the flats and lower slopes

waterlogging developed at two depths. A perched water table formed at about 35-55 cm in the A or B horizon and also groundwater rose to within 150 cm of the soil surface. High lateral flows on the lower slopes were probably related to high groundwater preventing vertical drainage. On the mid and upper-slopes, a transient watertable developed generally, but not exclusively, within the clay B horizon. In the mid-slopes, there was a slow leakage of perched soil moisture through to the C horizon, however in the upper-slopes, better drainage (probably through macropores), allowed a more immediate transfer across the C horizon boundary.

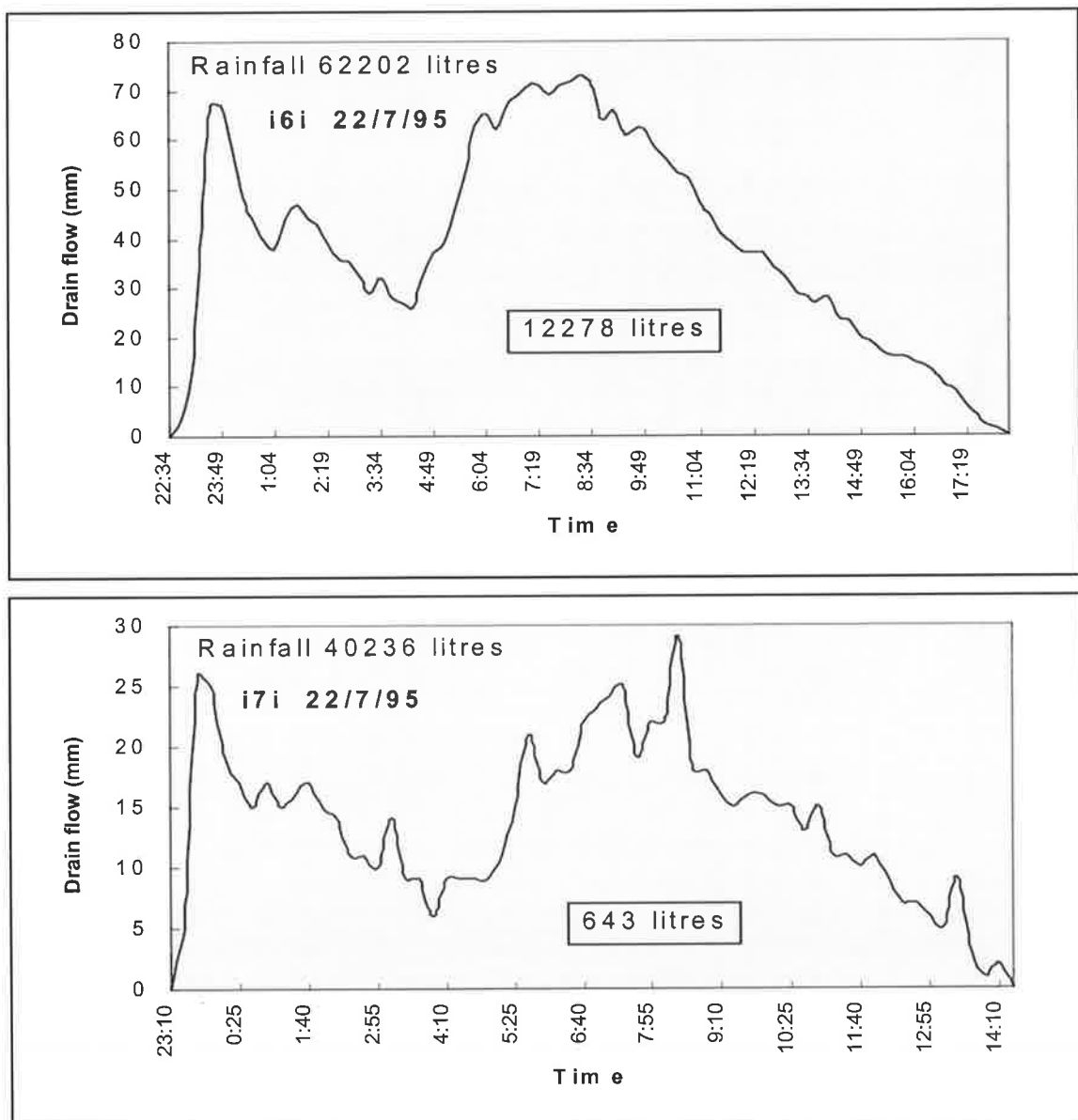


Fig. 4.12 Hydrographs of throughflow from Site L1 (phalaris-lower slope), through drain 6i, and Site Uc (cocksfoot-upper slope), through drain 7i, on 22 July 1995.

4.4.3 Tipping bucket flow meters on buried drains

All flows in the buried tube drains were delivered to tipping bucket flow meters installed in covered sumps (see Plate 3.6). The lag time until flow commenced in the tipping buckets was generally greater than that of the throughflow in the interceptors, primarily because there was no batter flow component involved. Once throughflow to the tipping bucket commenced, it was likely to continue for an extended period, often at very low rates that could not be maintained in an open drain where evaporation was a factor. In 1995, there was no flow recorded in the tipping buckets. During 1996, a mean of 3.3% of annual rainfall was recorded in the tipping buckets (Table 4.15) whilst in 1997 the flows were insignificant (Table 4.17).

4.4.4 Soil water content

Figure 4.13 shows representative examples of the variability in soil water content between the upper, mid and lower slopes adjacent to the drains (see the NMM sites in the undrained area adjacent to the study plots in Fig. 3.1), over a five-year period (1994 to 1998). This includes data for one year prior to and one year following monitoring of the drain flows. On the upper slopes, a perched watertable developed at about 0.5 to 0.6 m, well within the clay B horizon of all plots.

The duration and thickness of the saturated zone was related to rainfall (e.g. soil saturation and rainfall were highest in 1996). On the mid slope, soil saturation in each plot generally occurred nearer the surface than on the upper slope, at about 0.4 m, the upper B horizon. Surprisingly, the occurrence, duration and depth of saturation were usually much less on the mid slopes than on the upper slopes to crests. Soil saturation on the mid slopes also occurred at depth (below 1 m) in some plots, in 2 out of the 5 years, due to a deeper perched watertable. On the lower slope plots a perched watertable did not develop within the A or upper B soil horizon. Rather, soil saturation occurred below about 1 m and was caused by groundwater (groundwater is inferred from SWC and correlates with adjacent piezometer readings inserted as part of another project).

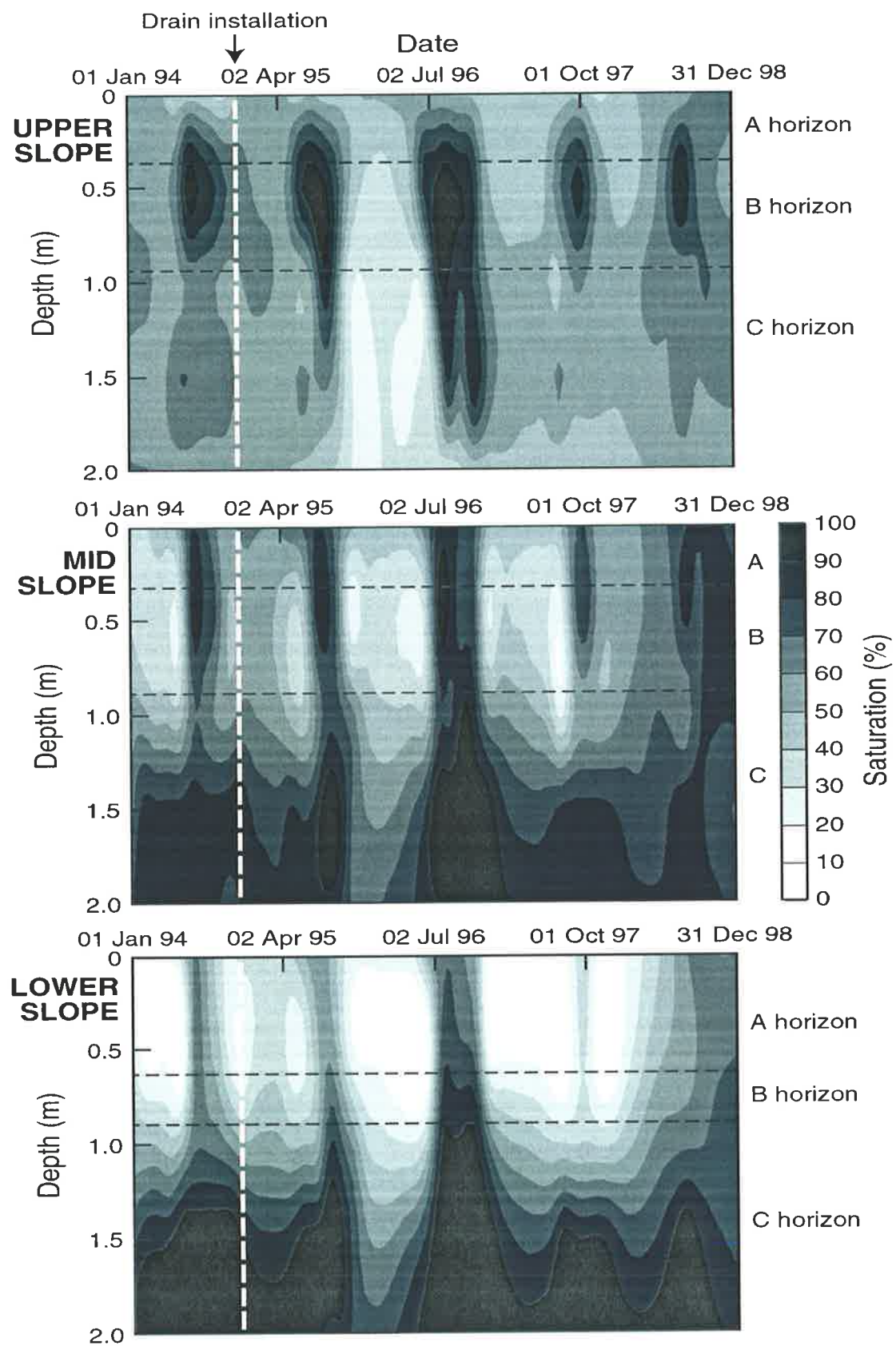


Fig. 4.13. Examples of the variability in soil water content between the upper, mid and lower slopes from 1994 to 1998.

4.5 Conclusions

The soils of the site are extremely variable, particularly in the presence or absence of some of the horizons as well as their depth and extent. Some general similarities are apparent dependent on the slope position. The upper slope profiles predominantly have a brown or reddish brown topsoil and a mostly uniform reddish subsoil, indicating very infrequent topsoil saturation and a free draining subsoil. The C horizons show signs of redoximorphic conditions by their yellow and grey colouration (McMurray 1994; Cox *et al.* 1996^a).

In the mid-slope positions, the profiles show A/E horizons with considerable yellow and grey colouration as well as low chromas (<2) and some nodulation (manganese and iron), indicating periodic saturation. The subsoils display substantial mottling and graded into the yellow, grey and white of the C horizon at depth. Cox *et al.* (1996^a) suggest that this implies periodic saturation by water perching on the subsoil clay, with mostly lateral flow occurring.

The lower slopes display redox conditions in both top and subsoils, indicating mostly lateral flows, periodic saturation of the topsoil and increased level of subsoil saturation with both perched water and rising groundwater. Variable, but predominantly low K_s values in the mid and lower slopes, declining as one proceeds down the profile, and high bulk densities, again increasing generally downslope and down the profile, support the above observations predicting poor drainage and perched water in these soils.

This high variability in subsoil point K_s (measured with constant head well permeameter) is not uncommon in texture contrast soils and McFarlane and Cox (1992) attribute this to preferred pathway flows through the soil. These preferred pathways may lead to lateral as well as vertical flows, and seepage areas along the drain walls and at discrete spots on the surface, were indicative of this. McFarlane *et al.* (1990^a) reported some observed flow in the upper subsoil in WA and recommended extending drains below the subsoil interface, and this was done at Keyneton.

Given that in these soils there is an indication that perching may occur on the B/C boundary, it is possible that some lateral preferential flows were travelling below the interceptor drain. Brouwer and van de Graaf (1988) noted a western Victorian texture

contrast soil where flow was near the top of the C horizon, which could have “considerable implications for soil management decisions”. If this is also the case at Keyneton, it adds to the complexity of predicting drain flows and drains may have to be installed to a greater depth to ensure the interception of lateral flows.

Variability in mean seepage interceptor flow across catchments

Figure 4.14 shows the relationship between the average total flow measured in seepage interceptor drains in several catchments in southern Australia. All drains were placed into the upper B horizon over c.40 m spacing. Despite the enormous variability in the limited published drain flows in catchments across southern Australia, there appears to be a simple relationship between the average annual drain flow (overland flow and throughflow) and rainfall for drainage from pastures and winter cereals. In a region (or season) with 400 mm annual rainfall, drains spaced at 40 m or more will average 10 mm flow (2.5% of annual rainfall). Drain flow (from this relationship) will increase to 100 mm in a 700 mm rainfall year (14% of annual rainfall).

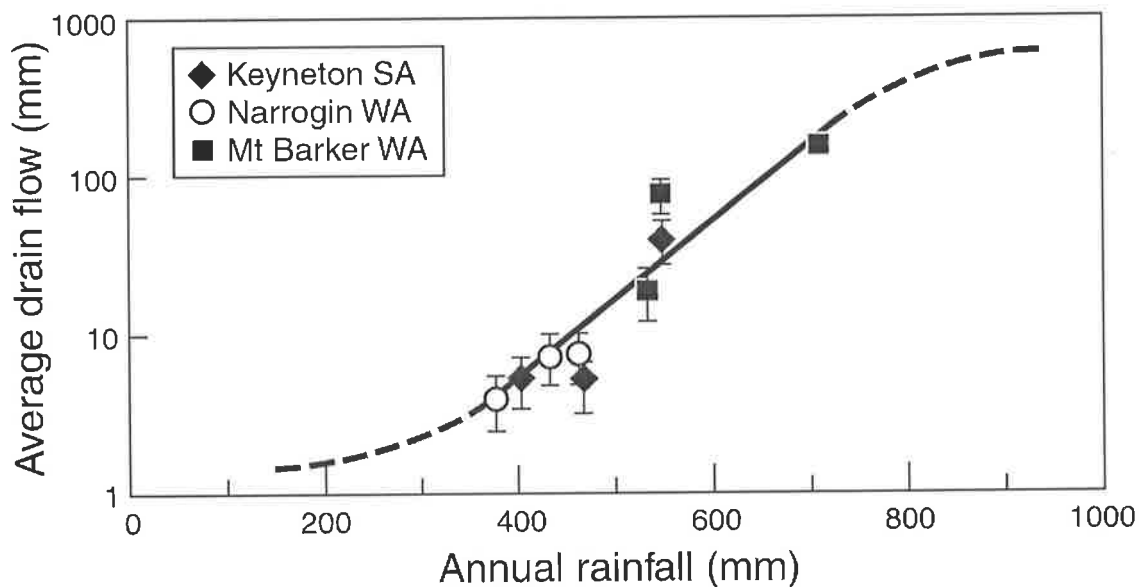


Fig. 4.14. Mean and standard error seepage interceptor drain flow and catchment rainfall (data for Narrogin and Mt Barker are from Cox 1988).

Drain performance

Variability in the performance (volume of drained water) of seepage interceptor drains was partly due to variability in texture (loamy sand to sandy loam) of the horizon(s),

which moved throughflow. Also, the coarse-textured (high K_s) A and E horizons were thickest on the lower slope and thus these landscape positions had the largest storage capacity for drainable soil water. K_s of the B horizon on the lower slopes was generally much lower than elsewhere on the hillslope, which caused reduced vertical drainage. These factors combined were conducive to high throughflow. In addition, on the lower slopes, there was a seasonal contribution of deep, perched water or groundwater to the drains or, at least, a further restriction to vertical flow.

K_s of the topsoil varied by several orders of magnitude between drains and in some positions on the slope, the standard deviation was as high as the mean. K_s of the B horizon clays in the mid to upper slopes was surprisingly high, and there was similar variation about the mean. K_s of the subsoil clays on the lower slope were an order of magnitude lower than elsewhere on the slope, most probably due to the soils being sodic and thus easily dispersive. Because K_s of the A horizon was similar on all parts of the slope and less water drained vertically through B horizons on the lower slope with low K_s , drains on the lower slope had highest flows (with the exception of a drain through a Dermosol).

Because of the extreme variability in depth to a clay B horizon and K_s , the drains would not necessarily have been installed into a low permeable layer along their entire length. Thus, the drains would probably have discharged perched watertables along some sections and recharged perched watertables along others. In addition, throughflow must have passed under the drains in some parts of the landscape as the barriers to flow were only installed to about 0.6 m depth. This may help explain why Cox (1988) found the duration of waterlogging on the down slope side of seepage interceptor drains in Western Australia was higher than upslope in 13% of cases.

Due to the seasonal contribution of deep, perched water (on the upper slopes and crests) or groundwater to upper soil horizons (on the mid to lower slopes), the drains on the lower slopes intercept some of this water most years. Some drains on the mid slopes intercept this water in average to above-average rainfall years (with the number of drains intercepting this water and the volumes intercepted increasing with annual rainfall). Therefore, the drains on the lower and mid slopes in particular are most effective (on a

percentage basis) in wet years and the risk of over draining in dry years is small. In contrast, drains on upper slopes have not intercepted this water in any year.

The use of reverse interceptor drains instead of the conventional type is probably justified despite their higher installation costs (as discussed in McFarlane and Cox 1992). In the majority of drains in years of average to below average rainfall, less overland flow than throughflow was measured. However, the work of McFarlane and Cox (1992) and Stevens *et al.* (1999) and Figure 4.7 suggest that in above-average rainfall years, overland flow will be the dominant flow pathway in most of the drains.

The enormous variability in drain flows within the study catchment was due to a number of complex processes and not just to the standard suite of hydrological processes incorporated into current drainage models. The variability in flow processes helps explain why it has been difficult to model flows in interceptor drains installed in texture-contrast soils in southern Australia. The assumptions made in the modelling scenarios have been invalid in some instances. Also, the models used have not adequately described the hydrological processes operating within some parts of these landscapes.

The results show the importance of predicting not only when and where groundwaters or deep, perched waters will rise into the root zone, but also where sodic subsoils have formed within catchments. Sodic subsoils with low K_s have a major impact on drainage volumes, particularly throughflow. Unfortunately this adds to the complexity of predicting drain flows in texture-contrast soils. Previous studies in Western Australia have suggested that preferential lateral pathways, mostly infilled root channels of the previous native vegetation, are responsible for rapid throughflow to drains in texture-contrast soils. This research suggests that these pathways may be underlain by sodic subsoils that further facilitate lateral flow.

Chapter 5

Water use, rooting depth and productivity of three perennial pastures

5.1 Introduction

The aim of this section of the study was to compare the water use, rooting depth and productivity of three perennial based pastures at three positions along a toposequence, upper, mid and lower slope. The perennials, lucerne, phalaris and cocksfoot (a component of the existing pasture), were established on texture contrast soils at a site typical of much of the Mt Lofty Ranges (SA).

5.2 Results

5.2.1 Dry matter production

The dry matter production was a measure of total herbage, consistent with local management practice (ie. there was no attempt to create 'pure' pasture swards). In general, annual weeds (grasses and broadleaf) made up a similar proportion of production regardless of the principal pasture species, or the landscape position. Clover, particularly the Balansa, began to emerge around June-July and became a significant component of the phalaris treatments throughout Spring. In July 1996, clover comprised between 3% and 10% of these pastures but by October had increased to between 20% and 45% of production.

The cocksfoot treatments had between 3% and 6% clover in their composition in July 1996, increasing to 5-50% in October of that year, and for the same periods lucerne had 1-4% and 5-25% respectively. Clover seedling persistence in the lucerne was very marginal, probably as a result of the drier surface soil there at the time of emergence (Dear *et al.* 1998), given the soil water use shown by this perennial.

After rainfall of about 45 mm in February 1997, there was an early emergence of sub-clovers, which reached the 3-5 leaf stage before wilting a fortnight later. At this time cocksfoot and phalaris both broke summer dormancy, phalaris starting to green up about

a week prior to the cocksfoot. For the next two months there was virtually no rain, and by early May, numerous dead leaves were appearing on the lucerne, the cocksfoot was showing no sign of growth and the phalaris had only c.12-15 cm of green tinge showing in its tillers. Rain in mid and late May led to good clover emergence by mid June that persisted through to late November. By early December both cocksfoot and phalaris were approaching dormancy.

Statistically, lucerne-based pasture was significantly more productive over the two year study period than the phalaris-based ($P < 0.05$) or cocksfoot-based in the upper and mid-slope positions, but was less so on the lower slopes (see Table 5.1 and Fig. 5.1). Toposequence position in this regard was highly significant ($P < 0.01$). From Table 5.1, the summer growth (Oct 96- April 97) advantage of the lucerne is apparent, when compared to the less summer active cocksfoot and phalaris-based pastures. Following establishment, both the lucerne and phalaris maintained their respective densities (Table 5.2). The phalaris crown size developed such that it occupied a higher proportion of measured groundcover relative to cocksfoot, and only slightly less than that of the lucerne.

Table 5.1. Mean dry matter production (kg/ha) of lucerne-, phalaris- and cocksfoot-based pastures, at three toposequence positions, over four time periods from March 1996.

Slope position	Lucerne sites				
	Aug-96	Oct-96	Apr-97	Oct-97	Total
Upper	1211.6	2851.5	2393.9	2860.3	9317.3
Mid	1123.9	2954.2	1518.9	2461.1	8058.1
Lower	796.5	1871.2	1229.4	1796.7	5693.8
	Phalaris sites				
	Aug-96	Oct-96	Apr-97	Oct-97	Total
Upper	1033.5	2732.4	936.1	2440.5	7142.5
Mid	902.2	2623.6	888.9	1491.4	5906.1
Lower	938.2	2618.4	905.0	1764.8	6226.4
	Cocksfoot sites				
	Aug-96	Oct-96	Apr-97	Oct-97	Total
Upper	941.4	2904.6	552.1	1796.1	6194.2
Mid	902.8	2295.0	599.2	1667.1	5464.1
Lower	997.4	2098.6	767.9	1637.7	5501.6

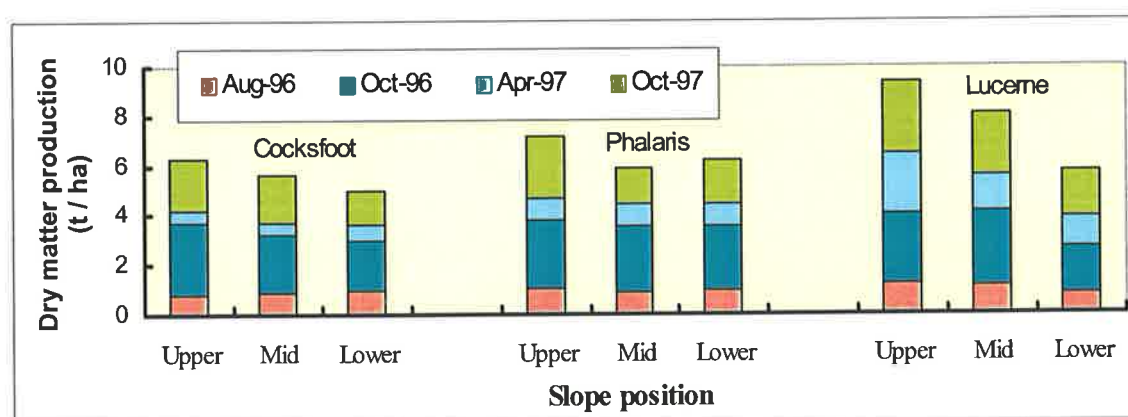


Fig. 5.1 Graphical representation of dry matter production over the period March 1996 to October 1997.

Table 5.2. Lucerne, phalaris and cocksfoot plant densities over the study period.

Density (plants/m ²)	Jul-96			Feb-97			Dec-97		
	Lucerne	Phalaris	Cocksfoot	Lucerne	Phalaris	Cocksfoot	Lucerne	Phalaris	Cocksfoot
Upper slope	64	25	36	65	21	33	78	28	41
Mid slope	65	24	34	74	23	30	77	31	38
Lower slope	72	33	41	83	23	36	78	28	36

5.2.2 Soil moisture

The soil water content of the A horizon (c. 0-35 cm), followed a similar pattern under all treatments (Figs. 5.2-5.4), responding rapidly to rainfall events (see Figs. 5.5-5.9). There are no real differences attributable to the overlying pasture, with the three lines representing SWC under the different treatments all within a narrow band.

During the winters of 1996 and 1997, some degree of a perched watertable developed on all treatments, most within the B horizon, probably around the B1/B2 boundary (see Figs. 5.5-5.9 in conjunction with Figs. 5.11 & 5.12, to interpret saturation at volumetric water contents). The extent and duration of these saturated zones were related to rainfall, with rainfall and soil saturation highest in 1996. The influence of the saturated zones extended into and through the A horizon to the surface, and was followed by its eventual redistribution into the lower soil profile after a time lag of about 30 days.

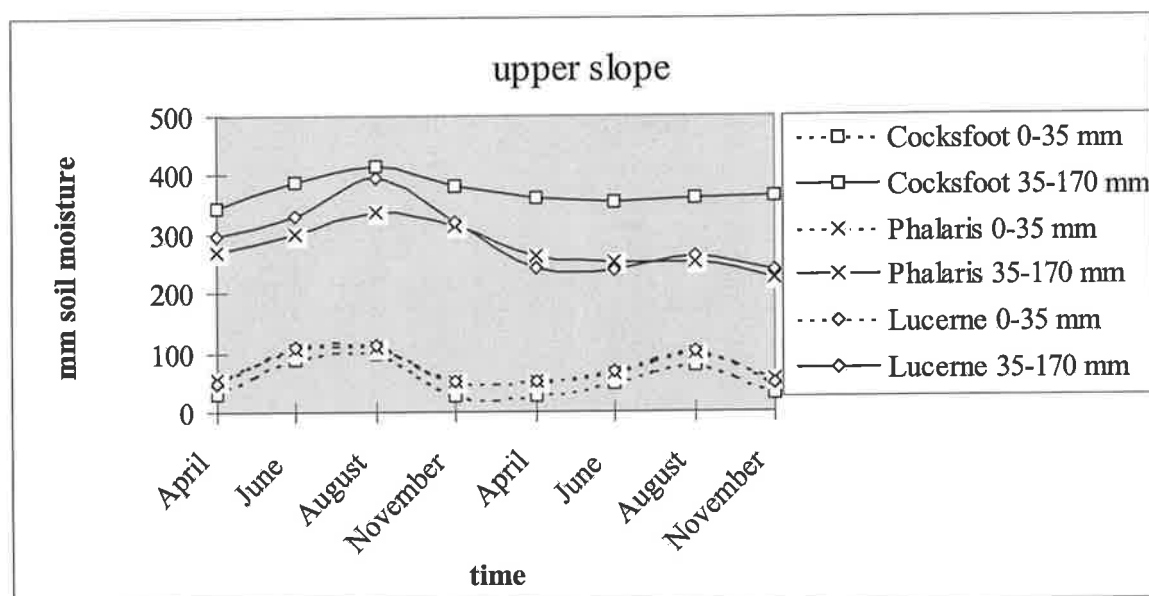


Fig. 5.2 Soil moisture content (mm), from April 1996 to December 1997, under pastures in the upper slope position.

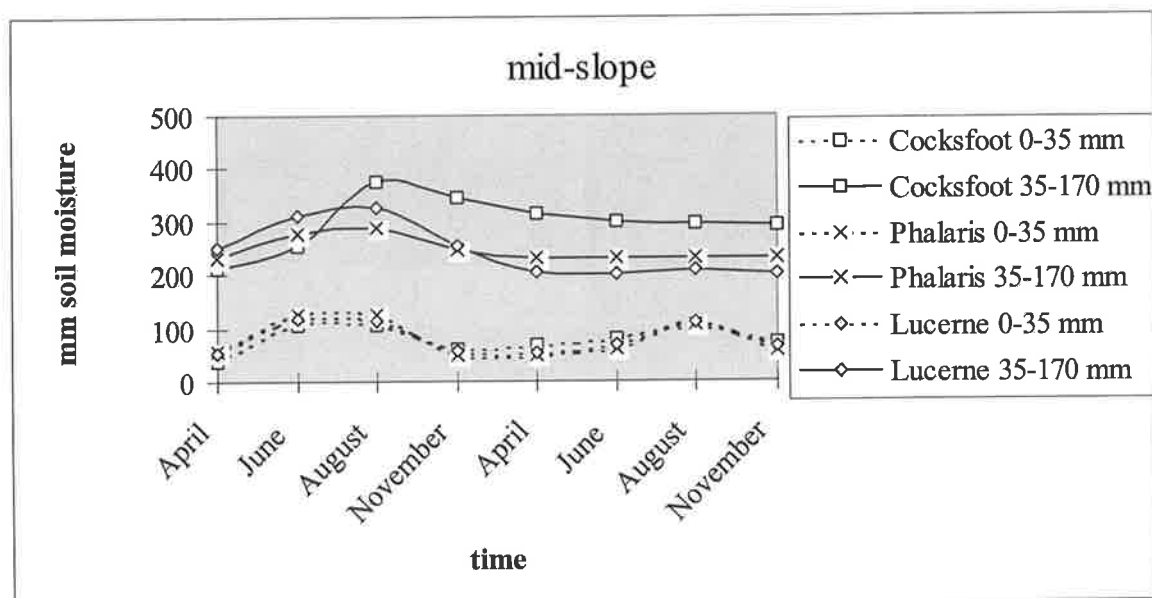


Fig. 5.3 Soil moisture content (mm), from April 1996 to December 1997, under pastures in the mid-slope position.

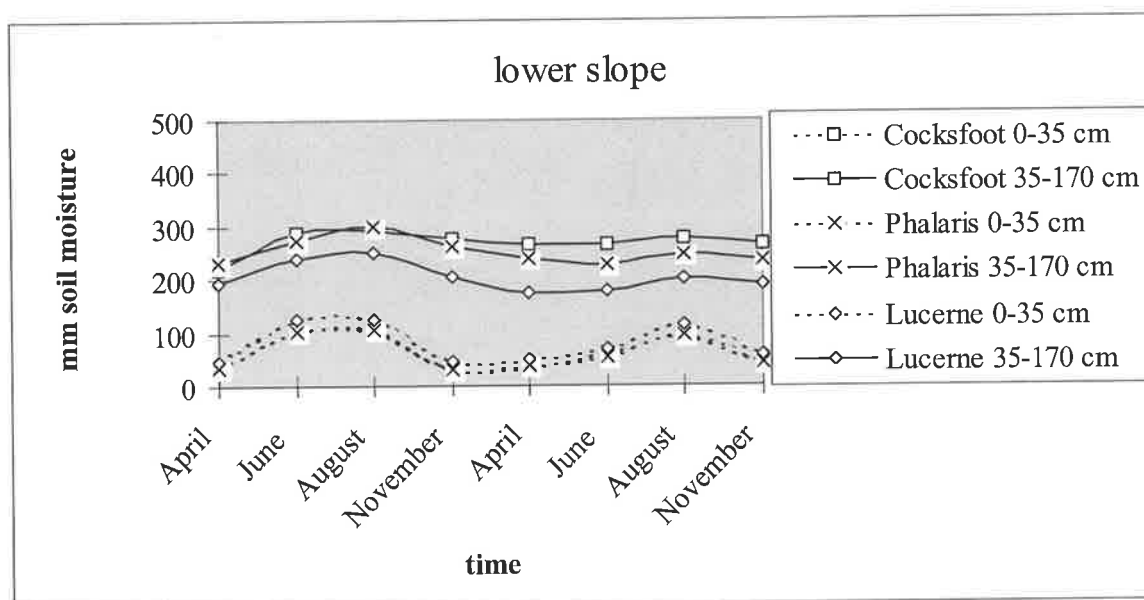


Fig. 5.4 Soil moisture content (mm), from April 1996 to December 1997, under pastures in the lowerslope position.

In the soil profile below 35 cm, changes in SWC were more gradual, and less influenced by the surface weather conditions. Over the period of study, SWC (35-170 cm) under cocksfoot based pasture increased at all slope positions (Table 5.3) but particularly in the mid-slope (see Tables 5.3 & 5.4, and Fig. 5.10), based on end moisture content minus the initial moisture content.

Under phalaris, there was an overall SWC decrease at all slope positions, with the greatest deficit in the upper slope position, followed by the mid and lower slopes. However, from Fig. 5.10 it is apparent that most of the mid-slope decrease was in the 20-80 cm section of the profile, with an increase at depth from an initial 15% to c.27% (volume %) by late 1997.

The overall decrease in SWC under lucerne was also greatest in the upper-slope region, again followed by the mid and lowerslopes. From Fig. 5.10 it can be seen that SWC under mid and lower slope lucerne also increased marginally at depth.

Table 5.3 End minus Initial SWC (mean values) in the soil profile below 35 cm, over the study period March 1996 – December 1997.

	Upper slope (mm)	Mid slope (mm)	Lower slope (mm)
Cocksfoot	+14.9	+81.8	+48.9
Phalaris	-58.0	-11.2	-1.8
Lucerne	-74.8	-57.8	-10.5

The ability of the three pastures to create an autumn soil water deficit is reflected in Table 5.4, showing the changes in subsoil SWC from immediately before the season break in 1996, to the same pre-break period in 1997. After only two years, lucerne had established substantial deficits at all slope positions. Given that the cocksfoot pasture had been established for a number of years, it was probable that it had reached a species-dependent equilibrium, and was unlikely to change greatly. Lolicato (2000) showed that this equilibrium was established under cocksfoot, phalaris and lucerne by the fourth year following pasture development.

Table 5.4 Changes in SWC in the soil profile below 35 cm, from May 1996 to May 1997 (ability to produce an autumn soil water deficit).

	Upper slope (mm)	Mid slope (mm)	Lower slope (mm)
Cocksfoot	+17	+94	+51
Phalaris	0	-6	-0.5
Lucerne	-48	-38	-21

Regarding the above discussion, it should be noted that although the lucerne and phalaris plots were established in May 1995, soil water content was not measured on the plots until March 1996. It is likely that by that time, both pastures would have already had an impact on SWC and therefore the figures quoted above should be viewed in relative rather than absolute terms.

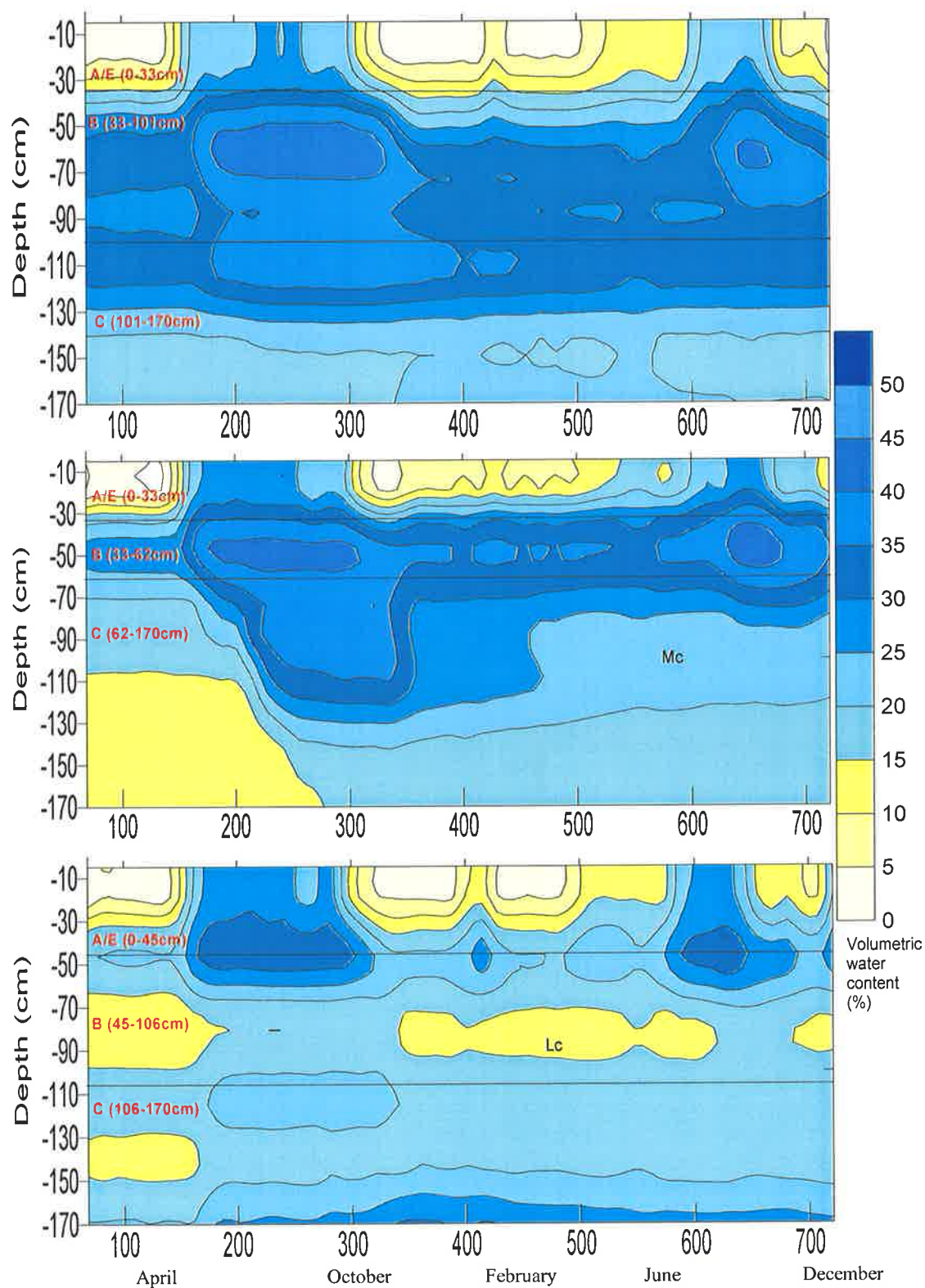


Fig. 5.5 SWC for the toposequence Uc-Mc-Lc (see Fig. 4.6), March 1996-December 1997. All treatments are cocksfoot. The upper diagram represents upper slope, mid diagram the mid-slope and bottom diagram the lower slope. X axis is Julian days commencing January 1, 1996.

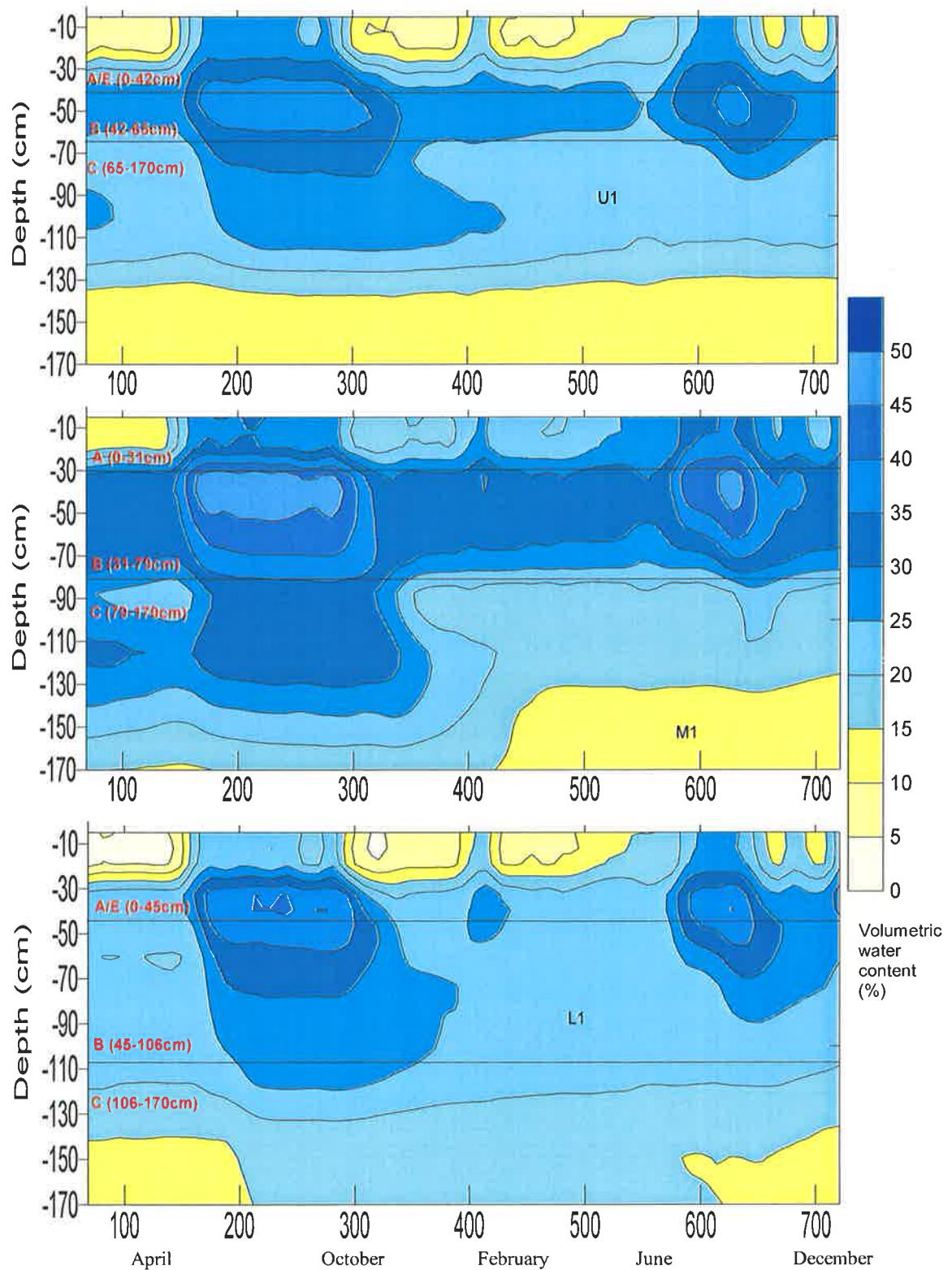


Fig. 5.6 SWC for the toposequence U1-M1-L1 (see Fig. 4.6), March 1996-December 1997. Slope representation is from upper to lower slope moving down the page. Upper and lower slopes are phalaris, and the mid-slope is lucerne. X axis is Julian days commencing January 1, 1996.

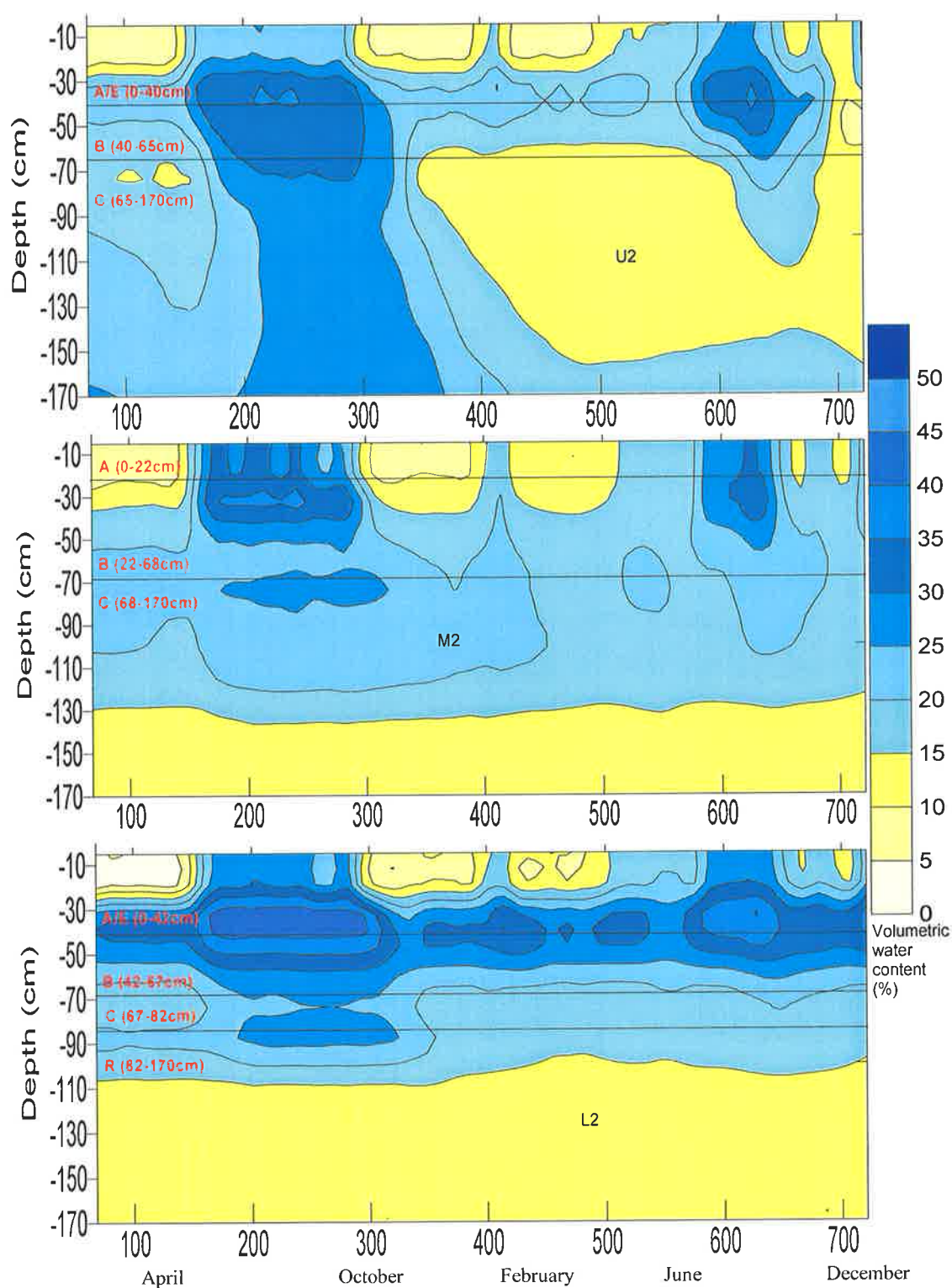


Fig. 5.7 SWC for the toposequence U2-M2-L2 (see Fig. 4.6), March 1996-December 1997. Slope representation is from upper to lower slope moving down the page. Upper and lower slopes are lucerne, and the mid-slope is phalaris. X axis is Julian days commencing January 1, 1996.

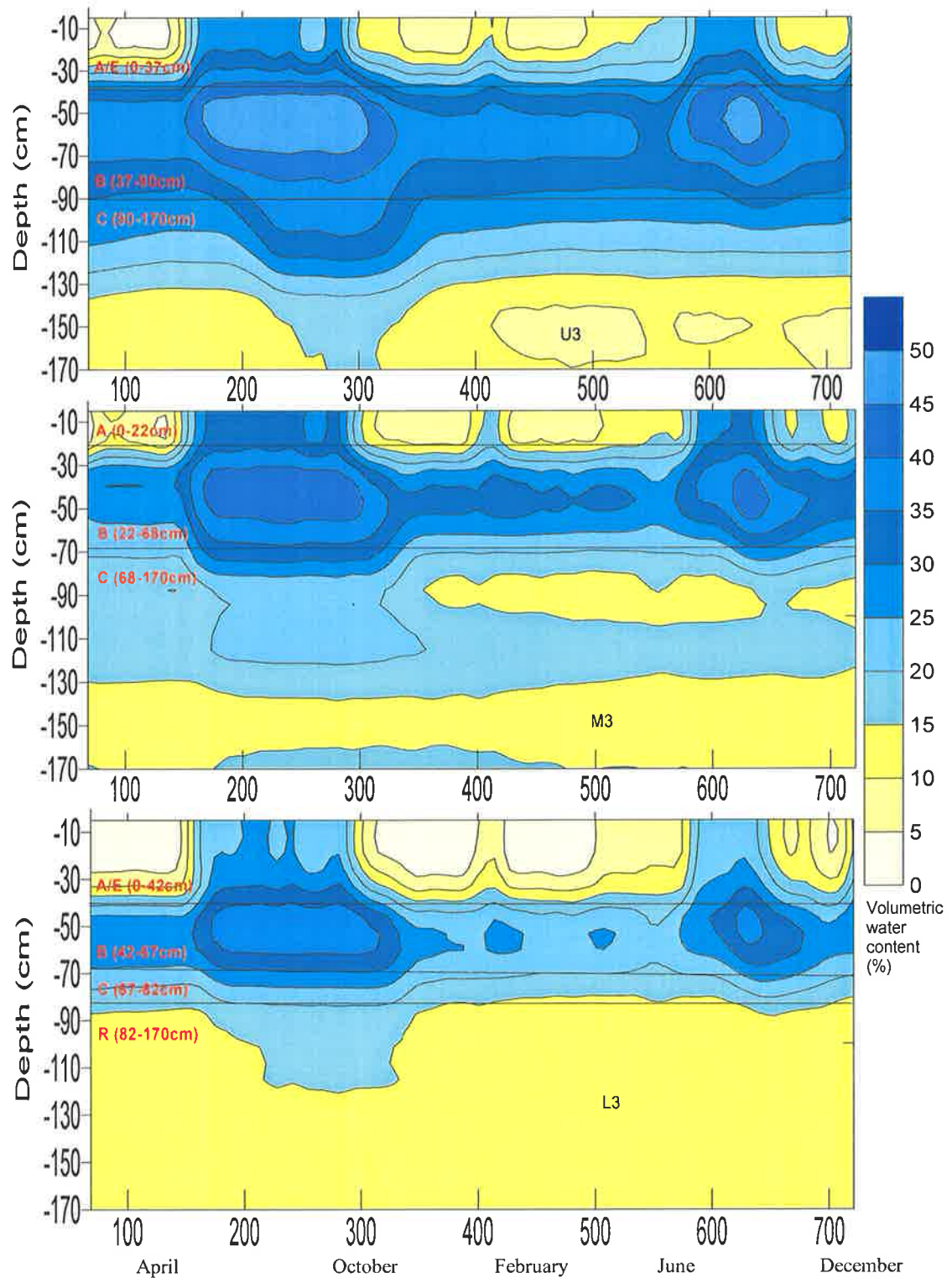


Fig. 5.8 SWC for the toposequence U3-M3-L3 (see Fig. 4.6), March 1996-December 1997. Slope representation is from upper to lower slope moving down the page. Upper slope is lucerne, and the mid and lower slopes are phalaris. X axis is Julian days commencing January 1, 1996.

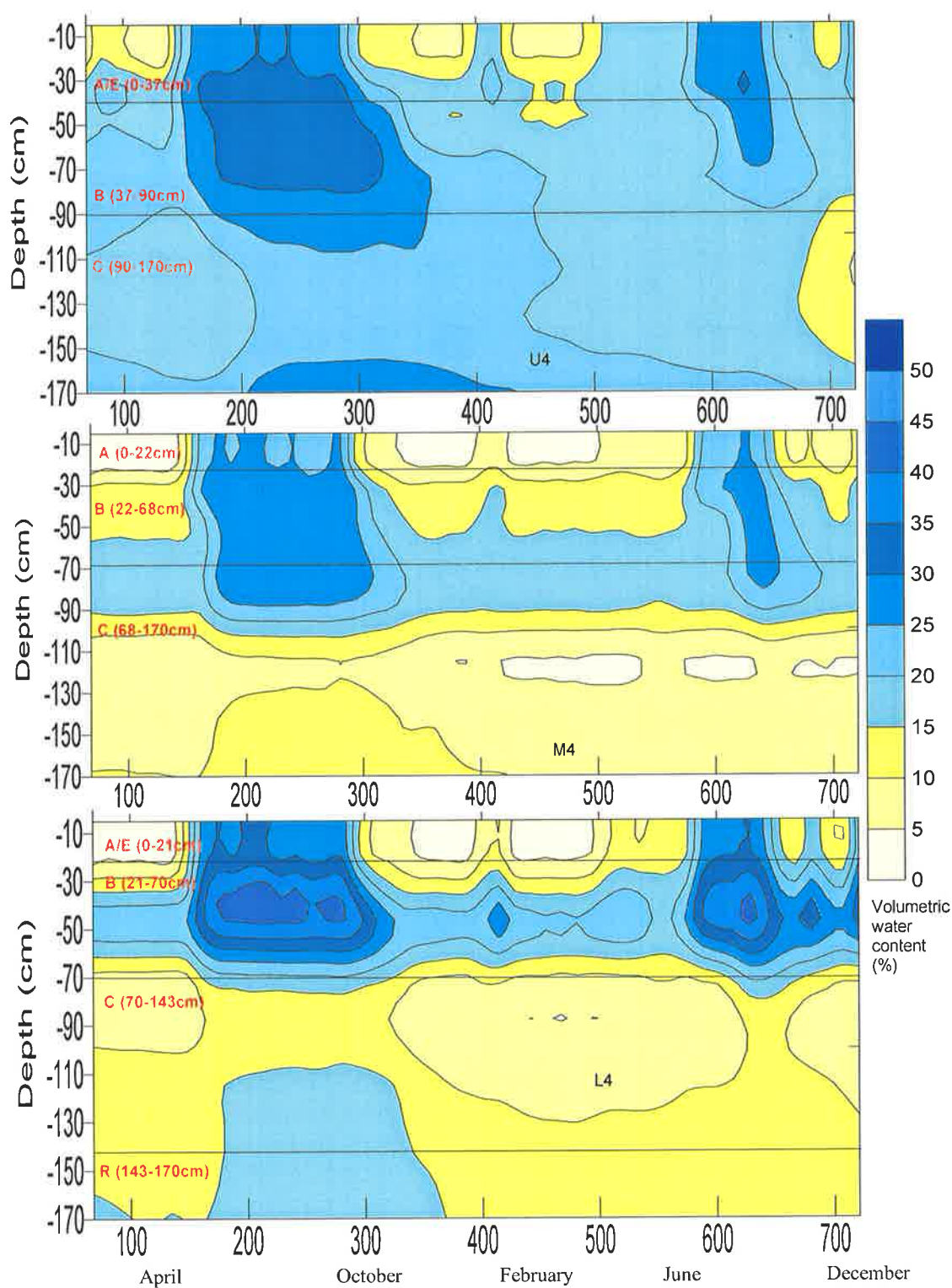


Fig. 5.9 SWC for the toposequence U4-M4-L4 (see Fig. 4.6), March 1996-December 1997. Slope representation is from upper to lower slope moving down the page. Upper slope is phalaris, and the mid and lower slopes are lucerne. X axis is Julian days commencing January 1, 1996.

The preceding Figures (5.5-5.9) are a representation of the changes in SWC under all treatments over the period for which NMM data was available (March 1996-December 1997). As a generalisation, this form of display gives important indications of the soil-water dynamic, and its changeability in time and space. It is apparent from these displays that the A/E horizons were the regions of greatest SWC fluctuation as these soils responded to the external wetting and drying processes. The B and to a lesser extent the C horizons were generally areas of more gradual change, and the genesis of that change either as drainage percolating from above or from ground water influence is apparent. The point of initiation of seasonal perched water tables was also definable with this form of display.

Toposequence Uc-Mc-Lc (Figure 5.5)

Points to note from this, are that perching developed within the B horizon in both the upper and mid-slopes, and on the A/B horizon in the lower slope. The pasture on this toposequence was cocksfoot and there was no apparent drying of the profile at any slope position. Also evident was the influence of rising groundwater (inferred from SWC measurements) in the lower slope from about May 1996 onwards.

Toposequence U1-M1-L1 (Figure 5.6)

Perching developed in the B horizon in the upper and mid-slopes, and above the A/B boundary in the lower slope. There was little long-term change in SWC over the time period for either phalaris treatment, whereas the lucerne in the mid-slope started to dry the profile below 130 cm around February 1997. It was unusual that this site (ie. mid-slope) also showed a relatively wet A horizon in comparison to all others (between 15 and 40 volume %), for the bulk of the recording period.

Toposequence U2-M2-L2 (Figure 5.7)

In both the upper and lower slopes perching developed in the A horizon, whilst in the mid-slope it was initiated in the B horizon. There were also indications that another transient water table developed near the B/C boundary in the mid and lower slopes. From January 1997, lucerne in the upper slope dried the profile between 60-150 cm, and lower slope lucerne treatment started to dry the profile above the 110 cm mark in the underlying sandstone.

Toposequence U3-M3-L3 (Figure 5.8)

All perching occurred in the B horizon. In the upper slope, lucerne began to dry the profile around the 150-170 cm level to <10 volume %. Phalaris in the mid-slope started drying at around 1 metre depth from about November 1996. Groundwater influence at depth in the mid-slope was noticeable from winter 1996 to autumn 1997.

Toposequence U4-M4-L4 (Figure 5.9)

All perching was in the B horizon, although whether the increased SWC in the upper and mid-slopes warranted that description was questionable. Groundwater influence between May 1996 and February 1997 was apparent in upper and lower slope positions. Phalaris began to have a drying effect in the upper slope between 80-150 cm late in the period (November 1997). Lucerne dried the C horizon in the mid-slope around 120 cm, and in the lower slope between 70-130 cm.

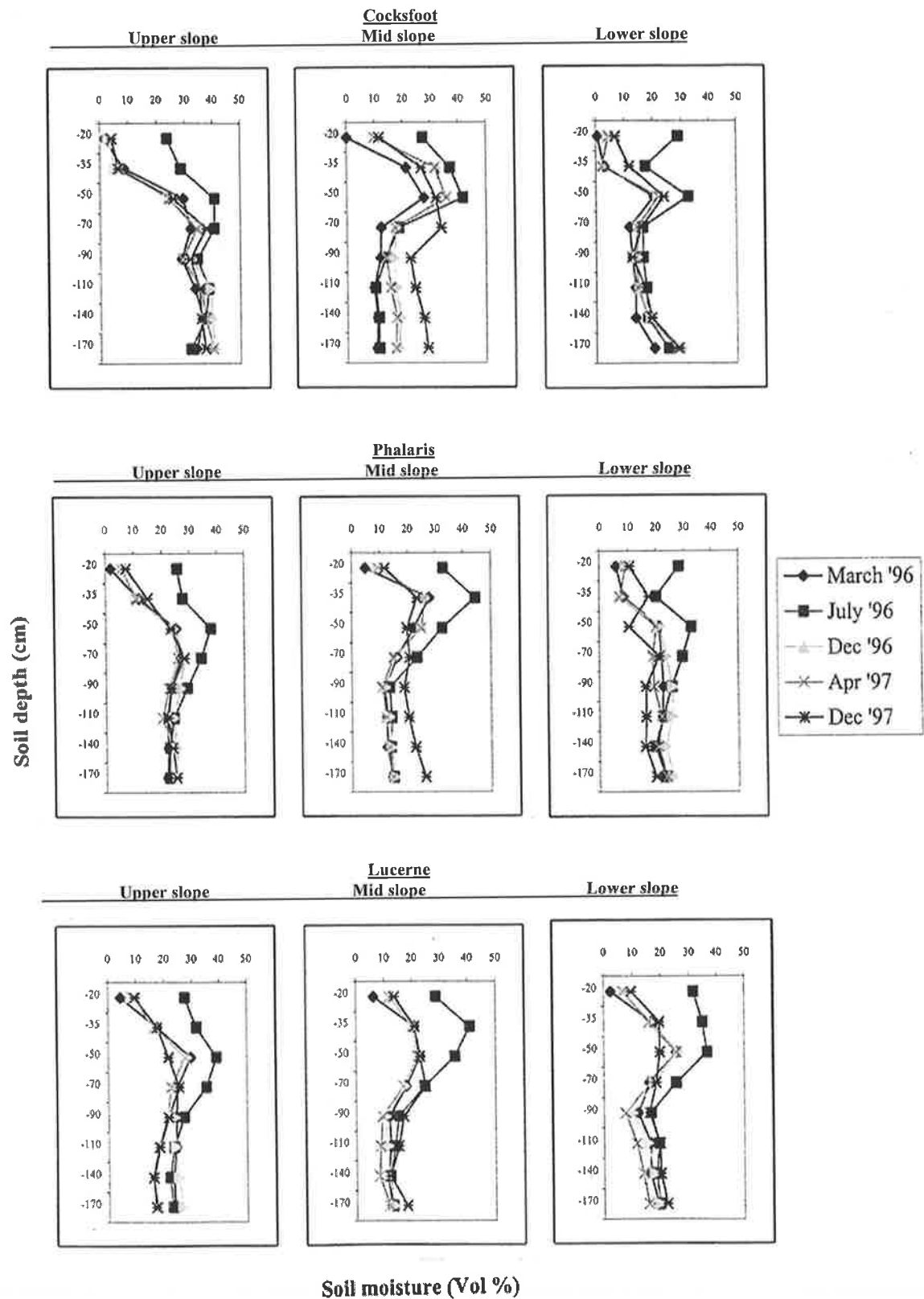


Fig. 5.10 Volumetric Soil Water Content for discrete days over the study period.

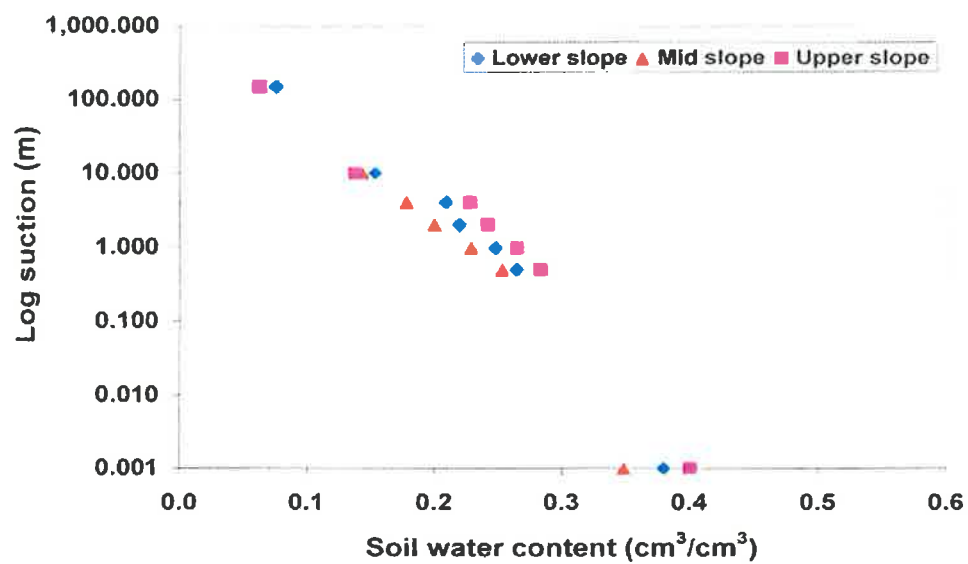


Fig. 5.11 A horizon data indicating saturation at 0.36 to 0.41 cm^3/cm^3 .

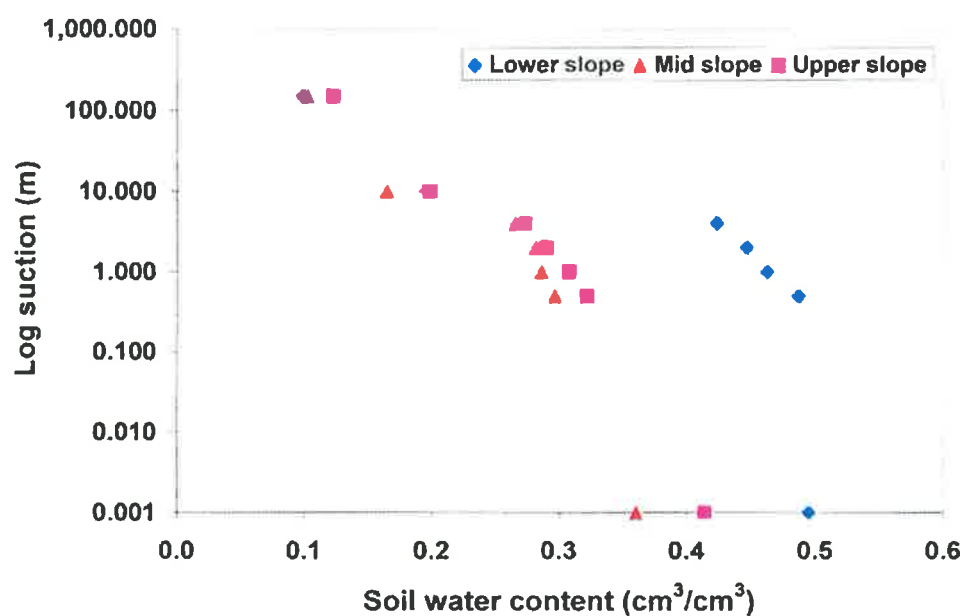


Fig. 5.12 B horizon data indicating saturation at 0.36 to 0.49 cm^3/cm^3 .

5.2.3 Root analysis

Examination of backhoe cuts, and quantitative results confirm that the bulk of roots in all three treatments, occurred in the top 30 cm of the soil profile (Table 5.5), and this is consistent with other studies (eg. Evans 1978; Ridley and Simpson 1994; Gregory 1998). From visual inspections, there was a dense mat of roots in the surface 10 cm under all treatments. These samples (ie. 0 - 10 cm) were not analysed as the quantity of extraneous organic material within the samples would have made sample preparation too lengthy a process (see Ridley 1995), and there was the possibility of sample deterioration due to time in storage whilst these samples were being prepared (Phil Ward, pers.comm.).

Table 5.5 Root length densities (cm / cm³) at discrete depth intervals for the three species tested, in October 1996 (18 months old). Values in parentheses are from Ridley & Simpson (1994). *(Note: the bulk of root material in the cocksfoot and phalaris samples below 90 cm appeared to be dead).

Depth interval (cm)	Root length density (cm / cm ³)		
	Cocksfoot	Phalaris	Lucerne
10 - 30	3.7 (4.6)	6.0 (3.7)	6.5
30 - 50	0.9 (3.1)	1.8 (2.6)	1.9
50 - 70	0.5 (0.8)	0.8 (0.5)	0.7
70 - 90	0.2 (0.2)	0.4 (0.3)	0.5
90 - 110	0.2* (0.1)	0.3* (0.1)	0.3
110 - 130	0.1*	0.2*	0.3

Given the quantity of root material recovered, particularly in the 10-70 cm region of the soil cores, it was impractical to attempt to measure this by the scanning process. Sub-samples from each core section (eg. 10-30 cm, 30-50 cm) were weighed and then scanned, and a relationship per depth interval derived between root length and weight ($R^2=0.9082 - 0.9408$). Total root length per depth interval was then calculated from known weights.

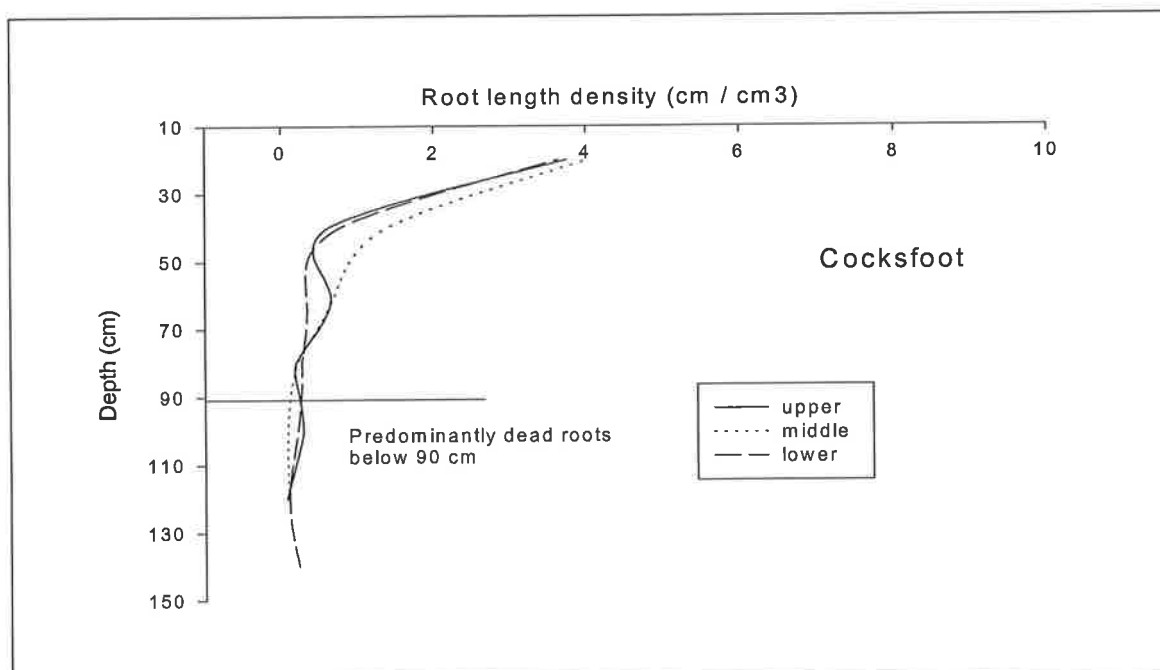


Fig. 5.13 Cocksfoot root length density at three toposcquence positions in October 1996.

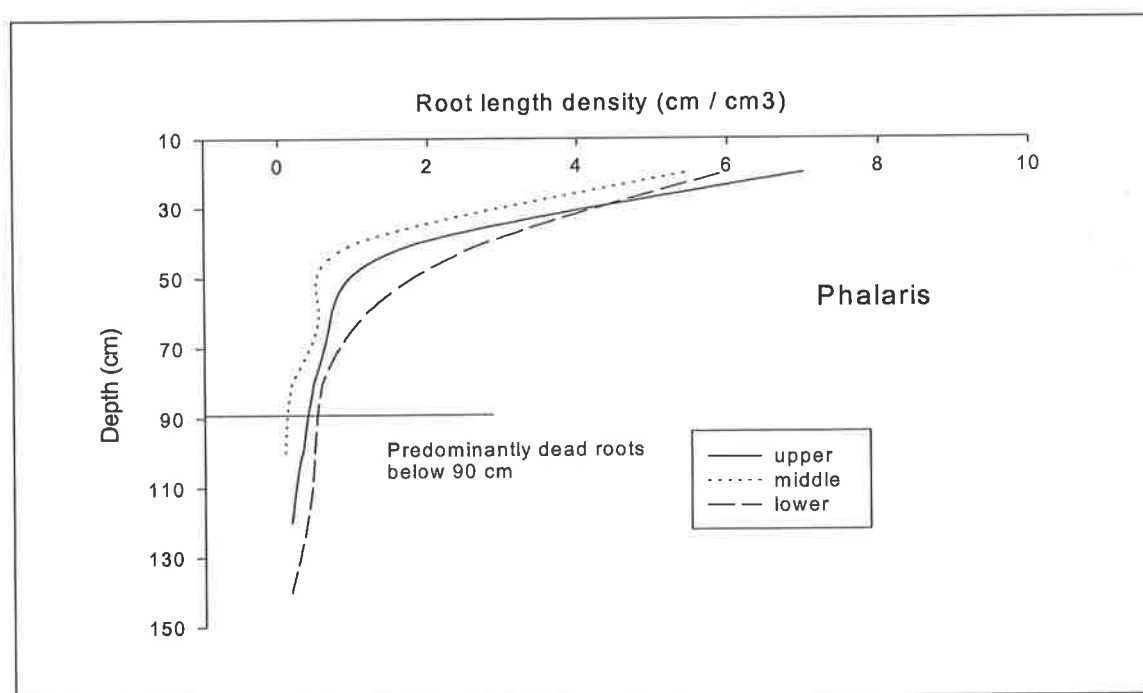


Fig. 5.14 Phalaris root length density at three toposcquence positions in October 1996 (18 months old).

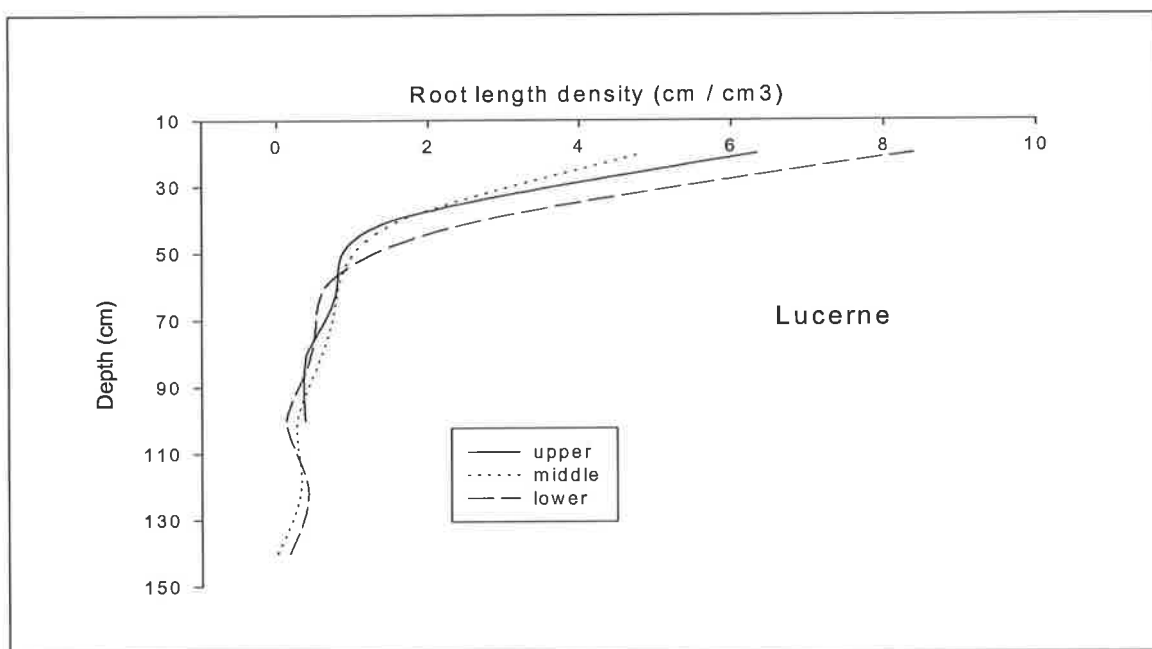


Fig. 5.15 Lucerne root length density at three toposequence positions in October 1996 (18 months old).

The data presented in Figs. 5.13-5.15 are mean data (from eight cores per treatment per slope position) from the scanned root samples taken in October 1996, approximately 18 months after pasture establishment (the cocksfoot was in the pre-existing pasture). Cores were taken to the maximum depth possible, given soil conditions and the limitations of the drilling rig, with most samples not exceeding 140 cm depth. As indicated in Figs. 5.13 and 5.14, the bulk of the cocksfoot and phalaris root material below c.90 cm was dead.

In Figs. 5.16 to 5.18, the variability of RLD across the different slope positions within and between pasture types is shown. Phalaris RLD ranged from 6.5-8.8 cm/cm³ in the 10-50 cm section of the soil profile, with similar values in the upper and lower slopes (8.8 and 8.7 cm/cm³ respectively). Below 50 cm the lower slope RLD was consistently higher, and also higher than lucerne for this part of the profile, probably accounting for its better dry matter productivity in this position. Lucerne RLD was greatest overall in the lower slope and 86% of its roots were in the 10-50 cm region compared to 68-70% in the upper and mid-slopes, indicating that in this potentially saline region, it was drawing most of its moisture from the upper part of the soil profile. Across the mid-slope there is little difference at any profile depth between the three perennials.

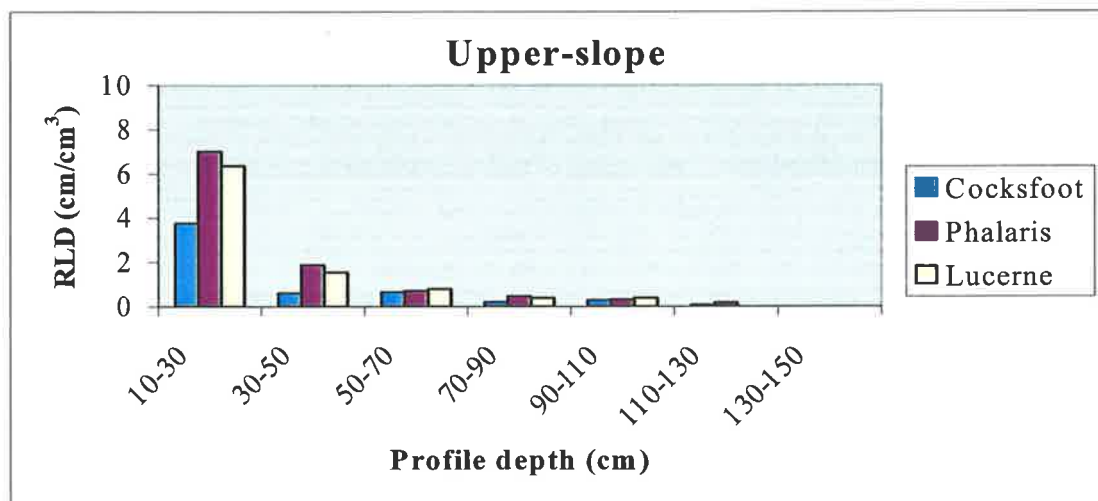


Fig. 5.16 Root length density (cm/cm^3) for three perennials in the upper-slope position at Keyneton.

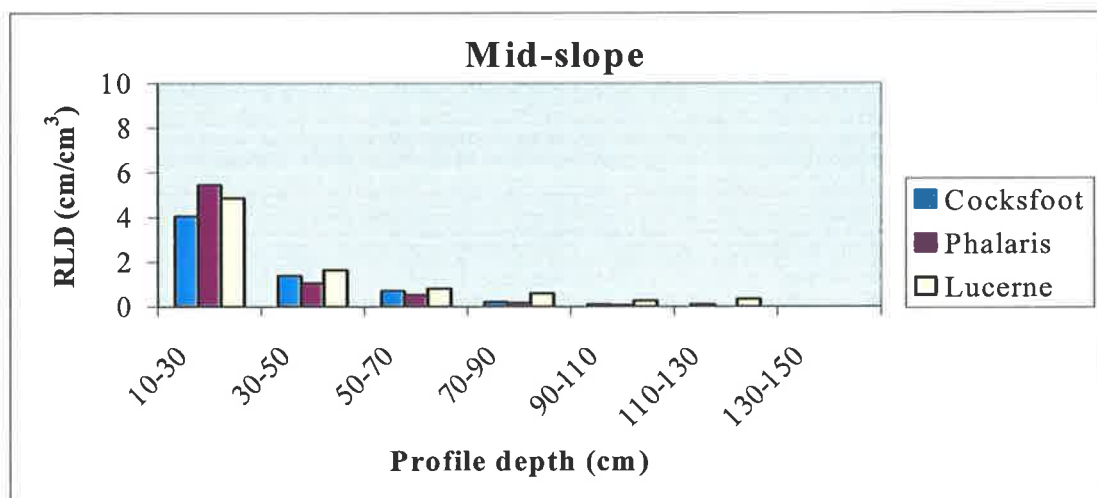


Fig. 5.17 Root length density (cm/cm^3) for three perennials in the mid-slope position at Keyneton.

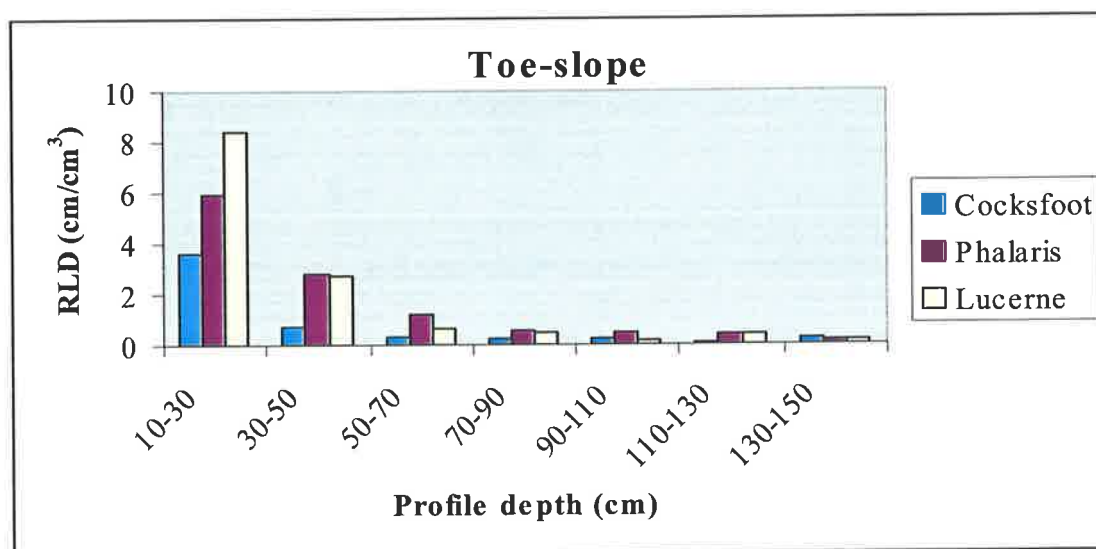


Fig. 5.18 Root length density (cm/cm³) for three perennials in the toe-slope (lower slope) position at Keyneton.

5.3 Discussion

5.3.1 Dry matter production

Although overall lucerne production was greater than that of phalaris or cocksfoot in the upper- and mid-slope positions, this was largely due to the summer dominant growth of lucerne, as reflected in the April 1997 production figures. Generally, phalaris and cocksfoot showed similar production figures, again the difference being the degree of summer activity, with phalaris showing a slight advantage in this respect. Of the three pasture types, only lucerne returned a consistently lower productivity in the lower slope position (relative to its DM production in the other two slope positions), and this is probably a reflection of the increased salinity and/or waterlogging in this toposequence position.

5.3.2 Soil water dynamics

The soil water content of the A horizon was similar under all pasture types, and as noted previously, it responded rapidly to the wetting and drying influences operating at the soil surface. Below 35 cm, in the B and C horizons, changes were both buffered and delayed. In the autumn/winter of 1996 there was a consistent development under all pastures and at all slope positions of a perched water table, mostly on or within the B horizon.

By the autumn of 1997, the lucerne, and to a slightly less degree the phalaris, had commenced to dry the profile below 35 cm, and this combined with below average rainfall reduced the development and duration of perching at this time. Under the cocksfoot, the soil profile below 35 cm remained at a higher SWC, and perching developed earlier and lasted longer than for the other two pasture treatments. Lolicato (2000) also noted that cocksfoot consistently displayed higher SWC values than phalaris or lucerne.

5.3.3 Water Balance

The known (measured) components of the water balance were rainfall, run off (overland flow and throughflow) and changes in stored soil water. Evapotranspiration (ET) and deep drainage were unknown terms and needed to be calculated. Two methodologies were trialed, the composite approach of Ward and Dunin (2001) and that of Ridley *et al.* (1997) based on winter mean water content (WMWC).

The composite approach

Ward and Dunin (2001) presented a relatively simple methodology for estimating the growing season ET (ET_{gs}) from the shallow duplex soils of south-western Western Australia with growing season rainfall >250 mm. The two main assumptions are:

- from the break of season until there is a rapid decline in SWC, actual ET is assumed to be equal to potential ET, and
- following this seasonally dependent date (generally in September-October), changes in SWC (plus any rainfall) will provide a reliable estimate of actual ET.

The model is described mathematically as:

$$ET_{gs} = Et_i + Et_j$$

$$Et_i = \sum(\text{potential ET}) \text{ for the period when } S > S_c$$

$$Et_j = \Delta S + \sum P \text{ for the period when } S \leq S_c$$

Where S is soil water storage, P the precipitation, and S_c is some critical value of SWC above which ET occurs at potential rates (Ward and Dunin 2001). S_c is estimated (within a range) from fortnightly SWC measurements and the date of the last SWC

measurement before the beginning of the rapid decline is used as the changeover date from potential ET to ET estimated from SWC changes. ET_{gs} is then calculated as above.

Ward and Dunin (2001) state that the main advantage of this composite approach “is that a robust estimate of ET can be obtained from routinely measured soil and meteorological variables, without recourse to labour-intensive and expensive installations”. They also see the estimate as being free from the biases associated with some other estimations of ET, and there is no need to assume zero drainage as many estimates wrongly do. In this model, deep drainage is most likely to occur during the period when ET is equal to potential ET, not during periods of rapid SWC decline.

If lateral flows are also measured, an estimate of deep drainage is possible using the estimated ET and seasonal rainfall data. However, Ward and Dunin (2001) reiterate the findings of Ward *et al.* (1998) that such an estimate is subject to great uncertainty given the range of errors in estimations of individual water balance components.

Winter mean water content

Ridley *et al.* (1997) estimated that deep drainage would occur once the soil profile of interest reached its WMWC, which corresponds approximately to the soil's field capacity. They defined WMWC (for Rutherglen, Victoria) as “the mean soil water content between July and September when the soil was wet and evaporation low”. For periods when profile SWC is $\geq$ WMWC, both actual ET (ET_a) and drainage (D) have to be calculated.

For these periods, Ridley *et al.* (1997) estimate ET_a as potential ET using the modified Penman equation (Doorenbos and Pruitt 1984). The FAO have since reviewed this (Allen 1998) and 19 other estimation procedures under the auspices of the American Society of Civil Engineers (ASCE) and now recommend that their Penman-Monteith method is maintained as the sole standard method for ET computation from meteorological data. On the assumption that no drainage has occurred whenever the soil is drier than the WMWC, ET_a is calculated as the difference in the remaining terms of Equation 5.1, below:

$$D = P - ET_a - R - \Delta S \quad \text{Eqn. 5.1}$$

where D is drainage, P is rainfall, ET_a is the actual evapotranspiration, R is runoff and ΔS is the change in SWC.

Between 1990 and 1993, Ridley *et al.* (1997) calculated a mean annual ET (mm) from phalaris of 642 mm and for cocksfoot, 619 mm. Mean calculated annual drainage for the same period under phalaris was 66 mm and under cocksfoot 69 mm (see Table 5.6), and compared well with previous estimates for the Rutherglen site.

Table 5.6 Calculated ET and drainage under phalaris and cocksfoot at Rutherglen, Victoria (from Ridley *et al.* 1997).

Year	Calculated evapotranspiration (mm)		Calculated drainage (mm)	
	Phalaris	Cocksfoot	Phalaris	Cocksfoot
1990	547	538	123	126
1991	513	495	2	3
1992	689	691	117	107
1993	820	754	20	38
per year	642	619	66	69
Total	2568	2477	262	274

Calculation of Evapotranspiration

The composite approach (Ward & Dunin 2001)

The growing seasons for 1996 and 1997 were deemed to be from May 29-Dec 9 and May 28-Dec 4 respectively, to fit SWC measurement dates and match the early June to early December approach of Ward and Dunin (2001). These break of season dates corresponded well with actual rainfall data in both years.

The timing of the rapid decline in SWC (and the corresponding S_c) was variable across the 15 plots over a period of about three weeks in 1996 and about six weeks in 1997 (Table 5.7). ET calculated for the growing season (Table 5.8) averaged 252 mm in 1996 and 233 mm in 1997, and rainfall for the corresponding periods was 447 mm and 259 mm respectively. Growing season ET was in the range reported by Scott and Sudmeyer (1993).

ET for those periods outside of the (annual) growing seasons above was estimated as ET_j (ie. it was assumed to equal rainfall and changes in SWC), bearing in mind the caution urged by Ward *et al.* (1998) and Ward and Dunin (2001). Given that these measurements related to perennial pastures rather than annuals, this assumption is probably reasonable. In any case it provides for a comparison with the WMWC methodology which also assumes that ET equals rainfall and SWC changes in periods when drainage is unlikely to occur.

Ward and Dunin (2001) noted that for their lucerne treatment, ET was probably underestimated as water uptake was occurring from below the depth of SWC measurement. This error would also apply at Keyneton where lucerne roots have penetrated below 1.8 m. The root zone of the other two perennials falls within the zone of measured SWC.

Table 5.7 Dates of S_c (Ward & Dunin 2001), as well as WMWC (mm) and the period of >WMWC (Ridley *et al.* 1997) for all plots at Keyneton, 1996-1997.

Plot	Sc 1996	Sc 1997	WMWC (mm)	days >WMWC (1996)	days >WMWC (1997)
Uc	Oct-18	Sep-26	587	Jul 19-Oct 18	
Mc	Oct-18	Oct-24	354	Aug 9-Oct 24	Sep 17-Sep 26
Lc	Oct-18	Oct-24	355	Jun 27-Aug 30	Sep 17-Sep 26
U1	Nov-07	Nov-14	429	Jun 7-Nov 7	
U2	Nov-07	Nov-14	422	Jun 27-Nov 21	
U3	Nov-07	Nov-14	500	Aug 2-Nov 21	
U4	Nov-07	Nov-14	395	Jul 12-Nov 7	
M1	Oct-24	Nov-14	391	Jun 27-Oct 24	
M2	Oct-24	Nov-14	415	Jun 27-Oct 18	
M3	Oct-24	Nov-14	390	Jun 27-Oct 18	
M4	Oct-24	Nov-14	273	Jun 27-Oct 24	
L1	Oct-24	Nov-14	468	Aug 2-Oct 24	
L2	Oct-24	Nov-14	322	Jun 27-Oct 18	
L3	Oct-24	Nov-14	436	Jun 27-Oct 24	
L4	Oct-24	Nov-14	442	Jun 27-Oct 24	

Table 5.8 ET and drainage calculations for Keyneton plots using methodology of Ward & Dunin (2001) and Ridley *et al.* (1997)

	June - December		March 1996 – December 1997			
	ET _{gs} (mm)		ET (mm)		Deep Drainage (mm)	
	1996	1997	Ward	Ridley	Ward	Ridley
Uc	278.5	198.3	709.7	704.7	137.0	137.6
Mc	233.6	167.9	606.8	681.1	147.8	102.0
Lc	259.5	177.9	625.9	604.3	97.6	119.2
U1	212.4	223.1	755.9	707.8	148.4	200.5
U2	233.0	250.4	906.2	823.8	65.1	138.7
U3	251.0	257.2	867.4	863.3	49.9	44.0
U4	220.7	265.9	805.9	776.6	109.5	111.8
M1	253.5	236.4	706.4	625.8	72.0	155.9
M2	254.3	247.0	772.6	685.6	100.5	199.7
M3	260.0	245.1	740.3	617.7	52.9	175.7
M4	285.6	265.4	753.6	621.3	60.3	181.6
L1	245.7	234.6	761.5	748.2	31.1	41.6
L2	245.4	211.8	607.4	576.8	155.5	194.7
L3	249.0	241.4	847.2	741.9	84.7	180.7
L4	300.3	266.6	974.0	724.9	-50.7	212.8
Mean	252.2	232.6	762.7	700.3	84.1	146.4

The WMWC approach (Ridley *et al.* 1997)

The WMWC of the different plots was quite variable (Table 5.7) and although this was expected given the degree of heterogeneity of the soils there, the range of variability was nevertheless surprising. This variability is reflected in the timing and duration of those periods predicted by the WMWC as being when drainage could occur (Table 5.7), and consequently on the periods when ET is calculated as being equal to P and ΔS .

Table 5.8 shows the ET calculated for the 22 months of the study period for which SWC measurements were available, and it can be seen that this methodology generally produced a lower estimate of ET (mean of 700 mm) than that of Ward and Dunin (2001) (mean of 762 mm). Although non-significant, there was a slightly improved linear relationship between calculated ET and dry matter production using the Ward and Dunin (2001) approach ($R^2 = 0.44$ versus 0.39).

Calculation of deep drainage

Using the values of ET determined by both methodologies above, deep drainage was calculated by difference of the other terms of the water balance (see Eq. 5.1). The method of Ridley *et al.* (1997) generally predicted higher drainage (mean 146 mm) than

that of Ward and Dunin (2001) (mean of 84 mm) as would be expected given their respective estimates of ET.

Drainage predictions under the different perennials are completely different under the two methods for ET calculation (Table 5.9). From other studies (eg. Lolicato 2000; Whitfield 2001) the expected pattern of drainage estimates would be a decrease from cocksfoot to phalaris to lucerne, and this was the case following the Ward and Dunin (2001) approach. Using the methodology of Ridley *et al.* (1997) however, completely reversed this trend

Table 5.9 Deep drainage estimates, by difference, after calculating ET by two different methods for plots at Keyneton.

	Deep Drainage (mm)					
	Ward & Dunin (2001)			Ridley <i>et al.</i> (1997)		
	Lucerne	Phalaris	Cocksfoot	Lucerne	Phalaris	Cocksfoot
	65.1	148.4	137.0	138.7	200.5	137.6
	49.9	109.5	147.8	44.0	111.8	102.0
	72.0	100.5	97.6	155.9	199.7	119.2
	60.3	52.9		181.6	175.7	
	155.5	31.1		194.7	41.6	
	-50.7	84.7		212.8	84.7	
Mean	58.7	87.9	127.5	154.6	135.7	119.6
% of ET	10%	15%	20%	22%	19%	18%
% of P	7%	10%	14%	17%	15%	13%

The amount of drainage predicted, particularly by the Ridley *et al.* (1997) methodology, is generally greater than would be expected given the suggestion by Nulsen (1993) that recharge is likely to be approximately 10% of the average ET of the crop or pasture. The results predicted by the Ward and Dunin (2001) method are certainly closer to expectations, and drainage as a percentage of rainfall is within the range reported by Hatton *et al.* (2002) for the 'Ucarro' site in WA. As well the WMWC approach has a large influence on any subsequent values of water balance components whereas the composite method is more robust, showing only minor effects in subsequent values by varying the S_c date by one or two weeks. Given the greater confidence in the 'reality' of the estimates from the Ward and Dunin (2001) approach, these results will be deemed to be representative of the Keyneton site.

Deep drainage has occurred under all perennials and at all slope positions with only one exception (a lucerne plot in the lower slope). Drainage was highest in the upper slope plots (mean 102 mm), slightly lower at mid-slope (mean 87 mm) and down to a mean of 54 mm in the lower slopes. This follows expected trends with freely drained soils in the upper slopes, and the lower permeabilities of the mid and toe-slope soils. As well, in the lower slopes raised groundwater tends to reduce vertical drainage.

Substantial drainage has occurred in soils that have shown reduced SWC over the study period. However, this is consistent with other studies (eg. Ward and Dunin 2001; Ward *et al.* 2002; Hatton *et al.* 2002) that have shown that although lucerne may achieve greater annual water use than annuals or some other perennials because of its greater rooting depth and longer growing season, it still has an annual recharge rate in the order of 10–30 mm (Hatton *et al.* 2002). At Keyneton, it is likely given the prevailing conditions, that summer thunderstorms may be a significant contributor to recharge via preferential macropore flow under all pastures.

It is quite apparent from this and a number of other studies that lucerne uses more water than most conventional pastures in many areas of Australia (Crawford and Macfarlane 1995; Lolicato 2000; Ward *et al.* 2001, 2002; Hatton *et al.* 2002). With its greater root distribution it extracts more water from deeper layers of the soil profile, producing a greater depth of relatively dry soil to act as a buffer for the following season. It is this increased storage zone for subsequent rainfall (Ward *et al.* 2002), that reduces the amount of water leaving the root zone as deep drainage.

5.3.4 Root analysis

At Keyneton, the decline in root material below 30 cm was quite marked for all species. Ridley and Simpson (1994) reporting a study on a duplex red soil with a similar A/B horizon at c.30 cm, found a more gradual decline, with substantial root length densities to 50 cm (approximately 2/3 of that in the 10–30 cm interval). The Keyneton data suggested that the high bulk densities of the B horizon in this environment, may encourage root proliferation in the surface soil.

From visual inspection of soil pits, there were distinct differences in rooting depth between species, with cocksfoot roots being observed to 100-120 cm, phalaris to c.150 cm and lucerne > 200 cm. The cocksfoot had been established for many years, so had probably established some equilibrium rooting depth for that environment, and was therefore unlikely to penetrate beyond its current limit. Soil pits dug some 18 months after the establishment of the lucerne and phalaris pastures, showed root extension to the above depths had already occurred in that time. Subsequent pits did not indicate that phalaris was increasing its penetration to any extent at that time, and it was not possible to excavate to the full extent of the lucerne root system, as its tap roots were following cracks into the underlying sandstone. Both the phalaris and the lucerne roots had taken advantage of macropores, cracks and other faults to extend into the lower depths. There was little evidence of roots elsewhere in the soil matrix at depth.

From Figs 5.13-5.15 showing root length density (RLD) for each toposequence, the relatively low RLD of cocksfoot in the 10 - 30 cm range is obvious. In the 30 - 70 cm section of the soil profile, cocksfoot RLD is in the range of 0.25 - 0.5 cm/cm³, whereas phalaris and lucerne are closer to 0.5 - 1.0 cm/cm³. Phalaris and lucerne have more roots in the 10 - 30 cm region, and phalaris appears to show differences related to toposequence position with a higher RLD below 30 cm, in both the upper- and toe-slope positions. This is consistent with production trends for phalaris, with the mid-slope position generally recording lower pasture cuts. Lucerne RLD seems consistent below 50 cm, at all toposequence positions, its main advantage over the other two pastures being that there are in fact substantial live roots extending down the profile. At depth, many of these roots are still >1 mm in diameter compared to the fine hair-like roots of both cocksfoot and phalaris (Figs 5.19 & 5.20). Decreases in SWC under the three pasture types are consistent with the densities and morphology of their root systems.

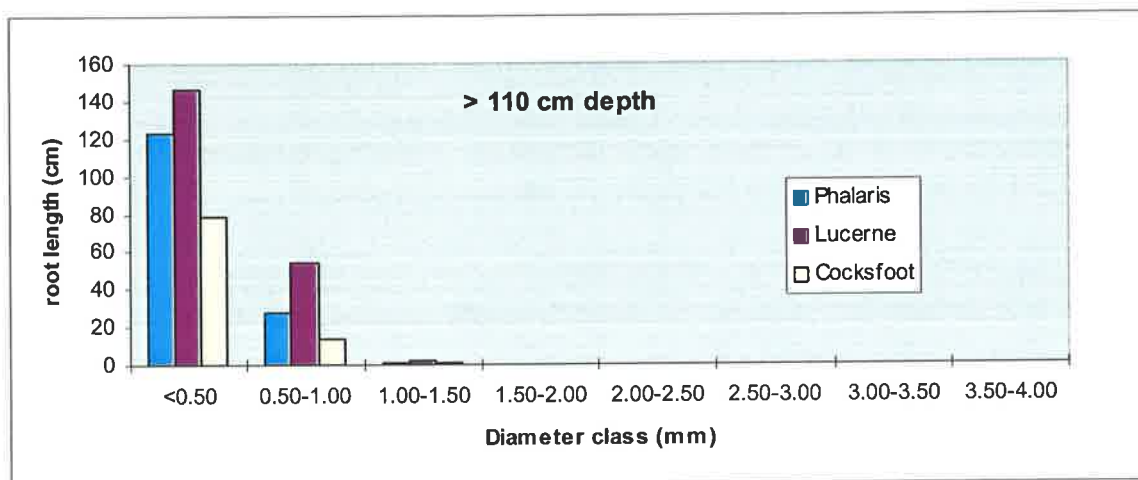


Fig. 5.19 Core samples showed a consistently higher lucerne root length in the diameter classes up to 1 mm, in the profile below 110 cm.

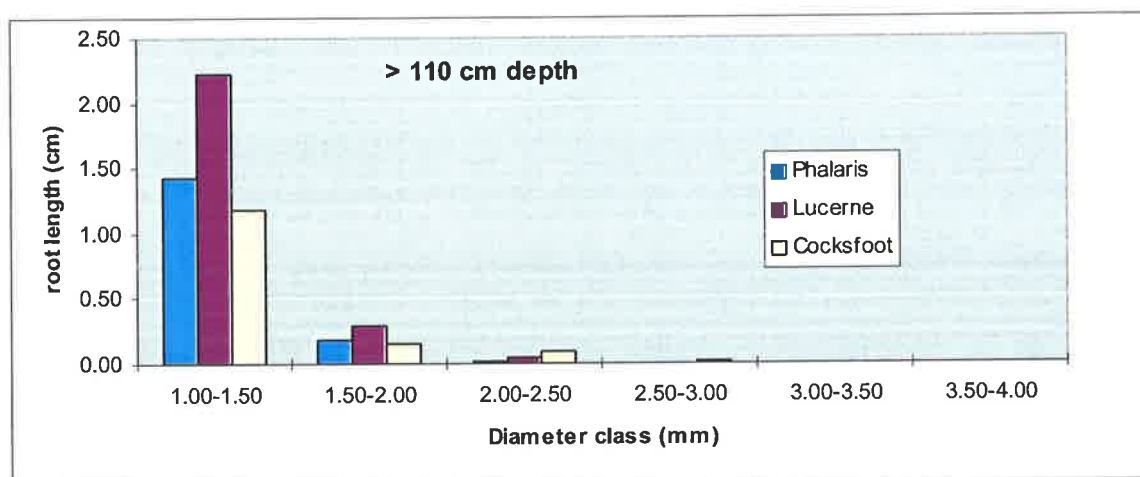


Fig. 5.20 As above, lucerne root length was generally higher in the diameter classes >1 mm below 110 cm. The exception in the 2-2.5 mm diameter class is unusual. It must also be remembered when trying to relate root distribution to soil water dynamics, that only the lucerne roots were likely to be alive.

5.4 Conclusions

The results of this study show potential for the introduction of summer active perennial pastures into the farming systems of the Mt Lofty Ranges (SA). It has been shown that both lucerne and phalaris will produce a higher amount of dry matter than cocksfoot in this environment, regardless of topographic position. Additionally, both lucerne and phalaris will reduce the soil water content whereas under cocksfoot based pasture, SWC was shown to increase. Given the need to induce pre-winter SWC deficits to lessen the likelihood of waterlogging and salinity, the adoption of perennials other than cocksfoot is advocated in this grazing system. In this regard, over the period of this study, both phalaris and lucerne showed a reduction on the estimated recharge experienced under cocksfoot.

There were indications from production figures and root growth that toposequence position may have influenced species performance in this environment. Whilst in the lower slope position, lucerne did not appear to be at a disadvantage relative to phalaris, in terms of root characteristics and soil water uptake, there were differences in dry matter production favouring phalaris in that position when compared to lucerne. However, these differences were more marked when the comparison was just between the range of production figures of the lucerne pastures at the different slope positions. Whereas cocksfoot and phalaris production over the study period varied down the toposequence by about 1 tonne/ha, lucerne production in the lower slope position was around 3.5 tonnes/ha less than the upper-slope and 2.5 tonnes less than the mid-slope production. Salinity and waterlogging are known to increase in the lower slopes, and this may have impacted on lucerne growth and development, indicating perhaps that phalaris may be a better choice for these positions in the toposequence to maximise productivity.

The considerable volumes of runoff, both overland flow and particularly throughflow, that were generated in the mid and lower slopes from all pastures highlighted the need for a range of measures to reduce recharge. Drains are probably the only way of coping with the excessive volumes of flow generated over a small timeframe by single storm events. One such chain of events produced most of the large runoff values that appeared

in the water balance. In this case there had been 45 mm of rain over a two week period in September 1996, followed by three days in which another 54 mm was recorded (12.6, 14, and then 27.6 mm). With many of the site profiles saturated, down-slope water movement on and within the soil was almost guaranteed.

One of the difficulties of recharge control is this inability to cope with episodic rainfall events that occur just when conditions are optimal for deep drainage. Over the long term, if lucerne were established over the entire slope, it is likely that there would be a gradual lowering of watertables, given the data from this study. The size of the SWC autumn deficit, and the increased ET give good indications of this. However, even with this advantage, there are still going to be substantial sporadic deep drainage events. Without a change in the current system though, there is no reason to expect anything other than a steady decline in catchment health, and this should provide sufficient incentive for further investigation of agronomic and other options.

Chapter 6

Comparison of Water Quality in runoff from different pastures

6.1 Literature Review

6.1.1 Introduction

Vegetation clearance in Australia has not only altered the amount of water available to move through catchments, but also its relative partitioning. With reduced canopy interception and evapotranspiration, more water is available to move as both overland flow, and into the profile where it may move laterally as throughflow, or drain beyond the root zone to the groundwater (Passioura 1992; Cox and Fleming 1997; Fleming and Cox 1998; Stevens *et al.* 1999).

The quality of water in a catchment is intimately linked to the hydrologic processes operating within it, with the mobilisation of salts, nutrients, particulate and organic matter related to the movement of water on and through the soil. Given the duplex soils of the Adelaide Hills, South Australia, this often leads to on-site problems of salinity, acidification and low fertility. Off-site there may be concerns over nutrient transport to waterways, particularly those feeding into public water supplies. P, N and DOC cause problems with water quality and treatment that are both difficult and costly to rectify (Nelson *et al.* 1996).

Nitrogen and phosphorus are the two main nutrients of concern for their potential to exacerbate eutrophication of standing water bodies. As well, the leaching of nitrate beyond the root zone and into groundwater, may lead to unacceptable levels in potential drinking water. Nitrogen applied to crops and pastures as organic waste or fertiliser (generally as NH_4^+) is highly soluble and unlikely to be adsorbed to soil particles (White and Kookana 1998) unless there is an excess of positive exchange sites (eg. Fe or Al oxides at low pH). Further losses can be attributed to N fixed by legumes, mineralised during summer/early autumn and leached by autumn rains (Simpson 1962, cited in Ridley *et al.* 1999). Excretal N, particularly that from urine, is in a labile form and vulnerable to loss (White *et al.* 2000; Ridley *et al.* 1999).

Phosphate solubility in the various forms of fertilisers, and in organic wastes, is particularly variable, and with a high propensity for adsorption, P concentrations are generally low in the soil solution. Phosphate is generally considered immobile in all but very sandy soils. Its presence in relatively high quantities in streams and water storages of southern Australia is attributed to surface soil erosion adjacent to the water course or point source pollution. However, recent work on hillslope plots demonstrated that relatively high fluxes of P and dissolved organic carbon (DOC), were moving through the soil profile of texture contrast soils as throughflow, in addition to overland flow (Smettem *et al.* 1994; Fleming and Cox 1998; Stevens *et al.* 1999; Cox *et al.* 2002).

Shallow drainage is recognised as one of a range of important management strategies that can be incorporated in the early stages of remediation of waterlogged/saline catchments (McFarlane and Cox 1992; Cox and McFarlane 1995; Cox *et al.* 1996^a). Artificial drainage of waterlogged soils in the Mt Lofty Ranges however, is complicated by the fact that any discharge may enter rivers and reservoirs used for public water supplies, and little research has been done on whether these drains enhance the movement of any nutrients and DOC.

The aim of this component of the research was to determine if pasture type affects the concentrations of chemicals (including nutrients, DOC and metals) in overland flow and throughflow. Thus the best management options could be suggested to minimise pollution potential. Additionally, the form (particulate or dissolved) of the chemicals was determined, as control methodologies may be form-dependent.

6.1.2 Phosphorus

The loss of phosphorus (P) from a catchment may lead to surface water eutrophication (Yeates 1993; Nelson *et al.* 1996; Kirkby *et al.* 1997; Baldwin 1998). Under suitable conditions this may induce algal blooms including the toxic microalgae (eg. blue-green algae) (Sharpley *et al.* 1994; Baldwin 1998). Nelson *et al.* (1990) report that Adelaide Hills' streams contain high concentrations of P thought to come from point sources and riparian soil erosional processes (Cox and Kirkby 1995). Sharpley *et al.* (1995) and

Nelson *et al.* (1996) suggest that recent controls on point source pollution, shift the focus of P inputs to diffuse sources such as agricultural runoff.

Leaching of P is normally thought to occur only in coarsely structured soils due to rapid infiltration, and in sandy soils due to a lack of adsorption sites (Mansell *et al.* 1977; Cox and Kirkby 1995; Kirkby *et al.* 1997). It is also thought that P will not leach more than a few centimetres in a profile containing significant amounts of clay, with fixation onto Fe and Al oxides and hydroxides (Rajan *et al.* 1974; Bolt 1976; Kirkby *et al.* 1997). Most P export from agricultural catchments is thought to be via overland flow (Trughill 1990) attached to soil particles.

Some studies now show considerable amounts of P may move off pastures in the dissolved state (Sharpley *et al.* 1994; Fleming and Cox 1998). Kirkby *et al.* (1997) note that macropore flow which bypasses the matrix, reduces P adsorption opportunities during infiltration therefore leading to the chance of significant transport via subsurface pathways (Cox and Kirkby 1995; Cox *et al.* 1996^b; Kirkby *et al.* 1996). In this instance, despite the clay content of the profile, there is the potential for environmental problems.

Conditions for macropore flow may be less restrictive than previous research indicated. Earlier studies suggested that the soil must be saturated (with surface ponding) to initiate macropore flow (Bouma 1981; Beven and Germann 1982). Later studies showed that flows could occur without ponding, however this was dependent on the formation of near surface saturated layers (Adreini and Steenhuis 1990) or saturation on either textural or structural boundaries (Steenhuis and Muck 1988).

Growing evidence however suggests that macropore flow is possible regardless of the above constraints. Bootlink and Bouma (1991) showed that more than half of the water applied to undisturbed cores of a clay soil, was transmitted via macropore flow. Other studies indicate the likelihood that macropore flow will bypass the unsaturated soil matrix, given conditions conducive to its initiation, such as the saturation of individual peds or ped-surface filming (Bouma *et al.* 1979; Russell and Ewel 1985; Seyfried and Rao 1987; Sollins and Radulovich 1988; Booltink and Bouma 1991; Trojan and Linden 1992).

Smythe (1995) quantified the amounts of dissolved and particulate P leached through soils in the Adelaide Hills and found that P movement was dependent on slope position

and independent of soil P adsorption properties. This was explained by the short residence time of solutional P, as it rapidly passed through macropores thus bypassing the soil matrix (Kirkby *et al.* 1997).

6.1.3 Nitrate

Nitrate from N-fertilisers, legume N fixation and manure/urine, is a notorious pollutant (Bronswijk *et al.* 1995), and losses from farmland are increasingly recognised as contributors to environmental problems (Skaggs and Chescheir 1999). Nitrate is particularly problematic due to its high solubility, and its weak bonds in the soil (Troeh *et al.* 1980) compared with NH_4 , K and P ions. Generally, the rate of nitrate movement through soil is equal to the mean velocity of water (White and Kookana 1998), and normal processes of organic matter decomposition may produce significant quantities of leached nitrate, regardless of fertiliser application (Troeh *et al.* 1980).

As well as movement via overland flow, in solution or attached to soil particles, substantial quantities of nitrate (NO_3) may be leached below the root zone (Chapman and Mason 2000; White *et al.* 2000). A combination of enhanced nitrification and NO_3 leaching is considered the major contributor to accelerated soil acidification in south-eastern Australian soils (Helyar and Porter 1989; Ridley *et al.* 1999). Dowdell *et al.* (1987) in a UK study, reported high nitrate losses through subsurface drains compared to surface runoff rates while Skaggs and Chescheir (1999) refer to studies showing that improved subsurface drainage increases N losses to surface waters whilst reducing P, sediment and pesticide losses. Passioura and Ridley (1998), in discussing the problems of nitrate in water, raise the possibility of “public concerns about the environmental acceptability of agriculture”, given problems of nitrate leaching and eutrophication of streams. With the move to more intensive agricultural systems, there is an increasingly serious risk of groundwater pollution.

In studies on duplex soils at Book Book, in southern NSW, nitrate-N concentrations in soil water were generally low in surface flows, but up to 50 g/m^3 in subsurface flow and between $5\text{--}20 \text{ g/m}^3$ in deep drainage (Passioura and Ridley 1998). Chapman and Mason (2000) and Ridley *et al.* (2000) (see also Williams 1998) report losses via deep drainage and sub-surface flows from high input pasture systems in the range of $2\text{--}30 \text{ kg N/ha/yr}$. They state that losses of the higher magnitudes pose a threat to stream health, and the

long-term sustainability of pasture systems because of soil acidification. In light of the above, the design and management of agricultural drains, will need to consider the quality of water being collected and transported (Skaggs *et al.* 1995).

6.1.4 Dissolved organic carbon

Dissolved organic carbon (DOC) degrades the colour and taste of water (Naidu *et al.* 1993), and its removal is a costly process in public water supplies (Cox *et al.* 1998). The streams of the Adelaide Hills have relatively high levels of DOC, averaging 10-13 mg/L and reaching 20-25 mg/L during storm events (Chittleborough *et al.* 1992). These levels are high enough to impact on water quality as noted above, creating difficulties in flocculation processes (Nelson *et al.* 1996), and through reaction with chlorine in the chlorination process, the production of high levels of carcinogenic trihalomethanes (Bursill *et al.* 1985).

Although the reasons for high DOC are unknown, various reports have shown an effect of soil type on DOC movements (Herbaults 1980; McDowell and Wood 1984; Moore 1989). Nelson *et al.* (1990) sampled streams receiving drainage from a range of soil types and found that drainage from catchments with very coarse surface textures (sands, loamy sands) over a clay B horizon were a significant source of DOC (Chittleborough *et al.* 1992).

Studies show that in winter, most DOC is transported via macropores and subsequent lateral flow. In summers, the highest concentrations are in overland flows, due to the hydrophobic nature of the dry organic material on and near the soil surface and it was found that the longer the dry period, the higher the DOC in the subsequent flushing flow (Chittleborough *et al.* 1992).

6.1.5 Metals

Fleming and Cox (1998) is the only published Australian study (at the time of writing) that assesses the mobility of metals in both overland flow and throughflow from texture contrast soils. This study of two grazed dairy pastures at Flaxley, South Australia, found that Al and Fe loads were the highest of the elements measured, a reflection of their relative abundance in the soil. Concentrations were similar in overland flow and throughflow, although the highest load was in overland flow (88-96% of runoff was

overland flow), and they suggest that these quantities moving off pasture are unlikely to seriously impact water quality.

Potassium (K), then Calcium (Ca), Magnesium (Mg) and Sodium (Na) were the next highest loads. K moved mostly in the dissolved form in overland flow, but in throughflow the particulate and dissolved forms were variable. Whilst overland flow concentrations of Ca and Mg were evenly divided between particulate and dissolved forms, throughflow carried more in dissolved forms. Na moved mainly in the dissolved form, and the authors suggest that this was probably associated with Cl^- as salt (Fleming and Cox 1998). All four elements had substantially higher concentrations in throughflow.

The micronutrients Boron (B), Copper (Cu) and Manganese (Mn), were recorded as low loads. Zinc (Zn) loads were higher than expected, but it was felt that this might have been a result of the cattle being fed Zn supplements.

Generally the study concluded that, whilst in environmental terms the levels of P and N moving off-site may have been significant, there was no similar concern with the other chemicals (Fleming and Cox 1998).

6.2 Materials and Methods

6.2.1 Water sampling

Water samples (overland flow and throughflow) were collected from all plots during or following rainfall events (within 24 hours), stored in polyethylene bottles and frozen prior to analysis (see Fig. 6.1). Ten samples were collected in 1995 and 1996, whilst only two were available in 1997 due to the extremely low number of rainfall events that produced runoff or throughflow. From the reverse interceptor drains and the contour banks, overland flow samples were collected from flow through the V-notch or from the centre of the stilling pool. Throughflow (reverse interceptor) was directed through a series of pipes (5 mm diam) set through the weir at different stage heights, and transported via plastic tubing to storage bottles in a covered collection area. Tipping bucket throughflow (from the buried drains) was collected by pipette from the bucket mechanism within the covered sumps.

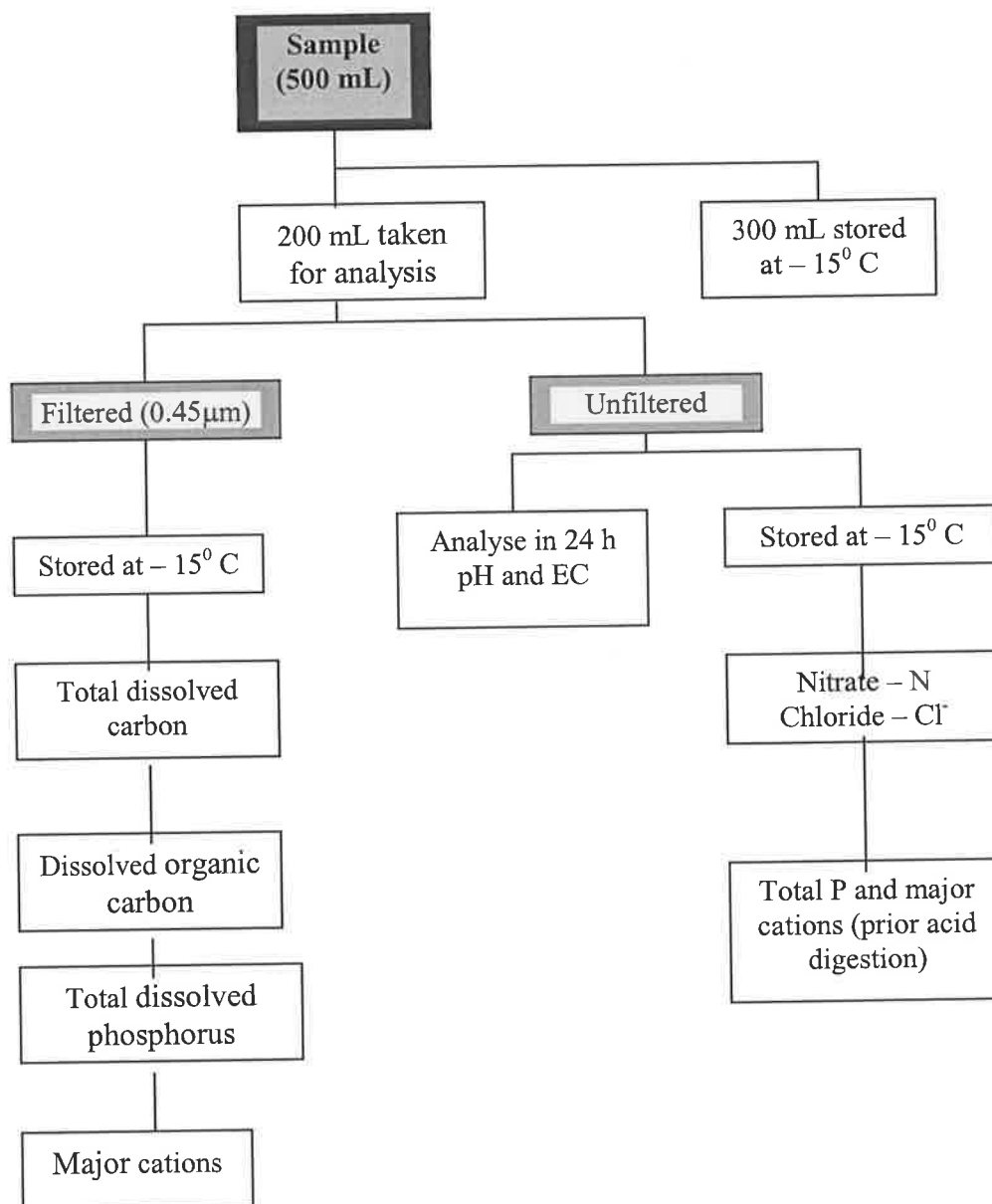


Fig. 6.1 Methods for chemical analyses of the water samples.

Samples were analysed for pH and electrical conductivity (EC) using standard techniques (Rayment and Higginson 1992). Nitrate and Cl were measured on the unfiltered leachate using automated colorimetric methods (Rayment and Higginson 1992). Major elemental composition including P were measured by inductively coupled plasma-atomic emission spectroscopy analysis (ICP) on both filtered ($<0.45\mu\text{m}$) leachate ('dissolved') and unfiltered, acid-digested ('total') leachate. The acid digest

which removes organic matter (and partly dissolves very fine clay) involved adding 5mL of 0.1 M nitric acid (HNO_3) to 10 mL of sample, and heating for 9 hours at 110°C, 5 hours at 115°C, and 2 hours at 130°C (Varcoe 1995).

Particulate elements were determined by difference (total-dissolved) as discussed in Stevens *et al.* (1999). Metal concentrations were determined as a product of the ICP analysis. Concentrations of total dissolved carbon (TDC) and dissolved organic carbon (DOC) were measured on filtered samples using a Dohrmann DC-180 carbon analyser

Total chemical loads were calculated by multiplying the chemical concentration by the instantaneous discharges between sampling periods then integration of the load versus time curve.

6.3 Results

Tables 6.1-6.4 show the average chemical analyses (taking flow volumes into account) of drain flows for the three years of the study.

Table 6.1 Average chemical analyses (pH, EC, DOC, NO₃, P) of reverse interceptor flows, 1995-97.

Analysis	Cocksfoot		Phalaris		Lucerne	
	Overland flow	Through-flow	Overland flow	Through-flow	Overland flow	Through-flow
1995						
pH	6.4	7.2	7.2	7.5	5.3	7.3
EC	0.02	0.1	0.08	0.2	0.04	0.12
DOC	4.4	12.3	5.5	11.0	6.6	12.7
Nitrate	0.1	0.8	1.1	5.3	0.3	2.1
P	0.8 (30)	0.9 (50)	1.3 (30)	0.4 (100)	1.4 (20)	0.5 (100)
1996						
pH	6.6	6.8	6.5	6.8	6.6	6.7
EC	0.07	0.10	0.09	0.15	0.09	0.13
DOC	4.8	10.9	14.1	15.9	10.0	13.5
Nitrate	0.3	0.9	0.2	1.5	0.2	0.8
P	0.5 (40)	0.5 (5)	0.3 (30)	0.3 (15)	0.3 (30)	0.4 (10)
1997						
pH	No flow	7.1	No flow	7.0	No flow	7.0
EC	No flow	0.09	No flow	0.13	No flow	0.09
DOC	No flow	12.8	No flow	15.3	No flow	14.1
Nitrate	No flow	0	No flow	0	No flow	0
P	No flow	0.2 (0)	No flow	0.4 (0)	No flow	0.6 (100)

pH in pH units, EC in dS/m, all other data in mg/L. Values in parentheses are the percentage dissolved (<0.45µm) fraction (to the nearest 5%).

Table 6.2 Average chemical analyses (pH, EC, DOC, NO₃, P) of contour bank/buried tube drain flows, 1995-97.

Analysis	Phalaris		Lucerne	
	Overland flow	Through- flow	Overland flow	Through- flow
1995				
pH	6.8	6.4	6.7	7.9
EC	0.1	0.3	0.1	0.2
DOC	7.9	13.1	6.4	9.6
Nitrate	0.8	18.6	0.7	11.1
P	1.0 (25)	0.1 (100)	0.7 (65)	0
1996				
pH	6.4	6.7	6.5	6.5
EC	0.10	0.17	0.08	0.43
DOC	12.6	21.6	11.4	14.1
Nitrate	0.1	2.5	0.2	23.1
P	0.3 (40)	0.3 (25)	0.4 (35)	0.1 (15)
1997				
pH	No flow	6.9	No flow	No flow
EC	No flow	0.13	No flow	No flow
DOC	No flow	21.5	No flow	No flow
Nitrate	No flow	0.4	No flow	No flow
P	No flow	1.0 (100)	No flow	No flow

pH in pH units, EC in dS/m, all other data in mg/L. Values in parentheses are the percentage dissolved (<0.45µm) fraction (to the nearest 5%)

Table 6.3 Average chemical analyses (Na, Cl and metals) of reverse interceptor flows, 1995-97.

Analysis	Cocksfoot		Phalaris		Lucerne	
	Overland flow	Through-flow	Overland flow	Through-flow	Overland flow	Through-flow
1995						
Na	5.1 (50)	17.2 (60)	15.4 (50)	27.3 (85)	7.5 (50)	18.1 (90)
Cl	3.4	11.7	8.0	24.7	5.2	17.8
Al	102.2 (0)	286.4 (0)	9.5 (0)	83.0 (0)	115.6 (0)	82.4 (0)
Fe	57.2 (0)	138.2 (0)	55.2 (0)	40.6 (0)	61.2 (0)	37.7 (0)
K	15.2 (5)	25.6 (10)	14.5 (20)	10.5 (15)	16.0 (10)	9.9 (15)
Mg	4.1 (10)	9.2 (10)	5.9 (20)	6.8 (45)	4.9 (10)	4.4 (40)
1996						
Na	6.4 (90)	14.5 (90)	9.1 (90)	23.2 (90)	6.6 (80)	18.6 (90)
Cl	7.3	13.5	7.8	22.6	6.4	19.9
Al	17.8 (0)	108.4 (0)	10.9 (0)	48.8 (0)	11.5 (0)	45.1 (0)
Fe	10.9 (0)	57.9 (0)	6.5 (0)	26.6 (0)	6.9 (0)	21.5 (0)
K	7.1 (55)	14.2 (15)	6.8 (75)	7.9 (25)	6.1 (70)	7.9 (30)
Mg	2.0 (65)	5.1 (30)	1.9 (75)	4.9 (50)	1.9 (75)	3.8 (55)
1997						
Na	No flow	7.1 (100)	No flow	8.5 (100)	No flow	5.0 (100)
Cl	No flow	8.04	No flow	12.6	No flow	6.8
Al	No flow	22.8 (0)	No flow	35.5 (0)	No flow	13.7 (0)
Fe	No flow	11.9 (0)	No flow	19.7 (0)	No flow	7.0 (0)
K	No flow	3.6 (80)	No flow	4.9 (60)	No flow	2.6 (80)
Mg	No flow	1.7 (70)	No flow	2.3 (70)	No flow	1.3 (100)

pH in pH units, EC in dS/m, all other data in mg/L. Values in parentheses are the percentage dissolved (<0.45µm) fraction (to the nearest 5%).

Table 6.4 Average chemical analyses (Na, Cl and metals) of contour bank/buried tube drain flows, 1995-97.

Analysis	Phalaris		Lucerne	
	Overland flow	Through-flow	Overland flow	Through-flow
1995				
Na	11.5 (40)	40.3 (90)	8.8 (60)	27.6 (90)
Cl	5.8	31.4	5.5	12.8
Al	111.0 (0)	11.1 (0)	76.2 (0)	6.4 (0)
Fe	48.3 (0)	5.5 (0)	30.4 (0)	3.8 (0)
K	17.5 (10)	3.7 (50)	12.2 (20)	3.2 (40)
Mg	5.3 (10)	5.1 (85)	3.9 (30)	4.6 (80)
1996				
Na	12.6 (90)	26.2 (95)	9.0 (85)	50.4 (100)
Cl	9.3	19.8	7.4	43.8
Al	10.9 (0)	39.0 (0)	14.1 (0)	22.0 (0)
Fe	5.9 (0)	20.8 (0)	7.3 (0)	12.0 (0)
K	7.4 (75)	6.6 (30)	6.8 (65)	4.2 (40)
Mg	2.2 (60)	4.6 (55)	2.5 (55)	9.0 (95)
1997				
Na	No flow	8.4 (100)	No flow	No flow
Cl	No flow	15.5	No flow	No flow
Al	No flow	8.4 (0)	No flow	No flow
Fe	No flow	6.1 (0)	No flow	No flow
K	No flow	1.8 (75)	No flow	No flow
Mg	No flow	1.4 (100)	No flow	No flow

pH in pH units, EC in dS/m, all other data in mg/L. Values in parentheses are the percentage dissolved (<0.45µm) fraction (to the nearest 5%)

6.3.1 1995

The annual rainfall in 1995 of 467 mm, was 15% below average. Throughflow was generally alkaline and overland flow acidic which reflects the pH of the soil. EC of the throughflow (0.1 to 1.0 dS/m) was consistently, significantly ($P < 0.05$) higher than overland flow (0.02 to 0.1 dS/m) and salt concentrations (Na and Cl) were higher (by up to three times) in throughflow than in overland flow. DOC concentrations were generally around twice as high in throughflow (9.6 to 13.1 mg/L) as in overland flow (4.4 to 7.9 mg/L), and were similar for all pasture types.

Nitrate concentrations were up to 20 times higher in throughflow (0.8-18.6 mg/L) than in overland flow (0.1 to 0.8 mg/L), and concentrations in the tube drains were highest (up to 0.8 and 18.6 mg/L in overland flow and throughflow respectively). Nitrate concentrations were significantly ($P < 0.05$) higher in the improved pastures than in the standard cocksfoot pasture. Al (clay) concentrations were highest from the cocksfoot pasture, particularly in throughflow (286.4 mg/L). This reflects the higher loss of soil in runoff from unimproved pastures with low leaf area particularly at the start of the winter season. The concentrations in the buried drains were significantly lower. Phosphorus concentrations were generally lowest in throughflow (< 0.9 mg/L). However up to 100% of throughflow P (mean 70%) was moving in the dissolved state compared to a mean of 35% in overland flow. There was no P lost in throughflow from the tube drain under lucerne (due to lack of flow).

Fe concentrations were low in the tube drains and generally elevated elsewhere, with the highest concentration also recorded in throughflow from the cocksfoot pasture. As for high amounts of Al, this also reflects greater soil loss. K was low in the tube drains, however there was a high percentage moving in the dissolved state.

6.3.2 1996

The rainfall in 1996 was 548 mm, < 1 mm off the long term annual average. Throughflow was neutral to acid in 1996, whereas it was alkaline in 1995. EC concentrations increased slightly (but not significantly) in overland flow probably due to the deeper leaching of salts in wetter years. Nitrate concentrations under phalaris were significantly ($P < 0.05$) reduced compared with in 1995, however there was a two-fold increase in concentrations in the lucerne tube drain throughflow. Al concentrations increased in the tube drains but were reduced elsewhere, particularly in overland flows. Fe concentrations were lower than 1995, particularly in overland flow.

Whilst P concentrations were generally lower than the previous year there was proportionately more moving in the particulate form as the increased drainage of a wetter year resulted in greater soil movement. K concentrations were also lower, but there was generally a higher percentage in the dissolved state.

6.3.3 1997

The rainfall of 402 mm was 27% below average and was not sufficient to cause overland flow. Overland flow in these soils only occurred when the soil profile above the clay horizon was saturated (however it could occur under dry conditions if the rainfall was intense).

The pH concentrations of the throughflow were close to neutral in all treatments (6.9-7.1). EC levels were low and Na and Cl levels were considerably reduced from the previous year. DOC concentrations were similar to 1996, but very high in the phalaris tube drain (21.5 mg/L). Nitrate concentrations were low as were Al, Fe and K. Whilst P concentrations were similar to those of 1996, the forms were variable, with the total flow from a particular drain type being either all dissolved, or all particulate.

Particulate P flow increased during those periods when rainfall was most erosive. Whether this was because the drains had not grassed up or the slope in that particular area was slightly steeper is uncertain. One explanation could be that dissolved P had the opportunity to become fixed as particulate P when drain flows were turbid.

6.3.4 Water movement and chemical concentrations at individual rainfall events

Note: In this chapter, any reference to toe-slope should be regarded as synonymous with 'lower slope'.

In 1996, the wettest year, there was a late break to the season, with the first rainfall of any note (18 mm) falling on 4th June. Previous to this, total rainfall for the year was only 60 mm, with 28 mm of this falling in a false break in mid-March. The rainfall events (>1 mm) and nutrient concentrations, from early June to early October are shown in Table 6.5.

Table 6.5 Rainfall during period June to early October, 1996, and indicative concentrations of $\text{NO}_3\text{-N}$, DOC and P, in throughflow.

Date (1996)	Rainfall (mm)	$\text{NO}_3\text{-N}$ (mg/L)	DOC (mg/L)	P (mg/L)
4 June	18	0.6	28.0	0.4
18 June	23	0.8	63.0	0.4
27 June	33	1.5	19.7	0.3
5 July	30	11.8	19.6	0.3
19 July	8	1.9	9.8	0.7
2 August	3	0.9	13.1	0.2
23 August	11	0.1	14.0	0.2
13 September	18	0.3	17.6	0.4
2 October	2	0.2	16.2	0.3

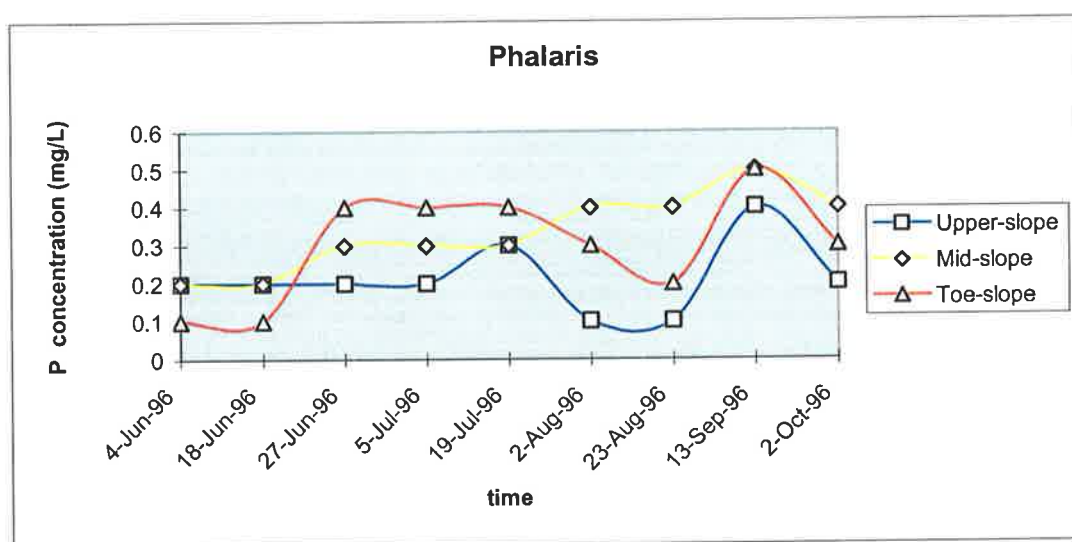


Fig. 6.2 P concentrations in throughflow from phalaris pastures, 4 June-2 Oct., 1996.

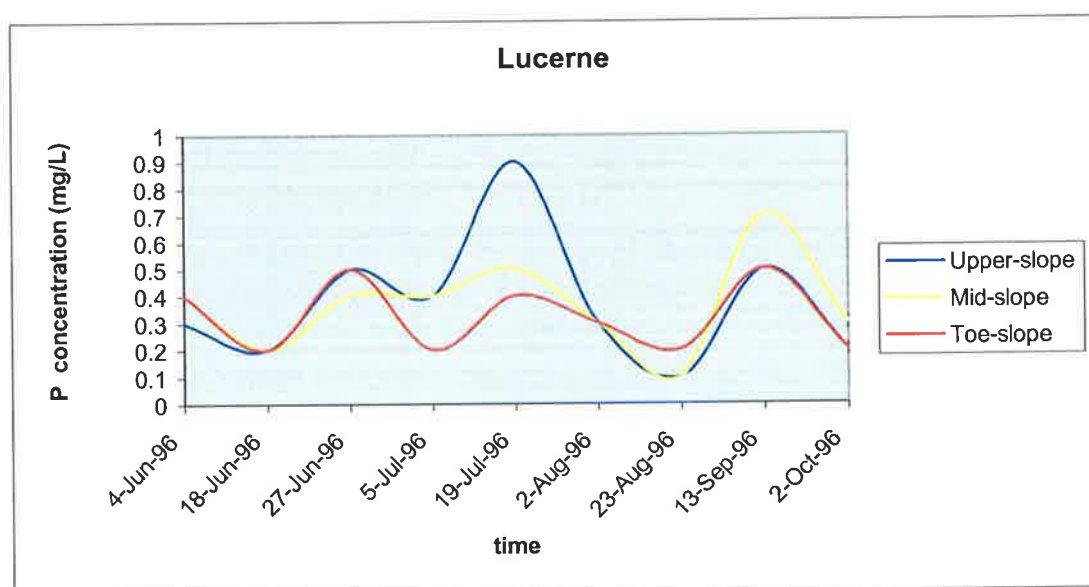


Fig. 6.3 P concentrations in throughflow from lucerne pastures, 4 June-2 Oct., 1996.

Minimal throughflow occurred on 4th and 18th June, and whilst NO₃-N and P concentrations were low in water samples (generally <1.0 mg/L), DOC levels were high (15-60 mg/L). Following a further 33 mm of rainfall on 27th June, increased amounts of throughflow were recorded, and by early July concentrations of NO₃-N were at or near their maxima (>10 mg/L), P levels were also peaking (0.7 mg/L) and DOC concentrations were decreasing (see Figs 6.2 and 6.3 indicating the variability of concentrations between perennials at different slope positions).

Overland flow did not occur until mid-July in most instances, when the A horizon was approaching saturation and throughflow output was at its peak. Chemical concentrations in early overland flow were low (NO₃-N <0.5 mg/L, P <1 mg/L) and remained relatively constant, whilst DOC levels were initially low (<10 mg/L) and increased over the next couple of rainfall events (>20 mg/L).

6.3.5 Chemical and nutrient loads

The average loads (averaged over plots) of chemicals and nutrients are presented in Tables 6.6 to 6.9.

Table 6.6. Average chemical loads (DOC, NO₃-N, P) of reverse interceptor flows, 1995-1997 (data in kg/ha).

Analysis	Cocksfoot		Phalaris		Lucerne	
	Overland Flow	Through-flow	Overland flow	Through-flow	Overland flow	Through-flow
1995						
DOC	0.073	1.082	Minimal	0.484	0.031	0.266
NO ₃ -N	0.002	0.038	Minimal	0.205	0.001	0.138
P	0	0.089	Minimal	0.056	0.004	0.019
1996						
DOC	0.303	3.449	1.897	2.173	1.559	4.498
NO ₃ -N	0.012	0.252	0.040	0.234	0.043	0.433
P	0.031	0.148	0.071	0.045	0.073	0.059
1997						
DOC	No flow	0.785	No flow	1.360	No flow	8.512
NO ₃ -N	No flow	0.012	No flow	0.001	No flow	0.782
P	No flow	0.010	No flow	0.046	No flow	0.103

'Minimal flow' = sample taken and analysed, but flow too low to attempt load determination.

Table 6.7. Average chemical loads (DOC, NO₃-N, P) of contour bank/buried tube drain flows, 1995-1997 (data in kg/ha).

Analysis	Phalaris		Lucerne	
	Overland Flow	Through-flow	Overland flow	Through-flow
1995				
DOC	Minimal flow	Minimal flow	Minimal flow	Minimal flow
NO ₃ -N	Minimal flow	Minimal flow	Minimal flow	Minimal flow
P	Minimal flow	Minimal flow	Minimal flow	Minimal flow
1996				
DOC	0.350	4.347	0.670	0.857
NO ₃ -N	0.042	0.379	0.544	0.020
P	0.011	0.069	0.013	0.031
1997				
DOC	No flow	0.047	No flow	No flow
NO ₃ -N	No flow	0.001	No flow	No flow
P	No flow	0.001	No flow	No flow

'Minimal flow' = sample taken and analysed, but flow too low to attempt load determination.

Table 6.8. Average chemical loads (Na, Cl and metals) of reverse interceptor flows, 1995-1997 (data in kg/ha).

Analysis	Cocksfoot		Phalaris		Lucerne	
	Overland flow	Through-flow	Overland flow	Through-flow	Overland flow	Through-flow
1995						
Na	0.083	1.150	Minimal flow	1.175	0.064	0.531
Cl	0.070	0.626	Minimal flow	0.947	0.046	0.611
Al	0.518	28.845	Minimal flow	12.939	0.158	3.154
Fe	0.337	13.905	Minimal flow	6.381	0.076	1.330
K	0.095	2.714	Minimal flow	1.193	0	0.417
Mg	0.025	0.885	Minimal flow	0.548	0.015	0.175
1996						
Na	0.239	4.726	1.626	3.405	2.396	4.954
Cl	0.188	5.649	1.392	4.224	3.562	6.148
Al	0.521	28.130	4.089	7.345	3.231	8.374
Fe	0.279	14.430	2.416	4.100	1.992	4.073
K	0.386	3.489	2.318	1.217	2.132	1.583
Mg	0.096	1.474	0.417	0.773	0.434	0.877
1997						
Na	No flow	0.419	No flow	0.839	No flow	8.279
Cl	No flow	0.535	No flow	1.309	No flow	10.132
Al	No flow	1.649	No flow	2.582	No flow	15.342
Fe	No flow	0.849	No flow	1.420	No flow	7.314
K	No flow	0.217	No flow	0.413	No flow	2.780
Mg	No flow	0.101	No flow	0.252	No flow	1.437

'Minimal flow' = sample taken and analysed, but flow too low to attempt load determination.

Table 6.9. Average chemical loads (Na, Cl and metals) of contour bank/buried tube drain flows, 1995-1997 (data in kg/ha).

Analysis	Phalaris		Lucerne	
	Overland flow	Through-flow	Overland flow	Through-flow
1995				
Na	Minimal flow	Minimal flow	Minimal flow	Minimal flow
Cl	Minimal flow	Minimal flow	Minimal flow	Minimal flow
Al	Minimal flow	Minimal flow	Minimal flow	Minimal flow
Fe	Minimal flow	Minimal flow	Minimal flow	Minimal flow
K	Minimal flow	Minimal flow	Minimal flow	Minimal flow
Mg	Minimal flow	Minimal flow	Minimal flow	Minimal flow
1996				
Na	0.792	5.132	3.374	0.990
Cl	0.212	3.913	3.516	1.046
Al	0.157	5.372	2.658	0.819
Fe	0.180	4.923	1.421	0.430
K	0.177	1.444	0.402	0.558
Mg	0.065	0.996	0.554	0.212
1997				
Na	No flow	0.020	No flow	No flow
Cl	No flow	0.040	No flow	No flow
Al	No flow	0.011	No flow	No flow
Fe	No flow	0.180	No flow	No flow
K	No flow	0.004	No flow	No flow
Mg	No flow	0.003	No flow	No flow

'Minimal flow' = sample taken and analysed, but flow too low to attempt load determination.

DOC loads were consistently higher in throughflow than in overland flow, in both the reverse interceptor and buried tube drains. The highest loads occurred in a year of average rainfall (1996), with the exception of that recorded in reverse interceptor throughflow under lucerne in 1997 (8.51 kg ha⁻¹). Annual loads of NO₃-N ranged from <0.001 to 0.782 kg ha⁻¹, and again the higher loadings were in throughflow. In the year of average rainfall NO₃-N loads were generally higher than those in the below average rainfall years, again with the exception of the maximum recorded load (0.78 kg ha⁻¹) in reverse interceptor throughflow under lucerne. The highest NO₃-N loads were consistently in drainage from lucerne treatments.

The loads of P were less variable over the study period (<0.001-0.148 kg ha⁻¹) with the higher loadings again in throughflow.

From Tables 6.8 and 6.9, loads of all chemicals are almost universally higher in throughflow than in overland flow with the notable exception of those from the lucerne treatment, contour bank/buried tube drain, in 1996 (Table 6.9). Al and Fe loads were the highest of all the chemicals and nutrients, indicative of their relative abundance in the soil. Their loads in throughflow below cocksfoot pasture were generally two or more times greater than those under the phalaris and lucerne treatments in 1995 and 1996. In 1997 however, all chemical loads under lucerne were significantly higher than those from the other two pasture types, probably due to higher production and higher N-fixation.

Cox and Ashley (2000) reported on a study of water quality in the gully drainage of Keynes catchment, which gives an indication of chemical concentrations and loads without engineered drains in place. The loads of P (0.005 kg/ha/yr) and NO₃ (0.01 kg/ha/yr) are in close agreement with the overland flow from the cocksfoot sites, and this is to be expected given that this is the established pasture in the catchment. The loads of these nutrients in overland flow from phalaris and lucerne however are higher than the cocksfoot drained sites (and the overall catchment) suggesting that this is a treatment difference. However, it is a limitation of this study that there are no undrained agronomic treatments. In any case, this would not allow a comparison of throughflow water quality unless a discharge site for lateral flows could be positively identified in the lower slopes or drainage lines.

6.4 Discussion

6.4.1 pH

Generally, the pH of throughflow was higher than that of overland flows and became alkaline in dry years. Thus the pH of the overland flow reflects the pH of the acidic topsoil. The pH of the subsoil reflects the alkalinity of the subsoil in dry years and the acidity of the humic substances (see DOC concentrations) which are leached into the subsoil in years of average (and above average) rainfall. In studies in high rainfall catchments on texture contrast soils throughflow has been shown to have the same or lower pH than overland flow (eg. Cox *et al.* 1998; Stevens *et al.* 1999). This suggests deeper leaching of humic and acidic material occurs under higher rainfall conditions on texture contrast soils and is consistent with these findings.

6.4.2 *EC_{1:5}*

Overland flow was generally non-saline (EC 0.02-0.1 dS/m) although there is a slight increase in EC in a wet year following dry years when surface soil evaporation would be higher (eg. 1996). The EC of throughflow was always higher than overland flow (0.09-0.43 dS/m), generally non-saline but occasionally slightly saline. Na and Cl concentrations were all well below the guidelines for drinking water (300 mg/L Na, 400 mg/L Cl; NHMRC and AWRC 1987).

6.4.3 *DOC*

DOC concentrations in overland flow were lower than those in throughflow, particularly in 1995, a below average rainfall year. In 1996, with an average rainfall, the concentrations were generally higher from phalaris and lucerne plots than from cocksfoot, and differences between flow types were less marked. Concentrations in 1996 were similar across a range of catchments in the Mt Lofty Ranges, with varying rainfalls and land uses (Cox *et al.* 1998; Fleming and Cox 1998; Stevens *et al.* 1999).

DOC concentrations were highest after long dry spells. Chittleborough *et al.* (1992) also reported high DOC concentrations at the start of the season, although not quite to the levels recorded at Keyneton. Over the 3 years of the study, DOC loads were consistently higher in throughflow, in contrast to other similar studies in the Adelaide Hills (Chittleborough *et al.* 1992; Fleming and Cox 1998), where overland flow was the dominant form of drainage.

6.4.4 *Nitrate*

Losses of nitrate were high in comparison to those reported at other catchments with duplex soils in South Australia (eg. subclover and ryegrass dairy pastures; Fleming and Cox 1998). In dry years, nitrate concentrations in tube drains below lucerne and phalaris/subclover pastures were 11 mg/L and 19 mg/L respectively. These concentrations are two to three times higher than those reported from dairy pastures (Cox *et al.* 1998; Fleming and Cox 1998) although it is to be expected that pastures with a high grass content should have a high N uptake and so less capacity for leaching (Ridley *et al.* 1999).

The following year (average rainfall), the mean concentration in the tube drains below phalaris reduced to 2.8 mg/L whereas under lucerne it increased to 23 mg/L. These levels are probably linked to increased dry matter production, with the greater amount of phalaris leading to a higher N uptake, and the greater lucerne production to increased summer mineralisation and subsequent autumn/winter leaching (Ridley *et al.* 1999). In 1997, drain flows were minimal and the concentrations were negligible.

The World Health Authority limit for nitrate levels set for human drinking water is 10 mg/L (OECD 1986) and some of these findings were well in excess of that value. However, from an agricultural context these losses are at the low end of the range for low input pastures. Rainfall inputs of N of 5 kg/ha/yr are reported by Ridley *et al.* (2000) suggesting that the reported losses in this study indicate a system conservative of N. White *et al.* (2000) commenting on the reduction in drainage under perennials felt that the reduced loss of solutional N (up to 12 kg/ha/yr) “suggested soil acidification under perennials was reduced by about 1 kmol H⁺/ha/yr”. Ridley *et al.* (2001) estimated N losses under perennial pasture relative to an annual pasture of 8 kg N/ha/yr. None of the loads at Keyneton reached this level.

6.4.5 Phosphorus

In an average rainfall year, P concentrations were similar (0.1 - 0.5 mg/L) in overland flow and throughflow, in all drain types and under all pastures. There were however differences in the forms of P moving, with approximately twice as much in the dissolved state in overland flow compared to throughflow (Cox and Pitman 2001). This is in contrast to other catchments in the Ranges where about half of the P moved in each form (Fleming and Cox 1998). The amount of P moving in the dissolved state poses potential management difficulties in reducing loads before the runoff enters a watercourse.

In drier years, concentrations were significantly higher overall, although more variable. Concentrations in throughflow were generally lower than in overland flow reflecting P reactivity that restricts movement through the soil (Fleming and Cox 1998). There was a much higher percentage moving in the dissolved state (up to 100%) in the throughflow samples, again posing potential management difficulties in its reduction.

6.4.6 Metals

Losses of Al, Fe, K and Mg were generally much higher than those from higher rainfall agricultural catchments in South Australia (eg. Fleming and Cox 1998; Stevens *et al.* 1999) in average rainfall years. Throughflow losses (Al 108 mg/L, Fe 58 mg/L, K 14 mg/L, Mg 9 mg/L) exceeded overland flow losses considerably (Al 17 mg/L, Fe 11 mg/L, K 7 mg/L, Mg 2 mg/L), and in dry years, concentrations increased in both flow pathways (eg throughflow Al 286 mg/L and overland flow Al 102 mg/L). Concentrations of metals from unimproved pastures were only slightly higher than from improved and losses of all metals exceeded guidelines for drinking water.

Concentrations of most metals in throughflow were significantly greater in open drains than in tube drains (eg. Al 83 to 6 mg/L, Fe 39 to 3 mg/L, K 10 to 3 mg/L), and one (untested) explanation for this could be ochre formation, a common occurrence with buried pipe drains. This involves the precipitation of sesquioxides within the drain, in which case the same relative amounts of Al and Fe could be moving in throughflow in all of the equivalent agronomic treatments.

6.4.7 Water movement and chemical concentrations at individual rainfall events

Given the relative infrequency of overland flow throughout this study, and the low concentrations of the above nutrients in this type of flow, there was little likelihood of overland flow contributing to high nutrient loadings in local waterways. On the other hand, throughflow volumes were much higher, and in 1996, the highest concentrations of NO₃-N corresponded to the period of greatest flow, when the A horizon approached saturation.

However, the work of McFarlane and Cox (1992) and Stevens *et al.* (1999) and Figure 4.7 suggest that in above-average rainfall years, overland flow will be the dominant flow pathway in most of the drains. Due consideration should be paid to this likelihood in any management recommendations.

Overall, P varied little between pasture types and drain arrangement, whilst DOC was similar under all pastures, but higher in throughflow than overland flow. NO₃-N

concentrations were more variable between pasture types and at different rainfall events. Maximum $\text{NO}_3\text{-N}$ concentration from throughflow under lucerne was 67.4 mg/L, under phalaris 9.4 mg/L and under cocksfoot only 1.4 mg/L.

It is the co-occurrence of a high concentration of a chemical, and a large volume of water flow, that may lead to significant losses off-site. From this study it is apparent that substantial losses of $\text{NO}_3\text{-N}$ are likely early in the growing season, and in this instance, following rainfall of c. 100 mm. This was sufficient to wet up the A horizon to a degree that promoted variable, but generally substantial, throughflow.

Under these conditions, there were losses (in a single rainfall event) of $\text{NO}_3\text{-N}$ from under lucerne-based pasture of from 0.15 to 1.12 kg/ha, with the higher losses from the mid- and toe-slopes. Under phalaris, losses were in the range of 0.15 to 0.23 kg/ha, and again the lowest losses were from the upper-slope position. Losses under cocksfoot were in the range of 0.019 to 0.11 kg/ha, and the lower slopes had the highest loads.

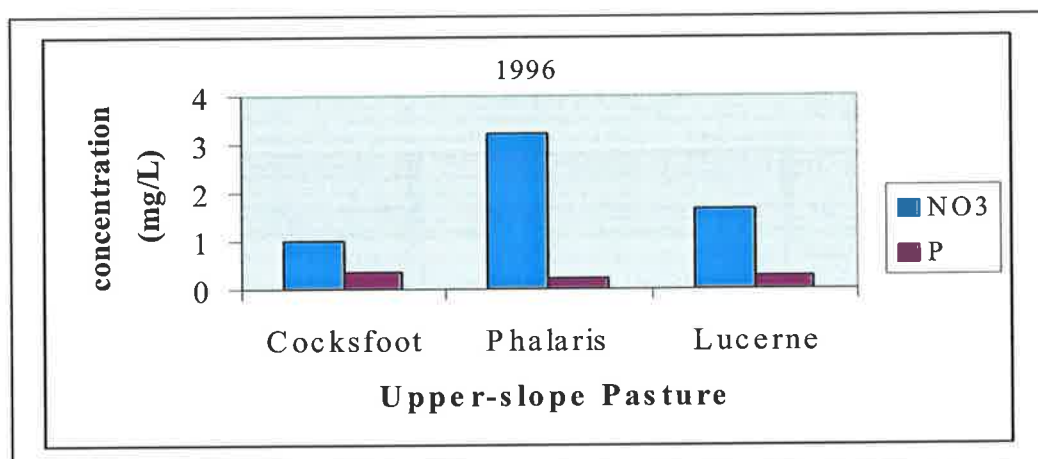


Fig. 6.4 Mean concentrations of N and P from upper-slope pastures in 1996.

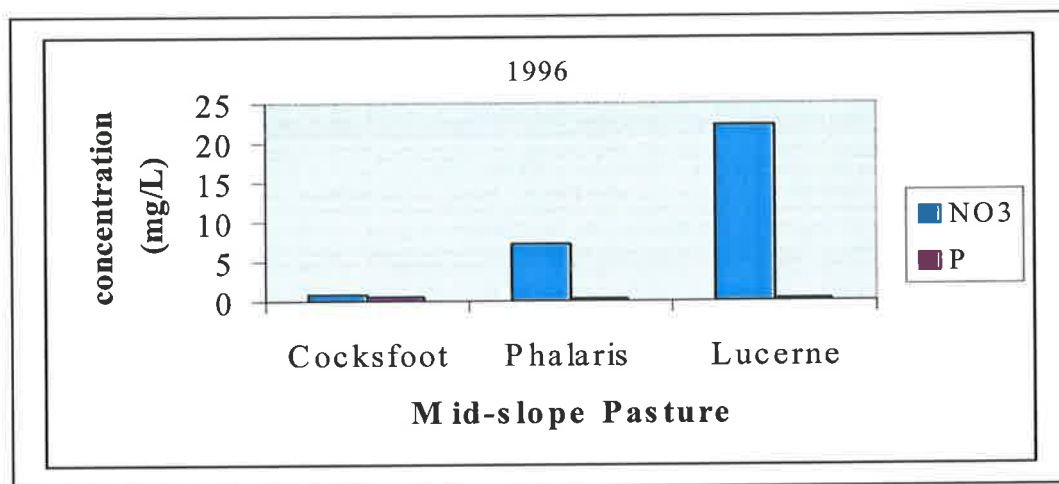


Fig. 6.5 Mean concentrations of N and P from mid-slope pastures in 1996.

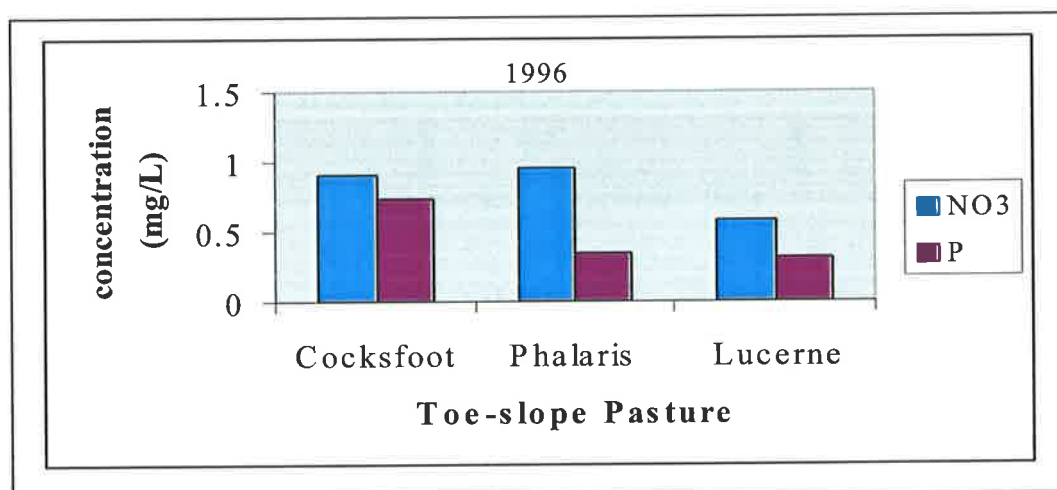


Fig. 6.6 Mean concentrations of N and P from toe-slope pastures in 1996.

Of the three years of the study, 1996 was an average rainfall year, and the above Figures 6.4 to 6.6 indicate the concentrations of NO₃-N and P over this period. The P concentrations from all pastures and slope positions were similar, and within the range expected (Greenhill *et al.* 1983^{ab}; Fleming and Cox 1998). Concentrations of NO₃-N from the upper-slope were only slightly below the expected range reported by Chichester *et al.* (1979) for pasture throughflow (4.5-7.1 mg/L). Toe-slope

concentrations were very low and probably a result of dilution, given that throughflow was greatest from this slope position.

The $\text{NO}_3\text{-N}$ concentration from mid-slope lucerne is of concern, and an indication of potential acidification under legumes. This mean value is heavily influenced by six of the seven analyses carried out throughout the year from a buried tube drain, where concentration ranged from 27.1-67.4 mg/L. $\text{NO}_3\text{-N}$ export from this particular site for the year was >1.2 kg/ha, which is well in excess of the values (0.088-0.091 kg/ha/yr) reported by Fleming and Cox (1998) for the Adelaide Hills. This supports concerns expressed by Dowdell *et al.* (1987) and Skaggs and Chescheir (1999) on the potential environmental problems from $\text{NO}_3\text{-N}$ in subsurface drains.

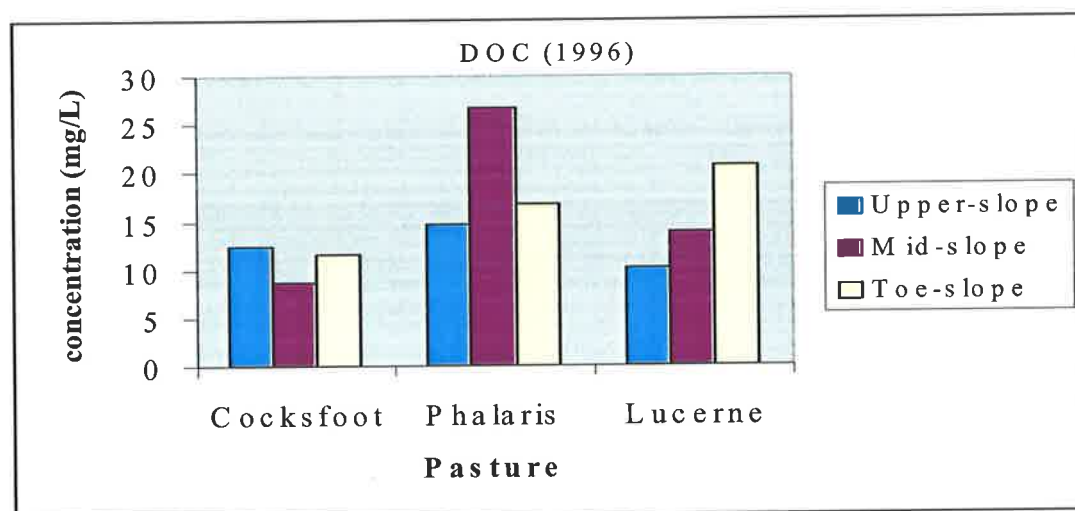


Fig. 6.7 Mean concentrations of DOC from Keyneton pastures in 1996.

DOC concentrations were within the range reported by Chittleborough *et al.* (1992), Nelson *et al.* (1996) and Fleming and Cox (1998) for catchments in the Mt Lofty Ranges. Drainage from duplex soils where a coarse textured A horizon (sand or loamy sand) overlies a clay B horizon, was found to be a significant DOC source (Nelson *et al.* 1996). However, generally overland flow concentrations would be expected to be higher than those in throughflow, with DOC decreasing as water moves through the soil.

Nelson *et al.* (1990) suggested an inverse relationship between the adsorptive capacity of the soil horizons and stream DOC concentrations.

In this catchment, most water movement was via throughflow, and Smythe (1995) showed that the prevalence of macropore flow tended to override adsorption characteristics in importance in this regard. The high DOC concentration shown for mid-slope phalaris (Fig. 6.7) may be linked to waterlogging, which was apparent on this treatment. Nelson *et al.* (1996) report that the anoxic conditions produced by waterlogging have been linked to high DOC concentrations. This may be correlated with the low $\text{NO}_3\text{-N}$ concentration of mid-slope phalaris (Fig. 6.5) as anoxic soil conditions with high levels of available C, lead to low soil nitrate concentrations due to denitrification (Gilliam 1994).

6.5 Conclusions

There were seasonal variations in flows from pastures, from slightly acidic to slightly alkaline. Although flows were generally non-saline, there were instances of slightly saline throughflow where shallow groundwaters were present or from the more saline soils of the footslopes. Drainage could be made shallower in the lower slopes to prevent interception of saline waters, but when these are the areas most likely to have high throughflow there is likely to be a high dilution factor in any case. Shallow drains are also likely to imply that the nutrients entering them are from a diminished thickness of the anoxic or reduced zone, a region where denitrification occurs, lowering N concentration in the drainage water (Skaggs and Chescheir 1999).

DOC concentrations were higher in flow from improved pastures compared with unimproved (14 mg/L and 5 mg/L respectively), and higher in throughflow than in overland flow. High DOC in throughflow probably reflects the preferential flow through these soils (Kirkby *et al.* 1996), and supports the findings of Chittleborough *et al.* (1992). An alternative viewpoint, and one untested in this study, suggests that high DOC reflects the movement of water as throughflow in intimate contact with organic carbon stores in the soil, whereas overland flow is not. If this were the case, whether flow is via preferential pathways or as massflow through the matrix is then irrelevant.

Nitrate concentrations were higher than those recorded from more intensively grazed pastures in higher rainfall zones of the Adelaide Hills and were above drinking water standards. The highest nitrate concentrations were found in throughflow via the tube drains beneath lucerne and phalaris pastures (67 mg/L and 19 mg/L), and concentrations were highest in dry years, probably reflecting the increased time period available for organic matter mineralisation before leaching occurred, and the reduced period of anoxic conditions during which denitrification could occur.

N losses are however low from an agricultural perspective and any potential acidification could be mitigated by liming if it proved economically feasible. The potential for off-site effects (eg. NO_3 concentration in ground and surface waters) still demands caution (Ridley *et al.* 2001).

The often higher loads in tube drain throughflow compared to interceptor throughflow is probably a result of the higher concentrations of chemical and often higher flow volumes they carry. In the open drains there is considerable potential for dilution by direct rainfall and also from batter flows. Additionally, it is likely that the tube drains were better located within the B horizon to intercept lateral flows, where some of the throughflow is likely to pass under the open drains.

Losses of metals in overland flow from these dryland pastures were as high as those from higher rainfall dairy pastures, and throughflow losses were even greater than those from overland flow. In dry years, the concentrations of all metals increased markedly in both flow pathways (eg. from 108 to 286 mg/L Al). Concentrations were only slightly higher in flows from the cocksfoot-based pastures than from those based on phalaris or lucerne. There were significantly lower concentrations in the tube drains than there were in the open drains (eg. Al 6 to 83 mg/L, Fe 3 to 39 mg/L, K 3 to 10 mg/L), and one suggestion for this is ochre formation in the tube drains.

Phosphorus concentrations were similar or only slightly higher in overland flow than in throughflow (<0.5 mg/L) and there were only minor differences in concentrations between pastures. There were however sizeable differences in the relative proportions of dissolved and particulate phosphorus recorded in the flows, with up to 100% movement in the dissolved state (and therefore likely to enter local stream flows).

Overland flow was negligible in this study under most conditions. Water moving as throughflow was the principal transport mechanism for chemical export off-site, and there is the potential for substantial amounts of $\text{NO}_3\text{-N}$ to be moved via this pathway, particularly in drainage from lower slopes. Any management regime introduced to try to reduce waterlogging in these environments must consider the potential consequences of transporting nutrients off-site, if sub-surface drainage (throughflow) is redirected by engineering solutions.

However as stated previously, overland flow is likely to be the dominant form of runoff in average and high rainfall years and management regimes should consider this. In this study the drains provide zones of lower water potential as exit points for throughflow. In an undrained slope throughflow will be constrained by saturation lower in the slope and will be regulated by the rate of discharge from the system either as base flow to the creeks, as surface seepage or potentially as deep drainage and subsequent recharge. Throughflow interception is an important consideration in that it may reduce both runoff and recharge with consequent beneficial impacts on potential erosion and salinity.

In a grazed catchment, tools to limit diffuse pollution involve pasture management options. Erosion can be minimised by ensuring maximum ground cover of pasture, and increasing the roughness factor by maintaining pasture height will slow particulate movement. Minimising grazing pressure particularly along drainage lines (Nelson *et al.* 1996) and maintaining good cover, will provide riparian buffer strips. These are often advised for nutrient movement control (Fleming and Cox 1998) although there is little information on their management in grazing country (Nelson *et al.* 1996).

Fleming and Cox (1998) suggested that “the effectiveness of buffer strips and riparian zones would largely depend on the settling velocities of the nutrient-containing materials” and that these would be higher for those in the particulate form than the dissolved. As there were worrying levels of nutrient movement (particularly P and N) from these pastures, a range of options including buffer strips and possibly wetland establishment should be considered to minimise the risk to local waterways.

Chapter 7

Simulation Modelling

7.1 Introduction

Over the last two decades or so there has been an increased interest in hillslope hydrology (Kirkby 1978) and many of the associated studies have incorporated the testing, application and validation of a range of ecohydrological models (Cox *et al.* 1994; Richardson and Narayan 1995; Pavelic *et al.* 1997; Clarke *et al.* 1998^a, 1998^b; Bronstert 1999; Zhang *et al.* 1999^a, 1999^b; Zhu *et al.* 1999).

With the increase in dryland salinity, soil acidification and other forms of land degradation related to soil hydrology, it has become increasingly obvious that land management practices have to be developed and implemented to contain, if not reverse, these problems. Recharge reduction utilising a range of options from trees to perennial pastures, is one remedial measure that could be investigated, but to do so at a catchment scale and over a time period that could produce meaningful outcomes, is problematic. For land management changes to be implemented, managers have to be shown that they are workable and feasible. Modelling provides the opportunity to test a number of options (eg. types of perennial) under a range of conditions that would probably be impossible in a field situation (Pavelic *et al.* 1997). Despite the disadvantages in modelling, Dawes and Hatton (1991) concluded that there was no other way “to describe and predict the complex interactions between land, vegetation and climate” (see also Clarke *et al.* 1998^{ab}).

Early studies often involved relatively simple models in terms of computational requirements (eg. simple digital elevation models), however this restriction is now less apparent and quite sophisticated modelling packages are in routine use. TOPOG, the model reviewed in this study, is a complex deterministic, terrain analysis based ecohydrological model, which uses distributed parameters to characterise landscapes and simulate “spatial and temporal water balances and plant growth across three-dimensional catchments” (Dawes and Hatton 1993).

7.2 Literature Review

Simulation models able to deal with the complex processes operating within the soil-water-landscape continuum of an ecosystem may be powerful tools in the evaluation of management options for field hydrology, plant water use, recharge investigations and water quality impacts from nutrient transport (Tsiros *et al.* 1998). Some models are relatively simple, whilst others are complex research models.

Models are classified as either deterministic or stochastic. In a deterministic model a certain set of events will produce a unique outcome, but in a stochastic model the outcome is accepted as uncertain and the model is structured accordingly (Addiscott 1995). Deterministic models can be further classified as mechanistic or functional. A mechanistic model tries to describe all of the processes involved, whilst the functional model focuses on describing key features of the system, not necessarily in full detail.

Maraux *et al.* (1998) compared mechanistic and functional models for estimating the soil water balance of successive crops, and although the results for the cumulative water balance components were in the same order of magnitude, the functional model did not accurately describe the time dependent water fluxes at the soil-atmosphere interface. They suspected that the differences between models were in the detail of the apportioning of the evaporation component into evapotranspiration, drainage and root uptake (Maurax *et al.* 1998).

Tsiros *et al.* (1998), also compared mechanistic and functional models in a study of the soil water regime in agricultural fields. They found that the mechanistic model based on Richard's equation for describing water distribution in the soil profile performed better overall than the functional model based more on water capacity terms. The drawback for the mechanistic model was the requirement for detailed, site specific soil characteristics. They felt the functional model was an acceptable alternative if resource and time restrictions applied.

Richard's water flow equation is the basis of several complex physically based models that describe catchment-scale flows of water (eg. WAVES, Hatton *et al.* 1995; HILLFLOW, Bronstert 1999; TOPOG, Dawes and Hatton 1993). Richard's equation

combines the water conservation equation (relating water fluxes, changes in storage, inputs/losses), with the Buckingham–Darcy flux equation (Barkle *et al.* 1998). The resulting equation has unknown matric potential (ψ) and water content (θ), one of which can be eliminated by using a measured relationship between the two (Barkle *et al.* 1998).

Beven (1989) questioned whether complex physically based models could be as mechanistic as the Richard's equation. His premise was that, in applying the model to a catchment represented as a large number of elements/grid squares, 'effective' values for parameters were chosen by spatially averaging highly variable soil properties such as hydraulic conductivity (Addiscott 1995). Beven's (1989) assertion was that these 'effective' parameters were no better than those used in a functional model. Addiscott (1995) however, suggests that it is not unlikely that mechanistic models may well fall into the functional category at large scales, and that what we should be seeking "is the most mechanistic model that is appropriate to, and has been validated for, the scale and purpose of the project at hand".

7.2.1 TOPOG

Early in the model's development, O'Loughlin (1992) discussed the role of TOPOG in demonstrating where in the landscape waterlogging was likely to occur. By combining a number of subsurface flow parameters that are relevant to waterlogging into a single number indicative of waterlogging potential, a "Wetness Index" is derived, and this is calculated in TOPOG. Across a catchment, wherever the Wetness Index reaches unity or more, waterlogging will occur. Other components of the model would allow prediction of the effects of vegetative change on the dynamics of waterlogging in the catchment. O'Loughlin (1992) predicted the commercial release of TOPOG later in 1992 (development commenced in 1987).

In 1998, active development of the model was still taking place (CSIRO 1998), and TOPOG consisted of more than 30 FORTRAN and C programs, some interactive, with mouse-driven menus and high-level graphics. Further debugging and updating has since occurred.

In Germany, Bronstert (1999) reviewed different objectives and concepts for hillslope hydrological modelling and tried to determine which hydrological problems had been approached successfully and what were the inherent limitations of hillslope models. In his paper, he summarises his hillslope model HILLFLOW, developed as part of a multidisciplinary research project to assess and model water and solute transport in small catchments. Part of the summary compares the simulation of the water budget of an Australian catchment by the two models, HILLFLOW and TOPOG.

Both models were applied to the Picaninny catchment, in the central highlands of Victoria, and the results of both discharge measurements and long-term (9 years) soil moisture measurements were compared. HILLFLOW modelled soil moisture well and predicted values coincided with measured values. The TOPOG simulation underestimated soil moisture by about 10% (volume) at the end of the period, following a progressively decreasing trend. Bronstert (1999) suggests that the different results are due to the hydraulic conductivity of HILLFLOW being based on the double porosity concept of macropores and matrix, whilst TOPOG represents the soil as a single porous medium. Given the high infiltration rates observed in the field, TOPOG must represent these as high matrix conductivity values whereas HILLFLOW can show moderate matrix conductivity and high macroporosity. Under these conditions, in the periods between rainfall events TOPOG's higher matrix conductivity continually over-estimates soil water recession and predicts lower volumetric soil moisture.

Later versions of TOPOG include a simple macropore flow algorithm, where each catchment element is given a macropore volume, depth and saturated hydraulic conductivity (CSIRO 1998). In the model, macropores can only fill when infiltration excess occurs, and at the end of each timestep, lateral flow via macropores is allowed at a rate determined by slope and the saturated hydraulic conductivity of the macropore. This process ignores unsaturated lateral flow and the determination of Beven and Germann (1982) that within macropores, gravity is the only driving force for water flow.

The accurate measurement of saturated hydraulic conductivity is a difficult proposition, and Davis *et al.* (1999) query the ability of modellers to estimate this parameter *a priori*

for catchment scale simulations. Their paper evaluated the effects of saturated hydraulic conductivity (K_s) measurement methods on simulated yields from a forested catchment. Using TOPOG_Dynamic, a series of simulations were conducted with K_s estimated by (i) constant head well permeameter, (ii) small cores (6.3 x 7.3 cm) and (iii) large cores (22.3 x 30 cm). Large core estimates of K_s were one to three orders of magnitude higher than those obtained by the other two methods (both of which fail to sufficiently account for preferential flow), and provided good simulations of water table depth and catchment discharge (Davis *et al.* 1999). The results indicated that caution is required in applying point soil hydraulic measurements to catchment scale models.

Zhang *et al.* (1999^a) and Zhu *et al.* (1999) both reported studies (in southeast Australia and the Loess Plateau of China respectively) using TOPOG_IRM to estimate soil moisture and groundwater recharge (Australia) and erosive stormflow (China). The model TOPOG_IRM (Dawes and Hatton 1993) predicts the dynamic interactions within the soil-vegetation-atmosphere system, and consists of three submodels simulating water, carbon and energy balances (Zhang *et al.* 1999^a).

In the Australian simulation, despite using vegetation parameters mainly based on the literature (only 3 of 13 were measured), and non-fitted soil hydraulic parameters, the authors found good agreement between measured and simulated soil moisture content, and “appropriate partitioning of soil evaporation and transpiration”. Ground water recharge was shown to be variable throughout the catchment and related mainly to soil type. The authors concluded that the “success of the model in simulating these variables showed that the assumptions inherent in the model capture the key processes relating to soil moisture dynamics and evapotranspiration” (Zhang *et al.* 1999^a).

Zhu *et al.* (1999) adapted TOPOG_IRM and TOPOG_STORM to simulate the slowly changing soil hydrologic states, as well as the vegetation cover, in studies on the processes involved in surface and subsurface flowpaths during and between storm events in a catchment on China’s Loess Plateau. Input parameters for the vegetation growth simulation submodel were all derived from the literature ie. the TOPOG user manual (Zhu *et al.* 1999). Some input variables (eg. initial soil moisture, soil structure, capillary length, water table depth) were adjusted within known variability.

The results of the above study (Zhu *et al.* 1999) indicated the value of combining some land management features (in TOPOG_IRM) with the simulation of stormflow (in TOPOG_STORM), showing the critical importance of managing the soil surface to reduce erosional events. However, the authors note that the results “highlighted the considerable variability in simulation accuracy”. Better simulation results were generated from large rainfall events than small, and the authors suggest that this may be due to errors in simulating water detention storage, antecedent soil moisture and surface infiltration. Accuracy also varied considerably when comparing the whole catchment with internal areas, and this indicated problems with both issues of scale and the inherent variability and uncertainty in parameterisation.

TOPOG_IRM is a complex mechanistic model, although of necessity it contains some assumptions and abstractions which affect both realism and ultimately, precision (Dawes and Hatton 1993). Theoretically, the model can be applied to any landscape as long as the assumptions and abstractions are relevant, whereas empirical models may be site specific. The model's precision will deteriorate as its abstract characterisations depart from the real world landscape.

7.3 Model description (incorporating methodology)

7.3.1 Digital elevation model

The catchment was aerially surveyed (University of SA), and photogrammetric data recorded along with aerial photographs. Using X, Y and Z values (eastings, northings and elevation) from this survey, a 5 x 5 m grid was derived for the entire catchment and a further 1 x 1 m grid across the area occupied by the drains. XYZ values were also determined for each drain arc, as well as the intersecting dividers and end zones, and a coverage of the drain arrangement was developed in ARCINFO, to be overlain on the contour coverage (digital elevation model). The original ASCII XYZ file had to be adapted to FORTRAN format to be used in the model

The XYZ file was the foundation for the development of a digital elevation model (DEM) in TOPOG, producing a grid of the elevation data (.*grd* file), the topographic

contours (*.tcn* file) and a range of other output files defining such things as high points and saddles (*.htp*, *.sad* files) and the data boundary (*.sys* file). Figure 7.1 shows the drain ARC coverage superimposed on the DEM (the drains are georeferenced so that their position on the DEM is a reflection of their placing in the real world).

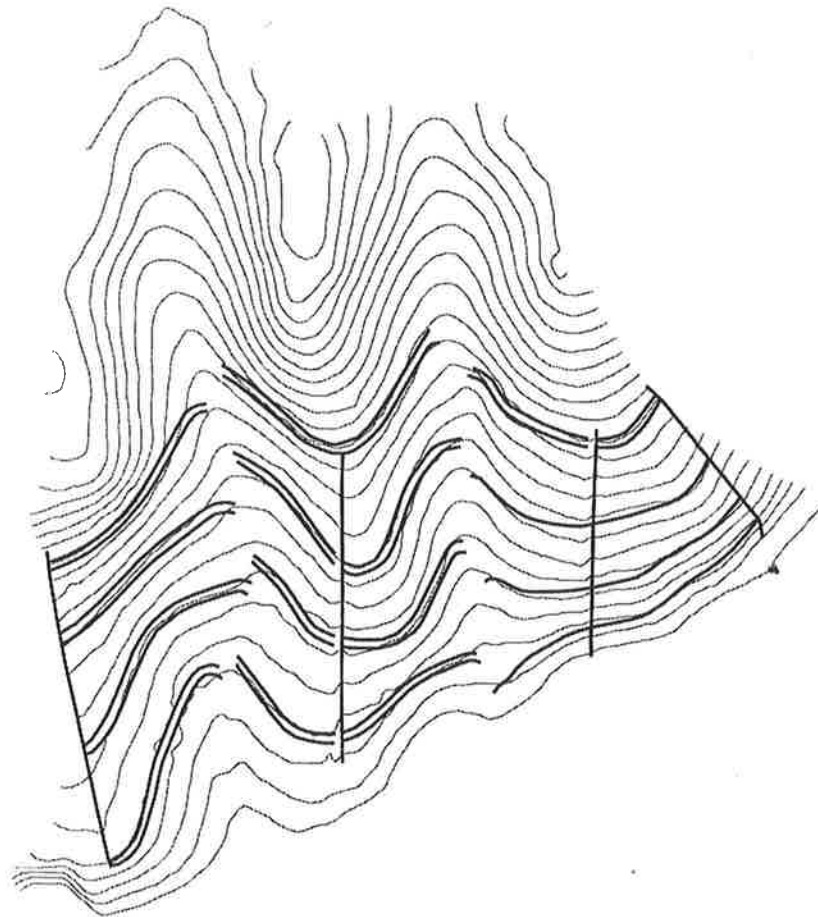


Fig. 7.1 Drain coverage from ARCINFO, overlaid on digital elevation model of study site, Keyneton.

7.3.2 Defining a catchment

Given the previously developed contour file, and by specifying the start and end points on the lowest catchment contour, as well as boundary high points and saddles, a catchment contour map was developed which calculated any internal watersheds, and partitioned the catchment on the basis of streams and ridges into a series of hillsides. A secondary segmentation process then calculated flow trajectories from one contour to the next upslope, creating catchment elements (Fig 7.2). Slope, aspect and potential

seasonal radiation were calculated for each of these catchment elements (Dawes and Hatton 1993; CSIRO 1998).

A drainage index (W) for each element (defined by its x, y location) was calculated in the model using the equation;

$$W(x,y) = (1/mbT)fqdA \quad \text{Equation 7.1}$$

where,

m = slope (m/m)

b = length of contour at element base (m)

T = local transmissivity (m²/d)

A = element area (m²)

q = net subsurface drainage flux (m/d) (*conceptually, q represents catchment baseflow*)

The drainage capacity of any point is affected by its slope (m), the size of its surface area (b), evaporation (a function of slope and aspect) and soil transmissivity (T). The drainage index (W) is a measure of each element's ability to drain a prescribed net lateral flux (q) taking into account the above factors CSIRO (1998).

7.3.3 Dynamic water balance modelling

The TOPOG_Dynamic component of TOPOG operated on the element network discussed above, and computations were performed in a number of modules to simulate a range of processes including;

- canopy interception (up to 2 plant layers) of rainfall and subsequent evaporation from the canopy and bare soil surfaces
- infiltration, vertical redistribution and lateral movement of soil water
- plant transpiration, and plant growth based on carbon allocations to the roots, stems and leaves.

The modules used in these simulations were;

- a soil water module
- an evapotranspiration module
- a plant growth module

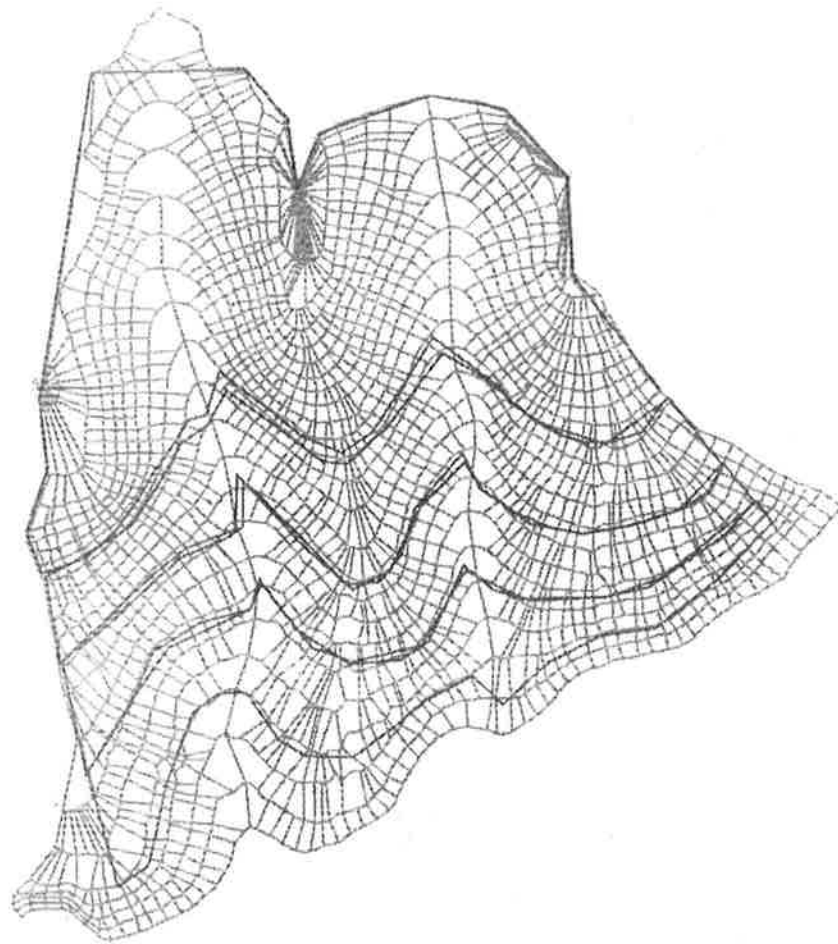


Fig. 7.2 Element network generated by TOPOG, for study site. Drains are super-imposed.

Soil water module

This module dealt with the infiltration and vertical movement of water in the unsaturated zone, as well as the dynamic interactions of perched water tables and possible subsequent lateral water movement in the saturated zone. Of the three choices for this module, the Newton-Raphson scheme was selected (Dawes and Hatton 1993; CSIRO 1998).

This scheme allowed for soil layering (up to 8 layers), and computed soil moisture based on a stacked series of nodes. "When a water table developed at any node, water was routed downslope through the TOPOG flow net using Darcy's law (ie. at a rate based on the local soil K_s value and the slope of the saturated water surface, assumed to be the same as the topographic gradient)" (CSIRO 1998). The Newton-Raphson scheme used a

‘mixed’ form of the Richard’s equation for an implicit solution (W. Dawes, pers. comm.).

Evapotranspiration module

Transpiration from vegetation, and soil evaporation, was predicted for all catchment elements at each daily timestep. The Penman-Monteith equation was used to calculate transpiration, with incoming short-wave radiation modified to correspond to the slope and aspect of individual elements. Canopy conductance in the transpiration model can be specified for each vegetation layer, from three methodologies;

1. scaled according to vapour pressure deficit,
2. proportional to the carbon assimilation rate and modified by surrounding air vapour pressure deficit, and
3. proportional to assimilation rate and moderated by transpiration rate (CSIRO 1998).

Soil evaporation was also calculated using the Penman-Monteith equation.

Plant growth module

Growth was modelled, using the Integrated Rate Methodology (IRM, Dawes and Hatton 1993), as a function of light, water and nutrients (the interactions of nutrients are difficult to quantify, so the model represents their availability, to supply nutritional requirements on a zero-one scale), where light is modified by ambient temperature, and water is modified by root zone soil salinity. These factors were combined to compute a carbon assimilation index and hence a growth rate, with assimilate dynamically partitioned across five carbon pools;

- leaf
- root (TOPOG does not model plant production of reproductive material)
- stem (live)
- litter
- a soluble pool (expressed as a fraction of leaf carbon) where, if in the growth routine there is an assimilate deficit, ie. the plant cannot cover respiration, then the soluble pool is reduced first. Similarly if there is an assimilate surplus and growth can occur, the soluble pool is again filled first (Dawes and Hatton 1993).

7.4 Data Inputs

7.4.1 *Climate*

Minimum requirements were (i) a referenced day number, (ii) daily maximum/minimum temperatures ($^{\circ}\text{C}$), and (iii) daily total rainfall (mm/d). Mean daily vapour pressure deficit (mbars) and total solar radiation on the horizontal ($\text{KJ/m}^2/\text{d}$) could be supplied, or generated through a climate simulator module. Weather data (modified to suit the model requirements) from the automated weather station at Keyneton was input to the model.

Elevation (400 m) and mean annual rainfall (549 mm) were used to scale the data, and the site specific latitude (-34° , where the minus sign indicates southern hemisphere) was input to scale horizontal surface radiation to the slope/aspect configuration of individual catchment elements.

7.4.2 *Soil properties*

Soil tables were built using a soil moisture accounting scheme based on the Richard's equation, representative of the Broadbridge and White soil property model (see Dawes and Hatton 1993). This assumes soil water retention curves with continuous slopes under all circumstances, thus promoting model stability (CSIRO 1998). Input requirements and typical values are shown in Table 7.1. Each soil type must be represented by a discrete soil file.

Table 7.1 Typical values of inputs used to develop soil tables based on Richard's equation (CSIRO 1998).

Inputs	Units	Typical values	Description
K_s	m/d	0.001-50.0	Saturated hydraulic conductivity
θ_s	m^3/m^3	0.30-0.80	Volumetric soil moisture content at saturation
θ_n	m^3/m^3	0.01-0.2	Volumetric soil moisture content at air-dry
C		1.001-2.0	The soil structure parameter; high values of C produce soil characteristic profiles indicative of well structured macropore soils, low values of C would be used for poorly drained soils with well developed capillary fringes.
λ_c	m	0.01-0.2	Capillary length scale, determined from sorptivity and C.
D	%	0.1	The most can change per iteration (% of $\theta_s - n$)
Ψ_{min}	m	-50 to -150	The driest psi value on the table
n		150 - 250	Number of points in soil table

Table 7.2 Initial input parameters, and values, in the development of soil files.

	<u>Upper-slope</u> horizon		<u>Mid-slope</u> horizon		<u>Lower-slope</u> horizon		<u>Universal</u>
	A	B	A	B	A	B	C
** K_s	0.639	0.08	0.136	0.06	0.372	0.006	0.07
** θ_s	0.40	0.37	0.43	0.35	0.41	0.33	0.32
# θ_n	0.05	0.05	0.05	0.05	0.05	0.05	0.08
# λ_c	0.10	0.80	0.10	0.80	0.10	0.80	0.10
#C	1.50	1.10	1.50	1.10	1.50	1.10	1.05

** = measured data

= TOPOG user manual data

Initially, seven soil files were created (Table 7.2), six to acknowledge differences in the hydrological characteristics of the A and B horizon soils in the three different slope positions, and a single 'universal' C horizon soil (see Tables 4.3-4.5 for morphological variations in horizon characteristics). K_s values were from measured data (see Table 4.9), and θ_s from neutron probe data at known periods of saturation, whilst the other parameter values were recommended default values (R.Silberstein, pers. comm.).

In later simulations, a range of other values, including maxima and minima, for the above measured parameters were trialed. In addition, there were variations on those default/literature derived parameter values from a range of sources. One of the difficulties was trying to determine just which of a wide range of values to use as suitable inputs for the simulations. For Lambda C for instance, values (in metres) ranged through 0.01-0.2 (CSIRO 1998) for the complete range of soil textures, 0.1 for coarse sandy loam to 0.5 for base clay (Dawes and Hatton 1993), 0.2 –0.5 for a duplex profile (Zhang *et al.* 1999^b), 0.1 for sand to 1.0 for clay (Hatton and Dawes 1993), and 0.02-0.05 for sand to 0.5-2.0 for heavy clay (R.Silberstein, pers. comm.).

Hatton *et al.* (1995), in a sensitivity analysis conducted on a range of model parameters, found that deep drainage was increased with increasing Lambda values (a 10% increase in λ_c leading to a 4.7% increase in drainage). Transpiration, soil evaporation and plant growth showed little sensitivity to any changes to this parameter. Increases in K_s also increased deep drainage (a 10% increase producing 11.7% change) and reduced soil evaporation in response to soil water moving away from the soil surface (Hatton *et al.* 1995).

7.4.3 Soil profiles

Every catchment element was matched with an appropriate *.nodes* file that described the element profile. Each profile type was allocated nodes at depths below the soil surface, with an associated integer value specifying that node's corresponding soil table. In order to model the soil surface/litter layer evaporation credibly, and for model stability, node spacings down the profile were small at the top and increased down the profile (CSIRO 1998).

By examination of the profile descriptions of each treatment site, it was possible to group sites with similar profiles (ie. horizon changes occurred at the same depths), and seven soil profile files were developed to represent these.

7.4.4 Vegetation

A *type2 .veg* file was used as plant growth was invoked. This model ran on a daily timestep. Inputs (and typical values) are shown in Table 7.3. Initial root carbon values for each node of each profile must also be listed in a *.rootc* file, a single column of values (kg/m^2) with identical line numbers to the *.nodes* file it is linked to. (This *rootc* file, combined with the *nodes* file describing the soil profile, was the means for manipulating the depth of plant water extraction).

In addition, for each vegetation type represented, the following information was required;

- a. initial leaf carbon values for each vegetation type
- b. initial leaf carbon to stem carbon ratios for each vegetation type (CSIRO 1998).

There was also the option to leave LAI static (ie. no growth and no death). However, even if growth does not occur, growth rate calculations are performed as an assimilation feedback on canopy conductance (Dawes and Hatton 1993).

For initial simulations, inputs were mainly from the values suggested below (CSIRO 1998), and on advice from those involved in the development of this model (R. Silberstein [TOPOG], pers. comm.). Maximum plant rooting depth was the only measured parameter. Later simulations also included suggestions from other modellers (A. Moore [Grazplan], pers. comm.), including (for phalaris) an increase in R_{int} to 0.001 and a reduction in A_{max} to 0.01.

Table 7.3 Inputs, typical values for a vegetation file (CSIRO 1998), and values applied in the early simulations (values in parentheses are changes in latter simulations).

Variable	Units	Typical values	Applied values			Description
			Lucerne	Phalaris	Cocksfoot	
*A _{can}	(0-1)	0.1-0.3	0.15	0.10	0.10	albedo of the canopy
A _{soil}	(0-1)	0.1-0.3	0.15	0.15	0.15	albedo of the soil
*R _{int}	m/lai/d	0.0001-0.002	0.0003	0.0007 (0.0003)	0.00006 (0.0003)	canopy interception coefficient
*L _{ex}		-0.3 to -1.0	-0.65	-0.65 (0.60)	-0.60	canopy light extinction coefficient
*A _{max}	kg C/m ² leaf	0.01-0.03	0.030	0.023	0.023	maximum carbon assimilation rate
g ₁		0.1-1.0	1.0	1.0	1.0	stomatal control coefficient; related to the vapour press. deficit dependence
lwp _{max}	m	-80 to -200	-200	-150	-150	max. plant available soil water potential
W _H	(0-1)	0.2/0.8	0.2	0.2	0.2	ratio of mesophyll to stomatal conductance; 0.2 for C3, 0.8 for C4 plants
W _N	(0-1)	0-1	0.5	0.5	0.5	nutrient weighting parameter; value of 1 means nutrients are limiting,
*T _{half}	°C	10-20	8	10	10	temperature half saturation rate
*T _{opt}	°C	20-30	15 (20)	20	20 (15)	optimal temperature for assimilation
jdgerm	day number	1-365	-1	-1	-1	germination day number for annuals; -1 if annuals not used
ddh	days °C	10000-20000	-1	-1	-1	degree day hours to senescence; -1 if annuals not used
Q _{psat}	μmoles/m ² leaf/s	1200-1800	1200	1200	1200	light saturation intensity
r _{max}	m	0.2-40.0	2.0	1.2	1.0	maximum plant rooting depth

Variable	Units	Typical values	Applied values			Description
			Lucerne	Phalaris	Cocksfoot	
sla	m ² leaf/kg leaf C	10-60	12	12	12	leaf area per unit mass of leaf carbon
*xleafres	kg/kg/d	0.001-0.004	0.003	0.002	0.002	Leaf maintenance respiration coefficient
xstemres	kg/kg/d	0.0001-0.0004	-1	-1	-1	Stem maintenance respiration coefficient; -1 for non-woody plants
*xrootres	kg/kg/d	0.001-0.004	0.0004	0.0002	0.0012 (0.0002)	root maintenance respiration coefficient
*xlmort	kg/kg/d	0.01-0.03	0.001	0.001 (0.004)	0.001 (0.004)	daily leaf mortality rate
*xsmort	kg/kg/d	0.01-0.03	0.001	-1	-1	daily stem mortality rate; -1 for non-woody plants
*xrmort	kg/kg/d	0.01-0.03	0.001	0.0001	0.00002	daily root mortality rate
*abpart	(0-1)	0.2-0.6	0.1	0.1 (0.2)	0.1 (0.2)	proportion of total fixed carbon allocated to stem and leaf
*saltf		0-4	1.5	1	1	salt sensitivity parameter; value of 1 means salt limits growth
*r _a	s/m	15-60	50	50	50	aerodynamic resistance for plant canopy
*r _s	s/m	50-110	80	100	60	aerodynamic resistance for soil surface
xrootwt	(0-1)	0-1	1	1	1	ignore; set to 1
type	Text		lucerne	phalaris	cocksfoot	vegetation type

* = parameter value recommended by R.Silberstein (pers. comm.), some of which are outside 'typical values' range.

7.4.5 Simulation control inputs

As a final step to preparing the simulation, a parameter file was built on a template provided. This contained the names of all input files, input parameter values, flags to start particular computational model routines, and a list of the data output requirements. TOPOG_Dynamic was then capable of running simulations.

7.5 Simulations

In the first simulations an endeavour was made to divide the model catchment in the same manner as the actual, by placing the drains in the landscape and dealing with the individual treatments as sub-catchments with their appropriate soil and vegetation characteristics. The difficulty that this raised, was that of distributing the drainflows. In reality much of the flow from the drains was directed off-site via sump drains, and this could not be accommodated by the model. Instead, the combined flows from each pair of treatments had to be delivered to the model elements representing the grassed waterways (drainage depressions), without any means of differentiating individual treatment drainage.

One method of dealing with this, was to assume for the sake of the model that the entire area of each treatment pair was in fact the one landscape unit, with one vegetation type and one representative set of soil features. This methodology then resulted in the delineation of an area of cocksfoot, phalaris and lucerne based pasture spatially represented across each of the three slope positions, upper, mid and lower slopes; nine virtual treatments, each with its unique soil profile file and related soil file. However, the time needed to perform such simulations (terminated after 18 hours when less than a quarter of the way into the process), particularly at the validation stage, could not be justified given limited access to the workstation.

To counter the above, Silberstein, R. (pers. comm.) suggested reducing the complexity for simulations by cutting out as much of the catchment as possible that was outside of the actual study site. He also recommended developing a coarser element network to reduce the number of elements within which mathematical computations were necessary, and working with a single vegetation/soil type at any one time.

Following this advice, individual simulations were run for the following;

- Cocksfoot, with soil files representing the upper, mid and lower slopes
- Phalaris, similarly in the upper, mid and lower slopes
- Lucerne, also in the upper, mid and lower slopes

Climate data was input for the three years of the study with daily timesteps in the model numbered from 1-1095. As NMM data, and hence SWC and its changes were not recorded until 1996, simulations were commenced on Day 400 (4th January 1996) and run until Day 1095 (31st December 1997). Rainfall over the period was c. 910 mm.

Numerous simulations (>500) were trialed in an attempt to validate the model with the range of measured data, and Tables 7.4 and 7.5 below illustrate the earlier and latter outputs respectively. In the early simulations it was apparent that there was considerable distance between reality and the predicted values, however there were some promising trends. If interception losses were interpreted as a surrogate indicator of plant growth, then predicted interception losses indicated that relative LAIs were in accord with qualitative field observations. This suggested that plant growth/development might be reasonably represented in the model.

Table 7.4 Results for a range of parameters from early TOPOG simulations.

	Slope position	Inter-ception (mm)	Net rainfall (mm)	Total transpiration (mm)	Total evapo-transpiration (mm)	Change in soil moisture storage (mm)
Cocksfoot	Upper	224	685	747	1039	-129
	Mid	205	704	754	1048	-139
	Lower	200	709	742	1042	-132
Phalaris	Upper	234	675	829	1142	-233
	Mid	224	685	812	1131	-222
	Lower	221	688	785	1108	-198
Lucerne	Upper	346	563	718	1165	-225
	Mid	333	576	715	1143	-223
	Lower	326	583	661	1104	-194

Measured changes in stored soil water (ΔS) over this period were generally positive but predicted values were all negative (Table 7.4), and all quite substantial. As there was no

runoff, throughflow or recharge predicted, all soil water loss was via evapotranspiration (ET). This indicated that soil hydraulic properties (model values) needed to be adjusted (within the range of measured data) to reduce ET and redistribute the excess amongst the other components of the water balance.

K_s was increased in the range of 0.01 – 0.02 m/d to improve drainage and reduce soil evaporation as soil water moved away from the soil surface more quickly (see Hatton *et al.* 1995). Volumetric soil moisture content at air-dry was raised from 0.05 to 0.1 m³/m³, and at saturation, reduced by approximately 0.05 m³/m³, to reduce soil water storage capacity and hence moderate ΔS . The parameter, C, which indicates macroporosity, was reduced marginally (0.1) to reduce macropore flow in light of the increase in K_s above. Table 7.5 shows the results of these adjustments. ΔS was substantially lower and in the range of -1 to -7 mm and total evapotranspiration was also reduced (c.200-300 mm less than previously). The redistribution of its components seemed more realistic (although runoff was under-represented with only 1 – 2 mm of lateral flow in 3 sites) with substantial quantities of deep drainage. Drainage and ET were closer to calculated values and there was a very weak linear relationship between calculated and simulated drainage (Fig 7.3) ($R^2 = 0.202$), and calculated and simulated values for ET (Fig. 7.4) ($R^2 = 0.302$).

Table 7.5 Results for a range of parameters following adjustments to soil hydraulic values.

	Slope position	Inter-ception (mm)	Net rainfall (mm)	Total transpiration (mm)	Total evapo-transpiration (mm)	Change in soil moisture storage (mm)	Recharge (mm)
Cocksfoot	Upper	47	863	581	763	-5	151
	Mid	53	857	576	766	-1	144
	Lower	77	833	662	838	-6	77
Phalaris	Upper	127	782	695	908	-2	0
	Mid	74	835	638	817	-6	98
	Lower	42	867	615	796	-3	116
Lucerne	Upper	76	833	645	890	-6	24
	Mid	79	830	670	890	-6	26
	Lower	77	833	709	881	-7	28

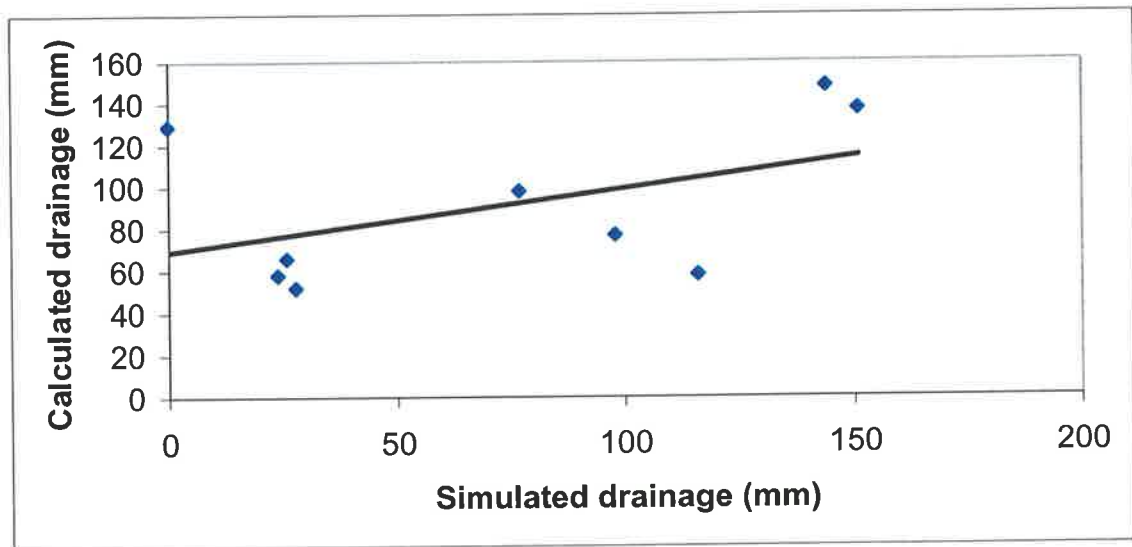


Fig. 7.3 Simulated versus observed values of drainage for a range of treatments (species x slope) ($R^2 = 0.202$).

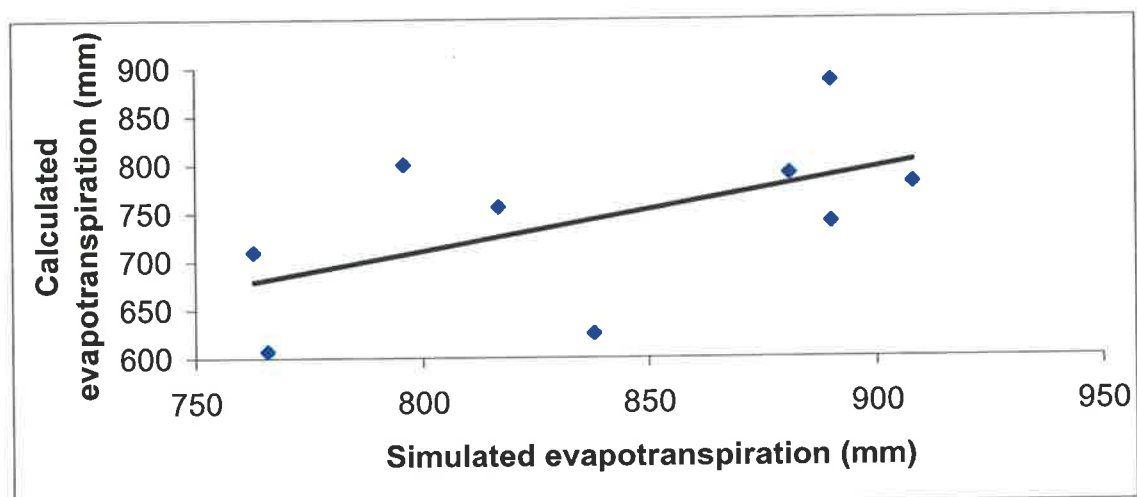


Fig. 7.4 Simulated versus calculated values of ET for a range of simulations ($R^2 = 0.302$).

Although the simulations showed some broad trends in agreement with some of the measured data, the outputs could not be considered a validation of the model, and this combined with other logistical constraints, precluded the running of long-term (25 year) simulations.

7.6 Conclusions

TOPOG is a data intensive model, and there is considerable reliance on literature sources when determining model inputs, particularly when trying to represent vegetation growth and development. In two studies reported above (Zhang *et al.* 1999^a ; Zhu *et al.* 1999), the authors primarily used these literature derived parameter values. Zhang *et al.* (1999^a) demonstrated good statistical comparison between observed and predicted evapotranspiration values, and by implication, good models of vegetation growth and water uptake.

Zhang *et al.* (1999^a) made the point that none of their parameter data were fitted to better match observed data, as “results must be seen in the light of the heterogeneous nature of the processes and unfitted parameters used” in the model. Zhu *et al.* (1999) however, adjusted some input variables “within their range of known variability”. In working with the model developers, it was apparent that data fitting was carried out as required (R.Silberstein, pers. comm.), and this fine tuning of parameters to give a better fit to observed data indicates problems in the generality of the model.

In the instance of this study, the first model simulations gave highly inflated evapotranspiration results, and badly predicted soil moisture storage changes (ΔS). This was hardly related to the inherent heterogeneity of the process; the input data were unsatisfactory. The problem then was to decide which parameters to change, often with no comprehension of a ‘real’ value, and unless fully conversant with the underlying mathematical assumptions, with no idea of the consequence of change. Under this situation the concerns of Zhang *et al.* (1999^a) of incorporating errors into the process representation by fitting one parameter to better match observed data, would be justified.

It was for this reason that any changes in parameter input values were only considered after referral to those involved in developing the model. The objective of this phase of the study was to try to extrapolate short-term sub-catchment scale results to longer-terms or larger areas by the use of a model. The results of the final simulations were unsatisfactory, and although the total ET values were in the right order of magnitude, it was obvious that the partitioning of the components of ET, and the derivation of some

of the moisture, was poorly represented in the model. The model needs a transparent mechanism for partitioning water losses from the system and should not have to be 'forced' to simulate say lateral flow rather than all ET or recharge.

The model could not adequately deal with the 'import' of water from below 1.8 m without extending the soil profile to this depth. Related to this was the assumption that as SWC was lowered in the 1.8 m profile, the DM production was reduced accordingly and this resulted in lucerne LAI approaching zero toward the end of the simulation period. Bulk densities were also higher than would normally be expected, with consequent lower porosities. Again, the model could not be configured to accept that there may still be adequate available water at depth outside of the soil matrix. One other major shortcoming was that although the trend of LAI increase/decrease was modelled, the seasonal growth patterns were out of synchrony with reality.

It was in the detail that the simulations seemed to fail, and this may have been due to an inability to represent reality in the model (eg. how to represent macropore flow before infiltration saturation, and if it could not be represented, how then to account for preferential lateral flows). As suggested by Bronstert (1999), by disallowing high macroporosity, TOPOG must necessarily show relatively high matrix conductivity with the potential to over-estimate soil water recession and hence show volumetric SWC at lower levels than reality. Whilst this may have been applicable to A horizon soils at Keyneton, SWC in the B horizon was more constant, and reality certainly differed from the model assumption.

Another factor alluded to previously concerned the use of measured data. The broad range of often contradictory parameter values either from the literature or given as default values by the model developers, was compounded in deciding just what form of any measured data to use. Given the level of heterogeneity encountered in the Keyneton soils, were mean values adequate representations to be used as model inputs? The compromise in representing this heterogeneity though, must be in the use of average data, as the prospect of smaller cells with measured data is not feasible at a landscape scale.

The Keyneton catchment is complex, with vertical macropore flow (Kirkby *et al.* 1997) and preferential lateral flow along the A/B horizon boundary and perhaps within the upper B horizon. TOPOG cannot explicitly deal with the macropore flow, and tries to handle it by approximating hydraulic conductivities, leading to limitations in the accuracy of the final result. TOPOG has shown the capacity to simulate catchment processes satisfactorily (Zhang *et al.* 1999^a; Zhu *et al.* 1999), although the latter authors noted considerable variability in simulation accuracy. As Dawes and Hatton (1993) stated, the precision of the model deteriorates as its abstractions depart from the real landscape.

Hatton *et al.* (2002) reported similar difficulties from the Ucarro catchment where the outstanding feature of their results was the enormous complexity and variability of hydrological processes operating within a small area (similar in size to the Keyneton site – c.5 ha). Qualities affected included the duration and extent of waterlogging, soil moisture behaviour and recharge. The complexity limited the potential of adequately representing the hydrological behaviour and management influences on-site with physics-based modelling.

Silberstein *et al.* (1999) concluded that the Ucarro site was “too complex and poorly understood to capture in a physical hydrological process model with typical assumptions made for such systems or with model parameters derived from field measurements”. Hatton *et al.* (2002) stated that even empirical interpretation of the soil or groundwater balances were problematic given such variability and complexity over a small area.

Silberstein *et al.* (2002) note that this extreme variability does not facilitate prediction at a local-scale with process-based models unless there is an equally extreme amount of data collected. However, at a broader scale this becomes less critical and Silberstein *et al.* (2002) found that variability in K_s for instance was no greater across the wider ‘Ucarro’ catchment than between measurements at replicate sites.

Stauffer *et al.* (2000), modelling on Eyre Peninsular, SA, reported uncertainty based on soil properties being truly representative of the site. TOPOG and similar models are

probably better suited to larger scale (or more uniform) catchments where 'average' input data are more appropriate.

In retrospect, TOPOG was not a good choice for this particular study for a number of reasons beside those listed above. At the time (1997-1998), TOPOG was still in its development stage, and was really only the province of serious modellers rather than those whose interest in it was simply as a tool for related research. Hatton (pers. comm. 1998) reiterated this in a presentation to a Salinity Forum at the State Landcare Conference (SA) in July of that year.

There was a high level of 'skill' necessary to operate the model efficiently and documentation was not user friendly. As the model was still evolving, there was no clear process for trouble shooting other than to record the problem on a CSIRO website for a response from the model developers. This often resulted in the provision of an updated model component that allowed the process to continue until the next 'bug' was found. However the compounding time delays inherent in this method of trial and error were considerable. Personal communication (including a two day visit) with the developers at Black Mountain (eg. Silberstein, Dawes, Vertessy, CSIRO, Canberra) whilst providing useful insights and advice, did not resolve the difficulty of model validation.

Chapter 8

Summary

8.1 Soil properties

From examination of soil pits and core samples, it was apparent that soils within the catchment were not homogeneous. There was a large degree of variability between soils across the slope and substantial variation from top to bottom of a toposequence. These differences are the result of ancient and more contemporary soil-water-landscape processes (Fritsch and Fitzpatrick 1994) and can be used to infer present day hydrology.

The upper slope reddish soils were freely drained with vertical flow paths, the yellow and grey soils of the mid and lower slopes were more prone to waterlogging and lateral flows (Fritsch and Fitzpatrick 1994). Bulk densities of these soils were high, and well in the range where root elongation might be affected, and they generally increased down the profile, and downslope. Low K_s in the B horizon led to perching and possible subsequent waterlogging, and this in concert with bulk density considerations was likely to impact on plant dry matter (DM) production. DM production decreased downslope for the three perennials in this study.

8.2 Water fluxes

In early 1995, when the construction works were being carried out for this project, ie. the grading of drains and the provision of barriers between sites, there had been an extended period of hot, dry weather. Plant growth on the site was negligible and subjectively, the A horizon soils appeared totally depleted of moisture. However, when the grader cut penetrated the clay B horizon, it was obvious that this section of the soil profile still held moisture.

Within 10 cm of the A/B horizon boundary, large earthworms (c. 8 mm diameter, >200 mm length) were found, and the clay could be easily formed into a bolus. Given the need for low subsoil moisture in autumn, to reduce waterlogging potential (Chapter 2),

it was apparent that this pre-condition was not being met under the current management routine. (Note that this qualitative comment cannot be supported by NMM data, as access tubes were not installed until 1996).

The literature at that time suggested that a 'throttle' existed at the A/B horizon boundary for these Adelaide Hills' soils, and between May and October a fresh perched water table developed in the A horizon soils on this boundary (Fitzpatrick *et al.* 1994; Fritsch and Fitzpatrick 1994). The reverse interceptor and the buried tube drains installed on site intersected the A/B horizon boundary to collect throughflow draining along this pathway (Cox 1988; Cox and McFarlane 1995). The drain flow was directed to grassed waterways leading downslope to the stream that drained the catchment.

Results from this study confirmed the development of a perched watertable within the B horizon in 10 of the 15 sites. Two of the lower slope sites and two of the upper slope showed perching that developed 5-10 cm above the A/B boundary, and in one lower slope site the development was on the A/B horizon itself. Given the depth of initiation of the perched watertable, the reverse interceptor and buried drains would intercept much of the throughflow as long as it was generated at or above that initiation depth. With the extension of the saturated zone further into the B horizon as shown in Figs. 5.5-5.9, it was unclear whether volumes of water were moving downslope as throughflow within the B horizon. It was more likely given relative hydraulic properties, that most movement when it did occur would be in the more permeable A horizon soils, and this was supported to some extent by the lack of additional flow volume in the more deeply placed tube drains.

Drains were effective in removing up to 16.4% of annual rainfall in an average rainfall year, and 5.4% in a dry year (Table 4.11). Flows were variable, with four sites recording overland flow in excess of 78% of total drainflow in 1996, however the majority of flow was from throughflow. In average and above average rainfall years, and particularly on sites without drains, overland flow is likely to be the dominant runoff mechanism.

It is worth noting that in all years, drainflow from cocksfoot pastures was more than from either of the other perennials (see Table 4.12). In the average rainfall year, drainflow from lucerne was about half of that from cocksfoot, and throughflow contributed about 26% of the lucerne flow (5.7 mm) compared to 81% of the cocksfoot flow (32.6 mm). If this were a consistent trend in average and above average rainfall years it would potentially remove the need for sub-surface (throughflow) drains in lucerne pastures in this environment. From an economic standpoint, and an environmental aspect given the problems of increased sub-surface drainage of nutrients from legumes, this could be significant.

Deep drainage under cocksfoot was 14% of annual rainfall whilst phalaris met the 10% of rainfall value generally attributed to annual pastures (CSIRO 1996). Drainage under lucerne was half that of cocksfoot and 7% of annual rainfall.

8.3 Perennial pastures and soil moisture

Deep-rooted perennials will remove soil moisture from B and C horizon soils. Lucerne roots had penetrated beyond two metres during the period of this study, and were highly effective in producing the desired autumn soil water deficit. Of the three perennials trialed, lucerne was the only one to maintain production throughout the year, both phalaris and cocksfoot becoming dormant through mid and late summer. Although lucerne root growth in the B and C horizons was primarily in macropores and other vertical faults, quantitative and qualitative results indicated their effect in extracting moisture from the surrounding soil matrix. A soil pit dug in lucerne pasture at the end of the study period, showed none of the B horizon moisture retention noted at the start, with instead what appeared to be a uniformly dry profile.

Phalaris, whilst not drying the soil profile to the extent of lucerne, still produced an autumn soil moisture deficit, and reduced waterlogging potential. Importantly, phalaris productivity was less influenced by site location than was lucerne. Whether new, more salt and waterlogging tolerant lucerne cultivars would perform better in the lower slopes needs to be considered. Cocksfoot has neither the production capabilities, nor the soil moisture mitigating effects, of lucerne or phalaris. Its root penetration is similar to

annual crops, and the present cultivar is of no benefit hydrologically in this environment as SWC did not reduce over the study period. Lucerne and phalaris (particularly lucerne) have greater capacity due to their rooting depth to remove soil moisture before it passes beyond the root zone.

It is an easy theoretical concept to partition the catchment into zones, dependent on topographic location, and plant it to the pasture type most suited. From a management point of view though this may be less attractive, due to increased cost (eg. fencing) or the extra difficulties imposed on the grazing regime. Even so, if there is to be any productivity gained from the saline lower slopes, there must be an introduction of salt-tolerant plants, and partitioning of the slopes is merely an extension of this concept. The Upper South East catchment plan for instance (CSIRO 1996) found that the value of increased production via saltland agronomy, as well as the flow-on effects, was expected to be greater than the entire cost of the plan itself.

Whether or not individuals will take up the concept of planting perennials will depend on their profitability relative to the agricultural enterprise they replace, or the area planted to perennials must protect a large area of additional (untreated) land (Bathgate and Pannell 2000). Figure 8.1 illustrates how direct profit from a perennial can be reduced as additional land is protected from salinity.

Based on a narrow financial perspective, Bathgate and Pannell (2000) suggest that with no additional land protected a perennial would need to generate a net profit of 78% of that of the enterprise it is replacing. As the area of protected land (ie. land that will no longer be subject to salinity) increases so the level of direct profit from the perennial can be decreased. With the relatively high costs of establishment for lucerne, as well as the potential for establishment failure, there are substantial risks involved for landholders to change enterprises for a resource conservation outcome.

Despite the awareness of farmers that increased planting of deep-rooted perennials is necessary if we are to prevent increasing salinity (MDBC 1999), it is unrealistic to expect that they would adopt this concept for altruistic purposes. The statement that “it isn’t easy to be green when you’re in the red” is often heard, and appropriate. Bathgate

and Pannell (2000) suggest that the most important issue in salinity policy at government level is that of financial incentives for the adoption of perennial production systems.

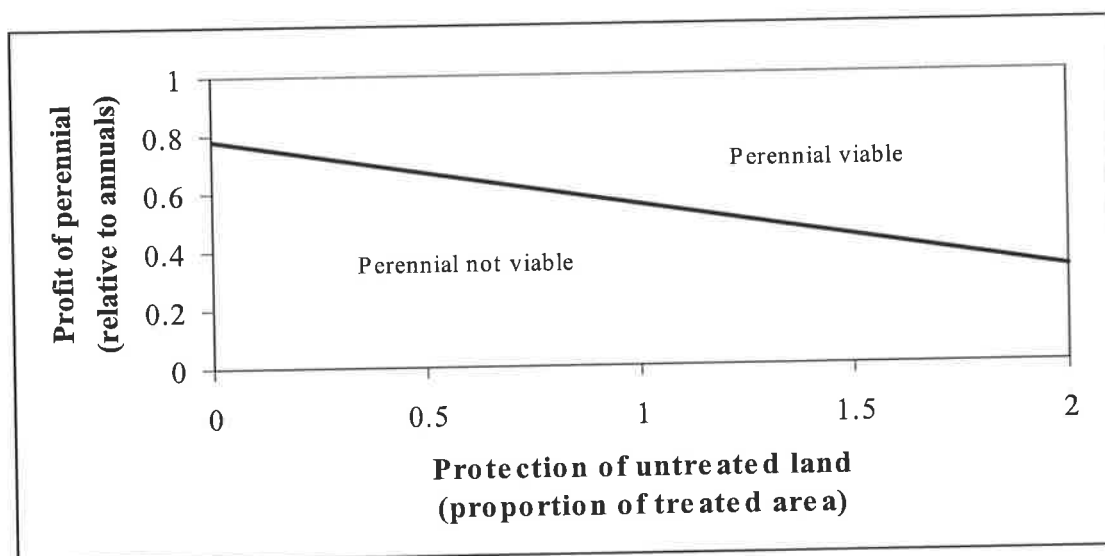


Fig. 8.1 Trade off between direct profitability and benefits from salinity prevention in calculating required net benefits of a perennial [from Bathgate and Pannell (2000)].

Given the increased recognition of salinity issues, and the importance being placed on reducing salt loads entering the Murray-Darling system (MDBC 1999), there is the potential in this area of the Adelaide Hills, to develop some cost-sharing mechanism that encourages the introduction of perennial and salt-tolerant pastures. Salinity credits (which could be an additional income source) are still at a formative stage, with no market mechanism yet established, although there appears to be an inevitability about their introduction given Federal and State government support. An indication of the private benefit of increased quantities, and quality of dry matter production, and the public (off-site) benefit from reduced recharge and hence salinity reaching the Murray River will be an important component of any future cost-sharing plan. The results of this study can assist in this.

A corollary of the above, particularly important in this era of high (and often competing) demand for water for production (irrigation) and/or environmental flows, is that better on-farm water management may reduce the quantity of water leaving a catchment and

entering river systems. The thrust to revegetate the landscape with trees and/or other deep-rooted perennial vegetation, to more closely mimic natural (pre-European) systems, will increase the water shortage problem. Vertessy (2003) referring to the loss of water following tree planting in the Murray Darling Basin over the next twenty years, states that the magnitude of this loss will be similar “to that which we are trying to claw back for environmental flows in the Murray”. Planting of deep-rooted perennial vegetation will bring environmental and economic benefits, however we need to factor the increased water usage into catchment planning and reduce allocations to users accordingly (Vertessy 2003).

As Young and McColl (2004) state, any assessment on the quantity of water needed to restore river health (eg. in the River Murray), involves the question of how “entitlements and allocations align with the hydrological processes”. They point to influences of land use changes that reduce recharge and runoff as well as impacts of increased water-use efficiency on river flows. Given the above, they suggest that irrigators need to understand that as well as clawing back access entitlements to increase environmental flows, some of them will also have allocations reduced because of these changes in land use, water use efficiency and other factors impacting catchment water balances.

8.4 Nutrient movement

Overland flow occurred infrequently in the Keyneton catchment, and generally nutrient concentrations in this flow were lower than those in throughflow (however, when it did occur it was significant). The combination of generally higher concentrations and potentially higher volumes of flow could lead to high chemical loads moving off site into waterways and reservoirs. A considerable percentage of phosphorus was moving in the dissolved state, and this was also cause for concern.

Under the present low input management regime (ie. cocksfoot based pasture without drains), nutrient and other chemical losses are probably of less concern than are those from phalaris and/or lucerne. Under cocksfoot, nitrate concentrations were usually lower, and there were lower P concentrations with most moving in the particulate form. Without drains to intercept flows, it is likely that low levels of P in the freely drained upper slope would leach through the profile to the watertable. In the lower slope

positions it is unlikely to pass beyond the B horizon as a result of physical and chemical changes in the soil profiles in these positions (Smythe 1995).

The high concentrations of $\text{NO}_3\text{-N}$ in tube drain throughflow under lucerne were of concern, and were above recommended levels for drinking water. The potential for these levels of leached $\text{NO}_3\text{-N}$ to lead to an increase in soil acidity in this landscape needs to be considered, although from an agricultural perspective the losses are quite small.. Removing the leachate via drains, simply moves the nutrient into the stream system, increasing the likelihood of eutrophication downstream.

If drains are likely to be transporting significant nutrient loads off-site, it is probably better that drain flows are retained in dams, or otherwise managed, to reduce the potential for environmental damage. As most of the soils in the lower slopes are sodic to some degree, it would be unwise to build containment dams in these areas. A better proposition perhaps, would be to establish a series of small wetlands with reedbed filters, at the bottom of the sub-catchment, and attempt to lessen nutrient loads in a more 'natural' manner.

8.5 Simulation modelling

The value of modelling is in the potential to extrapolate assumptions and abstractions from a known time period into the future, or from a known locality to one with similar characteristics. To have any faith in a model, it must be validated against known data and show reasonable correlation between observed and predicted behaviour. The model showed some relationship between calculated and simulated ET and drainage but was not really capable in this trial of partitioning water losses from the system, with preferential losses via interception and ET.

In other reported studies where TOPOG has been shown to have potential (Zhang *et al.* 1999^a; Zhu *et al.* 1999), lateral flow has not been considered to be of importance. Similarly, macropore flow occurring prior to infiltration excess has appeared not to be an issue, except in Bronstert (1999). Dawes and Hatton (1993) noted that model

precision deteriorates as the inherent assumptions and abstract characterisations depart from that of the real world, and in this instance, that was the case.

At Keyneton, both lateral flow and early preferential macropore flow (Kirkby *et al.* 1997) are important components of the local soil hydrology. This, and the extreme heterogeneity apparent in the soils at a similar landscape position, and particularly along a toposequence, makes the determination of model inputs difficult and imprecise. Although it was beyond the scope of this study to manipulate the model to better represent the reality than the abstraction, an attempt was made to do so in conjunction with the developers of the model in Canberra, but this also proved unsuccessful. TOPOG appeared unsuitable for the Keyneton hydrology and this is in line with the findings of Hatton *et al.* (2002) in their Ucarro catchment study in WA.

8.6 Further research

One of the realities of field research, is that it seldom delivers the weather you want/need for the purpose of the project. For a study on soil hydrology, it would have been beneficial to have one year of very high rainfall to compare potential waterlogging, as well as overland and lateral flows under these circumstances. It is probably a vain hope, but it would be sensible to think that after the often substantial costs involved in setting up and running a research project, that data collection would be on-going beyond the two or three years of initial study, to better identify trends over time. One of the benefits of a successful modelling exercise would be that these field constraints would be overcome.

From a management perspective it is important that landholders in this and similar environments are made aware of the economic and environmental consequences of a range of options in managing their enterprise. The results of this study potentially provide a number of options and indicative outcomes. These range from low DM producing cocksfoot pastures with high recharge, and high runoff with generally low nutrient concentrations, to high DM producing lucerne, that provides feed in the crucial late summer-early autumn period, reduces recharge substantially and lowers runoff, but may have negative impacts through nitrate leaching.

The data from this study indicate that sub-surface drains may be less important for lucerne than for cocksfoot given the reduced level of throughflow under the legume. Phalaris provides another option that delivers higher DM production than cocksfoot with improved hydrological benefits. A mixed lucerne-grass pasture may provide the hydrological benefits of lucerne whilst the grass component utilises any surplus $\text{NO}_3\text{-N}$ fixed by the legume so reducing any leaching potential.

Regardless of the above, there are some points that need further clarification, for both management considerations, and to better understand the processes operating in that environment. The perched water table development within the B horizon needs explanation. Conceptually, a 'throttle' along a horizon boundary is easier to accept than the occurrence of one within any particular horizon, and the factors leading to its genesis need to be understood. The ability to predict the frequency and location of waterlogging in relation to climate and landscape variation is needed to allow efficient and effective management intervention.

The use of better perennials than the current cocksfoot cultivar seems warranted, and trials of a range of perennial cultivars, particularly the less salt (and waterlogging) sensitive varieties would be of benefit. For example, *Melilotus albus* (sweet clover) is performing well in saline waterlogged soils near Hamilton, Victoria. In concert with this though, there needs to be a better understanding of the potential for $\text{NO}_3\text{-N}$ leaching in this environment. Finally, the capacity to trap mobile nutrients within the catchment, by reed bed filters or other mechanisms should also be looked at, to reduce or prevent downstream environmental consequences.

Appendix 1 Bulk density data from Keyne's catchment

ID	Date	Horizon	Bulk density (Mg/m ³)	Upper depth (m)	Mid depth (m)	Lower depth (m)
1	01/02/1994	Ap	1.32	0.00	0.05	0.10
2	01/02/1994	Ap	1.29	0.00	0.05	0.10
3	07/09/1994	Ap	1.49	0.02	0.07	0.13
4	07/09/1994	E1	1.28	0.12	0.17	0.22
5	07/09/1994	E1	1.49	0.23	0.28	0.33
6	07/09/1994	E1	1.59	0.33	0.38	0.43
7	07/09/1994	E1	1.88	0.52	0.57	0.62
8	07/09/1994	Bt1	1.97	0.75	0.80	0.85
9	07/09/1994	Bt1	1.87	0.82	0.87	0.92
10	07/09/1994	C	1.94	0.91	0.96	1.01
11	07/09/1994	C	1.78	1.00	1.05	1.10
12	07/09/1994	C	1.76	1.18	1.23	1.28
13	07/09/1994	C	1.39	1.39	1.44	1.49
14	07/09/1994	C	1.96	1.68	1.73	1.78
15	07/09/1994	C	1.92	1.76	1.81	1.87
16	01/02/1994	E1	1.62	0.50	0.55	0.60
17	01/02/1994	E1	1.82	0.50	0.55	0.60
18	15/02/1995	Ap	1.30	0.00	0.05	0.10
19	25/09/1996	E1	1.43	0.05	0.10	0.15
20	25/09/1996	E1	1.64	0.25	0.30	0.35
21	25/09/1996	E1	1.74	0.45	0.50	0.55
22	25/09/1996	E2	1.76	0.65	0.70	0.75
23	25/09/1996	Bt1	1.50	0.85	0.90	0.95
24	25/09/1996	C	1.78	1.00	1.05	1.10
25	01/01/1998	Ap	1.45	0.00	0.00	0.05
26	01/01/1998	Ap	1.45	0.00	0.00	0.05
27	01/01/1998	Ap	1.38	0.00	0.00	0.05
28	01/01/1998	Ap	1.37	0.00	0.00	0.05
29	01/01/1998	Ap	1.29	0.00	0.00	0.05
30	01/01/1998	E1	1.47	0.19	0.24	0.29
31	01/01/1998	E1	1.77	0.25	0.30	0.35
32	01/01/1998	E1	1.64	0.25	0.30	0.35
33	01/01/1998	E1	1.53	0.27	0.32	0.37
34	01/01/1998	E2	1.91	0.65	0.70	0.75
35	01/01/1998	C	1.79	1.05	1.10	1.15
36	01/01/1998	C	1.78	1.15	1.20	1.25
37	01/01/1998	C	1.66	1.32	1.37	1.42
38	01/02/1994	Ap	1.38	0.00	0.05	0.10
39	01/02/1994	Ap	1.46	0.00	0.05	0.10
40	01/01/1995	E	1.54	0.02	0.07	0.12
41	01/01/1995	C1	1.74	0.15	0.20	0.25
42	01/02/1994	C2	1.93	0.32	0.37	0.42
43	01/02/1994	C2	1.75	0.32	0.37	0.42
44	01/01/1995	C2	1.85	0.33	0.38	0.43
45	01/01/1995	C3	1.95	0.42	0.48	0.53
46	01/01/1995	C3	1.75	0.52	0.58	0.63
47	01/01/1995	C4	1.66	0.64	0.69	0.74

48	15/02/1995	Ap	1.42	0.00	0.05	0.10
49	01/02/1994	Ap	1.32	0.00	0.05	0.10
50	01/02/1994	Ap	1.36	0.00	0.05	0.10
51	01/02/1994	Bt1	1.90	0.45	0.50	0.55
52	01/02/1994	Bt1	1.80	0.45	0.50	0.55
53	15/02/1995	Ap	1.34	0.00	0.05	0.10
54	01/02/1994	Ap	1.46	0.00	0.05	0.10
55	01/02/1994	Ap	1.40	0.00	0.05	0.10
56	07/09/1994	A12	1.54	0.02	0.07	0.12
57	07/09/1994	A12	1.74	0.15	0.20	0.25
58	07/09/1994	E	1.85	0.33	0.38	0.43
59	07/09/1994	BE	1.95	0.42	0.48	0.53
60	07/09/1994	Bt21	1.75	0.52	0.58	0.63
61	07/09/1994	Bt21	1.66	0.64	0.69	0.74
62	01/02/1994	E	1.64	0.32	0.37	0.42
63	01/02/1994	E	1.76	0.32	0.37	0.42
64	15/02/1995	Ap	1.43	0.00	0.05	0.10
65	01/03/1995	Ap	1.38	0.00	0.03	0.05
66	01/03/1995	A12	1.60	0.05	0.07	0.15
67	01/03/1995	E	2.01	0.27	0.30	0.34
68	01/03/1995	E	1.99	0.34	0.41	0.48
69	01/03/1995	BE	1.97	0.48	0.50	0.52
70	01/03/1995	Bt21	1.90	0.52	0.56	0.64
71	01/03/1995	Bt22	1.81	0.80	0.83	0.87
72	01/03/1995	C	2.14	0.92	0.97	1.02
73	01/03/1995	C	2.14	0.87	1.10	1.07
74	01/03/1995	C	2.28	1.10	1.16	1.22
75	25/09/1996	A12	1.57	0.05	0.10	0.15
76	25/09/1996	E	1.89	0.25	0.30	0.35
77	25/09/1996	BE	1.81	0.45	0.50	0.55
78	25/09/1996	Bt21	1.66	0.60	0.65	0.70
79	01/01/1998	Ap	1.29	0.00	0.00	0.05
80	01/01/1998	Ap	1.21	0.00	0.00	0.05
81	01/01/1998	Ap	1.39	0.00	0.00	0.05
82	01/01/1998	Ap	1.38	0.00	0.00	0.05
83	01/01/1998	A12	1.68	0.13	0.18	0.23
84	01/01/1998	E	1.94	0.25	0.30	0.35
85	01/01/1998	E	1.61	0.26	0.31	0.36
86	01/01/1998	BE	1.68	0.41	0.46	0.51
87	01/01/1998	BE	1.62	0.42	0.47	0.52
88	01/01/1998	Bt21	1.63	0.62	0.67	0.72
89	01/01/1998	C	1.84	1.17	1.22	1.27
90	01/01/1998	C	1.81	1.40	1.45	1.50
91	01/02/1994	Ap	1.34	0.00	0.05	0.10
92	01/02/1994	Ap	1.41	0.00	0.05	0.10
93	01/02/1994	EB	1.74	0.29	0.34	0.39
94	01/02/1994	EB	1.86	0.29	0.34	0.39
95	15/02/1995	Ap	1.38	0.00	0.05	0.10
96	01/02/1994	Ap	1.32	0.00	0.05	0.10
97	01/02/1994	Ap	1.36	0.00	0.05	0.10
98	07/09/1994	Ap	1.43	0.00	0.06	0.11

99	07/09/1994	E	1.51	0.09	0.14	0.19
100	07/09/1994	Bt2	1.31	0.58	0.64	0.69
101	07/09/1994	C	1.52	0.68	0.73	0.78
102	07/09/1994	C	1.44	0.80	0.85	0.90
103	07/09/1994	C	1.50	0.95	1.01	1.05
104	07/09/1994	C	1.66	1.04	1.09	1.14
105	07/09/1994	C	1.82	1.14	1.19	1.24
106	01/02/1994	Bt1	1.79	0.29	0.34	0.39
107	01/02/1994	Bt1	1.74	0.29	0.34	0.39
108	01/01/1995	Bt2	1.40	0.45	0.50	0.55
109	15/02/1995	Ap	1.34	0.00	0.05	0.10
110	07/09/1996	EB	1.71	0.20	0.25	0.30
111	25/09/1996	E	1.56	0.05	0.10	0.15
112	25/09/1996	Bt1	1.75	0.25	0.30	0.35
113	25/09/1996	Bt2	1.67	0.45	0.50	0.55
114	25/09/1996	Bt2	1.52	0.60	0.65	0.70
115	01/01/1998	Ap	1.35	0.00	0.00	0.05
116	01/01/1998	Ap	1.31	0.00	0.00	0.05
117	01/01/1998	Ap	1.18	0.00	0.00	0.05
118	01/01/1998	Ap	1.34	0.00	0.00	0.05
119	01/01/1998	E	1.65	0.09	0.14	0.19
120	01/01/1998	E	1.83	0.15	0.20	0.25
121	01/01/1998	E	1.86	0.17	0.22	0.27
122	01/01/1998	E	1.78	0.19	0.24	0.29
123	01/01/1998	Bt2	1.83	0.50	0.55	0.60
124	01/01/1998	Bt2	1.69	0.56	0.61	0.66
125	01/01/1998	C	1.93	0.95	1.00	1.05
126	01/01/1998	C	1.78	1.15	1.20	1.25
127	01/01/1998		1.39	0.00	0.00	0.05
128	01/01/1998		1.49	0.00	0.01	0.06
129	01/01/1998		1.70	0.00	0.01	0.06
130	01/01/1998		1.46	0.00	0.02	0.07
131	01/01/1998		1.52	0.07	0.12	0.17
132	01/01/1998		1.77	0.17	0.22	0.27
133	01/01/1998		1.85	0.18	0.23	0.28
134	01/01/1998		1.60	0.29	0.34	0.39
135	01/04/1997	Ap	1.16	0.00	0.03	0.05
136	01/04/1997	A2/E	1.42	0.08	0.11	0.13
137	01/04/1997	Bt1	1.57	0.30	0.32	0.35
138	01/04/1997	Bt2	1.62	0.65	0.68	0.70
139	01/04/1997	C2	1.82	1.15	1.17	1.20
140	01/04/1997	C2	1.83	1.30	1.33	1.35
141	01/04/1997	A1	1.30	0.00	0.03	0.05
142	01/04/1997	B1	1.47	0.05	0.08	0.10
143	01/04/1997	A1	1.24	0.00	0.03	0.05
144	01/04/1997	E	1.55	0.10	0.13	0.15
145	01/04/1997	Bt1	1.52	0.50	0.53	0.55
146	01/04/1997	A1	1.68	0.00	0.03	0.05
147	01/04/1997	A1	1.65	0.00	0.03	0.05
148	01/04/1997	E1	1.71	0.05	0.08	0.10
149	01/04/1997	E2	1.83	0.10	0.20	0.30

150	01/04/1997	E3	1.74	0.30	0.38	0.45
151	01/04/1997	Btg1	1.97	0.45	0.53	0.60
152	01/04/1997	Btg2	1.73	0.60	0.75	0.90
153	01/04/1997	Btg3	1.72	0.90	1.00	1.10
154	01/04/1997	Btg4	1.77	1.10	1.20	1.30
155	01/04/1997	Ap	1.21	0.00	0.03	0.05
156	01/04/1997	E1	1.51	0.20	0.23	0.25
157	01/04/1997	E2	1.69	0.40	0.43	0.45
158	01/04/1997	Bt1	1.50	0.70	0.72	0.75
159	01/04/1997	C	1.79	0.90	0.93	0.95
160	01/04/1997	A1	1.34	0.00	0.03	0.05
161	01/04/1997	A2	1.96	0.10	0.13	0.15
162	01/02/1994	A1p	1.33	0.00	0.05	0.10
163	01/02/1994	A1p	1.27	0.00	0.05	0.10
164	07/09/1994	A1p	1.31	0.01	0.06	0.11
165	07/09/1994	A12	1.37	0.10	0.15	0.20
166	07/09/1994	2A	1.34	0.20	0.25	0.30
167	07/09/1994	2A	1.63	0.30	0.35	0.40
168	07/09/1994	2E	1.76	0.40	0.45	0.50
169	07/09/1994	2B	1.56	0.64	0.70	0.75
170	07/09/1994	2C	1.83	0.86	0.91	0.96
171	07/09/1994	2C	1.76	0.97	1.02	1.07
172	07/09/1994	2C	1.84	1.06	1.11	1.16
173	07/09/1994	2C	1.68	1.15	1.20	1.25
174	07/09/1994	2C	1.92	1.50	1.55	1.60
175	07/09/1994	2C	1.53	1.65	1.70	1.75
176	01/01/1995	2E	1.70	0.49	0.54	0.59
177	01/02/1994	2E	1.69	0.49	0.54	0.59
178	01/02/1994	2E	1.60	0.49	0.54	0.59
179	15/02/1995	A1p	1.30	0.00	0.05	0.10
180	07/09/1996	2E	1.66	0.51	0.56	0.61
181	07/09/1996	2C	2.17	1.79	1.84	1.89
182	25/09/1996	A12	1.07	0.05	0.10	0.15
183	25/09/1996	2A	1.53	0.25	0.30	0.35
184	25/09/1996	2E	1.73	0.45	0.50	0.55
185	25/09/1996	2B	1.77	0.65	0.70	0.75
186	25/09/1996	2C	1.72	0.85	0.90	0.95
187	25/09/1996	2C	1.74	1.05	1.10	1.15
188	25/09/1996	2C	1.78	1.25	1.30	1.35
189	25/09/1996	2C	1.92	1.40	1.45	1.50
190	01/01/1998	2C	1.65	1.19	1.24	1.29
191	01/01/1998	2C	1.77	1.30	1.35	1.40
192	01/02/1994	Ap	1.23	0.00	0.05	0.10
193	01/02/1994	Ap	1.38	0.00	0.05	0.10
194	07/09/1994	Ap	1.25	0.01	0.06	0.11
195	07/09/1994	A12	1.78	0.25	0.30	0.35
196	07/09/1994	Bt2	1.77	0.38	0.42	0.47
197	07/09/1994	B3	1.76	0.50	0.55	0.60
198	07/09/1994	B3	1.75	0.60	0.65	0.70
199	07/09/1994	C	1.66	0.70	0.76	0.81
200	07/09/1994	C	1.80	0.80	0.85	0.90

201	07/09/1994	C	1.72	0.97	1.02	1.08
202	07/09/1994	C	1.66	1.03	1.09	1.14
203	07/09/1994	C	1.66	1.15	1.20	1.25
204	07/09/1994	C	1.73	1.24	1.29	1.35
205	07/09/1994	C	1.88	1.35	1.40	1.45
206	07/09/1994	C	2.20	1.46	1.51	1.56
207	07/09/1994	C	1.79	1.53	1.58	1.63
208	07/09/1994	C	1.59	1.61	1.66	1.71
209	07/09/1994	C	2.40	1.75	1.80	1.86
210	07/09/1994	C	1.69	1.85	1.90	1.95
211	01/02/1994	Bt2	1.92	0.41	0.46	0.51
212	01/02/1994	Bt2	1.82	0.45	0.50	0.55
213	15/02/1995	Ap	1.30	0.00	0.05	0.10
214	25/09/1996	Ap	1.55	0.05	0.10	0.15
215	25/09/1996	A12	1.73	0.25	0.30	0.35
216	25/09/1996	Bt2	1.69	0.45	0.50	0.55
217	25/09/1996	B3	1.73	0.65	0.70	0.75
218	25/09/1996	C	1.60	1.05	1.10	1.15
219	25/09/1996	C	1.67	1.25	1.30	1.35
220	25/09/1996	C	1.88	1.40	1.45	1.50
221	25/09/2006	C	1.75	0.85	0.90	0.95
222	01/02/1994	Ap1	1.38	0.00	0.05	0.10
223	01/02/1994	Ap1	1.38	0.00	0.05	0.10
224	01/02/1994	Bt2	1.83	0.55	0.60	0.65
225	01/02/1994	Bt2	1.84	0.55	0.60	0.65
226	15/02/1995	A1p	1.38	0.00	0.05	0.10
227	01/02/1994	A1p	1.14	0.00	0.05	0.10
228	01/02/1994	A1p	1.12	0.00	0.05	0.10
229	07/09/1994	A1p	1.61	0.05	0.10	0.15
230	07/09/1994	A12	1.68	0.15	0.20	0.25
231	07/09/1994	E	1.94	0.25	0.30	0.35
232	07/09/1994	Bt1	1.97	0.36	0.42	0.46
233	07/09/1994	Bt2	2.05	0.53	0.58	0.63
234	07/09/1994	Bt3	1.85	0.70	0.75	0.80
235	07/09/1994	BC	1.72	0.80	0.85	0.90
236	07/09/1994	BC	1.90	0.90	0.95	1.00
237	07/09/1994	C	1.75	1.04	1.09	1.14
238	07/09/1994	C	1.21	1.16	1.21	1.26
239	07/09/1994	C	1.76	1.22	1.28	1.33
240	07/09/1994	C	1.86	1.27	1.33	1.38
241	07/09/1994	C	1.61	1.33	1.38	1.43
242	07/09/1994	C	1.42	1.38	1.43	1.48
243	07/09/1994	C	1.64	1.52	1.57	1.62
244	07/09/1994	C	1.89	1.60	1.65	1.70
245	07/09/1994	C	1.66	1.62	1.67	1.72
246	25/09/1994	Bt3	1.81	0.65	0.70	0.75
247	01/02/1994	Bt3	1.73	0.75	0.80	0.85
248	01/02/1994	Bt3	1.57	0.75	0.80	0.85
249	15/02/1995	A1p	1.13	0.00	0.05	0.10
250	25/09/1996	A1p	1.49	0.05	0.10	0.15
251	25/09/1996	E1	1.67	0.25	0.30	0.35

252	25/09/1996	Bt1	1.76	0.45	0.50	0.55
253	25/09/1996	BC	2.94	0.80	0.85	0.90
254	01/02/1994	Ap	1.27	0.00	0.05	0.10
255	01/02/1994	Ap	1.09	0.00	0.05	0.10
256	01/01/1995	Ap	1.25	0.01	0.06	0.11
257	01/01/1995	B1	1.78	0.25	0.30	0.35
258	01/01/1995	Bt21	1.77	0.38	0.42	0.47
259	01/01/1995	Bt21	1.76	0.50	0.55	0.60
260	01/01/1995	Bt22	1.75	0.60	0.65	0.70
261	01/01/1995	Bt22	1.70	0.61	0.66	0.71
262	01/01/1995	Bt22	1.70	0.61	0.66	0.71
263	01/01/1995	Bt22	1.66	0.70	0.76	0.81
264	01/01/1995	BC	1.80	0.80	0.85	0.90
265	01/01/1995	C	1.72	0.97	1.02	1.08
266	01/01/1995	C	1.66	1.03	1.09	1.14
267	01/01/1995	C	1.66	1.15	1.20	1.25
268	01/01/1995	C	1.73	1.24	1.29	1.35
269	01/01/1995	C	1.88	1.35	1.40	1.45
270	01/01/1995	C	2.20	1.46	1.51	1.56
271	01/01/1995	C	1.79	1.53	1.58	1.63
272	01/01/1995	C	1.59	1.61	1.66	1.71
273	01/01/1995	C	2.40	1.75	1.80	1.86
274	01/01/1995	C	1.69	1.85	1.90	1.95
275	15/02/1995	Ap	1.18	0.00	0.05	0.10
276	01/02/1994	A1p	1.30	0.00	0.05	0.10
277	01/02/1994	A1p	1.41	0.00	0.05	0.10
278	07/09/1994	A12	1.93	0.23	0.28	0.33
279	07/09/1994	E	1.54	0.31	0.36	0.41
280	07/09/1994	Bt1	1.65	0.37	0.42	0.47
281	07/09/1994	Bt2	1.34	0.48	0.53	0.58
282	01/02/1994	E	1.73	0.25	0.30	0.35
283	01/02/1994	E	1.87	0.25	0.30	0.35
284	15/02/1995	A1p	1.35	0.00	0.05	0.10
285	07/09/1996	Bt2	2.60	0.58	0.63	0.68
286	07/09/1996	Bt2	1.91	0.64	0.69	0.74
287	07/09/1996	BC	2.19	0.76	0.81	0.86
288	07/09/1994	A	1.35	0.00	0.05	0.10
289	07/09/1994	A	1.65	0.10	0.15	0.20
290	07/09/1994	E	1.77	0.20	0.25	0.30
291	07/09/1994	E	1.87	0.30	0.35	0.40
292	07/09/1994	E	1.63	0.39	0.44	0.49
293	07/09/1994	B	1.63	0.50	0.55	0.60
294	07/09/1994	B	1.78	0.60	0.65	0.70
295	07/09/1994	B	1.68	0.70	0.75	0.80
296	07/09/1994	B	1.43	0.80	0.85	0.90
297	25/09/1996	A	1.43	0.05	0.10	0.15
298	25/09/1996	E	1.80	0.25	0.30	0.35
299	25/09/1996	E	1.75	0.45	0.50	0.55
300	25/09/1996	B	1.77	0.65	0.70	0.75
301	25/09/1996	B	1.82	0.85	0.90	0.95
302	01/01/1998	A	1.34	0.00	0.00	0.05

303	01/01/1998	A	1.23	0.00	0.00	0.05
304	01/01/1998	A	1.34	0.00	0.00	0.05
305	01/01/1998	A	1.29	0.00	0.00	0.05
306	01/01/1998	E	1.82	0.25	0.30	0.35
307	01/01/1998	E	1.91	0.27	0.32	0.37
308	01/01/1998	E	1.77	0.68	0.73	0.78
309	01/01/1998	B	1.63	1.10	1.15	1.20
310	01/01/1998	C	1.51	1.35	1.40	1.45
311	01/01/1998	C	1.56	1.37	1.42	1.47
312	01/01/1998	C	1.70	1.38	1.43	1.48
313	01/01/1998	C	1.61	1.41	1.46	1.51
314	01/04/1997	A1	1.78	0.05	0.10	0.15
315	01/04/1997	E	1.92	0.25	0.30	0.35
316	01/04/1997	B/C	1.96	0.45	0.50	0.55
317	01/04/1997	C	2.03	0.65	0.70	0.75
318	01/04/1997	R	2.10	0.85	0.90	0.95
319	01/04/1997	R	2.25	1.00	1.05	1.10
320	01/04/1997	E	1.68	0.05	0.10	0.15
321	01/04/1997	B2	1.77	0.25	0.30	0.35
322	01/04/1997	B2	1.21	0.45	0.50	0.55
323	01/04/1997	C	2.24	0.65	0.70	0.75
324	01/04/1997	C	2.81	0.76	0.82	0.86
325	01/04/1997	Ap	1.44	0.05	0.10	0.15
326	01/04/1997	E	1.52	0.25	0.30	0.35
327	01/04/1997	B21	2.02	0.45	0.50	0.55
328	01/04/1997	B21	1.98	0.65	0.70	0.75
329	01/04/1997	B22	1.96	0.83	0.88	0.93
330	01/04/1997	A	1.60	0.05	0.10	0.15
331	01/04/1997	E	1.60	0.25	0.30	0.35
332	01/04/1997	B21	1.71	0.45	0.50	0.55
333	01/04/1997	B22	1.87	0.65	0.70	0.75
334	01/04/1997	C	1.98	0.85	0.90	0.95
335	01/04/1997	C	1.82	1.05	1.10	1.15
336	01/04/1997	C	1.90	1.25	1.30	1.35
337	01/04/1997	C	2.02	1.40	1.45	1.50
338	01/04/1997	A	1.57	0.05	0.10	0.15
339	01/04/1997	B21	1.71	0.25	0.30	0.35
340	01/04/1997	B21	1.80	0.45	0.50	0.55
341	01/04/1997	C	1.84	0.65	0.70	0.75
342	01/04/1997	C	1.90	0.77	0.82	0.87
343	01/04/1997	A	1.40	0.05	0.10	0.15
344	01/04/1997	E	1.43	0.25	0.30	0.35
345	01/04/1997	B2	1.75	0.45	0.50	0.55
346	01/04/1997	C	1.77	0.65	0.70	0.75
347	01/04/1997	C	1.89	0.85	0.90	0.95
348	01/04/1997	C	1.81	1.05	1.10	1.15
349	01/04/1997	C	1.88	1.23	1.28	1.33
350	01/04/1997	A	1.59	0.05	0.10	0.15
351	01/04/1997	E	1.79	0.25	0.30	0.35
352	01/04/1997	B2	1.84	0.45	0.50	0.55
353	01/04/1997	C	1.82	0.65	0.70	0.75

354	01/04/1997	C	1.68	0.85	0.90	0.95
355	01/04/1997	C	1.87	1.04	1.10	1.15
356	01/04/1997	A	1.42	0.05	0.10	0.15
357	01/04/1997	E	1.70	0.25	0.30	0.35
358	01/04/1997	B2	1.55	0.45	0.50	0.55
359	01/04/1997	B/C	1.80	0.65	0.70	0.75
360	01/04/1997	C	1.89	0.85	0.90	0.95
361	01/04/1997	C	1.92	1.02	1.08	1.13
362	01/04/1997	A1	1.77	0.05	0.10	0.15
363	01/04/1997	E	1.96	0.25	0.30	0.35
364	01/04/1997	B21	2.14	0.45	0.50	0.55
368	07/09/1994		1.93	0.25	0.28	0.30
369	07/09/1994		1.54	0.33	0.36	0.39
370	07/09/1994		1.65	0.39	0.42	0.45
371	07/09/1994		1.34	0.50	0.53	0.55
372	07/09/1994		1.91	0.68	0.69	0.70
373	07/09/1994		2.19	0.80	0.81	0.82
374	01/01/1999	A	1.37	0.04	0.07	0.10
375	01/01/1999	A	1.30	0.04	0.07	0.10
376	01/01/1999	E	1.79	0.26	0.28	0.30
377	01/01/1999	E	1.89	0.26	0.28	0.30
378	01/01/1999	B	1.69	0.51	0.53	0.56
379	01/01/1999	B	1.77	0.51	0.53	0.56
380	01/01/1999	A	1.59	0.07	0.10	0.13
381	01/01/1999	A	1.48	0.07	0.10	0.13
382	01/01/1999	B	1.91	0.38	0.40	0.42
383	01/01/1999	B	1.79	0.38	0.40	0.42
384	01/01/1999	B/C	1.85	0.59	0.62	0.65
385	01/01/1999	B/C	1.85	0.59	0.62	0.65
386	01/01/1999	A	1.46	0.06	0.08	0.11
387	01/01/1999	A	1.53	0.06	0.08	0.11
388	01/01/1999	E	1.68	0.25	0.27	0.29
389	01/01/1999	E	1.70	0.25	0.27	0.29
390	01/01/1999	B	1.89	0.50	0.52	0.55
391	01/01/1999	B	1.84	0.50	0.52	0.55
392	01/02/1994	E2	2.12	0.57	0.60	0.63
393	01/02/1994	E2	2.01	0.57	0.60	0.63

Appendix 2

Broad Correlation between Australian Soil Classifications and the US Soil Taxonomy

From: Charman, P.E.V. & Murphy, B.W. (2000). *Soils: Their Properties and Management*. 2nd Edition. DLWC/Oxford University Press, Melbourne, Australia.

Great Soil Groups	AUSTRALIAN SOIL TAXONOMY Australian Soil Classification	US SOIL TAXONOMY Order	Suborder
Alluvial Soils	Rudosols Tenosols	Entisols	Fluvents
Lithosols	Rudosols Tenosols	Entisols Inceptisols Aridisols	Orthents Ochrepts Orthids Argids
Calcareous Sands	Rudosols	Entisols	Psamments
Siliceous Sands	Rudosols	Entisols	Psamments Orthents
Earthy Sands	Tenosols	Aridisols Entisols	Argids Psamments Orthents
Grey-brown and Red Calcareous Soils	Calcarosols	Aridisols Alfisols	Orthids Argids Xeralfs
Red and Brown Hardpan Soils	Kandosols	Aridisols	Orthids Argids
Rendzina	Calcarosols (Rudosols)	Mollisols Aridisols	Xerolls Ustoll Orthids
Black Earths	Vertisols	Vertisols	Usterts Torrepts
Grey, Brown and Red Clays	Vertisols	Vertisols	Usterts Xererts Torrepts
Chernozems and Prairie Soils	Dermosols	Mollisols Entisols Alfisols	Ustolls Xerolls Fluvents Ustalfs Xeralfs
Wiesenboden	Vertisols	Vertisols	Usterts Xererts
Solodic Soils (solonetz, solodised solonetz, solodic soloth)	Sodosols	Alfisols Aridisols	Xeralfs Ustalfs Argids
Solonised Brown Soils	Calcarosols	Aridisols Mollisols Alfisols	Orthids Argids Xeroll Xeralfs

Great Soil Groups	AUSTRALIAN SOIL TAXONOMY Australian Soil Classification	US SOIL TAXONOMY Order	Suborder
Desert Loams	Sodosols	Aridisols Alfisols	Argids Xeralfs
Red-Brown Earths (sodic)	Sodosols	Alfisols	Xeralfs Ustalfs
Red-Brown Earths (non-sodic)	Chromosols	Alfisols	Xeralfs Ustalfs
Non-calcic Brown Soils	Chromosols	Alfisols	Ustalfs Xeralfs
Chocolate Soils	Dermosols Chromosols	Mollisols	Xeroll Ustoll Ustalfs
Calcareous Red Earths	Kandosols	Aridisols	Argids Orthids
Red Earths	Kandosols	Alfisols Oxisols Aridisols Ultisols	Ustalfs Xeralfs Torrox Argids Ustults Udults
Yellow Earths	Kandosols	Alfisols Ultisols Aridisols	Ustalfs Xeralfs Ustults Argids
Terra Rossa Soils	Dermosols	Alfisols Inceptisols Entisols	Xeralfs Ustalfs Ochrepts Orthents
Euchrozems	Dermosols/Ferrosols	Alfisols Inceptisols	Ustalfs Ochrepts
Xanthozems	Dermosols	Oxisols Ultisols	Orthox Humox Humults Ustults
Krasnozems	Ferrosols	Oxisols Alfisols Ultisols	Humox Orthox Ustox Ustalfs Humults
Red Podzolic Soils	Chromosols (Dermosols, Kurosols)	Ultisols Alfisols	Ustults Udults Xerults Ustalfs Xeralfs

Great Soil Groups	AUSTRALIAN SOIL TAXONOMY Australian Soil Classification	US SOIL TAXONOMY Order	Suborder
Yellow Podzolic Soils	Chromosols (Dermosols, Kurosols)	Ultisols Alfisols	Ustults Ustalfs Xeralfs
Lateritic Podzolic Soils	Kurosols	Ultisols Alfisols	Ustults Xerults Xeralfs Ustalfs Aqualfs
Podzols	Podosols	Spodosols Entisols	Orthods Psamments
Alpine Hummus Soils	Tenosols	Inceptisols	Umbrepts Ochrepts
Humic Gleys and Gleyed Podzolic Soils	Hydrosols	Ultisols Alfisols Inceptisols	Aquults Aqualfs Aquepts

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