

**APPROACHES TO MODELLING CATCHMENT-SCALE
FOREST HYDROLOGY**

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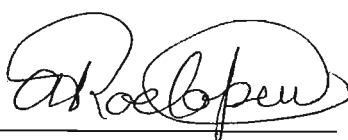
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DECLARATION

I hereby certify that these studies represent original work by the author for post-graduate degree purposes within the School of Bioresources Engineering and Environmental Hydrology, University of Natal, Pietermaritzburg from January 1999 to January 2002, under supervision of Dr. GPW Jewitt, and have not otherwise been submitted in any form or any degree or diploma to any University. Where use has been made of the work of others it is duly acknowledged in the text.

Signed: 

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ABSTRACT

South African commercial plantations occupy an estimated 1.5 million hectares of the country and as the demands for timber products increase, this area is expected to increase. However, further expansion is limited, not only by the suitability of land, but also by the pressures from other water users. As a result the need has arisen for simulation models that can aid decision-makers and planners in their evaluation of the water requirements of forestry versus competing land uses at different spatial scales.

Different models exist to perform such tasks and range from simple empirical models to more complex physically-based models. The policies of the National Water Act (1998) relating to forestry serve to highlight the requirements of a model used for the assessment of afforestation impacts and these are discussed in this document. There is a perception that physically-based distributed models are best suited for estimation of afforestation impacts on a catchment's water yield since their physical basis allows for extrapolation to different catchments without calibration. Furthermore, it is often stated that the model parameters have physical meaning and can therefore be estimated from measurable variables. In this regard, a review of physically-based modelling approaches and a comparison of two such hydrological models forms the main focus of this dissertation. The models evaluated were the South African *ACRU* model and the Australian topography-based Macaque model.

The primary objective of this research was to determine whether topography-based modelling (Macaque model) provides an improved simulation of water yield from forested catchments, particularly during the low flow period, compared to a physically-based model (*ACRU* model) that does not explicitly represent lateral sub-surface flow. A secondary objective was the evaluation of the suitability of these models for application in South Africa.

Through a comparison of the two models' structures, the application of the models on two South African catchments and an analysis of the simulation results obtained, an assessment of the different physically-based modelling approaches was made. The strengths and shortcomings of the two models were determined and the following conclusions were drawn regarding the suitability of these modelling approaches for applications on forested catchments in South Africa:

- The *ACRU* model structure was more suited to predictive modelling on operational catchments, whilst the more complex Macaque model's greatest limitation for application in South Africa was its high input requirements which could not be supported by the available data.
- Despite data limitations and uncertainty, the Macaque model's topography-based representation of runoff processes resulted in improved low flow simulations compared to the results from the *ACRU* simulations, indicating that there are benefits associated with a topographically-based modelling approach.
- The Macaque model's link to the Geographic Information System, Tarsier, provided an efficient means to configure the model, input spatial data and view output data. However, it was found that the *ACRU* model was more flexible in terms of being able to accurately represent the spatial and temporal variations of input parameters.

Based on these findings, recommendations for future research include the verification of internal processes of both the *ACRU* and Macaque models. This would require the combined measurement of both catchment streamflow and processes such as evapotranspiration. For the Macaque model to be verified more comprehensively and for its application in operational catchments it will be necessary to improve the representation of spatial and temporal changes in precipitation and vegetation parameters for South African conditions.

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LIST OF ABBREVIATIONS

APS	=	Afforestation Permit System
CCWR	=	Computing Centre for Water Research
CMA	=	Catchment Management Agencies
COAIM	=	Coefficient of Initial Abstraction
DEM	=	Digital Elevation Model
DSS	=	Decision Support System
DTA	=	Digital Terrain Analyst
DWAF	=	Department of Water Affairs and Forestry
E_m	=	Maximum Evaporation
E_r	=	Reference Potential Evaporation
E_t	=	Transpiration
ESU	=	Elementary Spatial Unit
F_t	=	Fraction of Evaporative Energy
$g_{cl, max}$	=	Maximum Leaf Conductance
GIS	=	Geographical Information System
I_l	=	Interception Loss
ISCW	=	Institute for Soil, Climate and Water
K_{cm}	=	Crop Coefficient
K_d	=	Daily Crop Coefficient
K_{depth}	=	Saturated Hydraulic Conductivity Depth
K_{min}	=	Minimum Saturated Hydraulic Conductivity
K_{sat}	=	Surface Saturated Hydraulic Conductivity
LAI	=	Leaf Area Index
LWP	=	Leaf Water Potential
MAP	=	Mean Annual Precipitation
MLR	=	Multiple Linear Regression
MMPI	=	Mean Monthly Precipitation Index
NWA	=	National Water Act
PEST	=	Model Independent Parameter Estimation
P_g	=	Gross Precipitation
REA	=	Representative Elementary Area

SAWB	=	South African Weather Bureau
$S_{\text{sat},0,\text{surface}}$	=	Zero Saturation Deficit
SFRA	=	Streamflow Reduction Activity
SIRI	=	Soil and Irrigation Research Institute
VPD	=	Vapour Pressure Deficit
WI	=	Wetness Index
θ_{max}	=	Maximum Volumetric Water Content

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1. INTRODUCTION

To alleviate the water shortages experienced in South African catchments where the demand for water often exceeds supply, particularly during periods of drought, the government has adopted a policy of implementing demand-side measures, which include the issuing of licenses to control water use by different water users such as commercial forestry.

In the past, commercial forest plantations have been targeted as being responsible for the drying up of streams. The major reason why afforestation resulted in the drying up of streams is because of phenology and dormancy differences between forest plantations and the indigenous grassland they frequently replaced. Furthermore, forestry was often situated in important water supply catchments which placed greater focus on their water use.

The area available for afforestation is found in the high rainfall areas (Mean Annual Precipitation greater than 650mm) of hilly or mountainous marginal agricultural land, which extends from the Limpopo Province, through Mpumalanga, KwaZulu-Natal, Eastern Cape to the south and south-western parts of the Western Cape Province. The total area utilised by commercial plantations was estimated as 1.5 million ha in 1998 (Scott *et al.*, 1998) and this area is expected to expand as the demand for timber products increases due to population growth and increasing per capita consumption as living standards improve (Dye, 1996). However, further expansion of plantations will be limited, not only by the suitability of land but also by pressures from other water users. The result is increasing conflict surrounding the allocation of available water to competing sectors and this continues to focus attention on forest water use in South Africa, in comparison with other land uses.

Considerable research has focussed on quantifying the impacts of afforestation on South Africa's water resources, primarily through the establishment of catchment experiments in a range of forestry regions. The results from these experiments have been evaluated extensively (Van Lill *et al.*, 1980; Bosch and Hewlett, 1982; Van Wyk, 1987; Smith, 1992; Dye, 1996; Scott and Smith, 1997; Scott *et al.*, 2000) and have been used to develop models that provide quantitative assessments of afforestation impacts on various components of the water regime. Impact assessments have progressed to include the impacts of forestry during low flow periods, since the impact during the dry season has the most pronounced effect on

downstream water users and the environmental requirements of the rivers that now need to be satisfied according to the South African National Water Act of 1998.

Furthermore, the interest in quantifying the impacts of forestry on streamflow is extending to include the evaluation of other types of agricultural land uses. The increased competition between land users for a share of the water resource and the goal of optimal utilisation of the water resource has predetermined the need for reliable models of land use hydrology. It is essential that such models are verified so that they can be used with the desired level of confidence.

In this document, aspects of the National Water Act (1998) are discussed in order to present the fundamentals of how South Africa's water resources are likely to be managed and what type of models water resource managers and planners require to fulfil this task. Currently the Scott and Smith curves (1997) are used to quantify the impacts of afforestation on streamflow. These curves were developed from the results of five paired catchment experiments in four different afforestation regions, but have since been extrapolated with the aid of the *ACRU* model to a wider range of afforestation regions (Gush *et al.*, 2001). However, a need has been identified for more physically-based, spatially distributed models for site specific applications as described in Chapter 2. Different process based models, varying in complexity, have been developed based on process studies undertaken in forested catchments and these studies and modelling approaches are reviewed in Chapter 3. Several models have previously been reviewed in the context of forest hydrological modelling at an operational catchment scale, both in South Africa (cf Gørgens and Lee, 1992) and in Australia (cf Watson, 1999a). The South African *ACRU* model has been identified as being appropriate for modelling forestry related land use change for South Africa. However, shortcomings in the model require that improvements or alternatives be sought. The model theory and shortcomings of the *ACRU* model are discussed in Chapter 4. The Macaque model has recently been developed for Australian forest hydrology applications (Watson, 1999a) and seems to provide an alternative to the *ACRU* model or at least some aspects that may be useful in addressing the shortcomings of the *ACRU* model as is discussed in Chapter 5.

The Macaque model is a topography-based model that uses a Digital Elevation Model (DEM) to drive lateral sub-surface water movement downslope and control baseflow release and saturation excess flow. Conversely, the *ACRU* model uses a more conceptual approach to represent the runoff generation dynamics in forested catchments. It was therefore anticipated

that the Macaque model's more physically-based representation of lateral sub-surface flow could result in improved low flows simulations in forested catchments.

The two main objectives of this dissertation were to:

- Evaluate if these modelling approaches were suitable for application in South Africa.
- Determine whether topography-based modelling provides an improved simulation of water yield from forested catchments, particularly during the low flow period compared to a model that does not explicitly represent the lateral sub-surface flow.

During the course of the research reported herein a number of problems were identified when the Macaque model was applied to South African catchments and these are discussed in detail in Chapter 6. It became evident that the limited range of hydrological measurement techniques and limited range of measurements in time and space create an impediment to the development of models, and their application and verification in South Africa and elsewhere in the world.

2. FORESTRY AND THE SOUTH AFRICAN NATIONAL WATER ACT

In the past commercial forest plantations have been targeted as being responsible for streamflow reductions. As a result strict measures were set in place to control the expansion of afforestation and research was initiated into the impacts of afforestation on catchment water yield. In the following section a brief overview of the history of legislative control over commercial forestry in South Africa is provided.

2.1 Review of Changes in the Water Law

South African perceptions of tree water use have changed over the years and consequently the water law has changed. In the early 1900s it was believed that afforestation led to increased water yields and forestry was therefore considered to contribute substantially to the maintenance of water supplies (Bands, 1989). This historical association between forestry and water resource conservation is reflected in early South African forestry legislation, which provided for protection of the mountain catchment areas and encouraged afforestation in these areas (Bands, 1989).

This perception began to change after the First World War. The increased demand for timber resulted in the initiation of an extensive afforestation programme and evidence to suggest that the drying up of streams could be associated with this expansion arose. This prompted the then Departments of Forestry and Agriculture to initiate catchment experiments in the mid-1930's in order to research the effects of afforestation on streamflow (Bands, 1989). By 1935 the Forestry Department had established the Jonkershoek research catchment (Western Cape) and a few years later the Cathedral Peak research catchment (KwaZulu-Natal) was founded, followed by the planting of the Mokobulaan research catchments (Mpumalanga) in 1955. In 1975 streamflow monitoring was initiated at Witklip (Mpumalanga), followed by the establishment of controlled multiple catchment experiment at Westfalia (Limpopo Province). The fact that forest plantations caused reductions in streamflow was anticipated by the Forestry Department since in 1932 a policy was adopted to protect the natural vegetation up to approximately 20 metres on either side of streams and around other surface waters within its plantations (Bands, 1989).

Following the well-known drought of the mid-1960s and consequent increased conflict between water users, committees were established in 1966 to investigate the link between forest plantations and water supply (Van der Zel, 1995). In 1972 the findings of these committees provided the basis for the amendment of the 1968 Forest Act to introduce statutory control over all afforestation, in order to curtail large scale tree planting in areas where maintenance of streamflow was of regional or national importance (Bands, 1989) by way of the Afforestation Permit System (APS). According to this legislation, afforestation on a land segment could only take place once a permit was issued. The country's catchments were classified in terms of water supply importance and a permit was issued according to whether the planned expansion in afforestation within the catchment would result in greater than permitted reduction in pre-1973 average streamflow (Versfeld, 1987). The predicted reduction in streamflow caused by afforestation was derived using the paired catchment experiment approach. The paired catchment experiment approach is based on the assumption that the relationship between streamflow of the two physiographically similar catchments (control and treatment catchments) will remain the same provided that the vegetation of these catchments remains the same or changes in a similar fashion (Scott and Smith, 1997). The streamflow prior to the afforestation of the treatment catchment is referred to as the calibration period (Scott and Smith, 1997). For this period a relationship between the 'treatment and 'control' catchment is derived by regressing the monthly streamflow total of the treatment catchment against those of the control catchment (Scott and Smith, 1997). The impact of afforestation is measured by the difference between recorded streamflow from the treatment catchment and that from the derived calibration relationship for the treated catchment (Scott and Smith, 1992).

The earliest predicted reductions in streamflow caused by afforestation, which were derived using the paired catchment experiment approach, were known as the Nänni curves (Nänni, 1970). The curves were derived from observations of changes in streamflow following the afforestation of grassland with pines at the Cathedral Peak research catchment. They represent an empirical relationship between rainfall and runoff from KwaZulu-Natal upland grassland and the estimated decrease in runoff that can be expected due to afforestation of a grassland area (Nänni, 1970). These curves were later improved by Van der Zel (Van der Zel, 1982) and became known as the Van der Zel curves. The Van der Zel curves' major improvement on the Nänni curves was that they were developed from a larger number of catchment experiments' results, therefore making them more representative than the Nänni curves. A

major criticism of the Van der Zel curves is that they did not account for climatic differences, tree species or forest management regimes. Despite these limitations the curves provided consistent and conservative overestimates of reductions in streamflow compared to predictions obtained from an extensive world survey of forest hydrological research results (Bosch and Hewlett, 1982). For this reason the curves were considered adequate for the use in the APS and were used for more than two decades to control afforestation expansion (Van der Zel, 1995).

The use of the APS has always been a contentious issue and the forest industry has argued that the application of unilateral restriction may favour other less efficient and often less profitable water users (Versfeld, 1996). The forest industry argue that returns from forestry are often good in terms of income generated, foreign exchange earned, and jobs created and supported, when weighted against cubic metres of water consumed (Versfeld, 1996). The APS has since been replaced by a new licensing procedure, described later in this Chapter, and permit allocation criteria changes have been instituted with the aim of developing an improved policy for the system. This included the replacement of the Van der Zel curves (1982) with the Scott-Smith statistical curves to estimate streamflow reduction. Since the Scott-Smith curves formed the most recent basis for permit allocation, is it necessary to examine them more closely. In the following section their advantages and shortfalls are discussed.

2.2 The Scott and Smith Curves

The Scott and Smith curves are an improvement on the Van der Zel curves, in the sense that they were developed from a more comprehensive dataset. They represent afforestation impacts for a wide geographical range of South African forestry, for different growth potentials, for different species and for estimating impacts on total flow regimes, as well as, low flow regimes (Scott and LeMaitre, 1993). The significance of low flow has become evident in recent years as a result of the growing attention paid to the environmental requirements of rivers (Smakhtin and Watkins, 1997). Provision has now been made for such requirements in the National Water Act (NWA). In addition, it is during low flow periods that reliable water supply is also crucial to downstream water users and water pollution becomes most critical (Smakhtin and Watkins, 1997). Studies to determine the effects of afforestation on low flow have been reported by Smith and Scott (1992) and these results were re-analysed by

Scott and Lesch (1995), with the additional aim of determining if flow reductions were relatively greater in drought years. Scott and Lesch (1995) showed that the percentage reduction of total flows and that of low flows did not differ significantly and that reductions were not significantly greater in drought years (Scott and Lesch, 1995). However, it is the absolute volume, and not a percentage reduction of water that is of relevance to downstream users during the dry season.

The Scott and Smith curves, which are shown in Figure 2.1 and Figure 2.2, were developed from the results of paired catchment experiments in different parts of the country. The catchments selected are part of ongoing research experiments regarding impacts of afforestation practices on streamflow and include Mokobulaan (Mpumalanga Province escarpment), Westfalia (Limpopo Province), Cathedral Peak (KwaZulu-Natal) and Jonkershoek (Western Cape). These regions include optimal sites and sub-optimal sites for forestry, which describes the growth zone optimality and the water availability (related to soil depth and rainfall) of the site (Scott *et al.*, 1998).

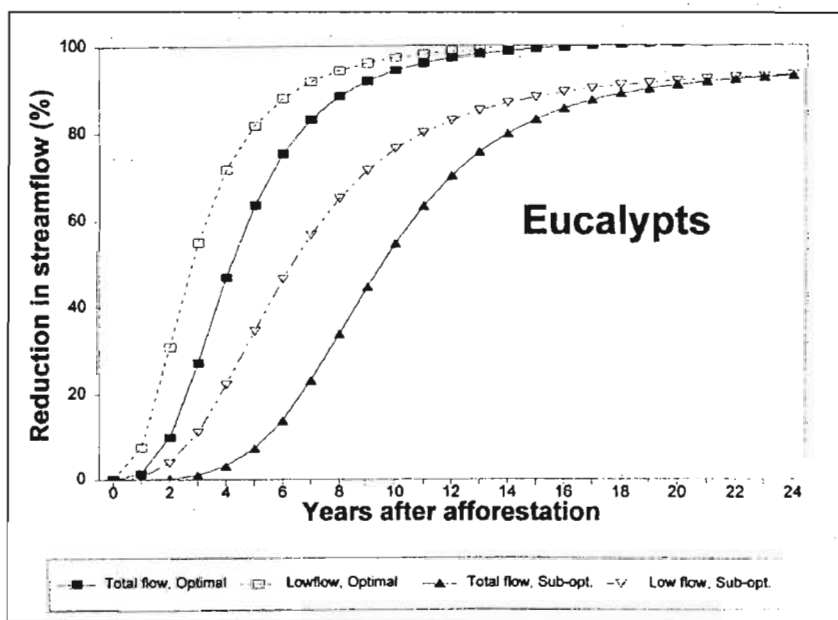


Figure 2.1. Scott and Smith curves after 100% afforestation with eucalypts (Scott and Smith, 1997).

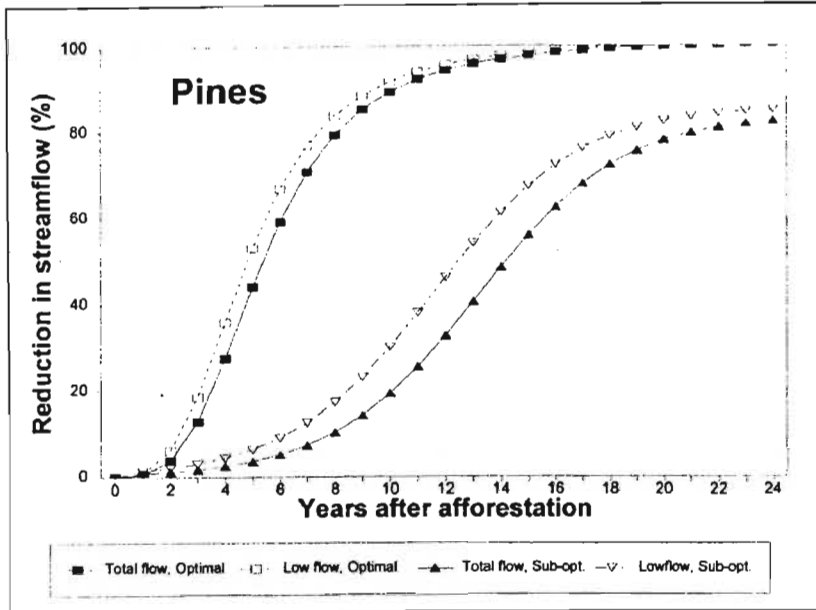


Figure 2.2. Scott and Smith curves after 100% afforestation with pines (Scott and Smith, 1997).

Whilst the curves are an improvement on the Van der Zel curves (1982), they have been developed with the objective of being simple to use, and as a result it was necessary to generalize the findings as far as possible (Scott and Smith, 1997). The inferences and assumptions made have been well documented by the model developers (Scott and Le Maitre, 1993; Smith and Scott, 1992 and Scott and Smith, 1997). They are as follows:

- A simple linear relationship is assumed between percentage of catchment afforested and the related reduction in streamflow at maturity. However, afforestation experiments indicate that the relationship may not simply be linear, but that the afforested part of the catchment may be able to exploit a greater proportion of a single catchment than it physically occupies (Van Wyk, 1987).
- Since no data were available for eucalypts growing under sub-optimal conditions it was necessary to synthesise curves for these conditions. The time delay observed between pines under sub-optimal as opposed to optimal conditions to reach the same flow reductions, was used to determine the flow reduction curves for eucalypts under sub-optimal conditions.
- Catchment results of Jonkershoek and Cathedral Peak were averaged according to the similarity in results, regardless of climatic or species distinctions.

- At Mokobulaan the riparian areas of the catchment were planted. This is no longer standard practice and therefore the curves may over-estimate afforestation impacts for this region.
- No site preparation or fertilisation took place in any of the treatment catchments, however, current practice is that soils are cultivated and fertilisers applied with the intent of stimulating rapid initial growth. These management practices enhance tree water use with the result that the curves may under-estimate flow reductions under modern establishment practices.
- For sawlog production, thinning takes place at certain intervals. Thinning, however, has not been considered and this may lead to over-estimates of tree water use.
- Subsequent re-analysis of the experimental results, indicate that flow reductions diminish towards the end of longer timber rotation. This finding is not reflected by the present curves (Scott *et al.*, 1999).
- These catchment experiments are all situated in areas with Mean Annual Precipitation (MAP) of greater than 1100mm and the curves should therefore not be extrapolated for use in marginal, drier forestry areas, which constitute the majority of all afforested land (Görgens *et al.*, 1999).
- The curves only consider two types of site suitability conditions, whereas in reality a wide range of site suitability conditions exist.
- The models remain empirical and their use in zones outside research areas where they were developed will constitute extrapolation.

Regardless of these assumptions the curves are easy to use and have helped to smooth the process of forest permit application by providing consistent, unambiguous and unbiased answers. It has been suggested that statistical methods such as the Scott and Smith curves will probably remain the backbone of the decision making process in catchment management (Vertessy *et al.*, 1993).

For site-specific applications in forested catchments, however, such curves remain empirical and cannot describe the interplay between multiple catchment attributes and processes which is essential for more specific application (Vertessy *et al.*, 1998). Furthermore, the curves do not consider streamflow reduction estimates for crops other than plantation trees. With a new approach to catchment management and water allocation outlined in the National Water Act

and the increasing competition of the water resource, the impact of Streamflow Reduction Activities (SFRAs) can no longer be viewed in isolation. In outlining the National Water Act (1998) and the policy for licensing SFRAs (DWAF, 1999), the application and limitations of the curves will become evident and provide an indication of what type of model is required for site-specific applications in forest and other landuse management. The model requirements under the National Water Act (1998) are highlighted in the following section.

2.3 Model Requirements under the National Water Act (1998)

A predicted inadequacy of available water resources to meet demands by 2030 has resulted in the movement from a supply-side approach to address the scarcity problem, to that of an integrated approach, which includes “demand-side” measures as one solution to the management of South Africa’s water resources. This approach to management of the water resources is reflected in the National Water Act of 1998 (NWA) which identifies and defines eleven different types of water use, including SFRAs, which will form the basis for the future allocation of water amongst users, once the requirements of the Reserve have been satisfied (DWAF, 1999b). The Reserve consists of two parts; the basic human needs reserve that provides for the essential needs of individuals and the ecological reserve, which relates to the water required for protection of the aquatic ecosystems of the water resource (DWAF, 1999b).

A streamflow reducing activity is defined as any activity, including the cultivation of any particular crop or other vegetation, that is likely to reduce the availability of water in a watercourse to the Reserve, to meet international obligations, or to other water users, significantly (NWA, 1998). The magnitude of reduction that can be considered significant has not as yet been defined, but this will differ depending on whether a catchment is under stress in terms of availability of water. By declaring a SFRA, allowance is made for the regulation of such land-based activities which reduce streamflow and enables the responsible authority to set or prescribe a method for determining the extent of the streamflow reduction caused by the activity (NWA, 1998).

Presently only commercial plantation forestry is identified as an SFRA (DWAF, 1999c) and is controlled by the licensing procedure that came into effect in October 1999. This form of water-use licensing will serve as a basis for the development of a comprehensive approach for the licensing of all SFRAs (DWAF, 1999c) and will be extended to include other water user

groups. In catchments where the demand for the water resource exceeds water availability, *all* users will apply for licenses and the allocation of the resource will be on the basis of the applications received and will be allocated according to a defined set of principles (DWAF, 1999c).

2.3.1 Comparisons with other landuses

Where previously forestry was singled out and the APS focussed on forestry and water use alone, without considering the economic return in terms of water or other resources related to forestry, the new water law requires that licenses for SFRAs will be evaluated by comparison with the available alternative landuses for that particular property. These evaluations will be based upon the relative benefit offered per unit streamflow reduction for the given application (DWAF, 1999c). Therefore the impact of afforestation will no longer be assessed solely relative to the impact of the natural vegetation. This approach suggests the need for models that can accurately predict the water use of different types of landuses, not just forestry. The equitable and efficient allocation of water to SFRAs will depend on whether the calculations of their likely effects on the allocatable water, and on the water resource, are accurate and acceptable (DWAF, 1999c).

2.3.2 Environmental requirements

Satisfying the requirements of the Reserve will inevitably place focus on the impact of forestry on streamflow during low flows periods. Since forestry expansion reduces low flows it is necessary that models are able to simulate accurately the impacts of forestry on all aspects of the flow regime, in order to establish whether the requirements of the Reserve can be met during the dry season of the year and in drought periods. The viability of afforestation in a catchment is currently assessed by DWAF using the concept of Mean Annual Low Flow which is defined as the mean of the driest 25% of monthly flows in a standard 70 year (1920-1990) simulated flow record (Smakhtin *et al.*, 1998). There does not, however, exist a clear definition of a low flow condition and the term has a different meaning for different groups of scientists and managers (Smakhtin *et al.*, 1998).

Where previously water resource assessments were made using monthly streamflow time series, the increasing importance of water quality and ecological considerations suggests a

need for daily flow information. It is preferential to obtain data at the daily timestep so as to obtain a complete representative time series from which a variety of streamflow characteristics describing different aspects of the flow regime of a river can be abstracted (Smakhtin *et al.*, 1998). The primary source of daily streamflow information is observed data. However, these data are often of inadequate quality and the availability differs in different parts of South Africa. This inevitably leads to a requirement for simulated daily time series. This method is the only feasible option at present for estimating daily runoff regimes for different scenarios of development. The ability of a daily model to reproduce specific aspects of the low-flow regime may be of critical importance for ecological or water quality purposes (Smakhtin and Watkins, 1997).

2.3.3 Scale considerations

A further issue to consider under the NWA is that of spatial scale. Currently the agreed scale for the assessment of forestry impacts is that of a quaternary catchment (Van der Zel, 1995). However, the NWA proposes the division of the country into approximately eighteen water management areas (Forest Industries Association, 1998). Each area will have a Catchment Management Agency (CMA), which is the responsible authority in a water management area and is responsible for issuing of licenses (Forest Industries Association, 1998). This implies that future management of water resources will be developed at local levels. Therefore, it will be necessary to simulate the effects of forestry-related landuse changes on runoff at scales varying between those of headwater hillslopes and developed drainage basins (Görgens and Lee, 1992), since the impacts of forestry on water resources may be significant at the regional scale, local scale or both (Versfeld, 1996).

In addition to the requirements of a model in terms of the NWA, water use models also have to account for changes in forestry management practices, for example the recent shift to short rotation forestry for pulp timber production (Versfeld, 1996). These differences in the rotation age may have an effect on the long-term average annual water use by the plantation and therefore need to be considered (Bosch and Gradow, 1990).

2.4 Critical Needs of the NWA

Schulze *et al.* (1997) and Watson (1999) highlight important questions relating to the ongoing hydrological debate regarding the extent to which upstream afforestation is detrimental to downstream water users and which water resource managers will have to address. These include amongst others:

- What are the hydrological response changes over the long term, in critically dry years and the critical low flow months?
- What are the hydrological response changes at various scales ranging from plantation scale to sub-quaternary level and linked quaternary catchment scale?
- To what extent do commercial forests use more water than other dryland crops?
- To what extent do management practices such as thinning, the timing of thinnings and degree of thinning have an influence on streamflow volume?
- How do the impacts of afforestation differ in different areas?
- Where within the catchment does forestry have the greatest impact?
- What is the impact of successive rotations on the streamflow? For first rotations there could be an accumulated store of water from the previous landuse, however after forestry plantations have been established this store will no longer be available for successive rotations. Furthermore, there could be a lag in streamflow recovery after clearfelling resulting from the replenishment of the soil water store (Scott and Lesch, 1997).

Under the NWA these questions need to be answered by water resource managers. The NWA, however, does not specify which methods should be used for determining afforestation impacts and the choice of model is likely to depend on the problem the CMAs are faced with and the resources available. For example, in situations where conflict arises between water users, models that can be applied to site-specific applications and which the stakeholders accept as providing accurate results will be required.

Physically-based models are considered to be suitable for site-specific applications and are widely used for predictive purposes and scenario modelling. Their physical basis allows for extrapolation to different catchments without calibration and model parameters have physical meaning and can be estimated from measurable variables. It is argued therefore that they can

be used with more confidence and are considered to be accurate outside the conditions for which they are developed and verified.

* * *

In Chapter 2 the changes in legislative control over afforestation expansion were reviewed and the methods used to estimate afforestation impacts on streamflow under the APS were described. The review of NWA of 1998 provided a means of highlighting the requirements of models that would aid water resource managers in the optimal allocation of South Africa's limited water resources. In the following Chapter the progress in forest process research is reviewed and the approaches to modelling catchment-scale forest hydrology are discussed.

3. HYDROLOGICAL PROCESSES DOMINANT IN FORESTED CATCHMENTS AND THEIR REPRESENTATION IN SIMULATION MODELS

It is evident from a comparison of pre- and post afforestation runoff records that the type of landcover has an enormous influence on the water yield and streamflow characteristics of a catchment (Hibbert, 1967; Bosch and Hewlett, 1982). The hydrological recovery of catchment from disturbances such a logging and afforestation can change the magnitude and distribution of flow paths as the canopy regenerates. The most significant change after afforestation is the decrease in “dry weather” streamflow (delayed flow) arising from increases in total evapotranspiration (Bonell, 1998). Total evapotranspiration consists essentially of three different processes, namely, interception losses, transpiration and evaporation from the soil surface (usually of less significant as a result of a litter layer).

3.1 Evapotranspiration

There are a number of factors responsible for the greater magnitude of evapotranspiration losses from forests than from grasslands or crops. There is increasing evidence to suggest that rainfall interception and evaporation of intercepted water is a major cause of observed differences in water yield of afforested and grassed catchments (Calder, 1976; Everson *et al.*, 1998). Harding *et al.* (1992) presented results showing interception losses for forests to be in the region of 40% of annual rainfall. Trees present rough surfaces to the wind and there is a greater degree of turbulent mixing within the forest canopy. As this is the primary mechanism for water vapour transport from the leaf surface to the atmosphere it leads to aerodynamic resistances an order of magnitude less than for shorter crops (Cain, 1998), resulting in evaporation from the canopy at rates in excess of reference potential evaporation (Calder, 1982). Interception losses in forests are, however, specific to their environment and near Sabie in Mpumalanga the interception component of evapotranspiration for young eucalypts was found to comprise only 4% of rainfall (Dye and Versfeld, 1991). Dye (1996) also found canopy interception losses in pine stands in Mpumalanga to be less than those reported in temperate forests, which reflect the less frequent, more intense rainfall characteristic of summer

rainfall regions in South Africa. Dye (1996) concluded that transpiration from dry canopies was the dominant evaporation process in South African plantations.

Trees also maintain evergreen canopies during the winter season when grasslands are dormant, and therefore continue to transpire (Dye and Bosch, 2000) resulting in more pronounced streamflow reduction during the low flow periods. The relatively deep roots of trees also enable trees to extract soil water from very deep in the soil profile (Dye *et al.*, 1997).

Although it is evident that water yield reductions are observed after forest regrowth, the response is unlikely to be linearly proportional to the extent of land cover change (Watson, 1999a). In the ensuing section the typical catchment responses to landcover change are discussed.

3.2 Runoff generation in forests and spatial variations in soil moisture

In many forested catchments, particularly those displaying marked topographical differences, there are distinct variations in soil water content, with wettest soils in the riparian zones and driest along the ridge tops or localised areas of shallow soil (Dye and Bosch, 2000). This is caused by redistribution of soil water in response to gravity, and significantly alters the soil water available to trees growing in different positions within the catchment (Dye and Bosch, 2000).

In 1965, at the International Forest Symposium, Hewlett and Hibbert (1967) put forward the so-called “variable source area” concept of runoff generation based on earlier research by Hewlett (1961), cited by Bonell (1998). This was based on the notion that infiltration is seldom a limiting factor in forested environments and that sub-surface flow rather than overland flow was capable of making a significant contribution to the storm hydrograph (Bonell, 1993). Typically, forest soils have a high infiltration capacity due to the presence of above ground vegetation and litter, which protect the soil from compaction caused by raindrop impact and below ground vegetation, which provides roots that keep the soil porous and highly permeable (Hornberger *et al.*, 1997). The soil is therefore capable of allowing essentially all of the incident precipitation to infiltrate. In such catchments overland flow develops not because of precipitation intensity (infiltration-excess overland flow or Horton type flow) but due to runoff

from temporary saturated areas (variable source areas) and is referred to as saturation excess flow (Hornberger *et al.* 1997). As a result of temporary saturation, trees in these areas have greater access to soil water, with the result that they grow faster than trees further upslope.

Hewlett's conceptual model (1961) also acknowledged the role of topography in determining the location of variable source areas through the downslope movement of water (Bonell, 1993). Stormflow moves downslope either above a subsoil-impeding layer or along the bedrock interface (Bonell, 1993). As part of the conceptual model of variable source areas of runoff generation, early hydrometric studies established that sub-surface stormflow was an important delivery mechanism to the storm hydrograph and in selected humid temperate forests it was the principle flow-path (Bonell, 1998). Hewlett's hypothesis initiated a considerable number of field experiments with a strong bias towards the humid temperate environments of the United States and Western Europe (Bonell, 1993).

It was initially believed, if the variable source area concept was sound, that for the storm hydrograph to be so responsive, the contributing sources had to be in the form of current rain or what was known as "event water". However, concurrent environmental tracer studies based on the two-component mass balance model for hydrograph separation, revealed the contrary i.e. that the storm hydrograph consisted of mostly pre-event water or previously stored catchment water. There has, however, been concern over the implicit assumptions on which these mass balance models are based and a number of these assumptions have been challenged (Bonell, 1998). For example the assumption that current rain (event) and groundwater (pre-event) were the "new" and "old" water contributing reservoirs, respectively and therefore soil water contributions were not accounted for. A further assumption was that the spatial variability of the isotopic signature of rainfall had not been considered, which may be acceptable at the hillslope scale but is less appropriate at larger scales (Bonell, 1998). While these concerns are legitimate, it is clear from the isotopic signatures that substantial portions of the storm hydrograph consist of pre-event water, at least within the framework of antecedent moisture conditions and prevailing low rainfall intensities experienced in the humid temperate latitudes (Bonell, 1998; Rice and Hornberger, 1998).

To accept the results from the hydrochemical studies it was necessary to establish what mechanisms could be responsible for allowing such amounts of pre-event water to be rapidly

released during a storm event. A frequent explanation is that of the “groundwater ridging mechanism”. This mechanism is considered to occur in situations where a capillary fringe or tension saturated zone above the water table is close to the surface. During rain only a small amount of infiltration is necessary to convert the small negative matric potentials to positive pressures, which causes the water table to rapidly rise (Bonell, 1993). However, this concept has increasingly come under challenge, as it has not been observed in the near-stream zone in several studies (Bonell, 1998). Another suggested mechanism is that of displacement (or translatory flow), whereby pre-event water is displaced laterally by new precipitation inputs. This involves the displacement of soil matrix water, which many researchers consider to be too slow unless combined with another mechanism. Nevertheless, studies in New Zealand showed increases in residence times of soil water downslope, supporting the displacement theory (Stewart and McDonnell, 1991). Other researchers have favoured macropores as the principle source of new water. Bonell *et al.* (1998), cited in Bonell (1998) presented field evidence to show that macropore flow gains more importance with increasing rainfall intensity.

What becomes evident from a review of hydrometric and hydrochemical studies of runoff generation is that;

- Such studies have been concentrated in the temperate forests of the northern latitudes with few examples from the tropics and subtropics.
- There are ranges of theories and conflicting conclusions drawn from studies, sometimes even within the same catchments.

In South Africa research of the impacts of afforestation on a catchment’s discharge has, up until recently, been confined to the paired catchment experiments (Scott *et al.*, 1999), described in Chapter 2, and the measurement of forest evaporation (Burger *et al.*, 1999). Only a limited amount of literature is available relating to the spatial variability in soil moisture and runoff delivery mechanisms in afforested catchments in South Africa. Reported studies include the hillslope process research in the Weatherly catchment in the North Eastern Cape (Hickson, 2001) and the research site at Seven Oaks in KwaZulu-Natal which aims to quantify the influence of stands of *Acacia mearnsii* growing at different distances from the stream channel on soil water storage and streamflow.

It is through runoff process studies, both hydrometric and hydrochemical, that attempts are made to gain a better understanding of runoff generating mechanisms. This in turn provides information for the development of models for prediction of land cover change. Since the 1980s increasing attention has been devoted to the development and testing of physically-based models to create a means of extrapolating from available measurements in time and space, to ungauged catchments and into the future to assess the impact of future hydrological change (Beven, 2000). In the ensuing section physically-based models are discussed and some of the problems encountered in their development and application are described.

3.3 Physically-based distributed modelling approach in forest applications

Spatial and temporal variations in catchment characteristics make it difficult to extrapolate well measured hydrological responses to vegetation change to sites where no catchment experiments have been performed (Vertessy *et al.*, 1993). Furthermore, uncertainty arises because catchment treatments, such as afforested area and type of species, vary between these experiments (Vertessy *et al.*, 1993). It is also unlikely that different sites will be affected in the same way by afforestation (Jewitt, 1991) and it has been shown, for example, that water yield decreases more rapidly in warmer climates than in cooler climates as a result of different rates of tree growth and canopy development. The lack of generic information has resulted in a growing demand for computer-based hydrological modelling to predict the impacts of afforestation and other land uses on the water resources in different catchments throughout South Africa.

It was believed that physically-based models were more suitable to address the practical issues of prediction. Such models utilised algorithms and variables that were believed to have physical meaning and required shorter periods of calibration and were therefore considered an improvement over statistically based models or empirical models (Smith *et al.*, 1994; Bonell, 1993; Refsgaard *et al.*, 1996). The explicit representation of physical processes and the consideration given to the spatial variations in catchment characteristics (Vertessy *et al.*, 1993) made it possible for such physically-based distributed models to describe the interplay between multiple catchment attributes and processes, which is considered essential in assessing the hydrological impacts of land-use change hydrology (Vertessy *et al.*, 1993). Physically-based, distributed models have also overcome the limitations of lumped-parameter conceptual models,

such as the need for calibration (Görgens and Lee, 1992), and can be extended to application in ungauged catchments where different catchments treatments apply.

However, in the past decade there has been considerable debate over the accuracy of physically-based models and whether these models produce better results than lumped parameters models in practice, particularly when they are used for predictive purposes (Beven, 1989; Grayson *et al.*, 1992; Beven, 1996a; Beven, 1996b). This debate is based on two major concerns, namely:

- The process description may not be valid since the model equations often do not apply at the appropriate scale.
- The input parameters are often not available or have been determined at the incorrect scale. The spatial heterogeneity of catchment characteristics is therefore not adequately represented in the model and the models are consequently applied as spatially lumped models.

In the following sections the concept and theory of physically-based modelling is discussed and consideration is given to the challenges and limitations of applying physically-based models for land use change applications in South Africa.

3.4 Model theory

Physically-based models attempt to account for all processes, storages, feedbacks and feedforwards in the hydrological system (Schulze, 1998). The models attempt to describe the behaviour of the hydrological cycle in terms of mathematical relationships which outline the interactions and linkages of the various components of the spatially and temporarily varying hydrological system with individual submodels representing each particular hydrological process (Schulze, 1998). Proponents of physically-based models believe that the use of equations, which are theoretically correct at a certain scale, and physically measurable input parameters should enable a universal application, so that the resulting model is of far greater use than a model requiring field calibration of empirical parameters (Grayson *et al.*, 1992).

However, the extrapolation of the understanding of hydrological processes based on small scale or point scale observations to larger scales may not be valid, as different physical laws will dominate at different scales (Watson, 1999a). Large-scale spatial models attempt to predict responses at larger scales than the observation scale and must overcome this scale discrepancy. In many hydrological models, an attempt to quantify hydrological variability that occurs at a range of scales is made by subdividing the catchment into smaller units (Jewitt and Görgens, 2000). Commonly these subdivisions are referred to as subcatchments (Schulze, 1995a) or Hydrological Response Units (Flügel, 1995) or Elementary Spatial Units (ESUs) (Watson, 1999a). The overall catchment response is therefore represented by the combined effect of the constituent process representations, and the spatial variability of a catchment can be represented by the distributed values of the model parameters at the model scale (Grayson *et al.*, 1992). There has been considerable criticism of the use of such representative elementary areas (REA) since the equations that describe hydrological fluxes in the field are typically derived under laboratory conditions and differ from those in the field. Furthermore, although these equations may be valid at the local scale they are not necessarily representative at the elementary scale at which they are applied (Beven, 1996a). A further major constraint of these models is the requirement of detailed meteorological, terrain, landcover and soil data for each REA. These data are often not available and the techniques for measuring these parameter values at the scale required by the model do not exist (Beven, 1996a). Therefore, although the models give representation to spatial heterogeneity of hydrological processes, in the absence of spatial input data they essentially operate as lumped conceptual models.

In an attempt to overcome these problems, conceptual elements have been introduced into physically-based models. Various methods have been used to determine whether certain preferred spatial and temporal scales exist at which simple conceptualisation of dominant processes are possible (Jewitt and Görgens, 2000). Topography-based models, such as the well-known TOPMODEL, are an example of this type of model approach (Bonell, 1993). They are based on the notion that catchment morphology controls water flow pathways and that spatial patterns of moisture repeat themselves in time (Bonell, 1993). These models, also referred to as distributed function models, have followed a process of simplification from the more complex models whilst adding geographic information and remote sensing data processing modules to give a more parsimonious representation of spatial heterogeneity (Band,

1993). These distributed function models can be parameterised over complex terrain whilst still retaining critical distributions and interactions of the key hydrological processes (Band, 1993).

For example, in the TOPMODEL framework, spatial differences in soil water deficits were calculated as a function of position within the hillslope using the $\ln\left(\frac{a}{\tan\beta}\right)$ wetness index concept, where a is the area draining through a point from upslope and β is the terrain slope. All points with the same index value are assumed to respond in a hydrologically similar way (Beven, 1997). This so-called TOPMODEL index is derived from Darcy's sub-surface flow theory (Hornberger *et al.*, 1997) with three fundamental assumptions;

- steady state conditions apply,
- there is an exponential decline in transmissivity as soil depth increases, and
- the hydraulic gradient of the saturated zone can be approximated by the local surface topographic slope, $\tan\beta$.

It has been shown that these assumptions do not always apply (Ambroise *et al.*, 1996) and therefore it has been accepted that TOPMODEL is a set of conceptual tools that need to be modified for the application at hand, rather than a defined model (Beven, 1997).

However, a number of limitations arise from the convenience and simplicity offered by distributed function modelling or any type of conceptual modelling approach. In particular, assigning physical meaning to distribution functions is difficult because they require over-simplified assumptions about sub-surface systems (Watson, 1999a).

Vertessy *et al.* (1993) suggested that although there may be a conceptual understanding of runoff generating mechanisms in afforested catchments, there is still a poor quantitative understanding of how different catchment characteristics interact to modulate water flow. Physically-based distributed models in hydrology therefore mediate between underlying theory, which may be partly qualitative or perceptual in nature, and quantitative prediction (Beven, 1996a).

3.5 Model verification

The only way of establishing the realism of process-based models is through internal testing or verification (Beven, 1996a, Grayson *et al.*, 1992) of the models, and this exercise should be a vital component in physically-based modelling studies (Watson, 1999a). By verifying internal state variables of the model, i.e. those components and processes within the system which are simulated *en route* to final estimations (canopy interception, transpiration and soil water status), the model user can be satisfied that the correct simulation results are obtained for the hydrologically correct reasons (Schulze, 1995b).

Although such internal verification can be achieved to a certain degree in well-monitored and researched catchments, the availability of detailed data in South African forested catchments is limited as is revealed in Chapter 6. Therefore, model developers and users rely essentially on catchment discharge for verification. Beven (1996a), however, stresses that discharge alone cannot be used as the only measure of assessment of model performance. It is possible to achieve a good simulation of discharge even though internal processes are modelled poorly. It is therefore essential to establish that differences in simulated streamflow for different land uses are a result of differences in the governing processes and not as a result of erroneous parameter adjustment.

3.6 Data limitations in South Africa

The need for predictions of afforestation impacts for decision-making has made it necessary for developers to construct models on the basis of best available knowledge, knowing that this basis can be improved in future (Refsgaard *et al.*, 1994). However, the concern is raised, that in the present scientific climate the collection and analysis of field data is undervalued (Grayson *et al.*, 1992; Smith *et al.*, 1994) and there is little doubt that “our modelling capabilities have surpassed our ability to gather meaningful field data for model parameterization and validation” (Vertessy *et al.*, 1993). This is particularly true for South Africa where spatial data of suitable accuracy and resolution for input into models is limited, as highlighted in later Chapters of this document.

If the required data inputs for the model cannot be obtained, particularly for large-scale applications, then the application of more complex models is questionable, since, without the appropriate input data, the physical equations lose their relevance (Beven, 1996b). Collection of model input data is a time-consuming and expensive operation. Measurement of input data is often restricted to point measurements with the implication that a measurement, taken at a point, is representative of the representative unit, such as the ESU, as a whole. The uncertainty of model predictions, therefore, is directly related to the uncertainty in the accuracy of the input data.

If models are to be used to predict the impacts of land-use change, such as afforestation, it is essential for model developers to critically evaluate their models and provide an indication of the capabilities of the model and the level of confidence that can be expected from the various outputs with consideration of the data requirements. In this respect, the limitations and assumptions of distributed predictions need to be made more explicit with the view of moving science ahead (Beven, 1996a).

3.7 The application of physically-based models for non-research purposes

Models are increasingly marketed to management agencies and authorities for non-research applications (Grayson *et al.*, 1992). Grayson *et al.* (1992) caution on the use of physically-based models for predictive purposes. They argue that if a model is to be used for predictive purposes it must have undergone extensive testing for well-defined conditions. However, in practice, the model's original purpose is often lost and the model is applied to situations beyond the scope of its capability or its intended purpose. Once models are being applied for non-research purposes, it is the responsibility of the individual user to apply the model in a responsible way (Smith *et al.*, 1994; Refsgaard *et al.*, 1996). However, Grayson *et al.*, (1992) believe that for a practicing hydrologist to determine the value of such a model, there should be a full and frank discussion of a model's capability and limitations. As models become more complex, it is unrealistic to expect the practising hydrologist to become fully familiar with all the assumptions and sub-model formulations (Grayson *et al.*, 1992). Whilst Görgens and Lee (1992) promote the idea of user-orientated interfaces for models, Grayson *et al.* (1992) caution against the use of user-friendly graphical interfaces and decision support system

models and believe they are being promoted as substitutes for knowledge and downplay the need for acquiring greater knowledge.

* * *

In Chapter 3 processes and runoff generating mechanisms dominant in forested catchments were described. The physically-based modelling approach that gives description to forest processes was described and some of the limitations of physically-based distributed models highlighted. Based on the discussion above, the question arises as to which models are available and capable of modelling forest hydrological processes and could be used as water resource management and planning tools required under the NWA. The South African *ACRU* model is probably the most widely used model for land use modelling in the region. The following chapter provides a critical review of the *ACRU* model structure with emphasis on the representation of those processes dominant in forested catchments.

4. THE *ACRU* MODEL

In 1992, Görgens and Lee (1992) produced an extensive report on the hydrological impacts of forestry in South Africa. The report included a review of models that were considered for use in simulating the impacts of forestry-related landuse change in South Africa. The only two South African models that were available for making water allocation decisions were the Pitman model, now referred to as the WRSM90 model (Pitman and Kakabeke, 1991), and the *ACRU* model. In the opinion of Görgens and Lee (1992) the WRSM90 model could only succeed as a forestry/landuse decision-making tool if the model's lumped character was amended and its link with physical processes was strengthened. On the other hand, they considered the *ACRU* modelling approach to have great potential to allow for affirmative answers to questions water resource managers needed to address. The model had been extensively applied in humid, sub-humid and temperate zones and its physical basis allowed it to be applied to ungauged catchments without calibration.

An advantage and strength of the *ACRU* model arises from the fact that it has been developed in South Africa and therefore consideration was given to the limited amount of hydrological data available. As a result substantial research has been undertaken to develop databases and derive parameter values and variables required by the model from existing data sources. A further advantage of the *ACRU* model is the user support and the comprehensive user manual that accompanies the model. Model documentation is a necessity when applying more complex models and is discussed in a later Chapter of this dissertation.

4.1 Background

The *ACRU* agrohydrological modelling system is a physical conceptual model - physical in the sense that it represents physical processes explicitly and conceptual in that important processes are idealised. The model also includes various databases to support user applications. The model operates at a daily timestep and revolves around multi-layer soil water budgeting routines (Schulze *et al.*, 1995a). It was initially developed by the erstwhile Department of Agricultural Engineering, now School of Bioresources Engineering and Environmental Hydrology (BEEH), at the University of Natal as an agricultural land-use model, but growing

concern of the impacts of forestry on catchment water yield initiated process studies in forested catchments and new routines were included in *ACRU* to account for forest related hydrological processes. These adaptations to the *ACRU* model include the addition of the von Hoyningen-Huene (1983) equation which uses gross daily rainfall and leaf area index (LAI) as inputs to calculate daily canopy interception loss (I_i), a routine to account for interception storage by the forest canopy which is estimated as being $0.5 I_i$ and lastly a wet canopy evaporation rate to account for the enhanced evaporation of the water stored on the canopy (Schulze *et al.*, 1995b).

In 1991, BEEH initiated process research to improve the understanding of the effects of forest litter layers interception and to gain insight into the effects of site preparation on tree water use (Jewitt, 1991). From this research it became evident that greater reductions in water yield were observed from well-prepared sites most likely as a result of enhanced root and canopy growth (Jewitt, 1991; Summerton, 1995). Further research on the effects of site preparation by Moerdyk and Schulze (1991) has shown that the higher the intensity of site preparation, the less surface runoff can be expected. In 1995 a series of workshops were held to establish the changes of LAI, interception and rooting patterns for different forestry regions, rotation periods, rainfall regimes, genera and management practices (Summerton, 1995) and how these changes would affect catchment processes and hydrological responses, and thus the *ACRU* variables used to describe them (Schulze *et al.*, 1995b). This collective research contributed to the development of the *ACRUforest* decision support system and an information base for use in *ACRU* forest simulations.

4.2 Operating Environment

ACRUforest decision support system (DSS) can be considered as a “shell”, containing pre-coded information sets, in which the standard *ACRU* water budgeting routines are imbedded. These information sets include a daily rainfall station, soil data, evaporation and minimum and maximum temperatures for each quaternary catchment. In addition to this, the database contains pre-programmed values of LAI, rooting depths and canopy interception that are adjusted according to management practices and rainfall regimes as described above. The *ACRUforest* DSS prompts the user with simple forestry related questions, such as where afforestation is to take place, planting date and which species is to be planted. The relevant

data are then automatically extracted from the pre-acquired quaternary catchment information sets. The benefit of such a support system is that it allows for a quick first impact assessment (Schulze *et al.*, 1996) without the user needing an extensive understanding of forest processes and parameter inputs.

However, to use the model for site-specific applications, the use of detailed on-site data is desirable as opposed to the more general quaternary information presented in the *ACRUforest* database. Therefore, for site-specific applications the stand-alone *ACRU* model is usually used, and the user is required to obtain and input the data for a particular catchment in combination with the information for forests available in the *ACRU* user manual.

In the forthcoming section a summary of the general structure of the *ACRU* water budgeting routines is provided, with an emphasis on the model developments that have taken place to account for forest hydrological processes. These are classified into horizontal and vertical process representations. The horizontal structure refers to the spatial disaggregation of the catchment into subcatchments and the movement of water between these units, whilst the vertical process representation refers to the vertical water fluxes within a subcatchment, as rainfall moves through the canopy and soil.

4.3 Horizontal process representation

The model user accounts for spatial differences in storages and fluxes caused by spatial heterogeneity in a catchment's physical characteristics by the subdivision of a catchment into more homogenous subcatchments. Different criteria can be used to divide the catchment into subcatchments for example soil types, type of landuse or topography. It is assumed that the processes within each subcatchment are uniform.

A multi-layer soil water budget takes account of the inputs into, redistribution within and output of water from each subcatchment. The runoff generated from each subcatchment is a combination of estimated stormflow and baseflow, with the stormflow consisting of a quickflow response (stormflow released into the stream on the same day as the rainfall event) and delayed flow response (post-storm interflow). The generated runoff is then routed to the next downstream subcatchment where it is added to the runoff contributions from that

subcatchment as shown in Figure 4.1. The timing and amount of runoff release from each subcatchment for a given day is controlled by a few, but effective parameters. This runoff generation component of *ACRU* can be considered a black-box type of approach whereby limited reference is made to the internal processes governing runoff generation, such as lateral flow, saturation excess flow or infiltration excess flow. Instead parameters are used to effectively re-produce the storm hydrograph observed in forested catchments by controlling the storage and time-delays of water release. This approach essentially ignores the complexity of the runoff generation processes observed in forested catchments which are often difficult to characterise and parameterise.

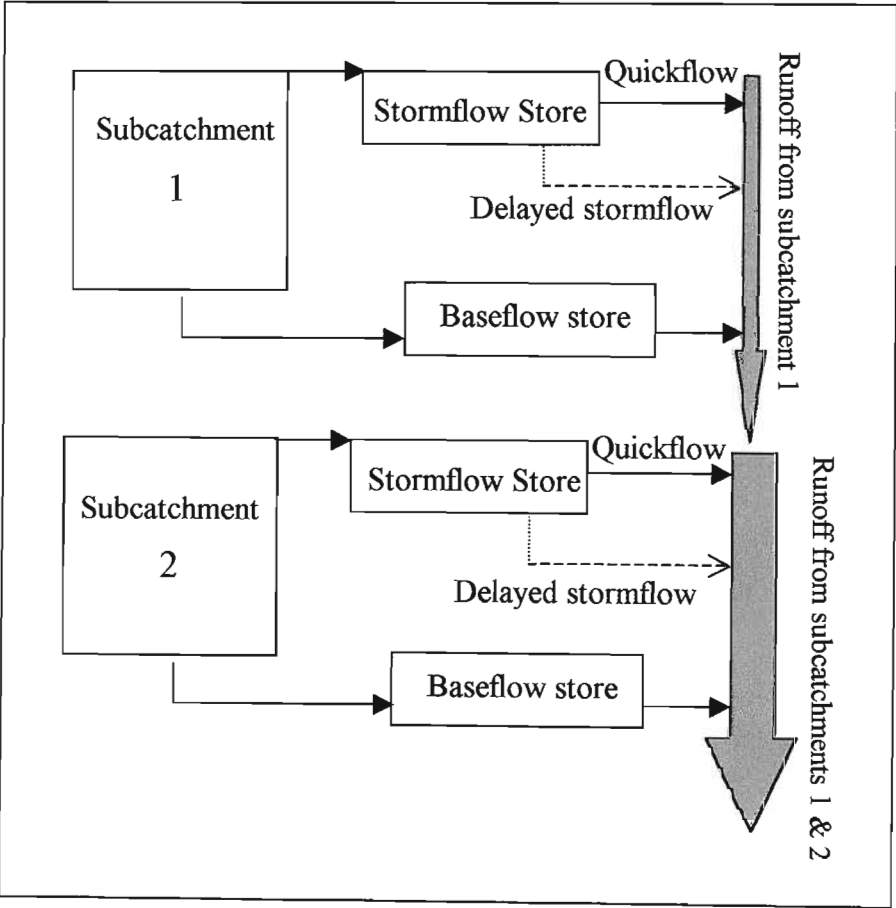


Figure 4.1. Runoff representation for a catchment with two subcatchments.

4.3.1 Runoff representation

Different runoff mechanisms may be dominant in different environments and in catchments with different characteristics of soils, vegetation and topography (Beven, 2000). Instead of incorporating detailed process descriptions of runoff generation, the *ACRU* model uses

catchment characteristics as a basis for the derivation of parameter values that control the amount of runoff generated on a given day. Although the parameters used cannot in reality be measured, they are linked to physical characteristics of the catchment such as its landcover climate and soil characteristics (Schulze, 1995a).

Stormflow – Quickflow and delayed stormflow

Stormflow in *ACRU* is generated when net rainfall (after interception – see Section 4.4.1) exceeds initial abstractions. Initial abstractions are calculated as the product of a coefficient of initial abstraction (COAIM) and the soil water deficit of the critical response depth (SMDDEP) of the soil (Schulze, 1995a). COAIM is dependent on vegetation, site and management characteristics and is increased to 0.3 or 0.4 for forest simulations due to factors such as the presence of a litter layer on the forest floor. SMDDEP is a conceptual depth designed to control how much water the catchment soil can absorb (i.e. infiltrate) before surface runoff is initiated. This depth parameter is set by the model user and is considered shallower in arid regions where high intensity storms would provide predominantly surface runoff, and is generally considered deeper in forested regions with dense canopy cover and deep organic layers, where more interflow and ‘pushthrough’ mechanisms dominate (Schulze, 1995a). What this means for forest simulations is that a larger amount of rainfall needs to infiltrate before the deficit is reduced and stormflow commences.

Once the initial abstractions are exceeded, stormflow is generated and added to the stormflow store. A fraction of this store, as prescribed by a quickflow response coefficient (QFRESP), is released and runs off on the same day, whilst the remaining stormflow is retained until the following day when the same fraction of the remaining stormflow is released (see Figure 4.1). The QFRESP coefficient is dependant on factors such as catchment size and slope, the density of the land cover and the interflow potential of the soil. The model is therefore able to reproduce the sustained “dry weather” flow or delayed flow following a rainfall event resulting from lateral sub-surface contributions.

Baseflow

A second response coefficient, known as the baseflow release coefficient, governs baseflow release from a groundwater store. This coefficient depends on factors such as geology, catchment area and slope. A baseflow release coefficient of 0.005-0.05 (0.5-5%) per day is suggested in the user manual (Schulze, 1995a). However, since the publication of the user manual a value of 0.09 (9%) per day was found to be more reasonable for most model applications (Pike, 2000, personal communication). The release from the baseflow store, however, is not constant and the “decay” in release from the store is dependent on the magnitude of the previous day’s groundwater store (Schulze, 1995a), as shown in Equation 4.1.

$$F_{\text{bff}} = F_{\text{bfi}} \left[\frac{[(S_{\text{gwp}})^2 - S_{\text{gwp}}]/1000 + 1.3}{11} \right] \quad \text{Equation 4.1}$$

where

- F_{bff} = final baseflow release coefficient
- F_{bfi} = input baseflow release coefficient
- S_{gwp} = magnitude of previous day’s groundwater store (mm)

The stormflow and baseflow response coefficients only control the timing of releases, and therefore impact on the day to day variability of streamflow and not the total soil water budget. The implication of prescribing a quickflow response coefficient is that lateral flow is implied by the controlled release of water from the stormflow store and therefore water accumulated in the stormflow store is not available for uptake by trees. In reality however, lateral flow occurs through the downslope movement of water in the soil, therefore this water would be available to trees and other vegetation. The same applies to water accumulated in the baseflow store. Once water enters this store, tree roots no longer have access to the water contained in the store.

In the opinion of Görgens and Lee (1992) these soil water translation parameters have only indirect physical meaning. They believed that this “black-box element” required a more physical basis. A further implication of using such conceptual parameters is that the model user would require a degree of experience and therefore it could be argued that the selection of

appropriate values, although based on soil climate and vegetation characteristics, introduces a certain degree of subjectivity to the modelling process (Scott, 1994).

A further criticism of the *ACRU* model highlighted by Görgens and Lee (1992), was that runoff production in *ACRU* is not sensitive to landscape forms and morphology that have been shown to control runoff production. They suggested the union of the *ACRU* model with elements of terrain sensitive models such as those discussed in Chapter 3. Topographic controls of runoff production were considered by Görgens and Lee (1992) to have a particularly important function in modelling forestry impacts since:

- Afforestation often occurs in headwater areas where topo-morphological controls of runoff can occur.
- Modelling the response of saturation zones would be useful for decisions regarding exclusion zones in permit allocations.
- Modelling the effects of riparian management policies might require topo-morphological controls to be explicitly represented, because riparian zones are the most common partial source areas of runoff generation.

To address these issues and account for partially saturated areas in the riparian zones of the catchment, *ACRU* developers included an option to re-direct runoff releases from upstream catchments into a designated riparian subcatchment, as described in the following section.

4.3.2 Riparian zone hydrology

Trees growing in the riparian zone are believed to use proportionally more water than those further away from the river (Rowntree and Byers, 1999; Dye *et al.*, 2000) due to the continuous availability of water in the riparian areas. A number of studies have shown that trees in the riparian zone transpire more water than indigenous herbaceous vegetation and that the replacement of riparian alien vegetation with indigenous vegetation results in increased water yields (Dye *et al.*, 2001). Earlier studies reported by Scott and Lesch (1995) also suggest that the removal of alien trees in close proximity to the stream results in greater streamflow increases than that resulting from the removal of alien vegetation away from the stream.

To account for this assumed enhanced soil water availability in riparian areas, the *ACRU* model can be configured to re-direct water from the upstream areas (subcatchments) into the riparian area (sub-catchment). In these cases the riparian zone of the river is usually defined as that area that extends 30m from all streams and wetlands visible on a 1:50 000 map sheet (Jewitt *et al.*, 2000) and is the lowest area of catchment topography. The model routes the stormflow from upstream catchments into the riparian area where it is infiltrated. The accumulated baseflow from upstream catchments is added to the riparian catchment's lower soil horizon where it is available to the trees, as shown in Figure 4.2. Once this horizon becomes saturated the upper horizon is filled and if this top horizon reaches saturation the excess water overflows and is added to the stormflow store of the riparian subcatchment (Meier *et al.*, 1997; Jewitt *et al.*, 2000). It has been shown from hillslope studies that these riparian saturated areas expand and contract as lateral sub-surface flow increases and decreases. However, such fluctuations are not represented by this modelling approach and the saturated zone is restricted to 30 metres.

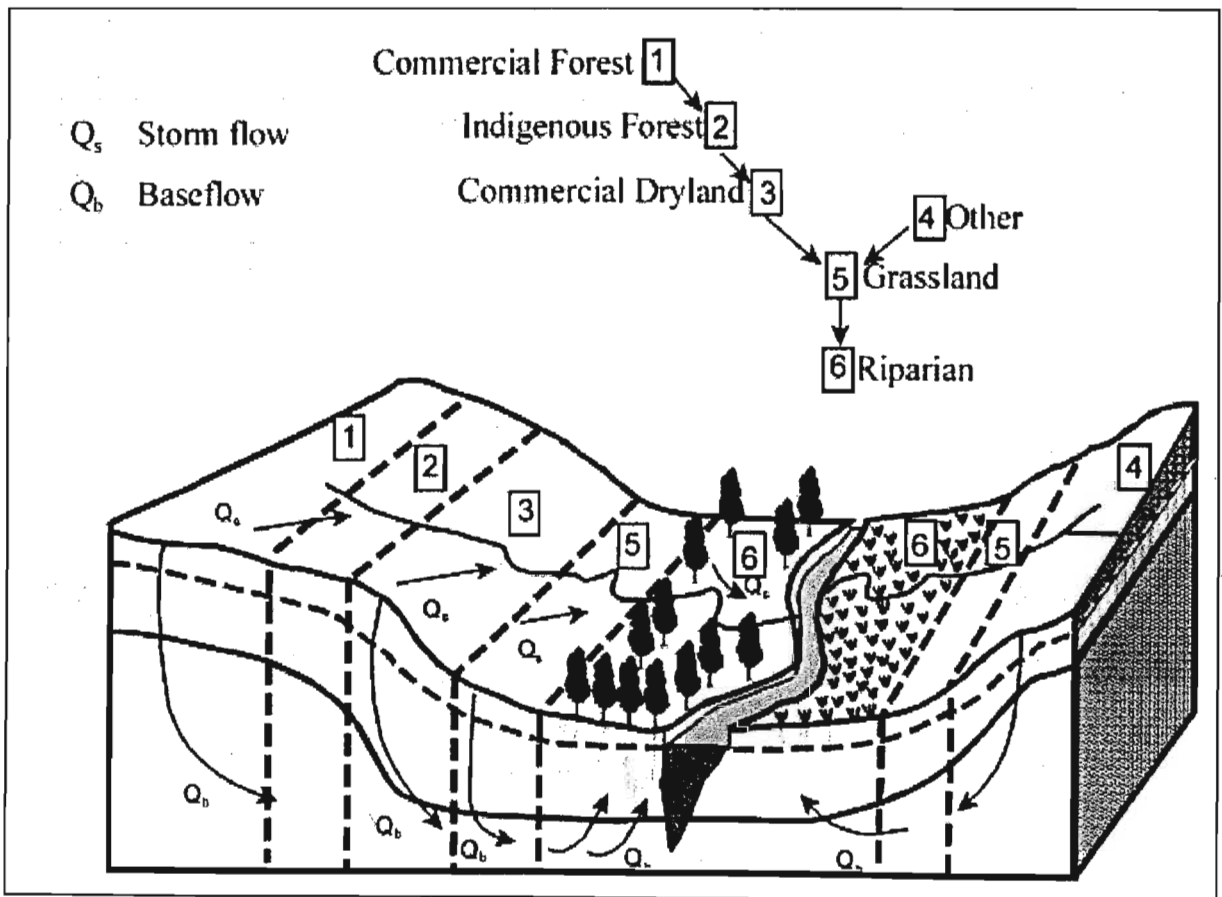


Figure 4.2. Example of the configuration of the *ACRU* model to simulate riparian zones (Meier *et al.*, 1997).

4.4 Vertical process representation

The vertical structure refers to the processes within a subcatchment which tend to have vertical orientation. In this section emphasis is given to the vertical processes dominant in forested catchments. Figure 4.3 below illustrates the vertical process representation in *ACRU*.

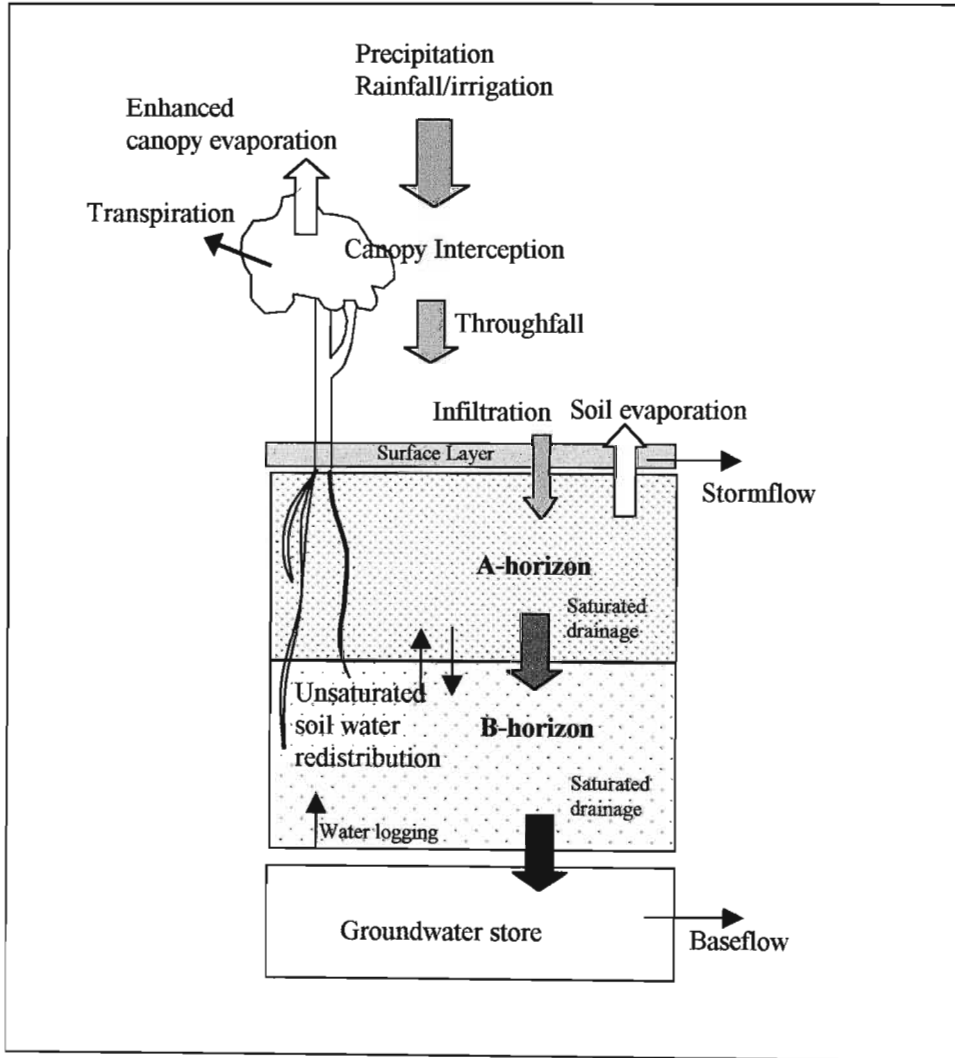


Figure 4.3 The *ACRU* agrohydrological modelling system: vertically orientated processes.

4.4.1 Canopy interception

Rainfall falling onto the vegetation cover is partially intercepted and vaporised back into the atmosphere. On a day with rainfall the net precipitation available to the soil water budget is calculated as the difference between gross precipitation and interception losses.

In the *ACRU* model the interception loss during a rainfall event is usually calculated using an approach developed by Von Hoyningen-Huene (1983), which provides an estimate of interception loss based on an empirical relationship between LAI and gross daily rainfall. Alternatively the user may input interception loss values manually, if they are known, or default values derived by Summerton (1995) which are provided in the *ACRU* manual.

The interception storage capacity of the canopy is estimated as being half of the interception loss (Schulze, 1995). To account for evaporation of the intercepted water stored on the tree canopy, which is known to proceed at rates well in excess of estimated reference potential evaporation due to advection and low aerodynamic resistance of the wet forest canopy, an enhanced wet canopy evaporation rate equation (Equation 4.2) has been incorporated into the model (Schulze *et al.*, 1995c).

$$E_w = E_r(0.267 \text{ LAI} + 0.33) \text{ for LAI} > 2.7 \quad \text{Equation 4.2}$$

where

- E_w = enhanced wet canopy evaporation
- E_r = estimated reference potential evaporation
- LAI = leaf area index

Therefore, a portion of intercepted water is evaporated on each rainday and the remaining water is stored until the next day. Water that is not intercepted or abstracted as stormflow, infiltrates the soil where it becomes available for transpiration and soil evaporation.

4.4.2 Transpiration and soil evaporation

Commercially grown species of trees in South Africa continue to transpire during the dry season (winter in summer rainfall areas), which is usually the dormant season for most plants, as a result of their deep rooting systems and evergreen canopies (Versfeld, 1994). Therefore, trees can potentially utilise more water than grasses that senesce during the winter season. In *ACRU*, the evaporative demand of the tree is estimated from both atmospheric demands and the LAI associated with the tree's stage of growth (Schulze *et al.*, 1997). The atmospheric demand is estimated from daily A-Pan equivalent evaporation and is used as the reference for

potential evaporation (E_r). The accurate estimation of daily E_r is essential, considering an estimated 91% of MAP is returned to the atmosphere by total evaporation. The *ACRU* model contains a number of different methods of estimating A-pan equivalent evaporation and the method selected is likely to depend on the data that is available.

That portion of potential evaporation not taken up by evaporation of intercepted water in the canopy is available for plant transpiration and soil water evaporation. The fraction of the evaporative energy (F_t) available for transpiration is dependent on LAI and is given by the adaptation of the Ritchie equation (1972);

$$F_t = 0.7 \text{ LAI}^{0.5} - 0.21 \quad \text{Equation 4.3}$$

where

F_t = fraction of evaporative energy

This equation only applies for daily LAI values between 2.7 and 0.1 and therefore transpiration by a tree with an LAI greater than 2.7 is limited to 0.95 of evaporative energy. Hence, even for dense vegetation, such as forests, at least 5% of available energy is used for soil evaporation. Although the fraction of evaporative energy (F_t) allocated to transpiration would be the same for trees with a LAI of 3.0 and trees with a LAI of 3.5 (i.e. both exceeding 2.7), a greater streamflow reduction would be observed from the catchment with trees of a higher LAI as a result of the greater interception loss. Another implication of this approach is that under non-water limiting conditions, regardless of LAI, soil evaporation and transpiration combined (total evapotranspiration) equal the day's reference potential evaporation (atmospheric demand) unless the canopy is wet.

The water use relationships associated with LAI in *ACRU* change over time. These were derived from fieldwork results, a review of relevant literature, and from expert opinions obtained from a four-day workshop of South African forest hydrologists and forest company scientists (Summerton, 1995). Concern has been expressed that these LAI-based functions, which drive the forest component of the model, have unavoidable subjective elements, stemming from the consultation process used in their derivation and are based on only limited fundamental research (Görgens *et al.*, 1999). Allied to this concern is the fact that the transpiration component of the model has not been extensively verified.

A restriction of using LAI values to estimate transpiration in South Africa is that LAI values are not readily available for vegetation types other than forestry. When the model is used in comparative studies to predict the impact of forestry relative to other vegetation types on catchment water yield, the maximum transpiration rates for these grasslands and crops would most probably be estimated from crop coefficients. The crop coefficient can be defined as the ratio of “maximum” evaporation (E_m) from the plant at a given stage of plant growth to a reference potential evaporation (E_r) (Schulze *et al.*, 1995c). The maximum is used to describe the evaporation from the plant that takes place under “well-watered” conditions when the effects of soil water shortages are negligible (Schulze *et al.*, 1995c). The fraction of E_m that is made up of transpiration (E_t) is estimated from;

$$F_t = 0.95 (K_d - 0.2)/0.8 \quad \text{when } K_d > 0.2 \quad \text{Equation 4.4}$$

$$= 0 \quad \text{when } K_d \leq 0.2$$

where

F_t = the fraction of total available transpiration

K_d = daily crop coefficient

Under conditions of limited available soil water E_t is adjusted to account for a deficiency of available soil water.

4.4.3 Rooting patterns and soil water availability

The *ACRU* model consists of two soil horizons, a topsoil and sub-soil horizon. It is assumed that roots absorb soil water simultaneously from both soil horizons in proportion to the root mass density of the respective horizons, except when low soil water conditions prevail and the wetter horizon contributes a greater proportion of soil water to the plant (Schulze, 1995c). The proportion of active roots in the topsoil is represented by a parameter which indicates from where within the soil profile the tree is able to extract its soil moisture (Schulze *et al.*, 1996). Therefore, the maximum evaporation available for transpiration from the plant is apportioned to the different soil horizons in proportion to the root mass distribution of the respective soil layers (Schulze, 1995c).

Typically trees have deep rooting systems and are able to extract soil water from deep in the soil profile (Dye *et al.*, 1997). Consequently trees are able to maintain high transpiration rates during the dry season when shallower rooted vegetation are experiencing stress (Dye and Bosch, 2000). Rooting depths are likely to depend on a combination of factors such as soil textural properties, depth of the soil profile and soil water availability. However, knowledge about the functional rooting depth of trees remains limited (Kelliher *et al.*, 1993). This limited knowledge of rooting depth, combined with the absence of actual soil depths, makes it practically impossible to simulate the response of trees to different soil water deficits in any model.

In *ACRU* the soil depth representation is based on agricultural soil depths rather than an actual soil depth. Most often soil depth estimates are obtained from the Land Type (soils) information published by the erstwhile Soil and Irrigation Research Institute, now the Institute for Soil, Climate and Water (ISCW). For simulations of forestry impacts with *ACRU*, it is suggested that an additional 0.25 metres is added to the Landtype depths (Pike, 2000, personal communication) as actual soils depths are often reported as being greater than the augering depth of 1.2m. Hence, the actual soil depth and the depth of soil from which tree roots can extract water may be deeper than is represented by the model. Under non-limiting soil water conditions this would have few implications. However, under conditions of soil moisture deficiency, when a critical soil moisture threshold is reached, simulated transpiration by the canopy is reduced, when in reality the trees may still be maintaining transpiration as a result of the roots having access to deep soil water reserves. Currently in South Africa the majority of afforestation takes place in marginally dry areas and therefore soil water availability is an important variable in establishing consumptive water use of trees.

4.5 *ACRU* model forest simulations

A number of verification studies have been undertaken to assess the performance of the *ACRU* model in simulating the catchment water yield from forested catchments. Recently the performance of the *ACRU* model was tested for simulations on forested catchments (Gush *et al.*, 2001). This was done as part of a study to develop streamflow reduction tables using a combination of the Scott and Smith curves and the *ACRU* model. In this study the *ACRU* model was tested on eight afforested and partly afforested catchments, which include the

research catchments (paired catchment experiments) and three additional sites namely Ntabamhlope, Cedara and Seven Oaks. The verification sites generally fell in the higher rainfall regions (MAP >1000mm) with the exception of Ntabamhlope and Seven Oaks. Therefore, the *ACRU* model's performance under more marginal rainfall conditions was not tested.

In terms of streamflow volumes generated by the simulations, two-thirds of the simulations generated greater streamflow than had been observed. This applied to both total flows and low flows, although regular under-simulation of the lowest flows was often evident (Gush *et al.*, 2001). In cases where streamflow was over-estimated this was due largely to the significant over-simulation of wet season flows (Gush *et al.*, 2001).

Scott (1994) used the *ACRU* model to simulate the impacts of fire in the afforested Ntabamhlope catchment. Comparison of pre-fire simulated and observed streamflow volumes revealed that the model under-simulated monthly totals of baseflow as the modeller could not get the slow but sustained dry season low flows to persist (Scott, 1999, Personal communication). In addition peak flows during the early part of the wet season were over-simulated when the catchment was wetting up after the long dry season (Scott, 1999, Personal communication). This feature of over-simulation of flows from a "dry" catchment had been observed in previous *ACRU* simulations (Schulze, 1993, personal communication, cited by Scott, 1994). It has also been observed that simulated stormflows were higher than observed stormflows. This was attributed to the fact that moderate to large storms (25-100mm) fell as low intensity rainfall events during the day and that surface runoff was over-estimated (Scott, 1994).

Other studies have shown that low flows in the relatively large Maritsane river catchment (221km²), situated in the upper reaches of Sabie River catchment, were under-simulated, whilst higher flows were over-simulated. According to Jewitt and Schulze (1999), this could be attributed to the incorrect simulation of timing of high and low flows and could be rectified using the flow routing module. Using the flow routing module would result in attenuation of peak flows. A similar trend was observed in the steep sloped and shallow soiled Cedara catchment (1.32km²) where simulated monthly streamflow exceeded recorded monthly totals during the wet season (Jewitt and Schulze, 1999).

The verification of the *ACRU*'s internal processes for forested catchments, however, remains limited and this can be attributed to a lack of measured data. Verification studies concerning forest hydrological processes which have been reported include the verification of the Von-Hoyningen-Huene canopy interception equation (Schulze, 1995b) and the verification of the evapotranspiration component of *ACRU* at Seven Oaks (Gush *et al.*, 2001). Simulations of the Von-Hoyningen-Huene equation were shown to closely follow observations of interception losses in a stand of 10-year old *Pinus patula*, at Cathedral Peak in KwaZulu-Natal (Schulze, 1995b). In the verification of the evapotranspiration component of *ACRU* at Seven Oaks in KwaZulu-Natal, simulated values of evapotranspiration rates for Eucalypts and Wattles were compared to observed evapotranspiration rates, measured for a two-year period with the Bowen ratio method. It was evident from this study that evapotranspiration for both Wattle and Eucalypts was under-simulated by *ACRU*, particularly during the winter season when *ACRU* estimated evapotranspiration to be half that measured by the Bowen ratio method. Although the limited number of studies provides inconclusive evidence that the model is simulating internal processes accurately, it does indicate a need for further testing of internal processes. The under-simulation of transpiration by *ACRU* would suggest that the model would over-simulate runoff, however, the model verifications show the opposite result, that low flows are typically under-simulated. This indicates that although less water is used in the transpiration process, it is not necessarily stored and available for release during the low flow period. It appears that the catchments described in this section have large soil storage capacities that is available for transpiration and ensures sustained baseflow releases during the dry season.

Although, in most cases, cumulative totals of daily streamflow over the entire simulation period are accurately simulated by the *ACRU* model, streamflow variances are not always well preserved and the model does often over-simulate high flows and underestimate low flows to some degree (Jewitt and Schulze, 1999; Gush *et al.*, 2001). Such trends could possibly be the result of the timing of low and high flows being incorrect, which in the large Maritsane catchment could be as a result of the flow routing module in *ACRU* not being used. However, in the smaller catchments, the effect of flow routing on the magnitude and timing of peak flow would be less significant. Therefore, it is likely that the over-simulation of high flows by the *ACRU* model is the result of runoff generating mechanisms not being fully represented by the model.

A further limitation of the verification studies described above is that the goodness of fit statistics were performed on monthly totals of daily streamflows. Although a model may perform well at the monthly timescale, the model may not necessarily perform as well at the daily timescale. Since the ability of a model to simulate daily streamflow is becoming more important, as discussed in Chapter 2, it would be beneficial to establish the performance of the model at this timescale.

In the light of the above discussion and in view of the suggestions made by Gorgens and Lee (1992), a topographically-based alternative to the *ACRU* model was tested for application in South African catchments in the hope that it would result in improved low flow simulations. The model was required to fulfil certain criteria to make it suitable for this application. These criteria were as follows;

- The model had to have a topographically-driven lateral flow component
- The model had to give representation to the processes dominant in forested catchments
- It had to have been developed for similar climatic conditions and represent processes prevalent in South African forested catchments
- It required a physical basis
- It had to have been applied and verified for forested catchments
- The model was required to be a catchment scale model
- User support was essential

Based on available literature, it became apparent that no South African models fulfilled all the criteria outlined above. The motivation for selecting the Australian Macaque model was that the model met most of the requirements listed above and became the focus of further study as is reported in subsequent chapters.

* * *

In Chapter 4 the *ACRU* model components relating to forest processes were described and some of the advantages and shortcomings of the model were highlighted. In conclusion the application of the *ACRU* model on forested catchments was reviewed. In the ensuing Chapter the Australian Macaque model is described with specific reference to forest processes.

5. THE MACAQUE MODEL

Australia has committed itself to a vision of trebling the country's area of timber production by the year 2020. The fulfilment of this vision will necessitate afforestation planning in which hydrological forecasting forms a key element (Vertessy, 1999). As is the case in South Africa, forested areas in Australia are often situated in important water supply catchments. Thus, there is a need to develop models to quantify the impact of various forestry treatments on water yield in important water supply catchments. The resulting research has led to tremendous advancement in forest process studies and model development in Australia. The Macaque model is a product of this initiative, a brief history and overview of which is provided in this chapter.

5.1 Model development history

Typically a curve, similar to those of Scott and Smith (1997) is used in Australia to describe the relationship between mean annual streamflow and forest age for mountain ash forest (*Eucalyptus regnans*). This curve, known as the Kuczera Curve, is based on the known hydrological response of eight large basins to fire (Vertessy *et al.*, 1998). A recognised limitation of the curve is that it represents a generalised representation and, as is the case with the South African Scott and Smith curves, ignores local site characteristics. It was suggested that physically-based models rather than a generalised curve could lead to improved management by accounting for large-scale, spatially variable hydrological responses to forest operations (Watson *et al.* 1999). In particular, a model was needed that could operate at the "larger" scales of 10-1000km² and was sensitive to spatial variability, in order to account for different management practices in the catchment.

In response to these suggestions, Watson (1999a) examined a number of international models and their applicability for predictive spatial modelling of the effects of land cover change on long-term water yields. He concluded that for the problem at hand, the ideal model should be able to;

- simulate water yield,
- identify which part of the catchment is at greatest risk to logging or wildfire,

- be able to simulate long term and at a large scale,
- simulate changes in water balance due to land cover change, as well as
- provide a realistic simulation,
- provide a framework for the organisation of the physical processes and spatial variability, and
- be parsimonious such that redundancy is minimised (Watson, 1999a).

From this review, it became evident that models suitable for large scale, long term modelling were limited. In particular the requirement of the model to simulate changes in landcover eliminated most candidate models (Watson, 1999a). Although selected models represented forest hydrological processes, such as evapotranspiration from complex forest canopies and multi-layered, multi-dimensional flow of unsaturated and saturated water in the soil in great detail, few models attempted to extend this detailed modelling approach to large scale application. The “bridging” of scales required careful consideration of parsimony and scale issues (Watson, 1999a), but was necessary if the model was to be applied for management purposes in operational catchments.

Watson (1999a) concluded that none of the international models reviewed were entirely suitable. Consequently, the development of a new model that accounted for complex forest processes, but had a vastly simplified soil moisture balance scheme, which would allow it to be applied at a larger scale, was initiated. The resultant model is known as the Macaque Model and its application to the South African situation constitutes a major part of this dissertation.

5.2 Operating Environment

The model operates within a visual modelling framework, known as Tarsier. Tarsier has been developed as a Geographical Information System (GIS) designed to support integrated spatial modelling (Watson, 1999b) and contains *inter alia*, analytical tools used for spatial interpolation, spatial statistics and terrain analysis. Tarsier’s digital terrain analyst, for example, allows for the derivation of various model inputs, such as aspect, slope, the delineation of hillslopes and the calculation of topographic wetness indices from a Digital Elevation Model (DEM).

The Tarsier framework allows the model user easy access to the tools used to derive the relevant model inputs, with relative ease and efficiency and without the requirement of an extensive knowledge of GIS. However, it is recognized that such automated procedures could result in undetected errors due to the possible limited understanding of the user.

The most fundamental component of the Macaque model is the spatial unit (Watson, 1999a). Spatial units are arranged in a hierarchical structure and typically range from one top-level unit, such as a region or catchment, with any number of levels of spatial units defined below this level. The lowest level in the hierarchical structure and also the smallest in area is that of the elementary spatial unit (ESU), which can be compared to the REA described in Chapter 3.

In the example presented in Figure 5.1 below, three spatial levels have been selected, i.e. the “world” level representing the catchment area, “hillslope” level representing the individual hillslopes (1) within the catchment and the lowest level being the ESUs. The model implements the hydrological processes (the combination of horizontal and vertical processes and fluxes) at the level of the ESU.

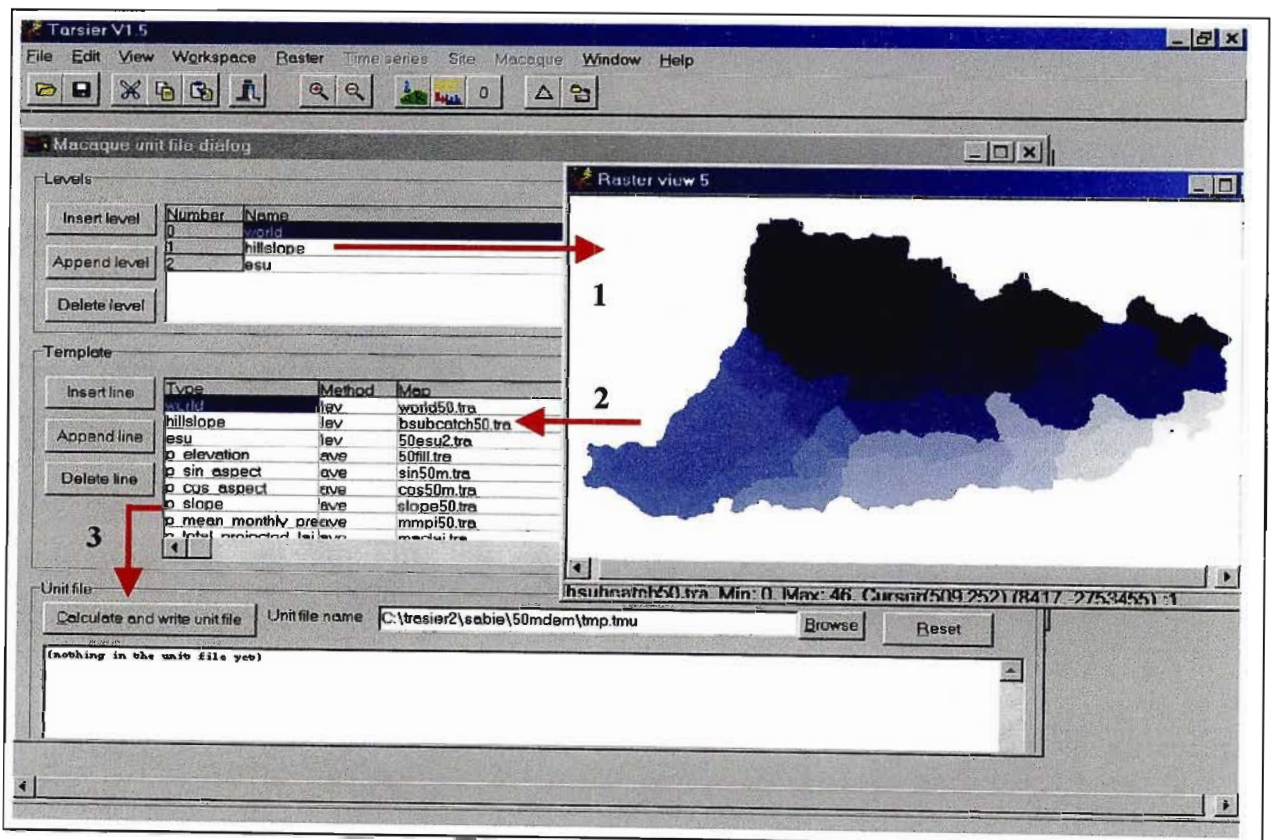


Figure 5.1. Macaque’s spatial template defining the spatial levels and input raster maps.

The methodologies used to derive the ESUs are described later in this Chapter. Decisions on how to structure the study area can be made at any stage and not necessarily at the onset of the simulation exercise (Watson, 1999a). All other spatial input data, for example slope, aspect and LAI, are presented as raster data (grids), shown in Figure 5.1 (2) and correspond to the resolution of the DEM. These spatial inputs into the model, at the DEM resolution, are then averaged for each ESU and written to what is known as the unit file (Figure 5.1, (3)). This unit file is then combined with the file containing the non-spatial model inputs.

A number of analytical tools are also available in Tarsier (1999). These include amongst others a spline analyst, which uses spline interpolation to derive spatially representative data from point measurements and a digital terrain analyst that, amongst other functions, removes pits and flats in a DEM and calculates slope from the DEM (Watson, 1999b).

Output from the model can either be in the form of a spatial display (i.e. raster map) or as time series data. The spatial displays can be run independently or run in conjunction with a running simulation, resulting in a dynamic spatial visualization (Watson, 1999b).

It is evident that Tarsier is a powerful framework providing the user with analytical tools and visualization support. The Tarsier operating environment also allows for the easy configuration of the model and re-running of simulations. Furthermore, the structuring of the spatial levels allows any model variable to be displayed for any spatial unit at any spatial level, which enables the model user to evaluate whether the model's internal processes are simulated realistically.

In the following sections the representation of forest processes in the Macaque model are described with particular emphasis on those components of the model that differ in approach to that of the *ACRU* model. The description of processes is divided into two sections i.e. horizontal and vertical process representation, as was the case with the description of the *ACRU* model in Chapter 4. The horizontal structure refers to the spatial disaggregating of the catchment into elementary spatial units and the lateral movement of water between the individual units. The vertical processes refer to the processes within a particular ESU.

5.3 Horizontal process representation

When considering the shortcomings of the *ACRU* model highlighted in Chapter 4, it has been suggested that the inclusion of a topographically driven lateral flow component could result in improved low flow simulations. Topographically-based models have become increasingly popular, not only because of the increased availability of DEMs (Ambroise *et al.*, 1996), but because topography is an important landscape feature controlling water flow (Hornberger *et al.* 1997). Therefore, the Macaque model which includes a lateral flow component which is dependant on topography (slope) and soil transmissivity was selected to test whether this modelling approach would result in improved simulations compared to models such as *ACRU* that do not incorporate topography or utilize a DEM.

Since the model's first documentation in 1999 (Watson, 1999a), developments to the lateral flow component of Macaque have been ongoing. Earlier versions of the model adopted lateral flow component parameters, based on the TOPMODEL concept, described in Chapter 3. However, the model developers felt that the parameters associated with this approach were too abstract (Watson, 2000, personal communication) and hence difficult to estimate. It was envisaged that the introduction of a more physically-based Darcian flow component could result in parameters with physical meaning, and thus be more easily derived. In the following section a comparison is made between the representations of these processes in the later version versus the earlier version of the model.

5.3.1 Limited distributed function modelling

In earlier versions of the model, the sub-surface lateral water movement component within Macaque had many features in common with TOPMODEL, reviewed in Chapter 3, and the RHESSys model (Band *et al.*, 1993), known as distributed function models (DFMs). Such models are based on the notion that spatial patterns of soil moisture repeat themselves in time as a function of the mean soil moisture status of the catchment (Bonell, 1993).

To extend this concept to the larger scale of 10-10000km², the RHESSys model and later the Macaque model utilised a catchment divided into hillslopes and grouped ranges of wetness indices, to form ESUs. Hillslopes may represent a small sub-basin or the flank of a large sub-basin (Watson, 1999a), whilst each ESU represents all parts of the hillslope with a certain

range of wetness index (Watson, 1999a). The individual ESUs within a hillslope are not necessarily contiguous. Although conceptually they might be thought of as lying one above the other, they are in fact statistical representations of the area within each hillslope at different points along the gradient representing the range of wetness indices for each hillslope.

Lateral redistribution of water was accomplished using a distribution function, which accumulated hillslope water into a single variable and then re-distributed it to each ESU according to its wetness index relative to the wetness indices of other ESUs in the hillslope, as shown in Equation 5.1. The saturation deficit of each ESU was therefore dependent on the wetness index of the ESU and the mean saturation deficit of the hillslope. ESUs with the highest wetness index received more water (and had a lower saturation deficit) than the others (Watson, 1999a). A limitation of this approach is that the disaggregation of the hillslope into wetness indices remained static subsequent to the initial calculation of wetness index, based on the assumption that the spatial pattern of soil moisture remains unchanged over time.

$$S_{sat, dist} = \bar{S}_{sat} + \Delta_{sat}(i_{wet} - \bar{i}_{wet}) \quad \text{Equation 5.1}$$

where

- $S_{sat, dist}$ = saturation deficit of the soil given by the distribution function
- \bar{S}_{sat} = the hillslope-wide area-weighted mean saturation deficit of the soil (metres)
- Δ_{sat} = rate of change of saturation deficit (according to distribution function) with wetness index
- i_{wet} = wetness index of an ESU
- \bar{i}_{wet} = mean wetness index of all ESUs on a particular hillslope

In this earlier version of the Macaque model, the method of calculating the wetness index was unconstrained, therefore the index did not have to be based on topography, as is the case for the TOPMODEL wetness index, but could be based on, for example, a combination of topography and soil heterogeneity. An added component to this earlier version of Macaque was the introduction of a limited distribution function (Equation 5.2) to allow for the water table to change shape - rather than to rise and fall uniformly in accordance with the changing

mean soil moisture status of the catchment, as was proposed by the TOPMODEL concept (Watson, 1999a).

$$q_{redist} = \delta (s_{sat, dist} - \text{previous } s_{sat}) \quad \text{Equation 5.2}$$

Where

- q_{redist} = implicit lateral flow redistributing saturated zone water from the current ESU to some other part of the hillslope
- δ = factor controlling lateral redistribution of water within a hillslope
- $s_{sat, dist}$ = saturation deficit of the soil given by the distribution function
- previous s_{sat} = saturation deficit of the soil.

The advantage of allowing for this change in the water table shape is that the subsurface response of the catchment can be accelerated, by allowing the water table in the saturated areas to rise, even though the water table away from these areas is inactive (Watson, 1999a). This limited distribution function, therefore, enabled the modelling approach to be applied to catchments with deep soils in a realistic manner (Watson, 1999a).

More recently, this DFM approach has been replaced by a more explicit representation of lateral flow between ESUs in the Macaque model. It was found that the parameters used to estimate lateral distribution between ESUs was too abstract, which made it difficult to parameterise and calibrate the model (Watson, 2000, personal communication). In addition, the Macaque model's distribution function and saturation deficit slope parameters were both very sensitive (Watson, 1999a). Further problems were also encountered during the application of the Macaque model on the South African, Maritsane catchment (Mpumulanga), as discussed in Chapter 6. It was found that the saturation deficit of individual ESUs increased without constraint, resulting in unrealistically deep water tables. Simulated recharge rates at depth were slow and consequently saturation deficits remained high (Watson, 2000, personal communication).

5.3.2 Explicit Darcian lateral flow component

In the most recent version of Macaque the sub-surface distribution function concept has been replaced by an explicit Darcian component, whereby water is routed from an upslope ESU to the one downslope as a function of transmissivity of the soil and hydraulic head.

Initially water table depth for each ESU is calculated as a function of saturation deficit and porosity of the soil. The saturation deficit, s_{sat} is the amount of water (in metres) required to fill the soil to saturation. When the water table is at the soil surface, the saturation deficit is considered to be zero and when the water table drops below the soil surface, the deficit is positive. Once the water table depth has been determined, the transmissivity of the soil is calculated by integration from the bedrock depth (z) up to the water table depth (d), as a function of four parameters of the theoretical saturated hydraulic conductivity variation curve, i.e. minimum saturated hydraulic conductivity, surface saturated hydraulic conductivity, hydraulic conductivity shape and saturated hydraulic conductivity depth (which is considered depth to bedrock). This is depicted in Figure 5.2.

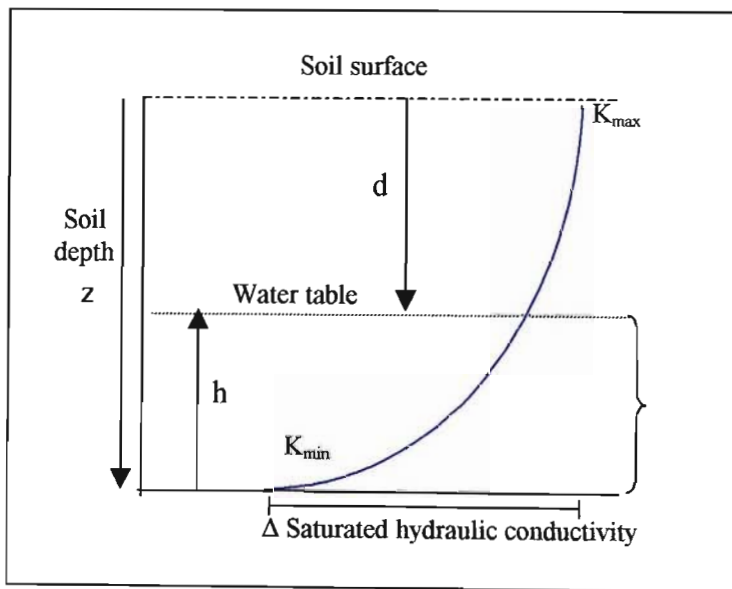


Figure 5.2. Soil transmissivity integrated from the bedrock depth up to the water table depth.

$$T = (h) \cdot K_{\min} + (K_{\max} - K_{\min}) \cdot (1/K_{\text{shape}}) \cdot \text{EXP}(-K_{\text{shape}} \cdot d) \quad \text{Equation 5.3}$$

Where

- T = transmissivity (m²/day)
- K_{min} = minimum saturated hydraulic conductivity (m/day)
- K_{max} = maximum or surface saturated hydraulic conductivity (m/day)
- K_{shape} = rate of exponential decline in saturated hydraulic conductivity with depth
- Z = soil depth (m)
- h = water table depth with bedrock as datum
- d = depth to water table from soil surface

The hillslope is assumed to be square. Although this assumption is never true, it is essential for the calculation of the hillslope width, which is derived from the hillslope area (Watson, 2000, personal communication). Further developments are likely to allow for the calculation of actual hillslope width using terrain analysis routines (Watson, 2000, personal communication). The length of an ESU is derived from the ESU width (which is the same as the hillslope width) and its area. The gradient of the ESU (as calculated from the slope), which is assumed to apply along its length, is used to determine the drop in elevation between successive ESUs. The mean elevation for an ESU calculated from the DEM cannot be used, since it is possible that an ESU may have a higher mean elevation than its neighbouring ESU upslope. Once the depth to the water table of each ESU has been determined, the drop in water table (hydraulic head) between successive ESUs can be calculated.

Macaque uses a description of flow through the soil based on Darcy's law, which states that flow is proportional to the gradient of hydraulic potential. The outflow (per unit area) from each ESU is calculated as the product of hydraulic potential (hydraulic head) and the transmissivity of the soil. This outflow then becomes inflow to the downslope, neighbouring ESU, as illustrated in Figure 5.3a and Figure 5.3b on the following page. Once the lateral movement of water has taken place, the saturation deficit of each ESU is re-calculated and the vertical movement (fluxes) of water within the ESU is initiated.

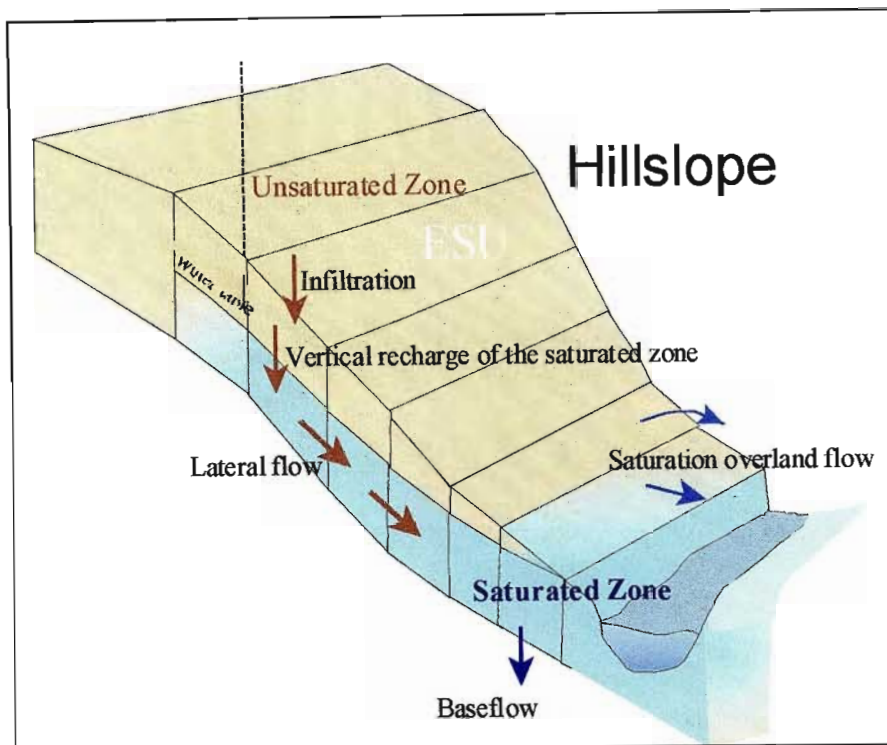


Figure 5.3a. Schematic representation of the horizontal structure of the Macaque model.

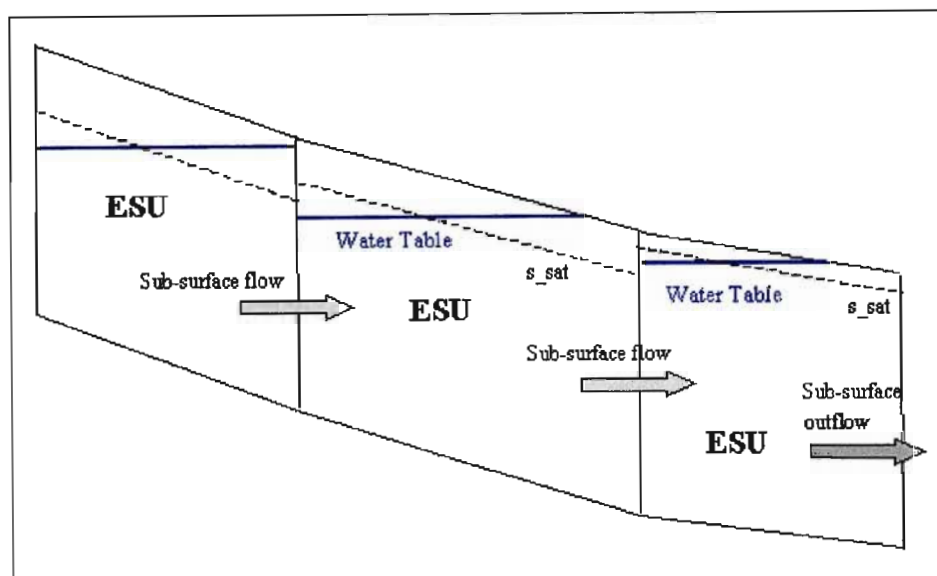


Figure 5.3b. Representation of sub-surface lateral flow in the Macaque model.

5.4 Vertical process representation

In essence the vertical structure of the model represents the processes that occur in a single ESU and which are vertically orientated. The model represents landcover as both canopy cover and understorey cover (see Figure 5.4). However, in this chapter, reference is made

only to canopy processes since understorey vegetation is often absent in South African plantations and was therefore not considered in model applications.

The vertical flux of water is implemented from top to bottom, beginning with precipitation and interception. That portion of precipitation not intercepted (throughfall) falls to the ground where it can infiltrate the soil. Each ESU is subdivided into two soil zones, representing unsaturated and saturated soil respectively. The interface between these is the water table, which moves up and down in response to vertical water movement, and inflows and outflows (lateral movement) from and to ESUs above and below, within the hillslope (Peel *et al.*, 2000), as discussed in Section 5.3.

If an ESU is partly saturated, then the water falling (throughfall) on the saturated portion becomes saturation excess flow. Water falling on the unsaturated portion infiltrates into the soil. Recharge from the unsaturated zone to the saturated zone follows and finally baseflow is released from the saturated zone. Once these fluxes are completed, transpiration and evaporation from the soil is calculated.

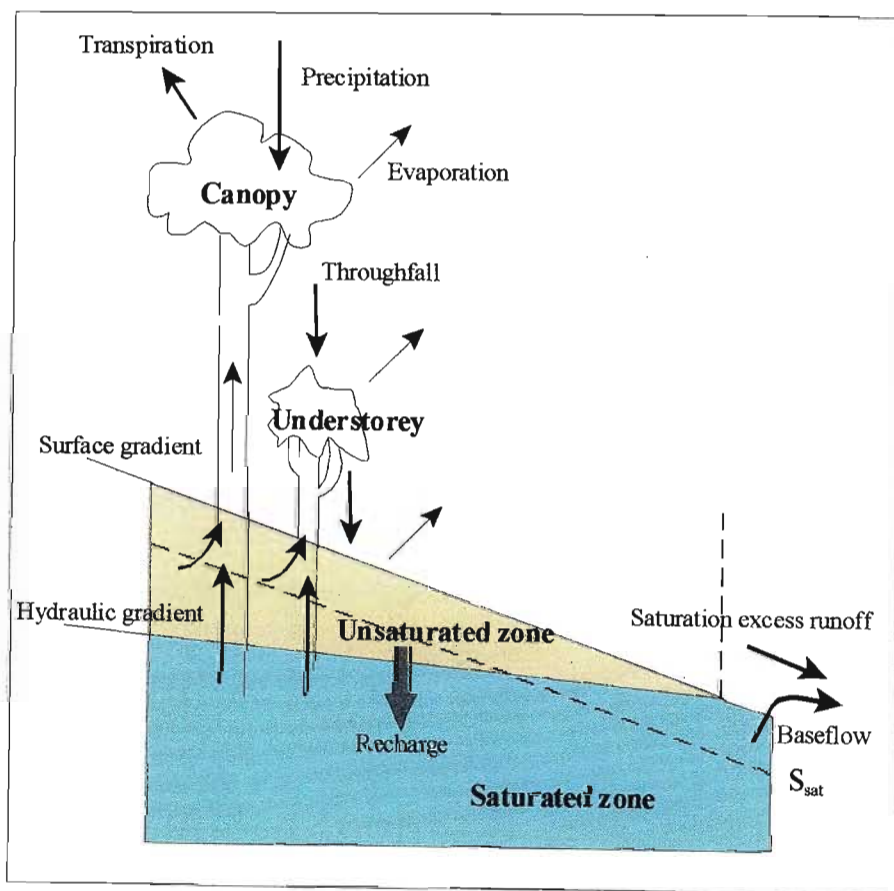


Figure 5.4 Schematic representation of the vertical structure of the Macaque model.

5.4.1 Precipitation and interception

Precipitation is the most sensitive input variable in any hydrological model. In several Macaque applications it has been shown that adjusting the precipitation scaling parameter was the single most effective means of improving simulations (Peel *et al.*, 2000). The technique used in Macaque to account for spatial variability in rainfall is very similar to that typically applied in *ACRU* applications. A station (known as the base station) with long continuous record is selected within or in close proximity to the catchment. For every month where a site has data, its monthly precipitation total is divided by the total for the corresponding month from the base station. An average ratio (mean monthly precipitation index) is then calculated for each rain gauge. A rainfall surface of these derived ratios or mean monthly precipitation index (MMPI) values is constructed by interpolating the MMPI values for each gauge (Watson, 1999a).

Rainfall or snow falling on the canopy is partially intercepted and vaporised back into the atmosphere. The canopy can intercept and store water up to a maximum level defined by a function of the LAI of the vegetation. The amount of stored water that is vaporised back into the atmosphere is limited by potential evaporation, and hence by the canopy absorbed net radiation and the reference level vapour pressure deficit (VPD) (Watson, 1999a). Excess water falls through the canopy and is either intercepted by the understorey or in the case of no understorey, falls to the ground.

5.4.2 Saturation excess runoff and baseflow

In Macaque runoff and baseflow exfiltration is generated from each ESU and not from the hillslope as a whole, which results in a more spatially realistic implementation of streamflow production (Watson, 1999a). Strictly speaking, saturation excess runoff and baseflow are not vertically orientated processes but have been incorporated in this section because they occur in succession to the processes described in Section 5.4.1

The releases from each individual ESU are illustrated in Figure 5.4. When water flows out of the groundwater into the stream (for baseflow to occur), it does so as a gradient diffusion process, with the hydraulic gradient resulting in a pushing force and the soil resisting movement measured by hydraulic conductivity (Watson, 2000, personal communication).

Although it is possible to measure hydraulic gradient using piezometers or observation wells it is costly and time consuming to install and maintain such equipment and monitor change. The developers of the Macaque model therefore derived a hydraulic gradient parameter ($p_ratio_hydraulic_to_surface_gradient$) defined as the ratio of the hydraulic to the surface gradient. In wet catchments with shallow soils the hydraulic and surface gradients correspond closely and the ratio is close to unity (Watson, 2000, personal communication) while in catchments with deep soils and an unresponsive water table the ratio is small.

The ratio of hydraulic to surface gradient parameter controls both the rate of baseflow release from an ESU and the proportion of the ESU that is saturated from the bedrock to the surface, which in turn determines the amount of saturation excess flow during a rainfall event. In equation 5.4 it is shown how this parameter determines the saturation deficit at which the partial surface saturation of an ESU begins ($S_{sat,0,surface}$).

$$S_{sat,0,surface} = \left(1 - \frac{\nabla_{hydraulic}}{\nabla_{surface}}\right) \Delta h \times \theta_{max} \quad \text{Equation 5.4}$$

where

- $S_{sat,0,surface}$ = the zero saturation deficit
- Δh = change in elevation for an ESU
- $\frac{\nabla_{hydraulic}}{\nabla_{surface}}$ = ratio of hydraulic to surface gradient
- θ_{max} = maximum available water

The saturated portion of the ESU will therefore increase as the actual saturation deficit continues to decrease beyond the zero saturation deficit ($S_{sat,0,surface}$). The greater the saturated portion the larger the contribution of saturated excess flow to streamflow.

The amount of baseflow released is calculated by multiplying the saturated portion of the ESU, the ratio of hydraulic to surface gradient, the surface saturated hydraulic conductivity and the surface gradient of the ESU.

In TOPMODEL applications it is assumed that the hydraulic gradient of the groundwater equals the gradient of the water table and that the gradient of the water table equals the

gradient of the surface of the soil. The Macaque model's ratio of hydraulic to surface gradient parameter would equal 1.0 in such cases. For Macaque applications, however, the ratio cannot be unity. In recent Macaque applications in the Maroondah catchments in Australia a strong correlation between this ratio and catchment area were observed (Peel *et al.*, 2000) as is discussed in Chapter 6.

5.4.3 Soil water availability and rooting patterns

The water holding capacity (maximum available volumetric water content) of soils and the depth to which roots are able to extract water from the soil profile are important factors controlling the amount of water available to the trees for transpiration, particularly in water limited environments. It has been shown that the absence of spatial soils information affects the accuracy of water yield predictions and the extent to which this occurs arises from the way in which soil water controls plant water use in limited environments (Peel *et al.*, 2000). For example, if the catchment is assumed to have a high water holding capacity when in actual fact parts of the catchment are characterised by shallow soil with low water holding capacities then the model would fail to simulate the plant water stress experienced by the trees in these areas. When trees experience plant water stress (typically during the dry season), evapotranspiration is reduced and the impact of forest clearing on a catchment is diminished (Peel *et al.*, 2000). The deep rooting systems of trees, however, do enable them to extract soil water from deep in the soil profile (Dye *et al.*, 1997) and this enhances their ability to maintain transpiration rates during the dry winter and drought periods.

Water falling on the saturated portion of the ESU becomes saturation excess runoff. There is no infiltration process representation in Macaque and the remaining precipitation is added to the unsaturated zone. Other than saturation excess runoff there is no other surface runoff representation in model. Hereafter recharge from the unsaturated zone to the saturated zone takes place and a fraction is released from the saturated zone as baseflow. The recharge of the saturated zone is calculated using the Clapp and Hornberger (1978) estimation of unsaturated hydraulic conductivity of the soil at the water table interface. The Clapp and Hornberger (1978) method uses texture parameters, such as percentage clay and percentage sand to derive hydraulic conductivity. In Macaque, the Clapp and Hornberger method (1978) of estimating unsaturated hydraulic conductivity using texture parameters, was preferred to other

conventional methods, such as the Van Genuchten model, since texture parameters are more widely available.

Once the soil moisture fluxes, such as recharge and baseflow release are complete, the trees are able to extract water for transpiration. The contribution of each soil zone to transpiration is determined according to two criteria:

- the relative availability of water from the two zones and,
- a parameter quantifying the extent to which the transpiration species is phreatic (able to draw water from the saturated zone).

Although the assumption is made that trees can potentially draw water from below the water table, there is a lack of documentation quantifying the extent to which trees are able to draw water from this saturated zone.

Based on these criteria different weights are assigned to control extraction of water from the saturated and unsaturated zones. The respective weights are then divided by their sum and multiplied by total transpiration to give the total transpiration drawn from each zone. The water available for transpiration in the unsaturated zone is dependant on the rooting depth and the actual volumetric water content of the soil. If the tree roots extend beyond the water table then the abstraction of water from the saturated zone is dependent on the depth of the roots beyond the water table and the maximum volumetric water content. If the transpiration requirement from the unsaturated zone is greater than the water available for transpiration, then transpiration is reduced to match the availability. If the transpiration requirement from the saturated zone is greater than soil water availability, an error is reported.

5.4.4 Transpiration and soil evaporation

The Macaque model represents transpiration from both the canopy and understorey vegetation, however in this chapter reference is only made to canopy processes since an understorey is largely absent in South African commercial plantations. There are 23 climatic and plant physiological parameters used in Macaque to reduce or enhance transpiration by the forest canopy, resulting in a far more complex representation of the transpiration process than that utilised by the *ACRU* model.

It has been shown that transpiration from coniferous forests is strongly dependent on vapour pressure deficit and canopy conductance (Whitehead and Jarvis, 1981). Other factors affecting the transpiration rate of plants include solar radiation and the soil water availability (Lindroth, 1993). In Macaque, transpiration is calculated using the Penman-Monteith equation (Equation 5.5) in which both these influences are represented as:

$$E_c = \frac{\Delta_{esat} \frac{\max(0, R_{trans,c})}{dayl} + c_p \rho_{air} D_{ref} \frac{1}{r_{aero,c,ref}}}{\lambda \rho_w (\Delta_{esat} + \gamma (1 + \frac{1}{r_{aero,c,ref} gc}))} \quad \text{Equation 5.5}$$

Where

- E_c = canopy transpiration rate
- Δ_{esat} = rate of change of saturation vapour pressure with temperature
- $R_{trans,c}$ = net radiation absorbed by canopy and used for transpiration
- c_p = specific heat of air at constant pressure
- ρ_{air} = density of air including water vapour
- D_{ref} = mean daytime vapour pressure deficit at reference level above canopy boundary layer
- $r_{aero,c,ref}$ = aerodynamic resistance to vapour transfer between canopy and reference level
- λ = latent heat of vaporisation of water
- ρ_w = density of liquid water
- γ = psychrometer constant
- gc = conductance to vapour transfer of entire canopy
- $dayl$ = number of seconds in a day for which the sun is above the horizon

The meteorological components of the equation are driven by humidity (specified as the vapour pressure deficit) and net radiation. Estimates of vapour pressure deficit, are obtained from Linacre's (1992) temperature based equations. Solar radiation is derived from daily temperature ranges, the time of year and topography (slope and aspect) of the terrain while these together with absorbed long wave radiation, are used to calculate net radiation.

The plant physiological controls of transpiration used in the Penman-Monteith equation are canopy conductance and aerodynamic resistance of the canopy. There are a number of factors than restrain or enhance leaf conductance (canopy conductance divided by LAI) as summarized in Figure 5.5. The effect of decreasing soil water availability on leaf conductance, for example, is correlated by pre-dawn leaf water potential (LWP). As the soil dries out, the LWP is reduced and leaf conductance decreases. Leaf conductance is further scaled by factors such as VPD, solar radiation, temperature (temperature range, night minimum temperature and optimal temperature) and atmospheric carbon dioxide concentration, using what is known as the product method. According to this method, simple optimality functions (ranging from zero to one) are derived for each controlling variable. The functions are then multiplied together to form a single optimality value, which is used to increase or decrease the leaf conductance.

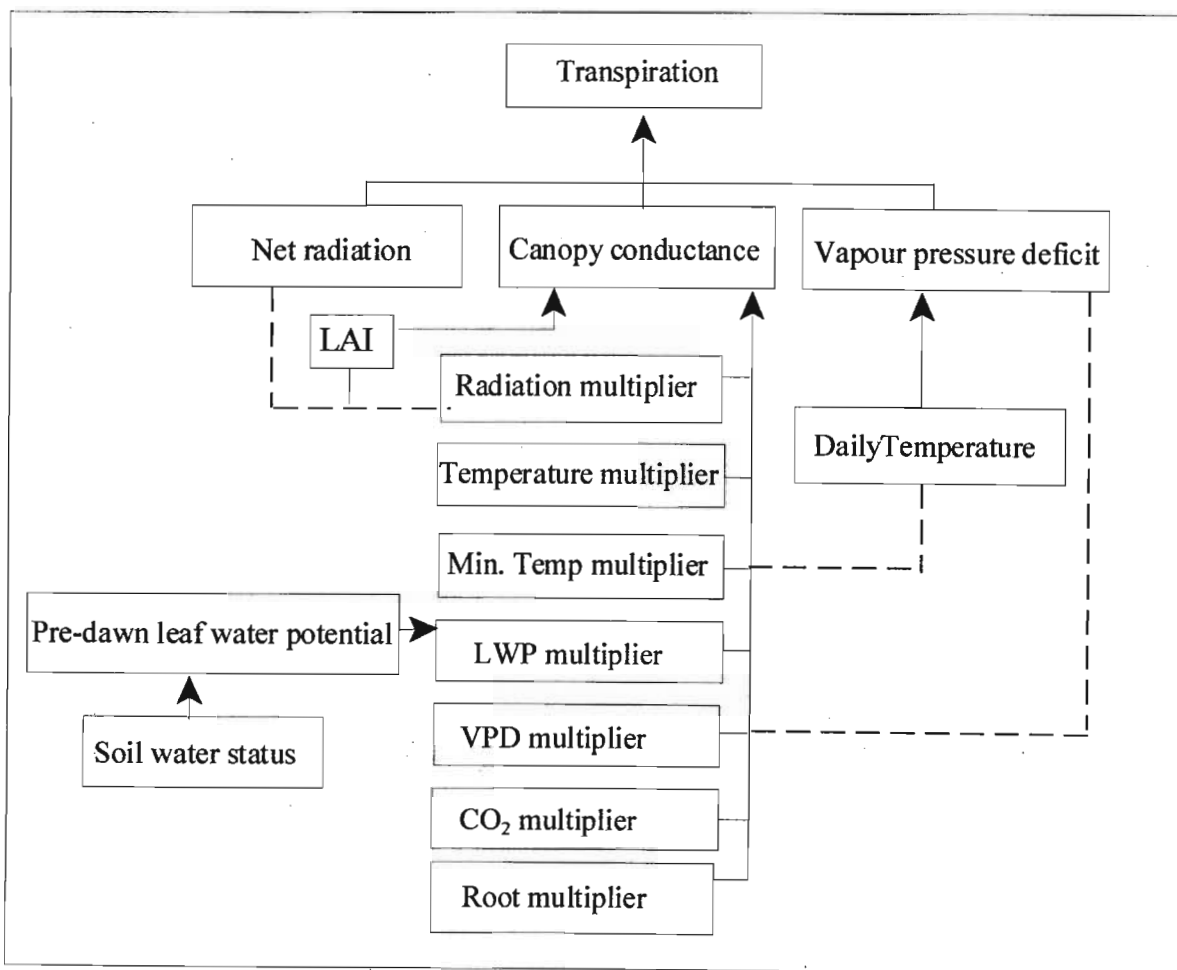


Figure 5.5. Transpiration controls of the Macaque model (after Watson, 2000).

Unlike *ACRU* where evaporative demand is linked directly to LAI and reference potential evaporation, LAI values in Macaque only control transpiration indirectly through the

calculation of canopy conductance, which is the product of LAI and leaf conductance and used in the calculation of radiation absorption by the canopy.

This approach is highly complex with numerous controls on transpiration being considered. It has, however, been noted that feedback within the transpiration/conductance systems of large eucalypt forests may be present, with the result that the conductance is always such that transpiration occurs at some potential, or at least at solely radiation-determined rates (Watson, 1999a).

Given the data limitations encountered in South Africa, the potential for using such a detailed approach to calculate evapotranspiration may not be feasible in South Africa. Although tree transpiration is seasonally affected by radiation (daylength), it is most highly correlated to vapour pressure deficit (VPD). The response to VPD rapidly tends towards a relatively constant transpiration rate, which is maintained by changes in canopy conductance - unless limited by soil water availability (Dye, 2000, personal communication). Thus, the accurate simulation of VPD and soil water available to trees is of great importance. Dye (1996), however, found it impractical to simulate the water balance of deep rooting systems at a site in Mpumalanga, South Africa for the purposes of estimating non-potential transpiration rates, since there were uncertainties concerning the depth of the rooting system, the soil water recharge mechanisms and the water retention characteristics of the deep subsoil strata.

In order to evaluate the performance of the Macaque model and its suitability for application in South Africa, the model was applied to a selection of forested catchments. In Chapter 6 the criteria for catchment selection are provided followed by a discussion on the model application and simulation results.

* * *

In Chapter 5 the Macaque model was described which served to illustrate the differences in modelling approaches between the Macaque model and *ACRU* model. Although both models are physically-based distributed models it is evident that Macaque's process representation is more detailed than that of the *ACRU* model. The Macaque model application to South African catchments described in the following chapter serves to highlight whether this additional model complexity can be supported by the availability of data.

6. APPLYING THE MACAQUE MODEL TO SOUTH AFRICAN CATCHMENTS

Applying the Macaque model to South African catchments and establishing whether the data requirements of the model can be met is the first step in establishing whether it is feasible to apply the model or extract components of the model for use in hydrological modeling in South Africa. Secondly, it is important to verify the model in order to establish whether it provides an accurate representation of the processes that are dominant in forested catchments. In the following section the criteria used to select catchments for the application and verification of the Macaque model are briefly discussed. This is followed by a detailed description of the configuration of the model for these catchments.

6.1 Catchment selection

The choice of catchments used in the application and verification of the Macaque model was largely dependent on data availability. The verification of any hydrological model requires reliable and long term climatic and streamflow data, as well as process measurements, such as estimates of transpiration and water table depths, for internal verification. Since the aim of this exercise was to test the ability of the model to simulate processes dominant in forested catchments, a further requirement was that these data needed to be collected for catchments for which the dominant land use was forest plantation. The model's input requirement of high-resolution digital elevation data placed a further restriction on the selection of suitable catchments.

Two catchments were identified as good candidates for initial application of the Macaque model in South Africa. These were the relatively large 212 km² Maritsane catchment and the small Witklip catchment (1.8km²), both situated in Mpumalanga Province as shown in Figure 6.1. The *ACRU* model had also been configured for these two catchments thus providing the opportunity to compare the simulation results of these two models.

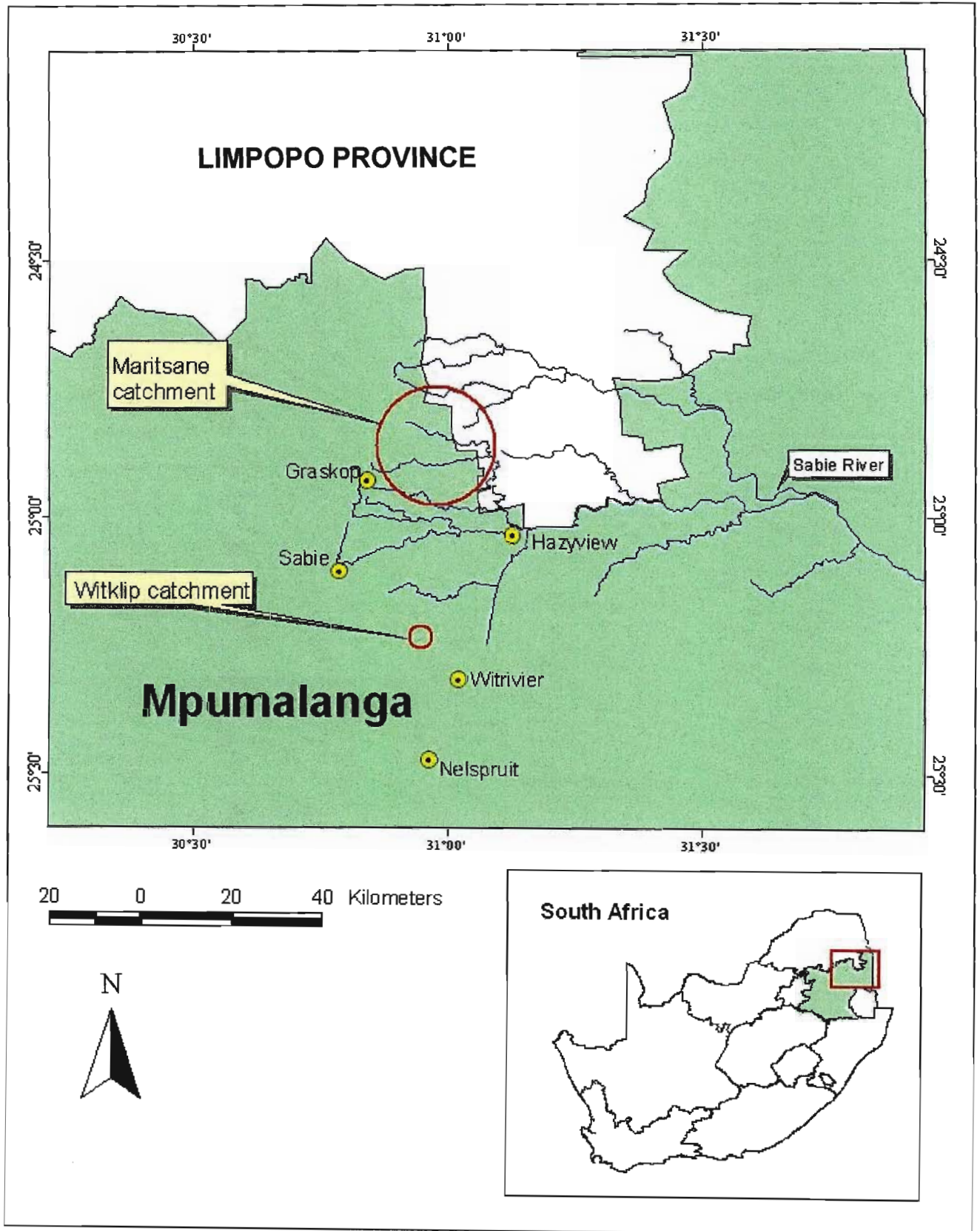


Figure 6.1. The locality of the Maritsane and Witklip catchments in the Mpumalanga Province.

6.2 The Maritsane Catchment

The Maritsane catchment is situated east of the town of Graskop in the Mpumalanga Province (Figure 6.1). The Maritsane River is a tributary of the Sabie River, which extends from the Eastern Drakensberg escarpment and continues into Mozambique. The Maritsane catchment area is 212 km² and extends upstream from the Department of Water Affairs and Forestry (DWAF) weir X3H011.

6.2.1 Description of Maritsane catchment

The catchment comprises mainly commercial pine forest plantation as illustrated in Figure 6.2, whilst the riparian areas and upper catchment escarpment are comprised of indigenous forest and scrub. Elevations range from 750m ASL at the catchment outlet to 1800m ASL at the top of the Drakensberg escarpment. The catchment is characterised by largely gentle terrain fringed by a steep escarpment. The Mean Annual Precipitation (MAP) is 1200mm.



Figure 6.2. Maritsane forested catchment

The Maritsane catchment is typical of forested catchments in the area and throughout South Africa, where commercial plantations consist of a large number of compartments with varying tree ages and rotational lengths and where management practices such as thinning are applied. Unlike the smaller research catchments, such as the Witklip catchment, reliable climatic, landcover and streamflow data were not always available. Therefore, the application of the model on this catchment offered some insight into the data limitations typically associated with the application of physically based models at a large scale in South Africa.

6.2.2 Model configuration and input parameters

The Macaque model requires a large number of input parameters due to the explicit representation of forest processes. The parameters required by the model and the parameter values selected for the Maritsane catchment are listed in Table A1 of Appendix A. A lack of detailed documentation of the latest version of Macaque made it difficult to assign parameter values since many of the added or enhanced processes and parameters have not as yet been defined. As a result, the model user is reliant on the source code to gain an understanding of these model processes and the parameters that are required.

Digital Elevation Model

At the time of the model configuration DEMs with resolutions finer than 200m were not yet readily available at a reasonable cost for all areas of South Africa. In the case of the Maritsane catchment, it was necessary to derive a finer resolution elevation model from digital contour data obtained from the Surveyor General. Initially the contour data was converted to a point coverage and the points were then interpolated using an inverse distance weighting technique to create a 25 by 25m DEM. On closer inspection of the DEM significant “contour steps” were evident in the DEM. This phenomenon may arise during interpolation as a result of the high density of data along the original contours. The high density of points along the contours resulted in a DEM with uniformly sloping terrain between the contours with a decrease in elevation occurring at the contour line. To rectify this problem an alternative interpolation technique in the ArcInfo GIS package, known as TOPOGRIDTOOLS (Hutchinson, 1989; Hutchinson, 1996), was utilised. This tool uses a thin plate spline interpolation technique to interpolate between contours and other elevation features, such as spot heights and lake depressions. Although this interpolation tool resulted in an improved DEM, not all the problem features were eliminated and finally it was suggested that the DEM resolution be halved to 50x50 metres to smooth out these effects (Watson, 2000, personal communication).

A considerable number of parameters and variables used in Macaque are derived from the DEM using Tarsier’s digital terrain analyst (DTA) and consequently these derived parameters are all affected by the choice of DEM resolution and DEM accuracy. Hence, inexpensive, accurate elevation data is essential if topographically sensitive models such as Macaque are to

be applied. All other spatial input parameters used in Macaque are represented as rasters (grids) corresponding to the raster size and resolution of the DEM.

Spatial disaggregation using Digital Terrain Analyst (DTAs)

Three spatial levels were chosen for the configuration of the model for the Maritsane catchment. The catchment represented the world level (top level) and contained a single spatial unit. The second spatial level contained the hillslopes and the Elementary Spatial Units (ESUs) were the third level.

Catchment delineation was achieved using the DEM. Once the DEM had been created and filled using the pit and flat filling routines in the DTA, the stream network was delineated using single flow direction algorithms. The stream cell representing the catchment outlet (weir X3H011) was identified and the overall catchment area and catchment area of each stream segment were then demarcated using a multiple flow direction algorithm to calculate upslope area (Watson, 2000, personal communication). The catchment area of each stream segment was then divided into hillslopes (illustrated in shades of blue in Figure 6.3), which are defined as areas bounded by stream segments (shown in white in Figure 6.3), stream intersections and ridgelines (Watson, 1999a).

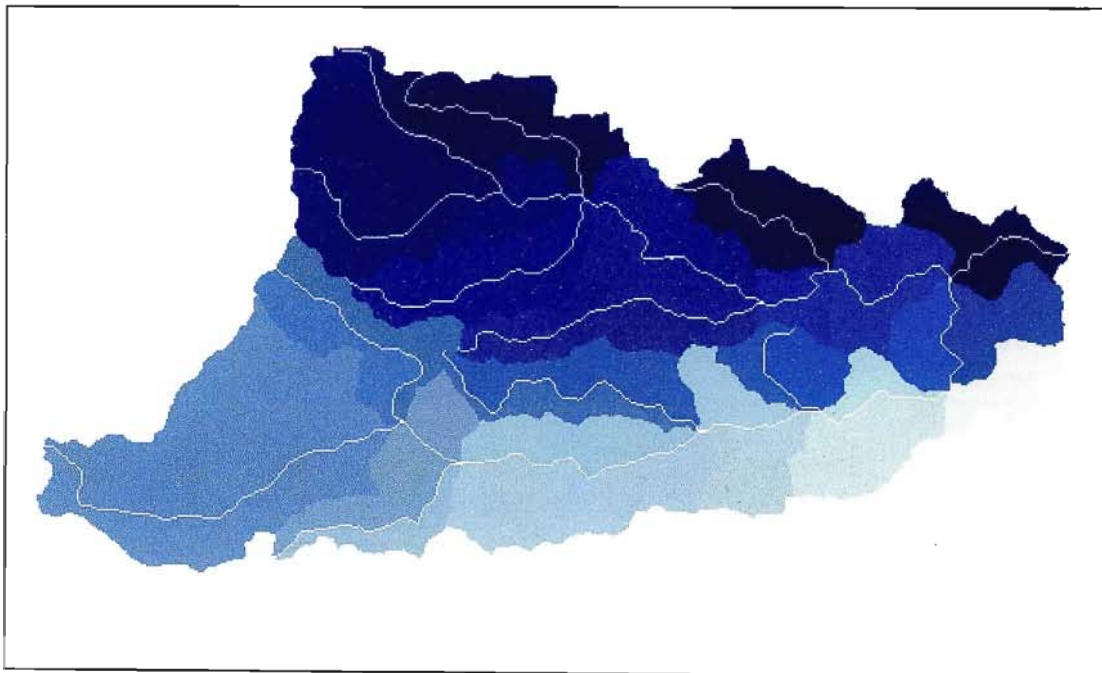


Figure 6.3. Hillslope delineation for the Maritsane catchment using Tarsier's (1999) digital terrain analyst.

The ESUs were defined as the areas occupied by specific intervals of the wetness index (WI). As discussed in Chapter 5, the most recent version of Macaque no longer has an association between WI and saturation deficit of an ESU and any method may be used to delineate the ESUs. For this simulation the TOPMODEL topographic wetness index namely, $\ln(a/\tan B)$, was used to define the ESUs and this choice of ESU delineation was based on the fact that firstly, the calculation of topographic WI is the only automated procedure within Tarsier to define ESUs and secondly this WI had previously been used in a number of Macaque applications. The intervals of the wetness index which were used to define separate ESUs within each hillslope are listed in Table 6.1 and illustrated in graphical format in Figure 6.4.

Table 6.1: Intervals of the $\ln(a/\tan B)$ wetness index which were used to define separate ESU.

Interval	ESU number
1-3	1
3-6	2
6-9	3
9-12	4
12-15	5
15-24	6

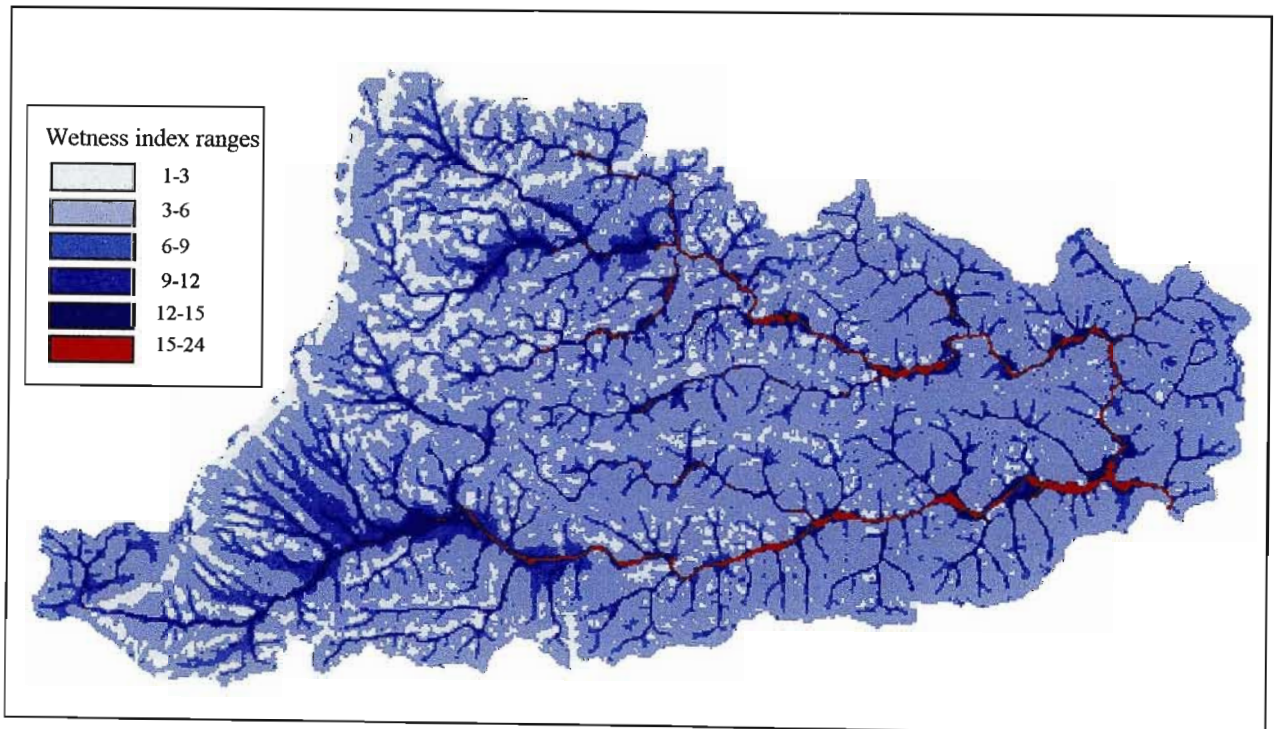


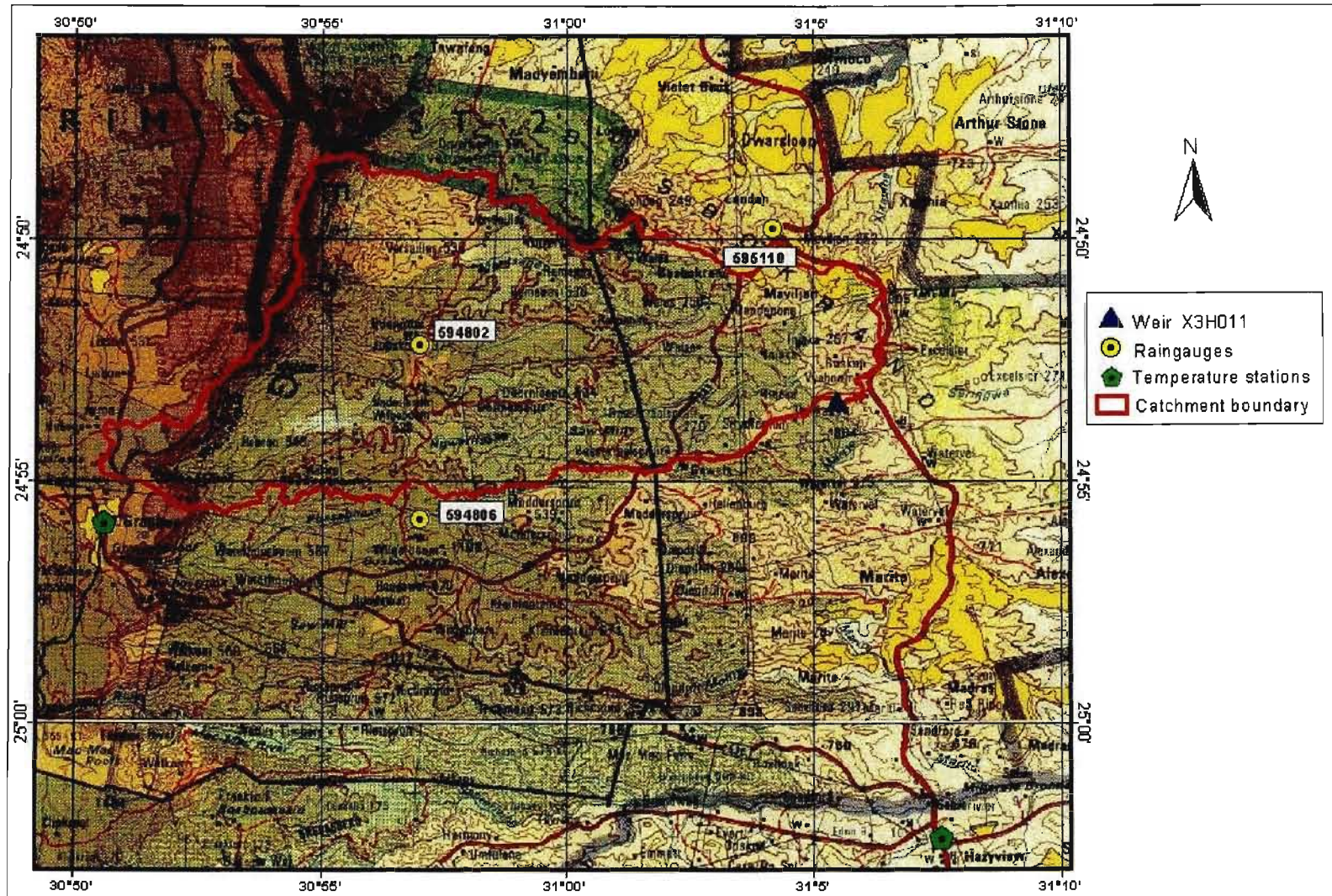
Figure 6.4. Graphical representation of ranges of wetness indices used for the delineation of ESUs for the Maritsane catchment.

The low WI ranges (ESU number 1) represent the upper portion of the hillslope, whilst the higher WI ranges (ESU number 6) represent the base of the hillslope or the permanently saturated areas. The derived wetness indices for the Maritsane catchment ranged between 3 and 24. A qualitative visual assessment of GIS output suggested that WIs of greater than 15 were associated with permanently saturated streams and therefore WI between 15 and 24 were considered a homogenous unit (ESU number 6).

Daily rainfall data

Rainfall and spatial variability of rainfall are the most sensitive input parameters into Macaque, as is the case with most hydrological models (Watson, 1999a). The Macaque model allows for different methods for representing spatial variability in rainfall as discussed in Chapter 5. The mean monthly precipitation index (MMPI) method, using one driver station, was selected for the Maritsane model simulation. In this approach a base station, a daily rainfall station with a long continuous record within or near the catchment, is selected. The monthly timeseries of the base station is then compared to the monthly rainfall of other rainfall stations within or in close proximity to the catchment. By dividing the base station monthly rainfall by that of each of the other stations and calculating the mean of the resulting quotients, a MMPI value for each station is derived. The MMPI values are then interpolated to create a surface of MMPI values.

In the Maritsane catchment there was only one rainfall station and a limited number of stations in the region as shown in Figure 6.5. It was therefore decided to select a single base station in the catchment and compare its median monthly rainfall to that of the median monthly rainfall 1'x 1' grids for South Africa (Dent *et al.*, 1989). The 1'x 1' median monthly rainfall grids were derived from daily and monthly rainfall records from 9049 stations using a combination of multiple linear regression and linear interpolation techniques (Dent *et al.*, 1989). The rainfall stations identified within or in close proximity to the catchment include South African Weather Bureau (SAWB) stations A0594802 (960m), A0594806 (980m) and A0595110 (822m).



6.5. Topographic map of the Maritsane forested catchment showing location of temperature stations and raingauges.

Station A0594802 was selected as the driver (base) station due to its long continuous record and its location within the catchment. The station's rainfall files were abstracted from the Computing Centre for Water Research (CCWR) database and patched using the Inverse Distance Weighting interpolation technique developed by the erstwhile Department of Agricultural Engineering (now School of Bioresources Engineering and Environmental Hydrology).

The Macaque model only uses a single MMPI surface for all months of the year and seasonal differences in spatial rainfall distribution are not accounted for. It was therefore necessary to select one of the twelve median monthly precipitation grids to derive a precipitation index. Since the majority of precipitation (>63%) falls during the summer months it was decided to use December's median monthly rainfall grid to derive a precipitation index. Each of the gridded median monthly rainfall values was then divided by the base station's December median monthly rainfall of 200.6 mm to derive a monthly precipitation index. The result was a grid surface of ratios at a resolution of approximately 1600m (one minute of a degree). Since the raster of median monthly precipitation indices had to match that of the DEM, the resolution had to be increased from 1600m to 50m. This was achieved by dividing the 1600m grid into 50m cells using the RESAMPLE command in Arcinfo. This method applies the nearest neighbour interpolation technique.

The limitation of the MMPI approach to representing spatial variability of rainfall is that temporal changes in spatial distribution of rainfall during the year are not considered. In the Macaque application by Peel *et al.* (2000) it was shown that the multiple linear regression technique (MLR) was able to capture more of the variability in precipitation than the MMPI method and therefore resulted in improved simulation results. However, it was necessary to apply a greater precipitation scalar for the simulation results to meet the desired modeling objective, which shows an inherent inaccuracy in the MLR method (Peel *et al.*, 2000). The approach to representing rainfall variability used in the *ACRU* model provides a more accurate representation of the changes in spatial distribution of rainfall during the year than Macaque's MMPI method. In the *ACRU* modelling approach, precipitation index values (PPTCOR) are assigned to each subcatchment and each month of the year and thereby both the variability in rainfall over the catchment and the changes in spatial distribution of rainfall during the different seasons are accounted for.

Daily temperature data

Daily maximum and minimum temperature data drive a number of processes within the Macaque model. Maximum temperature in particular, exerts a great influence on model operation. The Macaque model only requires one temperature base station and the temperature of each ESU is then adjusted according to the temperature lapse rate associated with an increase in elevation. Daily maximum and minimum temperatures were available for stations at Graskop (west of the catchment, altitude 1440m) and Hazyview (southeast of the catchment, 480m), as illustrated in Figure 6.5. No temperature stations are situated within the catchment. The Graskop and Hazyview stations were used to calculate the mean temperature lapse rate with altitude increase. The mean lapse rate was calculated to be $7^{\circ}\text{C}/1000\text{m}$ for maximum temperature and $4.1^{\circ}\text{C}/1000\text{m}$ for minimum temperature. However, the School of Bioresources Engineering and Environmental Hydrology has developed routines to derive daily minimum and maximum temperature for any location in South Africa from existing temperature stations within a specified radius (Schulze and Maharaj, 1996). It was resolved to use these derived daily temperature data for the centroid of the catchment at an elevation of 990m to drive temperature functions within the model.

Incident solar radiation data

In addition to temperature data, the model requires three radiation constants to calculate atmospheric transmission using the Bristow and Campbell method (1984). Atmospheric transmission is defined as the ratio between extra-terrestrial radiation that is attenuated by various atmospheric components and the actual radiation incident above a horizontal canopy (Watson, 1999a). These radiation constants are adjusted (calibrated) by comparing the modelled atmospheric transmission to the “observed” transmission. However, atmospheric transmission cannot be measured, but may be derived by dividing the measured daily global incident solar radiation by estimates of extra-terrestrial solar radiation produced according to Dingman (1994). Daily global incident solar radiation, however, is not measured at many sites in South Africa and the only radiation data available were daily sunshine hours measured at Skukuza, approximately 70 kilometers east of the Maritsane catchment. Sunshine hours can be translated into incident solar radiation using the Angström equation (1929), yet the result would be that intermediate models are used to derive appropriate input parameters. It was therefore decided to use the default parameters in Macaque.

Soils data

The Macaque model requires spatial data of soil physical properties, water retention characteristics, saturated hydraulic conductivity and soil depth. Detailed spatially representative soil data, however, are often not available and this was the case for the Maritsane application. The availability of detailed soils data has also been reported as a problem in other Macaque applications (Peel *et al.*, 2000).

The most comprehensive set of mapped soils information available to hydrologists in South Africa are the Land Type maps compiled by the former Soil and Irrigation Research Institute (SIRI), now the Institute for Soil, Climate and Water (ISCW). These Land Type maps were compiled by superimposing climate maps over a pedosystem map and identifying Land Type units (Schulze *et al.*, 1995d). A Land Type inventory was then compiled using data collected through field surveys (Schulze *et al.*, 1995d). This Land type database for South Africa, however, was not specifically developed for use in hydrological modelling and does not describe those soils characteristics required by the Macaque model for soil flux simulations. The information contained in the database included percentage soil series per terrain unit, soil textural data and soil depth measurements to a threshold depth of 1.2 metres as shown in Table 6.2.

Table 6.2: An example of soils data for Land Type Ab37 inventory.

SOIL TYPE	TEXTURE	SOIL DEPTH (mm)	TERRAIN UNIT (%)		
			1	3	5
			10%	85%	5%
Hu17	SaCl	500>1200	85	80	40
Gs19	Cl	300-500	10	5	10
Oa370		>1200	-	10	40
Ka10	SaCl	300-500	-	-	10
Rock			5	5	-

With the exception of soil texture and soil depth, none of the other soil properties relate to the Macaque model's required parameters. Furthermore, soil depth in the Macaque model is defined as the depth of soil from the surface to the bedrock-soil interface and since soils were not surveyed beyond the auguring depth of 1.2 metres these soil depth were not suitable as input into the Macaque model. In the case of the *ACRU* simulation (Pike *et al.*, 1997), this Land Type information had been "translated" into hydrological soils input properties as

required by the *ACRU* model using the AUTOSOILS program developed by Pike and Schulze (1995). From these derived parameters, porosity and the drained upper limit, which were further variables required by the Macaque model, were obtained. However, other parameters required by Macaque, such as surface and minimum saturated hydraulic conductivity, were still unknown.

It was therefore necessary to use information derived from soil data obtained from an extensive soil survey undertaken at Frankfort State forest, in the Sabie region (Lorentz *et al.*, 1996). From the Land type maps of the Maritsane catchment it was evident that Hutton soils (Hu17) are the dominant soil type. This pattern is consistent with that of the Frankfort site. The soil parameters required by Macaque were thus obtained from the Frankfort soil survey and included saturated soil moisture content and saturated hydraulic conductivity data that were measured at depths of 1.5-1.7 metres at the Frankfort sites. Soil texture data (fraction of clay and sand) were also obtained from the results of this survey. Soils depths of eight metres were measured at the Frankfort site (Lorentz *et al.*, 1996), but the variability of soil depths in the Maritsane catchment remained unknown. Consequently, it was necessary to assume a uniform soil depth. A “median” depth of four metres was input for the initial simulation: an unsatisfactory yet essential decision. This assumption was known to be incorrect in some parts of the catchment as the steeper slopes were reported as being shallow soils (<1.0 metres), but without performing extensive time consuming and expensive soil surveys of the catchment it was not possible to input spatially variable soil depths.

The lack of soils data and the assumption made that there was no spatial variation in soils results in the soil parameters essentially operating as calibration parameters in the model. In the sensitivity analysis provided in a later section of this chapter an indication of the sensitivity of the Macaque model to changes in soil depth and saturated hydraulic conductivity as input parameters is provided.

Leaf Area Index

The Macaque model can represent both spatial and temporal changes in LAI. Temporal changes in LAI reflect the variations in LAI with increasing vegetation age. The equation used in Macaque to approximate the changes in LAI with increasing forest age was developed from both ground measurements and remotely sensed values of LAI. The equation uses six

parameters to adjust magnitude and timing of peak, decay and climax LAI values. The equation was, however, developed for *Eucalyptus regnans*, indigenous to Australia, that reach ages of 200 years. The model developer, therefore, advised that these curves were not applicable for the representation of South African plantation species in the Maritsane catchment (Watson, 2000, personal communication). Alternatively the use of LAI look-up tables was suggested. Using this method the model user creates a file containing an LAI value, which is used on specified days to replace the fixed LAI input value for the simulation. A lookup file is created that contains the year and day of year on which the new LAI value replaces the existing LAI value. The implication is that a separate file needs to be created for each change in LAI and the lookup file then needs to reference these LAI files for each day of the simulation period. This approach is impractical for catchments, such as the Maritsane catchment where there are large variations in both spatial and temporal LAI values, due to the variability in age and management practices employed. The dynamic file used in the *ACRU* model is better suited to represent changes in both LAI, rooting depths and other physical characteristics of trees that differ with age and between species.

Although the age and species data of commercial compartments were available for sections of the catchment, there was a considerable amount of data that was unavailable. Commercial plantations were therefore identified from the landcover database derived from a LANDSAT TM satellite image of 1993 by the School of Bioresources Engineering and Environmental Hydrology. It was therefore decided to use fixed LAI for trees of an intermediate age which represented the average LAI over a rotation period. Although the catchment is largely afforested, other land uses within the catchment were also identified from 1993 LANDSAT TM satellite image. An LAI of 3.2 and 3.5 for pines and eucalypts of intermediate age (Summerton, 1995) were used, respectively for the forest plantation areas.

Transpiration parameters

The Macaque model uses the Penman-Monteith equation to calculate transpiration and evaporation as described in Chapter 5. Transpiration, as calculated by the Penman-Monteith, is very sensitive to leaf conductance estimates, particularly for forests due to their high aerodynamic conductance (Kelliher *et al.*, 1993). Consequently, a reduction of leaf conductance from the maximum can result in significant decreases in the canopy transpiration rate. Maximum leaf conductance has been widely recorded and it has been found that there is

little difference between maximum leaf conductances between different woody species (Körner, 1993). There are, however, a number of environmental factors, such as vapour pressure deficits, leaf water potentials and temperature that affect the leaf conductance. These influences differ between vegetation types and have been widely researched and recorded for coniferous forests (Cain, 1998; Kelliher *et al.*, 1993; Jarvis, 1980). Nonetheless, the findings from these research efforts and the derived relationships between the different environmental factors and leaf conductance do not directly translate into input parameters required by the Macaque model. Therefore, the estimation of these parameters remains partly qualitative and subjective.

A further input required by the Penman-Monteith equation for the calculation of transpiration is aerodynamic resistance. Aerodynamic resistance is a conceptual measure of the degree to which factors such as rough vegetation enhance the atmospheric mixing processes, which cause water vapour to diffuse upwards, away from the humid environment of a vegetated surface (Watson, 1999a). Typical values of aerodynamic resistances for forests, recorded in literature, range from less than 3 s.m^{-1} to 10 s.m^{-1} (Cain, 1998) and a value of 5 m.s^{-1} for coniferous forests (Kelliher *et al.*, 1993) was used for the Maritsane catchment simulation.

Radiation interception coefficients

Two radiation interception coefficients as well as the canopy LAI are used in Macaque to control the amount of solar radiation reflected, intercepted and absorbed by the canopy. These two radiation interception coefficients are the radiation extinction coefficient and the canopy reflection coefficient. The radiation extinction coefficient for a small leaf area is the ratio of the radiation intercepted by that leaf area to that which would have been intercepted if leaves faced directly towards the radiation source (Monteith and Unsworth, 1990). McMurtie, *et al.* (1990), cited by Watson (1999), used a un-sourced value of 0.5 for *Pinus Radiata* plantations and this value was assumed for the Maritsane simulation.

A portion of total solar radiation that reaches the canopy is reflected back into space and the degree of reflection is a function of the wavelength of light (Everson *et al.*, 1998). The terms “albedo” or reflection coefficient are used to describe this reflection of the solar beam, regardless of wavelength (Everson *et al.*, 1998). The albedo has an inverse control on all evapotranspiration processes (Watson, 1999a) and the greater the albedo, the less radiation

energy is absorbed and used to drive evapotranspiration. Forests reflect less shortwave radiation than for example grasslands (Kelliher *et al.*, 1993) and the mean daily albedo for plantation forests is generally in the range 0.1 to 0.15 (Jarvis *et al.*, 1976). A reflection coefficient of 0.1 defined by Monteith and Unsworth (1990) was adopted for the Maritsane simulation.

Rainfall Interception

In the Macaque model, the values of interception represent the capacity, in metres of water per unit LAI, of the canopy layer to store water (Watson, 1999a). A problem of representing interception storage in this manner is that storages are instantaneous, whilst rainfall is continuous. In climates with frequent, continuous rainfall, evaporation from the wet canopy is rapid due to high boundary layer conductance, despite low air saturation deficits (Jarvis and Steward, 1978). As a result, interception losses may be significantly greater than those predicted by the model's instantaneous interception capacity. However, in regions such as Mpumalanga, rainfall events are frequently of high intensity and short duration and it is unlikely that the canopy storage capacity will lead to under-estimation of evaporation losses from water stored on the canopy. McMurtie *et al.* (1990), cited by Watson (1999), used an interception value of 0.0005m/LAI for *Pinus radiata*, whilst Summerton (1995) reports higher interception values for Pines approximating 0.001m/LAI. An interception value of 0.0005m/LAI was used in the Maritsane simulation.

Vegetation rooting depths

For the Maritsane simulation rooting depths were considered the same as soil depth (4 metres). Therefore, water abstraction could take place from anywhere within the profile. The nominal proportions of canopy transpiration that should be drawn from the saturated zone was set as 0.1, implying that the trees only draw a small proportion of water from the saturated zone.

In the following section the results of the Macaque application on the Maritsane catchment are presented. The *ACRU* model results are also presented for comparative purposes and emphasis is placed on the models' ability to simulate low flows.

6.2.3 Simulation results for the Maritsane catchment

Verification of Macaque for the Maritsane was undertaken with the aim of showing how the model performed at the large scale (>100km²). As discussed in Chapter 2 it is the water yield of a forested catchment during low flow periods that is usually the focus of the water resources planner and for this reason the verification is centred around the ability of the model to reproduce observed low flows. The ability to verify the internal processes within Macaque was limited by a lack of field data and as a result the verification was only performed on model output (streamflow). In the ensuing section the statistics for daily simulated streamflow and monthly totals of daily simulated streamflows are presented for both the Macaque and *ACRU* model. The performance of Macaque at the daily timescale was important since environmental requirements as described in Chapter 2 are of increasing importance and require daily streamflow records.

6.2.3.1 Results

The accumulated streamflow simulated by the Macaque model and observed flow for the 18-year simulation period from 1979 to 1995 corresponded closely with only a 3.6% difference between totals. It is evident from Figure 6.6 that the cumulated flows did not differ significantly at any stage of the simulation.

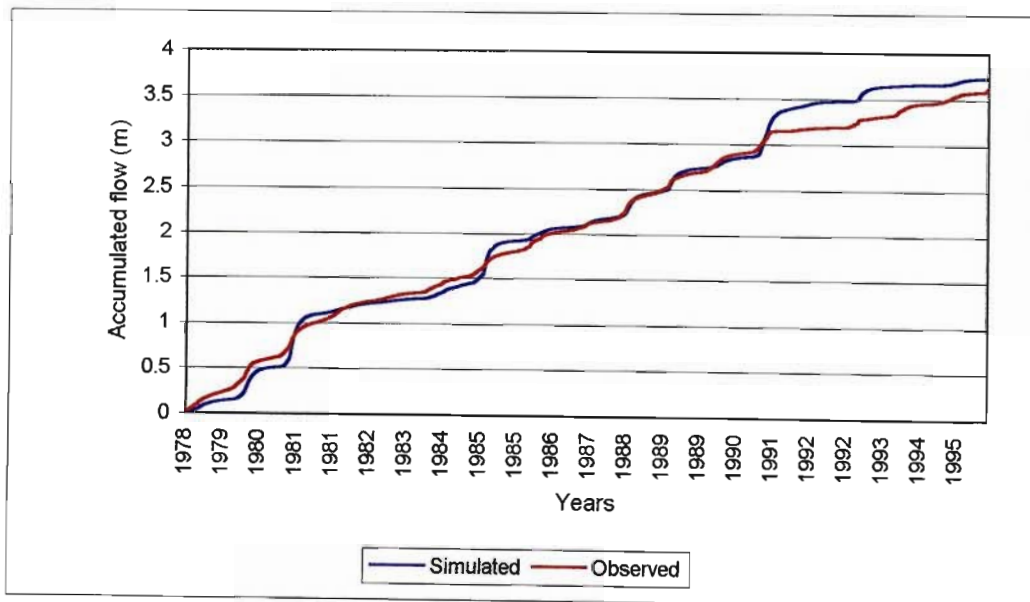


Figure 6.6 Accumulated totals of Macaque simulated and observed streamflow for the Maritsane catchment.

As shown in Figure 6.7, streamflow is over-estimated by the Macaque model during high rainfall years such as 1981, 1985 and 1991, whilst during below average rainfall years the model typically under-simulates streamflow.

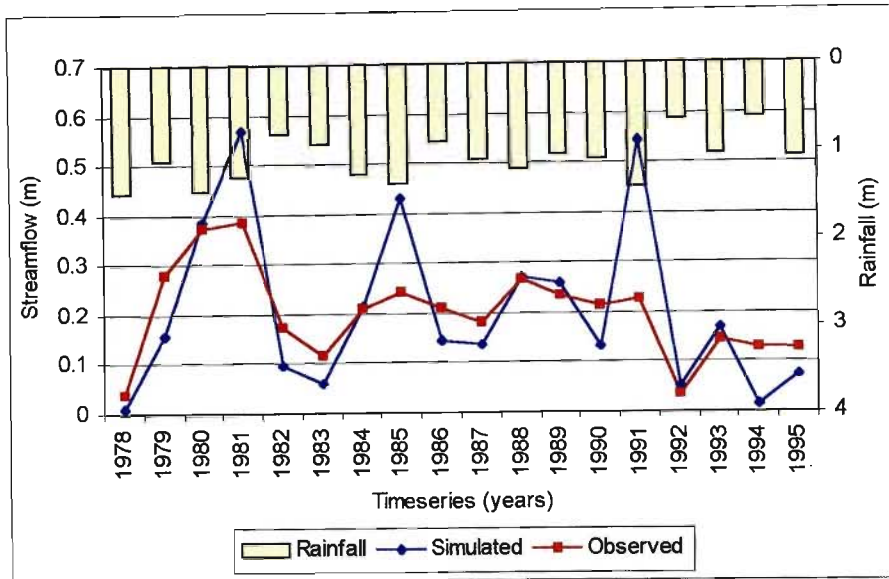


Figure 6.7. Comparison of annual totals of Macaque simulated daily runoff, observed runoff and rainfall for the Maritsane catchment.

At the daily timescale the Macaque model performed poorly as shown by the statistics presented in Table 6.3. The coefficient of determination was only 0.39 which shows a poor association between simulated and observed values. The poor results at the daily timescale are further reflected by the variance and standard deviation statistics (Table 6.3). Variance is not particularly well preserved in the simulated values and the regression coefficient of less than 1.0 suggests that higher flows are under-simulated. Although the mean simulated flow does not differ greatly from the observed mean, the standard deviation of simulated flow shows a far greater dispersion of simulated values about the mean than for observed flows. These statistics indicate that either the timing of high and low flows are incorrectly simulated or the volume of stormflow and baseflow release on a given day are incorrectly simulated.

Table 6.3. Goodness of fit statistics for simulation of daily streamflow from the Maritsane catchment X3H011 (1978 to 1995).

	<i>Complete record</i>	<i>Low Flows</i>
Total observed flows (depth in metres)	3.612	0.4319
Total simulated flows (depth in metres)	3.742	0.4635
Sample size (Days)	6045	1534
Mean of observed flows (depth in metres)	0.000598	0.00028154
Mean of simulated flows (depth in metres)	0.000619	0.00030212
Percentage difference between means	-3.6	-7.3
Correlation coefficient	0.625	0.433
Regression coefficient	0.916	0.4649193
Base constant for regression eqn. (m)	7.15E-05	0.0001712
Variance of observed flow	5.17E-07	2.868E-08
Variance of simulated values	1.11E-06	3.311E-08
Standard deviation of observed flow	0.0007193	0.00016929
Standard deviation of simulated flow	0.0010545	0.00018191
Percentage difference in standard deviation	-46.6	-7.45
Coefficient of Determination	0.391	0.187
Coefficient of Efficiency	0.387	-2.261

The low flow period was considered to extend over a three-month period from June to August. Low flows were very poorly simulated at the daily timescale and a correlation coefficient of only 0.43 was achieved, as shown in Table 6.3. The mean simulated flow was slightly higher than the observed mean for the low flow months, indicating that low flows were over-simulated by the Macaque model. The variance, however, was better preserved in the simulated values during the low flow months than for the complete flow record.

In Figures 6.8 to 6.10 shorter periods of the simulation record have been extracted and graphically represented. These selected periods reflect years with both above average rainfall and below average rainfall. The *ACRU* simulation results of the Maritsane catchment which forms part of the Kruger National Park River Research Programme (Pike *et al.*, 1997) are also graphically presented for comparative purposes.

In Figure 6.8a below the simulation results for 1983 are presented. This was a low rainfall year (914mm) and was also preceded by a low rainfall year (1982). On average 50% of annual rainfall falls within the first quarter of the year, however, only 30% of the annual total rainfall fell in the first quarter of 1983, which implies that the catchment moisture status would have

been low. Despite the dry catchment conditions the observed streamflow is very responsive to rainfall inputs. The Macaque model, however, was not able to reflect these stormflow contributions to streamflow. The simulated record shows only slight increases in runoff during a rainfall event, which indicates that saturation excess runoff contributions to streamflow were low. Such small contributions of saturation excess flow occur when the saturation deficit of the ESUs is high and only a small proportion of an ESU is saturated to the surface. It is possible that other runoff mechanisms, such as infiltration excess overland flow, occur within the Maritsane catchment, which result in such large runoff peaks and for which the Macaque model does not account. The under-estimation by Macaque of stormflow peaks during the rainfall season under dry catchment conditions was also observed in other low rainfall years.

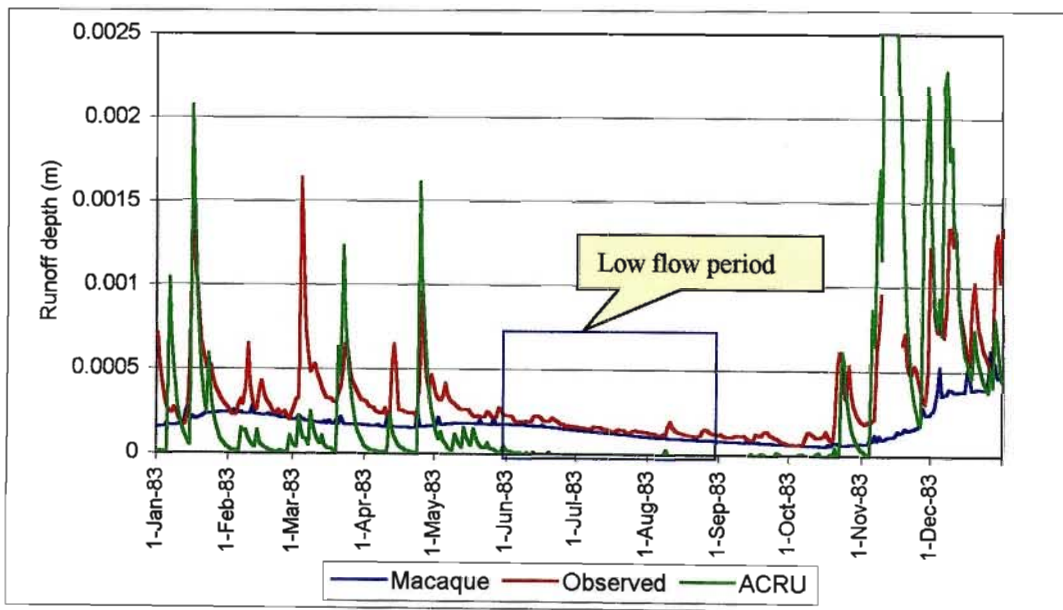


Figure 6.8a. Daily simulated and observed streamflows for the 1983 below average rainfall year.

The *ACRU* model simulated a greater streamflow response to rainfall inputs than Macaque during the wet season. On “dry weather” days streamflow is under-simulated by the *ACRU* model.

During the dry season of 1983, shown in Figure 6.8b, Macaque’s simulated streamflow corresponds closely to observed values, which indicates that simulated baseflow is similar to observed baseflow release. Observed streamflow is, however, more responsive to the

occasional dry season rainfall inputs than simulated streamflow. The *ACRU* model was not able to sustain dry season flow and runoff ceased during this period.

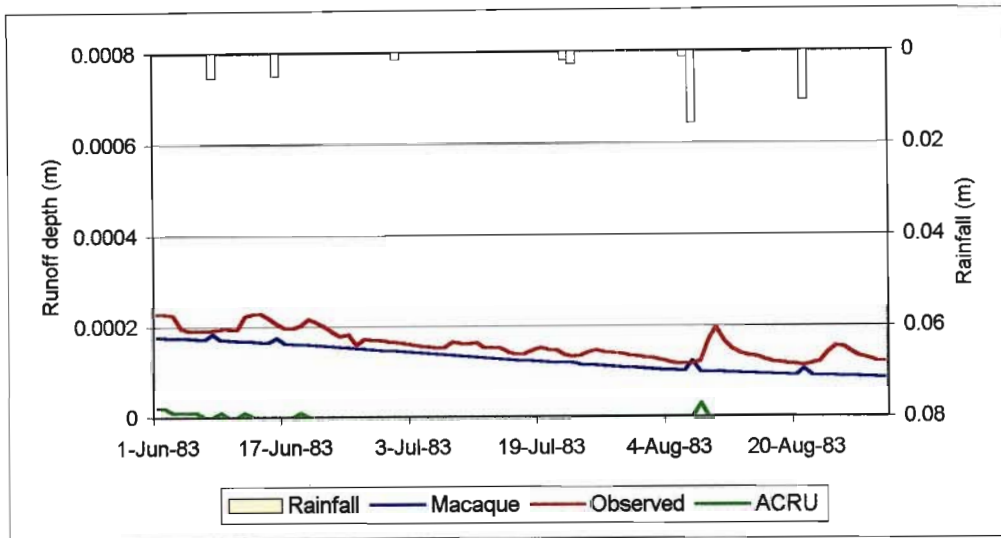


Figure 6.8b. Daily simulated streamflow by the Macaque and *ACRU* models for the 1983 low flow period.

In Figure 6.9a and 6.9b the simulated and observed streamflows for 1991 are presented. This year was an above-average rainfall year (1411mm) and was preceded by an average rainfall year in 1990. Sixty percent of annual rainfall fell within the first quarter of the year and the catchment soil moisture status is likely to have been high. Figure 6.9a shows that the Macaque model over simulates streamflow throughout the year.

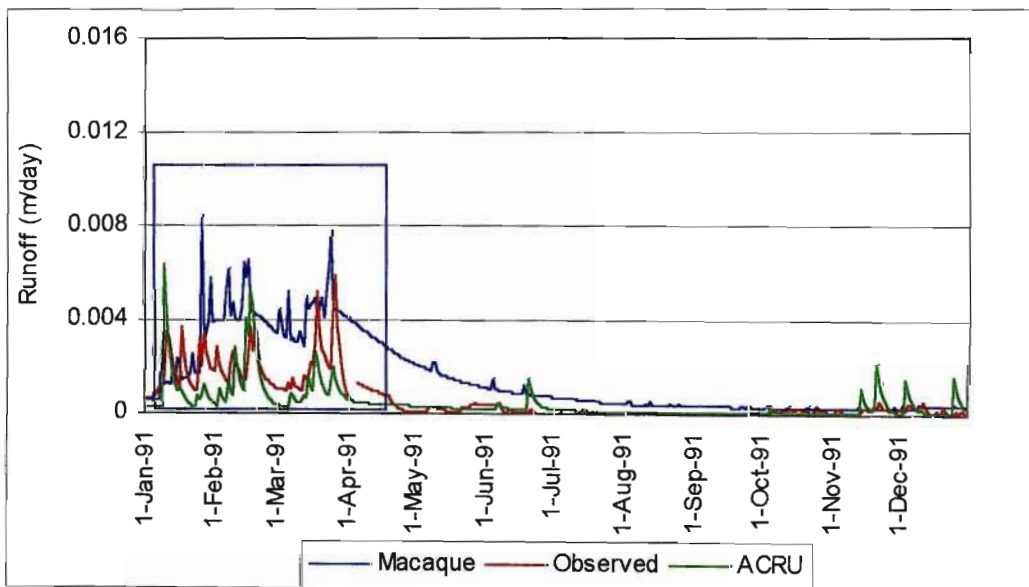


Figure 6.9a. Simulated and observed streamflow for the 1991 high rainfall year.

During the dry season Macaque maintained higher flows whilst actual (observed) streamflow ceased during the latter part of the dry season even though rainfall contributions were greater than during 1983.

In Figure 6.9b the observed and simulated streamflows during the 1991 wet season are shown. It can be seen that the observed and simulated hydrograph peaks are of the same magnitude and the model's streamflow response to rainfall inputs is therefore similar to that observed. Macaque's baseflow and sub-surface flow contributions on "dry weather" days are, however, greater than observed. It is further evident that simulated streamflow peaks occur sooner than observed peaks (observed streamflow peaks are lagged). This lag in the timing of the observed stormflow peak is likely to be the result of instream flow routing. When runoff is generated in the upper areas of a catchment there is a time delay for this increased volume of streamflow to reach the catchment outlet (point of flow measurement). The Macaque model does not account for this time delay and runoff from all ESUs instantaneously becomes catchment runoff at the basin outlet.

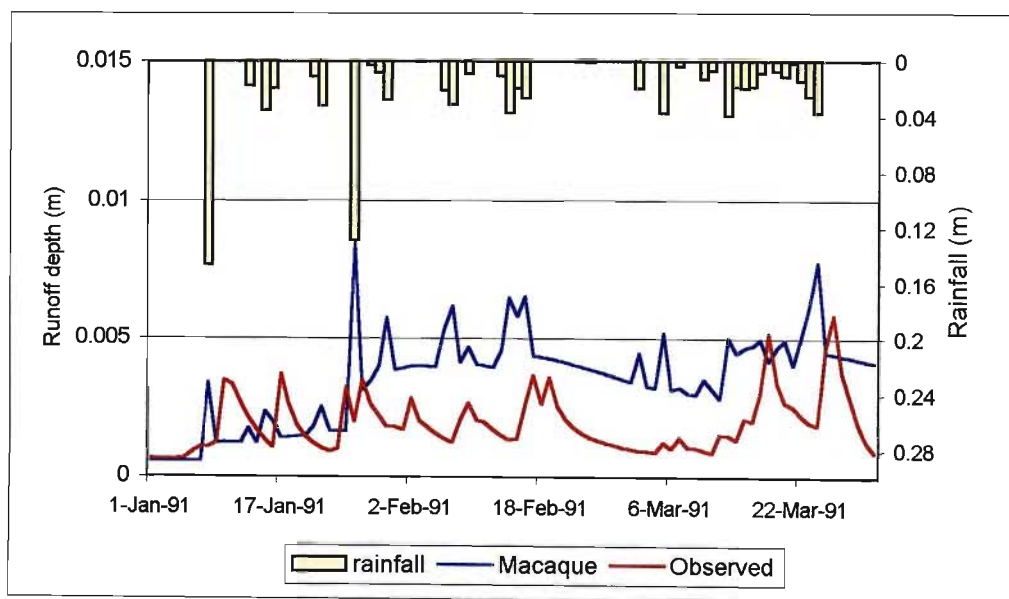


Figure 6.9b. Simulated and observed streamflow for the 1991 wet season.

In Figures 6.10a and 6.10b the simulation results for 1980 are given. This year is also an above average rainfall year and similar catchment response is observed to that in 1991. Streamflow was over-simulated by Macaque during the wet season and the simulated streamflow recession during the latter part of wet season is slower than observed. This

recession continues till the late winter season, whilst observed records show that streamflow recedes until the start of the dry season after which it remains constant for most of the low flow period.

The *ACRU* model over-simulates peak flows during the wet season, whilst baseflow releases are under-estimated by the model. It is evident that the *ACRU* model is not able to reproduce the observed slow recession of streamflow during the latter part of the wet season and early dry season.

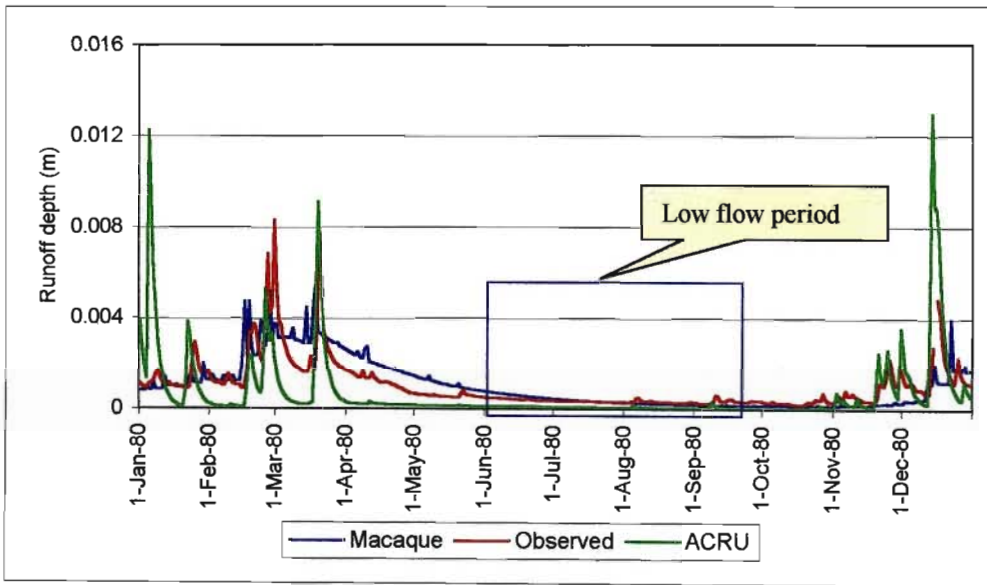


Figure 6.10a. Simulated versus observed streamflow for 1980 above-average rainfall year.

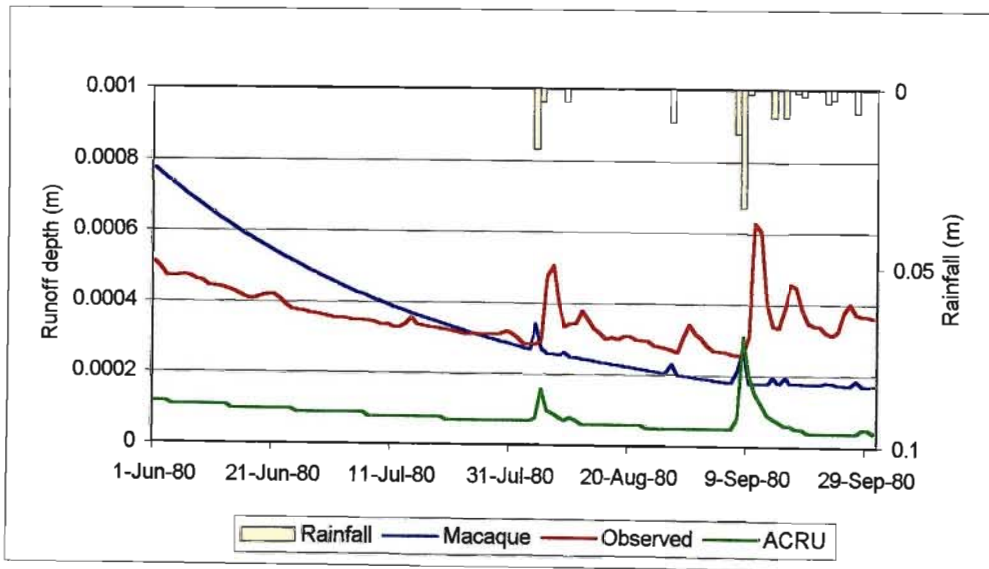


Figure 6.10b. Simulated and observed streamflow for the 1980 low flow period.

The Macaque model performed better at the monthly timescale than at the daily timescale as shown in Table 6.4. The correlation coefficient improved to 0.799 and the monthly mean of simulated streamflow corresponded closely to the observed mean. The difference between simulated and observed standard deviation was slightly higher at the monthly timescale, which indicates that there was greater dispersion of simulated monthly totals about the mean than for observed monthly totals. The regression coefficient higher than unity and the base constant approximating zero suggests that the Macaque model over-simulated higher flows.

The verification results of the *ACRU* simulation (Pike *et al.*, 1997), given in Table 6.4, indicate that the model did not perform well on the Maritsane catchment. Better results were achieved in a previous 10-year *ACRU* simulation of the Maritsane catchment (Jewitt and Schulze, 1993) for which a dynamic file was used to simulate the changes in tree physiology with age. The model under-simulated both monthly high flows and low flows and the low correlation coefficient shows a poor association between simulated and observed results.

Table 6.4. Goodness of fit statistics for simulation of monthly totals of daily streamflow from the Maritsane catchment X3H011 (1978-1995).

	Macaque		ACRU	
	<i>Complete record</i>	<i>Low flows</i>	<i>Complete record</i>	<i>Low flows</i>
Total observed flows (m)	2.867	0.4271	2.867	0.4271
Total simulated flows (m)	2.866	0.4481	2.116	0.21768
Mean of observed flows (m)	0.01638	0.008898	0.01638	0.008898
Mean of simulated flows (m)	0.01637	0.009335	0.0120911	0.004535
Percentage difference between means	0.04	-4.92	-26.2	49.0
Correlation coefficient	0.799	0.516	0.661	0.429
Regression coefficient	1.2437	0.614907	0.6906	0.615
Base constant for regression eqn.	-0.004	0.003864	0.00078	-0.00094
Variance of observed flow	0.00022	1.86E-05	0.00022	1.86E-05
Variance of simulated values	0.00054	2.64E-05	0.000244	3.91432E-05
Standard deviation of observed flow	0.01495	0.0043125	0.01495	0.0043125
Standard deviation of simulated flow	0.02329	0.005137	0.015621	0.0062565
Percentage difference in std deviation	-55.76	-19.12	-4.44	-43.56
Coefficient of Determination	0.638	0.267	0.437	0.184
Coefficient of Efficiency	0.613	0.1547	0.273	-231.16

Both the Macaque and *ACRU* model produced poorer results for the low flow periods than for the complete simulation record. In Figures 6.11a and 6.11b the monthly totals of daily

simulated streamflow for both the Macaque model and *ACRU* model are compared against monthly observed streamflow. It should be noted that during the Kruger National Park River Research Programme simulation it was suspected that overtopping of weir X3H011 occurred without the flags for suspect data appearing in the data record (Pike *et al.*, 1997). This combined with the fact that missing streamflow records were typically peak flows, makes it difficult to establish how well both models simulate peak flows.

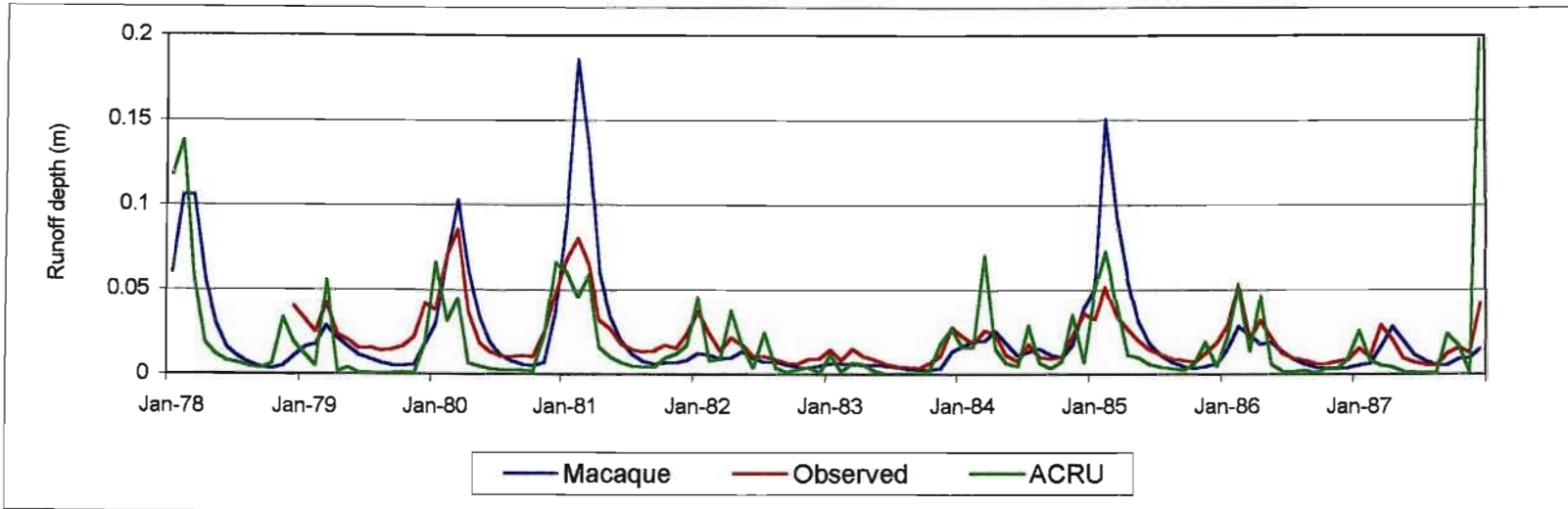


Figure 6.11a. Monthly simulated and observed streamflows at weir X3H011 for the period 1978 to 1987.

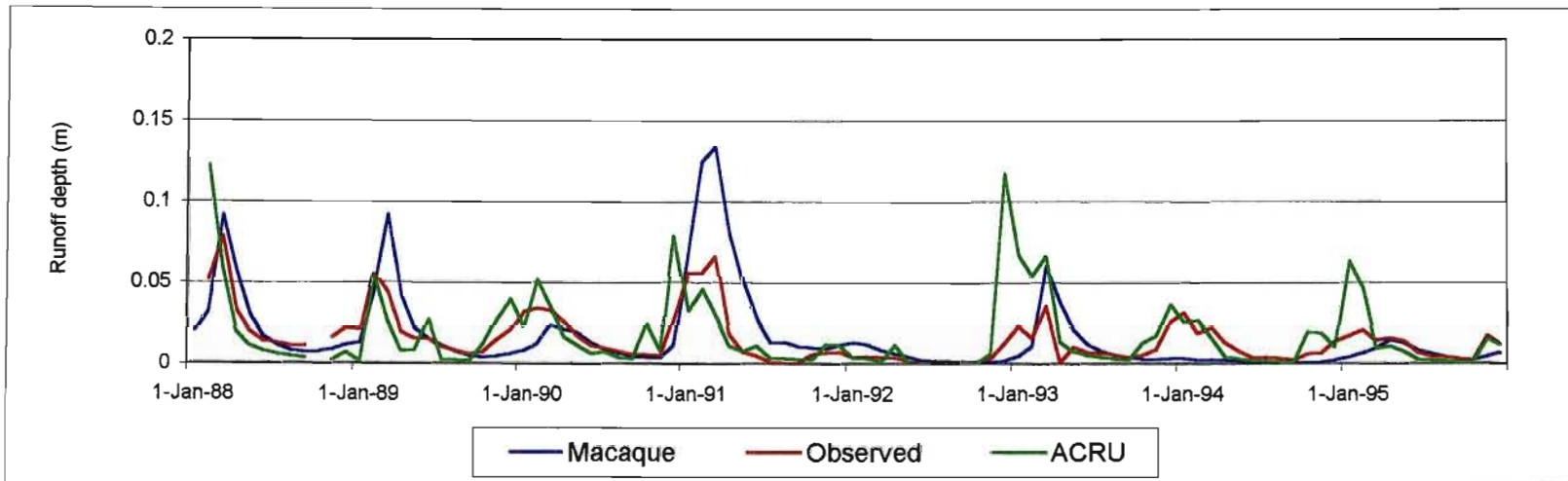


Figure 6.11b. Monthly simulated and observed streamflows at weir X3H011 for the period 1988 to 1995

6.2.3.2 Conclusions

The results of the Maritsane simulation show that the Macaque model performed poorly at the daily timestep and during low flow periods. Simulation results improved for the monthly simulation, but there is considerable uncertainty regarding the model performance due to the lack of hydrological data for verification of the internal model processes.

In a recent application of the Macaque model on the Maroondah catchments in Australia similar poor results were achieved for the initial model simulation. However, in this application the model was calibrated to attain a better fit (Peel *et al.*, 2000). By changing two of the model parameters, the precipitation scalar (a rainfall multiplication factor) and the ratio of hydraulic to surface gradient, the simulation results vastly improved (Peel *et al.*, 2000). By calibrating the model in such a manner the danger exists that better results are achieved for hydrologically incorrect reasons. The implication is that although the model performance improved by changing these parameters, the source of the problem that resulted in the poor initial results may not have been related to precipitation inputs or the hydraulic gradient. For this reason the Macaque model was not calibrated for the Maritsane catchment application to achieve a better fit with observed streamflow data. In section 6.4 the sensitivity of model output to changes in model input parameters is tested and serves to highlight which parameters that have the greatest influence on simulated streamflow output. However, the question remains whether the model inaccuracies could be improved by a better choice of parameters. In other words, it is not known whether some of the poorer results of the Macaque simulation are related to inaccurate process representations or an incorrect choice of parameter values. To mitigate against the arbitrary parameter selection it might be useful to use a parameter optimisation process, such as the Model Independent Parameter Estimation (PEST) system (Doherty, 1994). According to such an approach, parameters values can be set to vary within a specified and realistic range in order to obtain an optimal parameter set to fit the observed data. However, it has been noted that the model representation of certain processes, such as the spatial and temporal changes in LAI observed in commercial plantations is inadequate. Thus, optimisation of parameters such as LAI could improve the simulation results, however, it would not improve the realism of the input values.

The results described above indicate that the model cannot be used with confidence on operational catchments where the lack of input data influences the reliability of the model

results and the lack of hydrological data further prevents the verification of the model's internal processes. Although the *ACRU* model produced poorer results for both the complete simulation period and for low flows than the Macaque model, there is greater uncertainty regarding the model inputs used in Macaque. It cannot, however, be concluded that the topography-based modelling approach produces better results. What is encouraging about these results is the manner in which the Macaque model is able to sustain baseflow during the dry season, whilst the *ACRU* model was not able to sustain flows during the dry season. This is likely to be attributed to the fact that Macaque represents a deep soil profile with a greater soil storage capacity. This stored water is then available for release during the dry season. The *ACRU* model calculates the amount of effective rainfall (water infiltrated into the soil) by using the COIAM and SMDDEP parameters. However, it is evident from the Maritsane simulation results that peak flows were often over-simulated, indicating that although the COIAM parameter and SMDDEP are set high for forest simulations, the amount of water infiltrated on a rainday is less than in reality, resulting in greater than observed stormflow releases. As a result less water is stored for release during the dry season when rainfall inputs are low.

What has become apparent is that without the required model inputs the Macaque model cannot be used for application in South Africa unless changes are made to certain process representations within the model to accommodate the data limitations.

6.3. The Witklip Catchment

The Witklip catchment area is situated in the Mpumalanga Province of South Africa (25°14'S; 30°53'E) close to the town of White River as shown in Figure 6.1. The research catchments at Witklip form part of the Eastern Drakensberg escarpment and produce tributaries of the Witklip River (Scott *et al.*, 1999). The area has a humid, sub-tropical climate and the mean annual precipitation is 1475mm (Smith, 1991), which is considered a high rainfall area for forestry.

Afforestation of the Witklip catchments was initiated in the 1940s. Most of the plantation was established for sawlog production and harvested over a four year period from 1980 to 1983 (Smith, 1991) after which the plantations were progressively re-planted. The monitoring of streamflow and climatic data in these catchments was initiated in 1975 to test the effects of deforestation on catchment water yield using the paired catchment approach described in Chapter 2.

The Witklip catchment experiment comprises eight small catchments of which the so-called Catchment V (Figure 6.12) was selected for the model application and verification. Catchment V had previously been configured for an *ACRU* simulation and would consequently allow for a comparative study of the Macaque and *ACRU* model results. Although the Macaque model was specifically developed for large-scale applications, the model has successfully been applied to small catchments of less than 100 ha (Peel *et al.*, 2000).

6.3.1 Description of Witklip Catchment V

Catchment V faces north-west. The catchment covers an area of 1.08 km², of which 0.56km² is under exotic plantations, situated in the lower catchment areas. The plantation comprises mostly pines, whilst eucalyptus were planted as firebreaks (Scott *et al.*, 1999). The remaining 0.52km² consists of indigenous grassland in the upper catchment area and indigenous scrub in the riparian areas (Smith, 1991). Streamflow was recorded between 1975 and 1990 and the catchment history is summarised in Table 6.5 on the following page.

Table 6.5. Witklip, Catchment V history (after Smith, 1991)

Date	Treatment
1942 – 1944	Afforestation of 49% of catchment
1955 – 1956	Afforestation of 2.8% of catchment
Jan 1975	Raingauge A5 and streamflow measurement commences
Apr 1975	Upper part of catchment burnt
1980 – 1984	Clearfelling of plantation and re-afforestation

A topographic map of Catchment V is shown in Figure 6.12. The elevation ranges between 1000m a.s.l and 1470m a.s.l. The upper reaches of the catchment are a plateau and a steeply sloping scarp formed from Black Reef Quartzite. Granite outcrops are visible in the grassland slopes as shown in Figure 6.13.

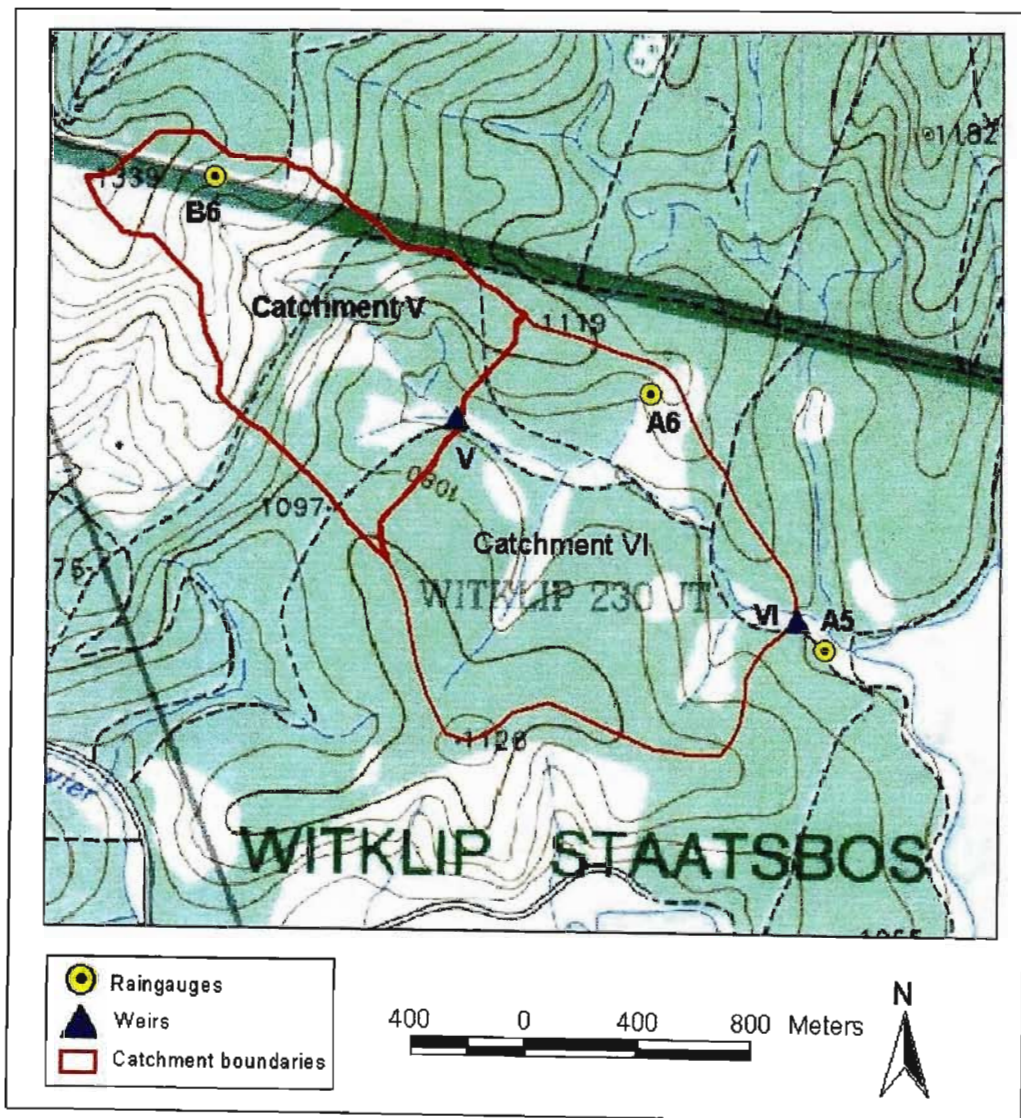


Figure 6.12. Topographic map of Witklip, Catchment V.

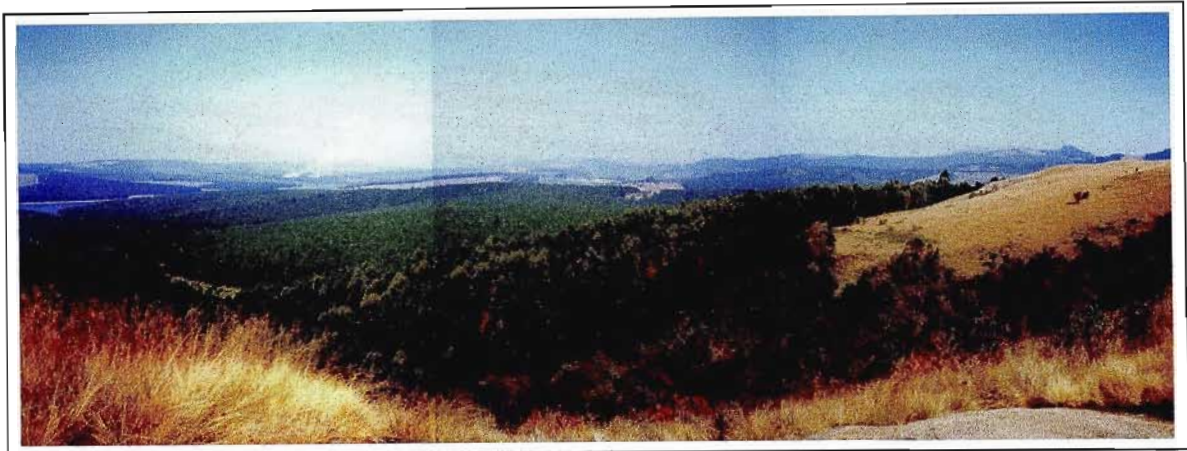


Figure 6.13. Witklip, Catchment V.

In the ensuing sections the model configuration for the Witklip Catchment V is described and a brief summary of the Macaque model inputs is provided. In Table B1 of Appendix B a comprehensive list of parameters required by the Macaque model and the parameter values selected for Witklip catchment is provided.

6.3.2 Model configuration and input parameters

Three spatial levels were selected for the configuration of the model for the Witklip catchment. As with the Maritsane catchment configuration, the catchment represented the world level. The hillslopes and ESUs represented the second and third spatial levels, respectively. Tarsier's terrain analysis procedures were used to delineate the catchment boundary and hillslopes using a 10m resolution DEM. The DEM was derived from contour data digitized off hardcopy 1:10 000 orthomaps. The Arcinfo routine, TOPOGRIDTOOLS (Hutchinson, 1989; Hutchinson, 1996), was again used to derive the DEM from digitized contour data and other height related features in the catchment.

The ESUs were delineated using the TOPMODEL wetness index $\ln(a/\tan B)$. The derived wetness indices (WI) ranged between 4 and 16. The intervals, which were used to define separate ESUs, are presented in Table 6.6 below. The low wetness indices represent the drier upper hillslope areas, whilst the high WI represent the saturated valley bottoms.

Table 6.6. Intervals of wetness index used to define separate ESUs in the Witklip catchment.

Interval	ESU number
4-6	1
6-8	2
8-10	3
10-12	4
12-14	5
14-16	6

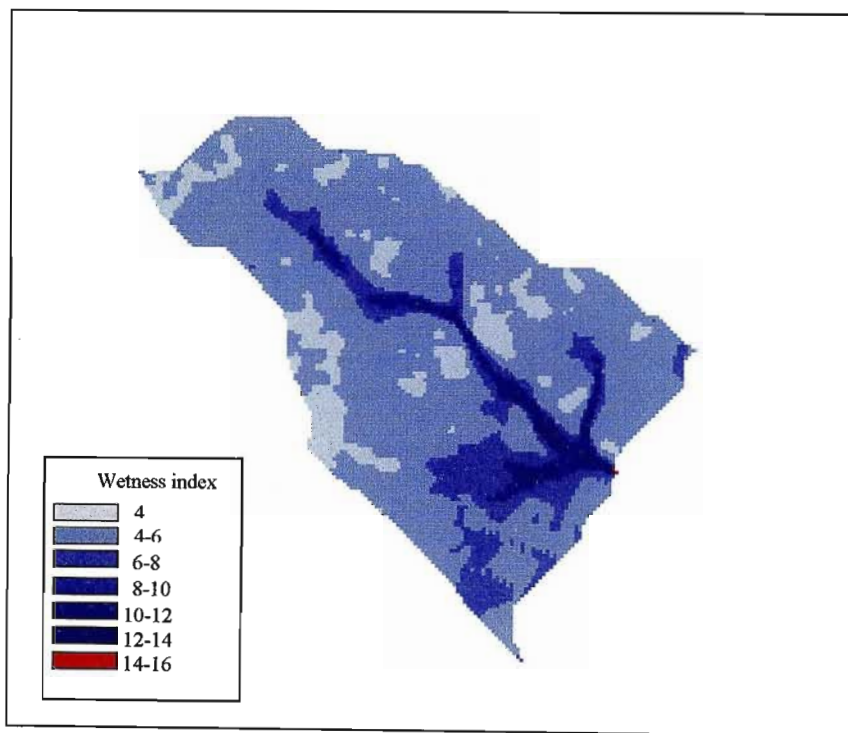


Figure 6.14. Wetness index ranges for Witklip Catchment V.

Daily rainfall data

Witklip receives predominantly summer rainfall. At Witklip a network of Casella recording raingauges, each paired with a standard Snowdon gauge with Nipher shield, record rainfall on a continual basis (Scott *et al.*, 1999). There is one rainfall station in Catchment V, namely A5 (altitude of 1300m) while raingauges A6 (altitude 1085m), and B6 are situated just outside the catchment, as shown in Figure 6.12. The MAP recorded at the upper rain gauge, A5, is 1137 mm whilst the MAP at the lower rain gauge, A6, is 1085 mm. Rainfall station A6 was the longest recording station and was used as the base station. Due to the small catchment size (1.04km²) and small difference in MAP between the upper and lower rain gauges, spatial variability in rainfall was not accounted for and rainfall was therefore considered to fall uniformly over the catchment. Missing rainfall data had already been patched for use in a previous *ACRU* model simulation (Gush *et al.*, 2001) using the inverse distance interpolation technique developed by the Department of Agricultural Engineering (now BEEH).

Daily temperature data

Temperature data at Witklip was only recorded for a six-year period from 1975 to 1980. Therefore it was necessary to supplement the short record with temperature data obtained from a nearby meteorological station at Nelspruit. A temperature correction factor, which had previously been derived for the *ACRU* model verification using the window of overlapping data (Gush *et al.*, 2001), was then applied to the daily maximum and minimum temperatures from Nelspruit. Since the temperature data from Nelspruit were used and a correction factor had already been applied to these data, it was decided not to apply a correction factor to account for variation in temperature with increasing elevation.

Incident solar radiation data

There was no daily global incident solar radiation data available at Witklip to derive the necessary parameters required by the transformed Bristow and Campbell equation (1984) used in Macaque to calculate transmission. Therefore, default parameters values had to be used.

Soils parameters

The soils at Witklip are formed on deeply weathered granites and are highly leached and well-drained (Scott *et al.*, 1999). Other than Landtype data, there were no other soils data available. It was therefore decided to undertake a soil survey along a transect of the catchment. Four sampling points were selected at which undisturbed core soil samples and disturbed core soil samples were taken.

The first site was in the upper catchment area, the second in the midslope section, the third in the forest plantation (footslope) and the fourth in the riparian zone. The samples were taken at 0.25 metre intervals up to a depth of 1.5 metres, after which the sampling interval was increased to 0.5 metres up to 3 metres below ground level. Soil textural analyses (hydrometer and sieve analyses) were undertaken on the disturbed samples, whilst the water retention characteristics of selected undisturbed samples were tested. Due to time constraints and limited resources not all the undisturbed samples could be tested. The results from these analyses are presented in Table B2 of Appendix B. The texture of the soil at all four sites was sandy loam and a greater variation in texture was observed with increasing depth than was observed between sites.

At each site, auguring was undertaken until either bedrock was reached or the soil depth exceeded the maximum auguring depth of 3 metres. In the upper grassland catchment, soil depths ranged between 1 metre and 2.5 metres, whilst soil depths in the midslope section varied greatly from visible outcrops to soils depths of greater than 3 metres. In the forest plantation, soil depths again varied between sampling sites. These variations in soil depths made it impractical, even at the small scale of the Witklip catchment, to derive spatially representative values of soil depth and other soil parameters for the catchment. An initial uniform soil depth of 4 metres was therefore assumed for the Witklip catchment. In the sensitivity analyses, described later in this Chapter, it was shown that simulated streamflow during the dry season was very sensitive to changes in soil depth. The implication is that by over-estimation or under-estimation of the actual soil depth, simulated monthly streamflow is increased or reduced, respectively.

The type of soil input parameters required by the Macaque model are seldom available and soil sampling, *in situ* soils measurement and laboratory analyses proved to be a very time consuming process. Furthermore, techniques do not exist to accurately extrapolate point

measurements to represent the spatial variability in soil physical and water retention characteristics over a catchment area. Therefore, in both the Maritsane and Witklip catchment applications these soil parameters could essentially be operated as calibration parameters.

Leaf Area Index

The lower section of Catchment V is managed as a commercial plantation and consists of nine compartments with trees of differing species and age. The Macaque model, however, only gives representation to the temporal changes in LAI for trees indigenous to Australia. It was therefore proposed that the model developer, Dr Fred Watson, encode the LAI tables that are used in the *ACRUforest* model (Summerton, 1995), into the Macaque model. However, due to the time constraints these tables could not be encoded and consequently the temporal changes in LAI were not accounted for. A fixed LAI for forests of intermediate age was therefore assumed for the entire simulation period. The implication of this assumption was that the Macaque model was likely to over-simulate runoff from the mature forest plantations prior to harvesting (in the late 1970s), whilst the model was likely to under-simulate streamflow during the early to mid 1980s when clearfelling and re-forestation of the plantations took place. The simulation results from the Witklip application are discussed in section 6.3.3.

The upper catchment area of Catchment V is grassland. Grasslands are winter dormant and therefore do not transpire during the dry season. The `leaf_on_day` and `leaf_off_day` parameters in Macaque allow the user to set the LAI of deciduous vegetation to zero for part of the year. During the winter months from June to August the grassland LAI in Catchment V was set to zero.

Transpiration parameters

The Macaque model accounts for a considerable number of factors that control stomatal conductance and hence transpiration rates. The implication is that for multiple land uses within a catchment a large number of parameters are required by the model. Although the responses of stomata to environmental factors for coniferous forests are well understood, the responses of other vegetation types, such as grasslands, are less widely documented. Furthermore, these publications are mostly based on studies undertaken in Western Europe.

The stomata in forest canopies react to the same stimuli as any other vegetation, such as grassland, however the relative importance of these stimuli is different (Cain, 1998). Despite the difference in canopy structure and stomatal response of grasslands and forests, differences between forest and grassland peak daily evaporation in mid-summer (non-water limiting conditions) is relatively small (Dye and Bosch, 2000; Kelliher *et al.*, 1993; Körner, 1993). Due to the limited availability of studies pertaining to the changes in stomatal conductance and hence transpiration of indigenous grasslands, the grassland's stomatal response to environmental controls was considered to be the same as that of the pine plantation.

Radiation interception

Forests reflect less shortwave radiation than grasslands (Kelliher *et al.*, 1993). The mean daily albedo of plantation forests is generally in the range 0.10 and 0.15 (Jarvis *et al.*, 1976) whilst that of grassland and many other crops is generally higher at about 0.25 (Jarvis and Steward, 1976; Everson *et al.*, 1998). These values were adopted for the Witklip application.

Rainfall interception

Studies in temperate forests have shown interception losses to be substantially greater than the reported interception losses in the summer-rainfall forested regions of South Africa (Dye, 1996). Hence, the difference in interception losses between grasslands and forests are less than is observed in temperate environments.

As with the Maritsane catchment application an interception value of 0.0005m/LAI was assumed for Witklip forest plantations. Interception losses of 2.2mm and 1.6mm per rainday for the flowering and seed development stages of *Themeda triandra* (De Villiers, 1980) were converted to metres/LAI and used in the Witklip simulations.

Vegetation rooting depth

Little is known about root zone depth in soil and the partitioning of root function into nutrient acquisition from chemically rich top soil and water uptake, possibly to great depth (Kelliher *et al.*, 1993). Furthermore, rooting depths between biomes differ, but also differ amongst trees species and according to site and soil factors. In water-limiting environments (and during the

dry season) the ability of plants to extract water from depths would enable deep-rooted vegetation to continue to transpire whilst vegetation with shorter roots would encounter stress. The ability of trees to continue to transpire when infiltration was inhibited was demonstrated by Dye *et al.* (1997). The limited knowledge of rooting depths makes it difficult to quantify the differences in water use and hence the impact on catchment water yield of short or deep rooted vegetation types. For the Witklip catchment, rooting depths of the grassland was taken as 0.5m and the rooting depth in the forest plantation were assumed to extend to bedrock (4 metres).

In the ensuing section the results of the Witklip simulation are presented. These results are compared with the results obtained from the *ACRU* simulation for the same catchment.

6.3.3 Simulation results for the Witklip catchment

The benefit of verifying hydrological models on small research catchments such as the Witklip catchment is that the availability and quality of data is better. In research catchments there is often a greater density of recording stations, collected data is thoroughly checked and missing data records are few. However, despite Witklip being a research catchment data for internal verification at Witklip was limited to meteorological data. In the ensuing section the simulation results of the Macaque model on Witklip are discussed, followed by a comparison between the *ACRU* and Macaque results at the monthly timescale.

6.3.3.1 Results

The accumulated simulated streamflow and observed flow for the 16-year simulation period from 1975 to 1990 corresponded closely with only a 1.3% difference between totals. However, it can be seen in Figure 6.15 that there is a considerable deviation in 1978 as a result of the Macaque model over-simulating streamflow during this year. During 1985 and during the four years thereafter the under-simulation of streamflow by the model equalized this deviation.

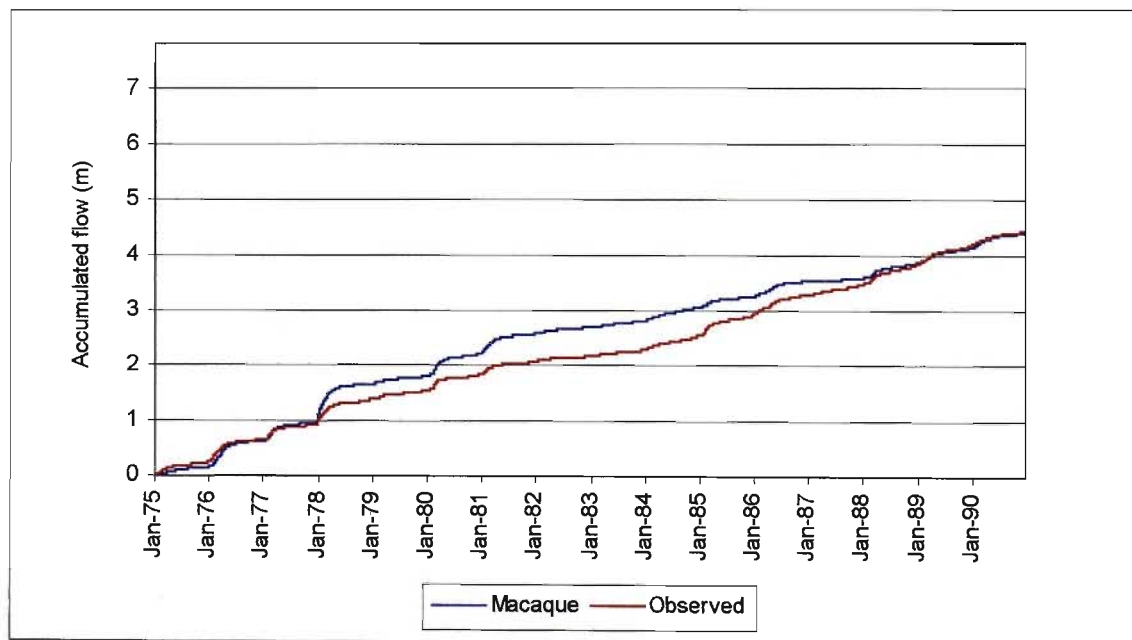


Figure 6.15 Accumulated totals of simulated and observed streamflow for the Witklip catchment.

As shown in Figure 6.16, 1978 was a high rainfall years and was preceded by high rainfall years, whilst the mid to late 1980s were characterised by low annual rainfall. This over-simulation by the model during wet years and under-simulation of streamflow during dry years was also observed during the Macaque simulation of the Maritsane catchment. It appears that both the Witklip and Maritsane catchments are able to store greater amount of water than is simulated by the model and that this greater storage allows for sustained flow releases during low rainfall years. However, an additional factor that could be responsible for the under and over-simulation of streamflow is the assumption of a constant LAI for the plantation area. The implication of this assumption is that the plantation LAIs and transpiration rates during the late 1970s, when the trees were mature, would have been higher than modelled since an LAI for intermediate aged trees was assumed for the forest plantation for the entire simulation period. Conversely during and after 1982 and 1983, when the majority of the plantation was clearfelled, the modelled LAI would have been higher than the actual LAI. Therefore, it would be expected that the model would under-simulate streamflow during these years as the higher LAIs would result in greater transpiration.

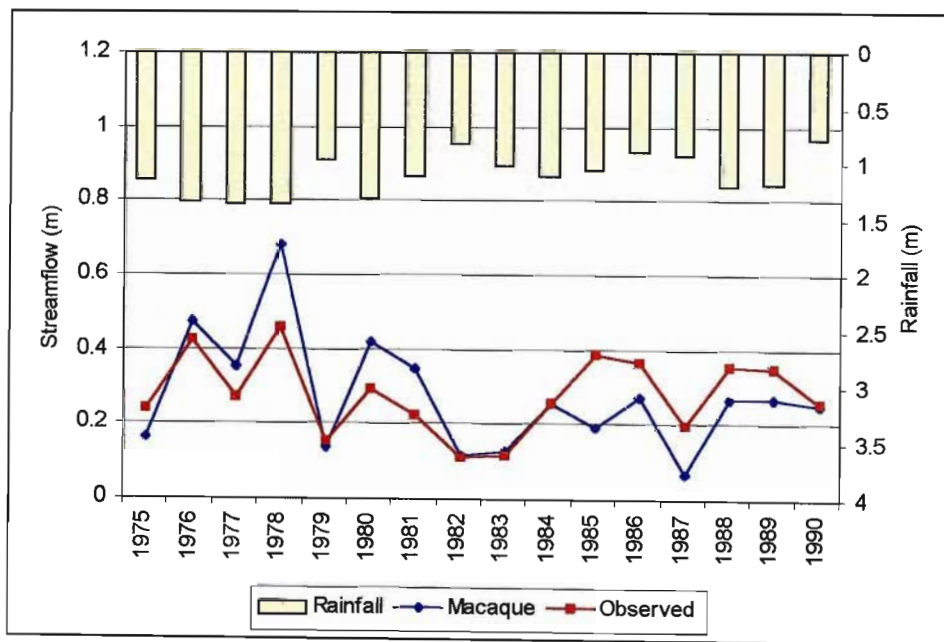


Figure 6.16. Annual rainfall and runoff for the Witklip catchment.

At the daily timescale better simulation results were achieved for the Witklip catchment than for the Maritsane catchment as shown in Table 6.7. Although the correlation coefficient was relatively low, the simulated mean corresponded closely to the observed mean and the variance was well preserved. The model's performance for the low flow period from June to August was slightly poorer than for the complete simulation record. The simulated low flow

mean compared well to the observed low flow mean, although the difference in standard deviations between observed and simulated low flows was greater than for the complete simulation record. For both the complete simulation record and the low flow record the regression coefficient (regression slope) was low indicating that high flows are under-simulated by the Macaque model.

Table 6.7. Goodness of fit statistics for simulation of daily streamflow from the Witklip catchment.

	<i>Complete record</i>	<i>Low Flows</i>
Total observed flows (depth in metres)	4.45	0.542
Total simulated flows (depth in metres)	4.39	0.570
Mean of observed flows (depth in metres)	0.000763	0.000365
Mean of simulated flows (depth in metres)	0.000753	0.000389
Percentage difference between means	-1.3	5.2
Correlation coefficient	0.73	0.65
Regression coefficient	0.756	0.595
Base constant for regression eqn. (m)	0.000177	0.000172
Variance of observed flow	9.84447E-07	3.5536E-08
Variance of simulated values	1.0438E-06	2.989E-08
Standard deviation of observed flow	0.00099	0.0001885
Standard deviation of simulated flow	0.00102	0.000173
Percentage difference in standard deviation	2.97	-8.29
Coefficient of Determination	0.538	0.421
Coefficient of Efficiency	0.482	0.206

In Figures 6.3.6 to 6.3.9 shorter periods of the simulation record have been extracted and graphically represented. These selected periods reflect years with both above average rainfall and below average rainfall. The *ACRU* model had also previously been configured for the Witklip catchment (Gush *et al.*, 2001) and simulation results of this *ACRU* model application are also graphically presented for comparative purposes.

In Figure 6.17a below the simulation results for 1978 are presented. As noted previously 1978 was a high rainfall year and 67% of the annual rainfall fell within the first quarter of the year. During the 1978 wet season, when the catchment saturation deficit was low, the Macaque model over-simulated streamflow. This over-simulation of streamflow during the wet season, was not as pronounced during the above average rainfall years in the late 1980s. This over-

simulation by the model could partly be attributed to the under-estimation of LAI and transpiration.

The *ACRU* model simulated a greater streamflow response to large rainfall events than was observed, whilst for days with small rainfall inputs or no rainfall the simulated streamflow corresponded closely to observed streamflow.

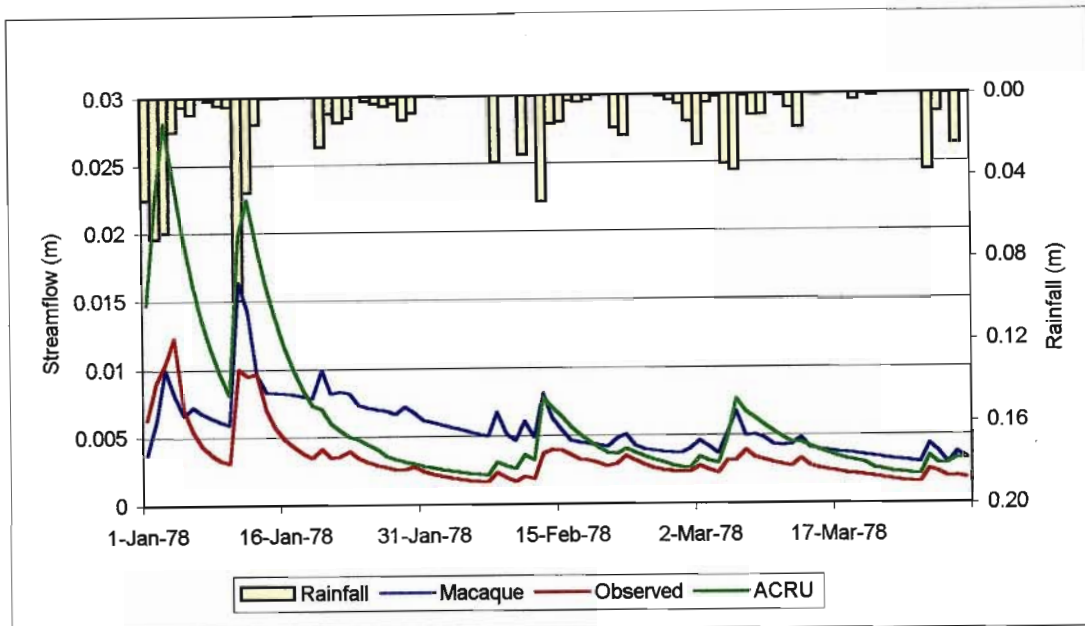


Figure 6.17a. Daily simulated and observed streamflows for the 1978 wet year.

In Figure 6.17b on the following page it is shown that the Macaque model over-simulated low flows during 1978 and showed a less pronounced response to rainfall inputs than was observed. The limited response of simulated streamflow to rainfall inputs was observed for all low flow periods.

The small runoff response simulated by Macaque suggests that the saturated portions of the ESUs were small, in other words the saturation deficits were large, and therefore saturation excess flow contributions to runoff were small. The observed records, however, show that even during the dry season rainfall inputs result in a rapid and significant streamflow response. This catchment response to rainfall inputs during the dry season was observed in most years, whilst the simulated streamflow did not respond to rain events.

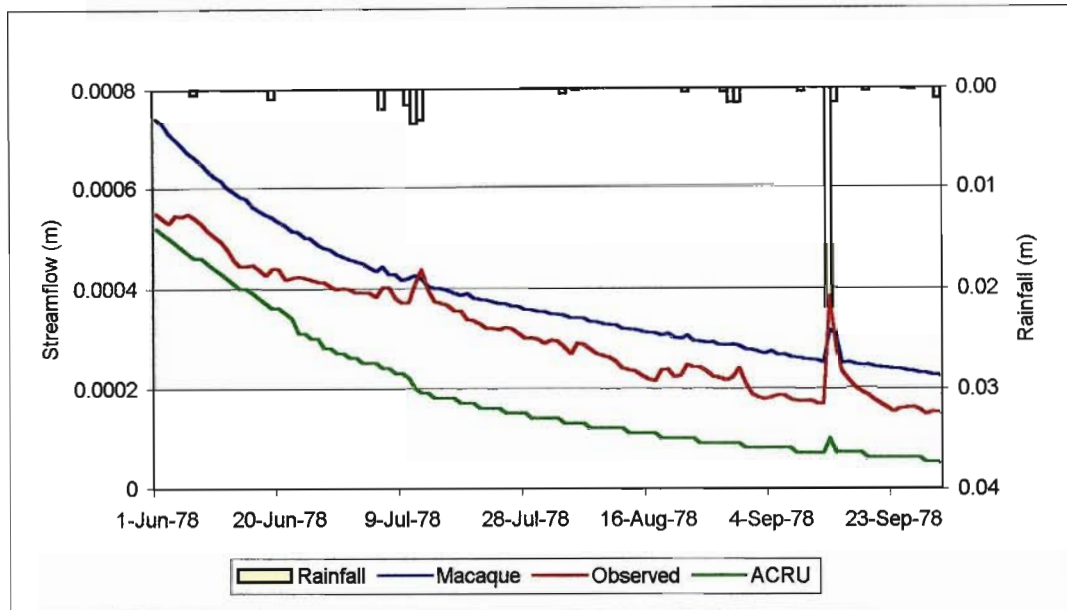


Figure 6.17b. Daily simulated and observed daily streamflows for the 1978 low flow season.

In Figure 6.18a and 6.18b the observed and simulated streamflow for the 1982 below-average rainfall year are shown. The annual rainfall in 1982 was 809.6mm of which only 40% fell within the first quarter. In Figure 6.18a it can be seen that the Macaque simulated streamflow remains almost constant and peak flows are less than observed, indicating that saturation excess flow contributions are small even during large rainfall events. Conversely the *ACRU* model over-simulated peak flows considerably during high rainfall events.

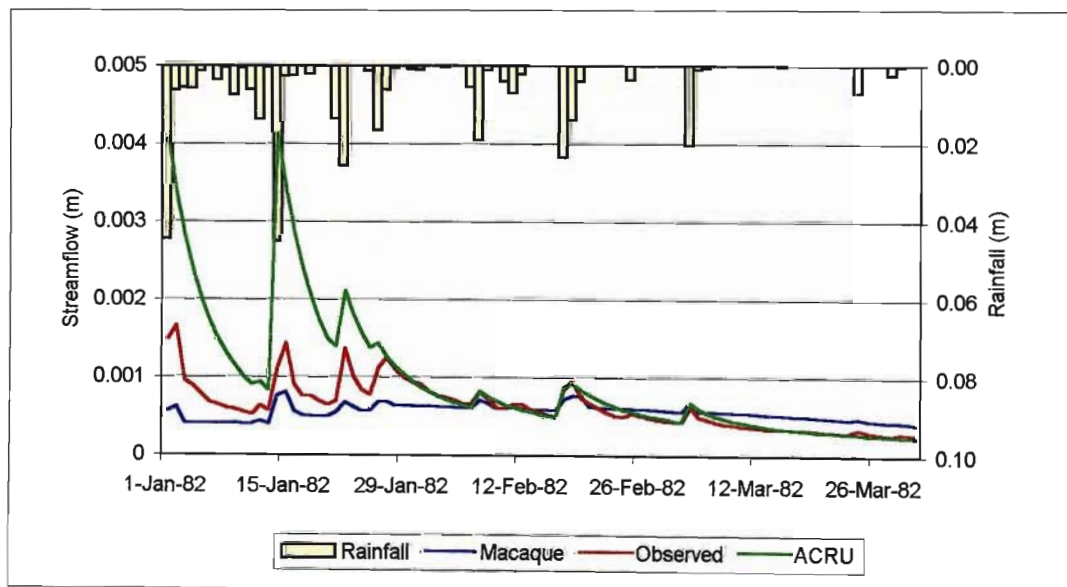


Figure 6.18a. Simulated and observed daily streamflow for the 1982 low rainfall year.

During the 1982 low flow period the Macaque model over-simulated streamflow, whilst during the low flow years after clearfelling the model under-simulated low flows. The simulated peak flow of the *ACRU* model following the rainfall event on 26 July 1982 shows how the quickflow response coefficient (QFRESP) (Schulze, 1995) releases a fraction of stormflow on the same day as the rainfall event, whilst the remaining stormflow is retained and the same fraction is released the following day. This results in a more gradual decline in flow. The observed data, however, revealed a rapid recession following the rainfall event, which is not typical of forested catchments.

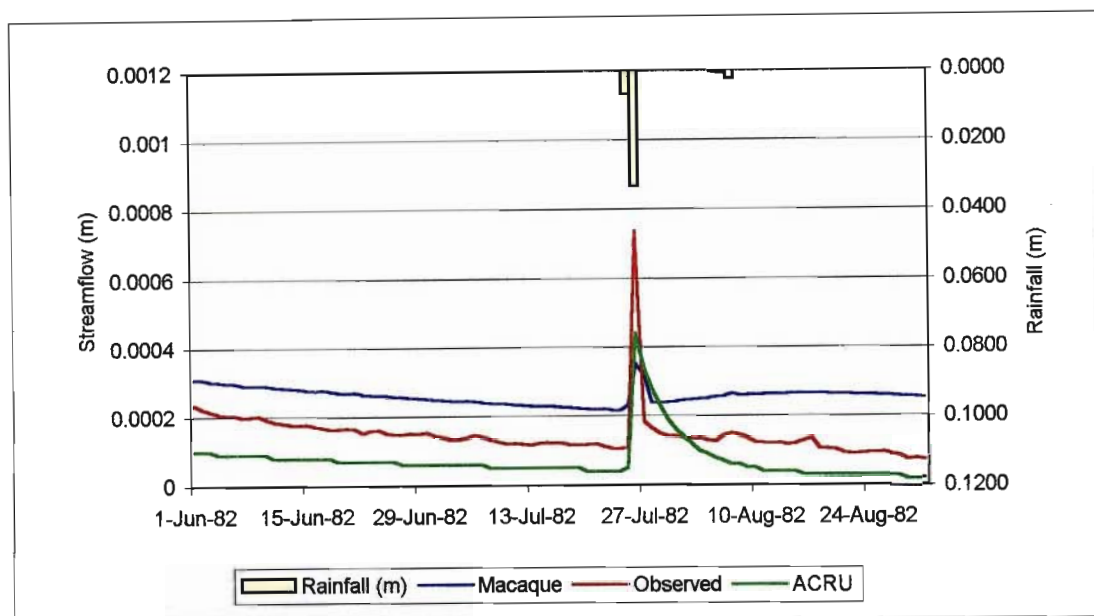


Figure 6.18b. Simulated and observed streamflow for the 1982 low flow period.

The statistical results, shown in Table 6.8, of the Macaque model's monthly totals of daily simulated streamflow versus monthly observed streamflow indicate that the model performance improved at the monthly timescale. The correlation coefficient improved to 0.83 and the monthly mean of simulated streamflow corresponded closely to the observed mean. The difference between simulated and observed standard deviation was slightly higher at the monthly timescale, which indicates that there was a greater dispersion of simulated monthly totals about the mean than for observed monthly totals. The regression coefficient was close to unity and a negative base constant suggested that the Macaque model under-simulated low flows. Low flows were not as well simulated and the correlation coefficient was reduced to 0.65. The simulated and observed means, however, corresponded closely and the variances of simulated and observed values were similar.

The verification results of the *ACRU* simulation (Gush *et al.*, 2001) indicated that the model performed well on the Witklip catchment at the monthly timescale. A correlation coefficient of 0.924 was obtained for monthly totals of daily simulated streamflow. However, the simulated mean was 26% higher than the observed mean and the variance was not well preserved. The regression coefficient of 1.7 indicated that high flows were over-simulated as was illustrated in Figure 6.17a and Figure 6.18a. Low flows were poorly modelled by the *ACRU* model and the mean simulated low flows were less than the observed low flow mean. This was opposite to that observed for the complete simulation record. In Figures 6.19a and 6.19b the monthly simulated and observed streamflow are presented for the periods 1975 to 1982 and from 1983 to 1990, respectively.

Table 6.8. Goodness of fit statistics for simulation of monthly totals of daily streamflow from the Witklip catchment for the Macaque model and *ACRU* model.

	Macaque		<i>ACRU</i>*	
	<i>Complete record</i>	<i>Low flows</i>	<i>Complete record</i>	<i>Low flows</i>
Total observed flows (m)	4.42	0.537	4.42	0.537
Total simulated flows (m)	4.39	0.574	5.61	0.241
Mean of observed flows (m)	0.0232	0.0112	0.0232	0.0112
Mean of simulated flows (m)	0.0230	0.0119	0.0292	0.0050
Percentage difference between means	0.73	-6.7	26.18	-55.22
Correlation coefficient	0.836	0.651	0.924	0.573
Regression coefficient	1.05	0.61	1.765	0.603
Base constant for regression eqn. (m)	-0.00128	0.00509	-0.0117	-0.01739
Variance of observed flow	0.000536	2.794E-05	0.000536	2.794E-05
Variance of simulated values	0.000853	2.491E-05	0.001936	3.092E-05
Standard deviation of observed flow	0.023144	0.00539	0.023144	0.00539
Standard deviation of simulated flow	0.02920	0.00507	0.044115	0.00569
Percentage difference in std deviation	26.2	-5.9	91.1	-5.3
Coefficient of Determination	0.69	0.421	0.853	0.329
Coefficient of Efficiency	0.688	0.229	0.674	-1.050

* Results as documented by Gush *et al.*, 2001.

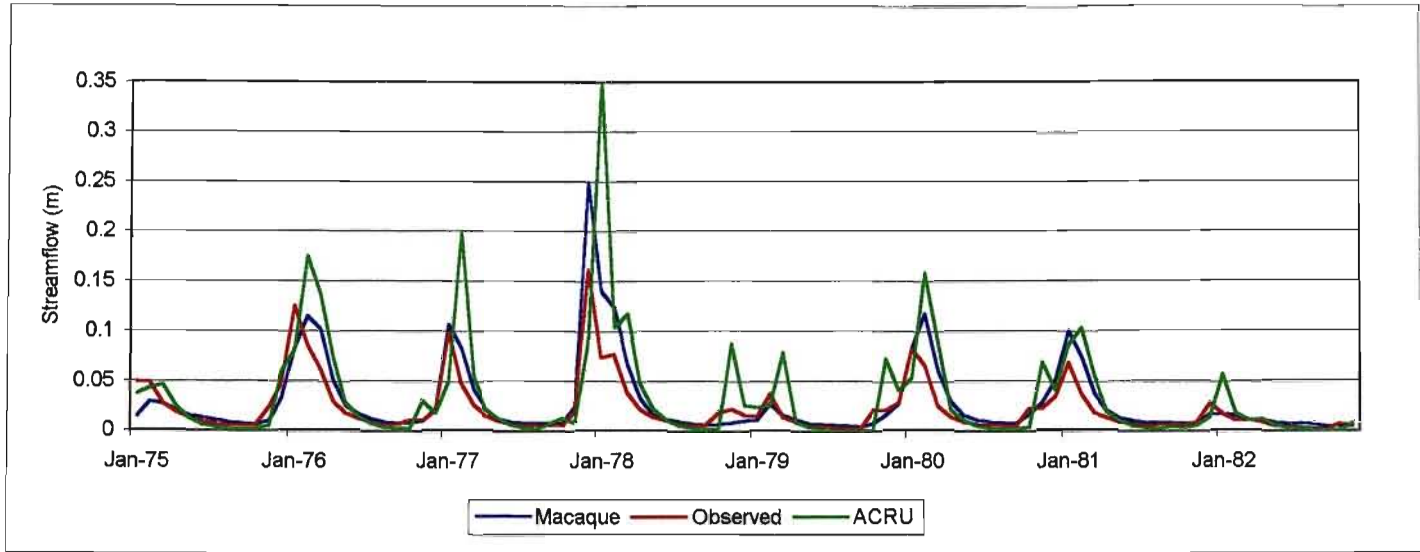


Figure 6.19a. Monthly simulated and observed streamflows for the period 1975 to 1982.

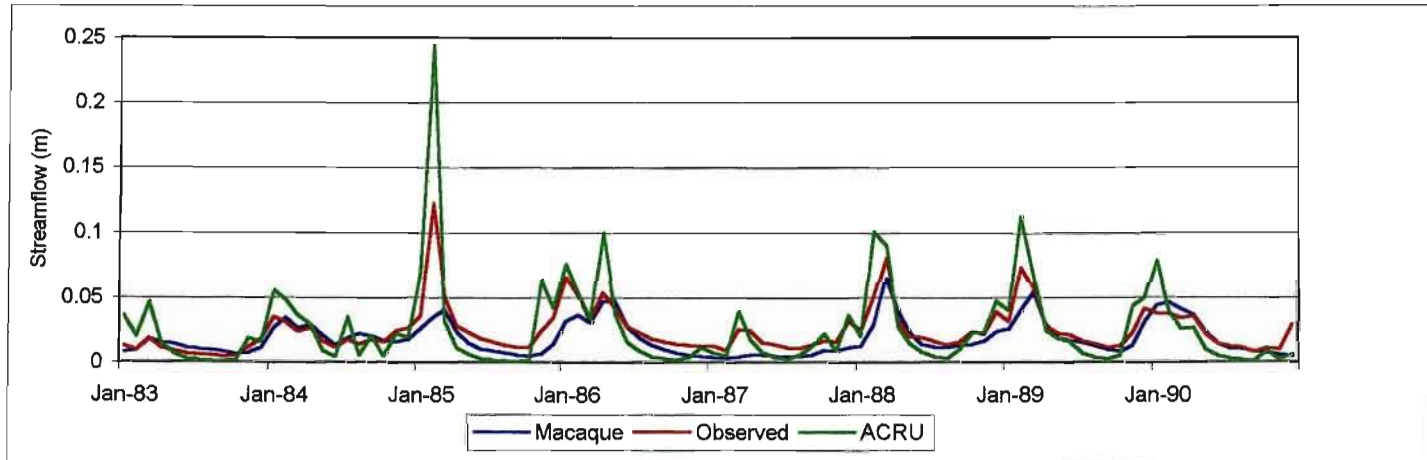


Figure 6.19b. Monthly simulated and observed streamflows for the period 1983 to 1990.

6.3.3.2 Conclusions

The results documented in the above sections indicate that although processes in both the Macaque and *ACRU* model operate at the daily timescale, both models' performance improved considerably when monthly totals of daily streamflow were compared. At the monthly timescale the positive and negative deviations from observed runoff tend to average out, resulting in better simulations results. As for the Maritsane catchment simulation no data, other than meteorological data, were available for internal verification of the Macaque model. Consequently, the sub-surface component of Macaque could not be verified.

Based on the simulations performed on the Maritsane and Witklip catchments, the main problems or limitations experienced during the application of the Macaque model to South African forested catchments were:

- A lack of spatially representative input data such as soil depth data
- Limited model documentation that made determining values for parameters difficult
- The spatial and temporal changes in landcover characteristics such as rooting depths and LAI could not be adequately represented.
- A lack of internal process observations

These problems and limitation are described and discussed in more detail in Chapter 7.

6. 4 Sensitivity of the Macaque model output to input

The most common method of assessing model prediction uncertainties is through sensitivity analyses (Refsgaard *et al.*, 1996). Rogers *et al.*, (1985) define sensitivity as the degree to which model output is sensitive to changes in input parameter values. The analysis of the sensitivity of a model is a useful tool as it gives an indication of the parameters whose values need to be known very accurately because they have a significant impact on the output of the model. The inclusion of such a parameter in a model is only desirable if it can be easily and accurately measured (Schulze, 1995b).

Sensitivity analyses can further be used to identify parameters and processes that are redundant (Franks *et al.*, 1997). By identifying which parameters have insignificant effects on the results, over-parameterisation of the model and equifinality can be avoided (Franks *et al.*, 1997). Equifinality means that the same end may be achieved equally well by a number of parameter sets, all of which may be physically reasonable (Franks *et al.*, 1997). In the following section the procedure followed in performing sensitivity analyses of the Macaque model in the Maritsane and Witklip catchments is described.

6.4.1 Procedure in applying sensitivity analysis

In order to investigate the sensitivity of model output to model input parameters, an input parameter is changed by a relatively small amount while the other parameters are maintained at constant values (Rogers *et al.*, 1985).

The estimation of parameter sensitivity requires a measure of the change in an output function represented by an objective function (Schulze, 1995b). The choice of objective function is very important and should reflect adequately the intended hydrological characteristic (McCuen, 1973, cited by Schulze, 1995b). Rogers *et al.* (1985), for example, used the sum of squared error between predicted and observed discharge as well as the estimated hydrograph peak as objective functions. In the sensitivity analysis of the *ACRU* model, Schulze (1995) used the percentage change in simulated mean annual runoff as the objective function. For the study of parameter sensitivity in the Macaque model the percentage change in mean monthly streamflow was selected as the objective function. An objective function based on monthly streamflow was chosen because an annual timescale would have masked the intra-annual

variations of the model output to changes in input, but analysis at a daily timestep is very complex and overly data intensive for long periods of simulation. The objective function is represented by the following equation:

$$\Delta O\% = \left(\frac{O - O_{\text{base}}}{O_{\text{base}}} \right) \times 100 \quad \text{Equation 6.1}$$

where

- $\Delta O\%$ = per cent change in mean monthly streamflow (output)
- O = mean monthly streamflow from a particular change in input parameter
- O_{base} = mean monthly streamflow from the base input

The same equation is used to calculate the per cent change in the input parameter ($\Delta I\%$) when changing the base input parameter (I_{base}) and the adjusted input parameter (I). For the sensitivity analysis of the Macaque model, the sensitivity ranking used in a sensitivity analysis of the *ACRU* model (Schulze, 1995) was adopted and is as follows:

- Extremely sensitive: the change in output ($\Delta O\%$) is more than twice that of the input parameter ($\Delta I\%$); $\Delta O\% = 2 * \Delta I\%$
- Highly sensitive: the output change is more than the change in input, but by less than 200%; $\Delta O\% > \Delta I\%$
- Moderately sensitive: relative output change is less than the input change, but more than 50% of input change; $\Delta I\% > \Delta O\% > 0.5 * \Delta I\%$
- Slightly sensitive: output changes between 10% to 50% of the input change; $0.5 * \Delta I\% > \Delta O\% > 0.1 * \Delta I\%$
- Insensitive: output changes by less than 10% of the input change; $\Delta O\% < 0.1 * \Delta I\%$

Prior to describing the results of the sensitivity analysis, the procedure followed to set up the Macaque model needs to be described. The Macaque model is set up by running a series of simulations in order to achieve “dynamic equilibrium” (Watson, 2000, personal communication). For the sensitivity analyses performed, dynamic equilibrium was considered to have been reached when the percentage difference between daily streamflow totals for successive simulations was less than 1%.

When a simulation is initialised, the saturation deficit of the soil (s_{sat}) and the unsaturated store (s_{unsat}) of each of the ESUs define the initial soil moisture status of the catchment. The model is then run and the s_{sat} and s_{unsat} of an ESU at the end of a simulation become the initial values at the start of the new simulation. By running a 10-year simulation repeatedly one is able to achieve dynamic equilibrium, whereby the values of s_{sat} and s_{unsat} at the start of the simulation are of little consequence. However, as described below this state of dynamic equilibrium was not achieved for all simulations.

A sensitivity analysis was initially performed using data for the small, 1.08 km² Witklip catchment. However, it became evident that changes to selected input parameters resulted in a failure of the model to reach dynamic equilibrium. In the example shown in Figure 6.20, the soil depth parameter was increased by 25% and the model repeatedly run to reach equilibrium. For successive simulations, with all parameters constant, percentage differences in daily streamflow between successive simulations of up to 4% were observed. These differences were alternatively negative and positive and usually observed during the low flow periods (dry winter months).

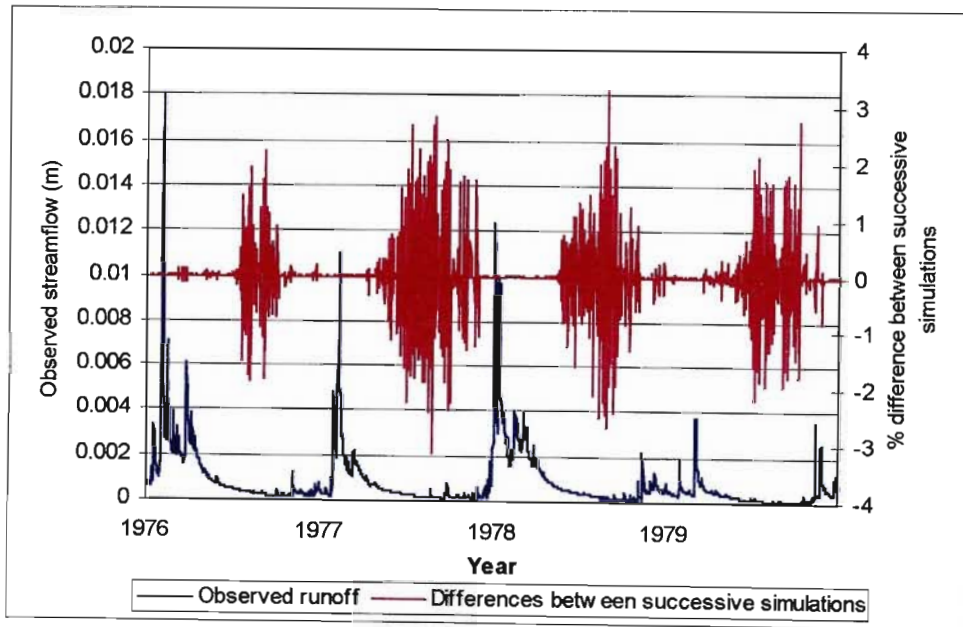


Figure 6.20. Percentage difference between successive simulations for all parameters kept constant for the Witklip catchment.

A possible explanation would be that soil water imbalances occurred during the dry season when soil water deficits were high and in certain ESUs transpiration exceeded soil water

availability. However, it is unclear how such soil water imbalances could be brought about. This trend was also observed when other parameters, such as the ratio of hydraulic to surface gradient, were adjusted during the sensitivity analyses on the Witklip data. During the Maritsane catchment sensitivity analyses this problem was not encountered and therefore these inconsistent results could be related to the small catchment size and scale considerations. Due to these anomalies the sensitivity analyses on the Witklip catchment was discontinued. The results presented below are from the sensitivity analyses performed using the Maritsane catchment's data.

6.4.2 Parameters tested in a sensitivity analysis of Macaque

During the application of the Macaque model to South African catchments, as discussed in section 6.1, it became evident that the absence of required input data was one of the principal limitations of applying the model in South Africa. It was therefore necessary to establish the sensitivity of parameters that were either difficult to measure in the field, or parameters for which values were not available. Furthermore, based on the understanding of the model structure, parameters were selected that were likely to have a more pronounced influence on model output. The following parameters were selected:

- Ratio of hydraulic to surface gradient
- Saturated surface hydraulic conductivity
- Saturated hydraulic conductivity depth parameter
- Maximum available volumetric water content of soil water
- Maximum leaf conductance

In the following section the sensitivity of the model output to changes in these parameters is described and discussed.

The ratio of the hydraulic to surface gradient

As discussed in section 5, the ratio of hydraulic to surface gradient has a large influence over the variability of the predicted monthly streamflow (Peel *et al.*, 2000). Watson (2000, personal communication) reports that model output is most sensitivity when the sub-surface parameter approaches 1.0. The results from the sensitivity analysis, illustrated in Figure 6.21,

verify this statement and an over-estimation of this parameter would result in large changes in mean monthly streamflow. The base input of hydraulic to surface gradient was 0.95 and the mean simulated streamflow prior to adjustment of the parameters, listed in Section 6.4.2, were 0.00038m/day and 0.00051m/day for the low flow months (June and July) and high flow months (January and December), respectively.

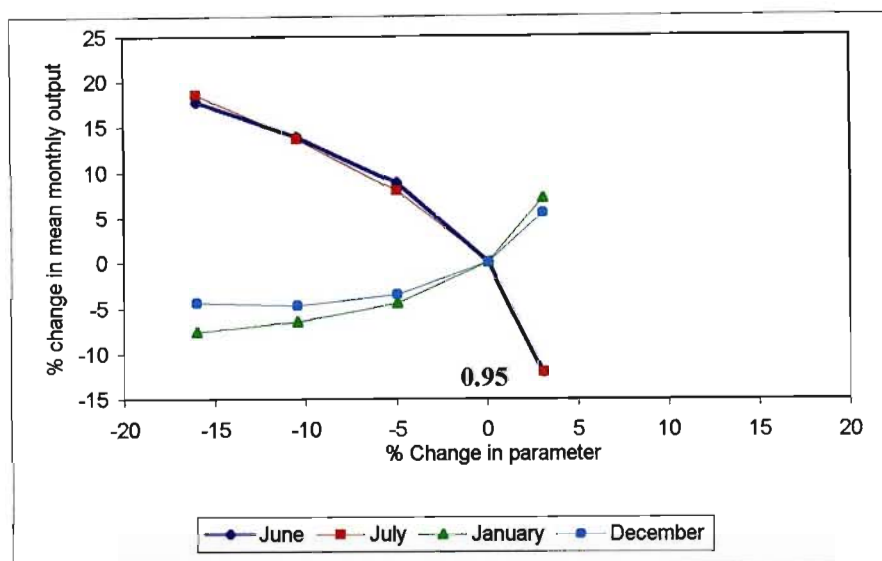


Figure 6.21. Sensitivity of mean monthly streamflow to changes in the ratio of the hydraulic to surface gradient.

The ratio of the hydraulic to surface gradient parameter is used in the calculation of both saturation excess runoff and baseflow release. Based on Equations 5.1 and 5.2 provided in Section 5, it can be proved that by increasing the ratio of the hydraulic to surface gradient, baseflow contributions to streamflow are reduced and saturation excess flow is increased and vice versa. Therefore, during the dry months (July and August) when the water tables are low, saturation excess flow is minimal and baseflow is the largest contribution to streamflow, an increase in the ratio of the hydraulic to surface gradient will result in significant decreases in streamflow. In the wet months when saturation excess flow is dominant, an increase in the ratio will result in increases in streamflow. The sensitivity of streamflow to changes in the ratio of the hydraulic to surface gradient parameter will depend on whether saturation excess flow contributes to streamflow on a given day during the wet season. The implication of whether or not saturation excess flow contributes to streamflow is that the overall sensitivity of streamflow, averaged over the month, is moderate.

From the results shown in Figure 6.21 it is apparent that streamflow output during the low flow period is highly sensitive to changes in the $p_ratio_hydraulic_to_surface_gradient$ parameter and a major shortcoming would be that this parameter, in reality, is not physically measurable. However, from Macaque simulations of the Maroondah and Thompson catchments (Peel *et al.*, 2000), it became evident that a relationship existed between the catchment area and $p_ratio_hydraulic_to_surface_gradient$ parameter. It was found that as the catchment area increases, so does the ratio of hydraulic to the surface gradient required to produce a good calibration (Peel *et al.*, 2000). For large-scale applications when the parameter approaches 1.0, the extreme sensitivity of the parameter will necessitate calibration of the model against the storm hydrograph, particularly to achieve accurate results during low flow period.

Surface saturated hydraulic conductivity (K_{sat})

The K_{sat} parameter is used in three processes in Macaque. Firstly, it is used to calculate the transmissivity of the saturated zone, described in Chapter 5, which controls the rate of sub-surface flow. Secondly, the hydraulic conductivity at the water table interface is also derived from this parameter and K_{sat} therefore controls the rate of recharge from the unsaturated store to the saturated store. Thirdly, the K_{sat} parameter determines the rate at which baseflow is ex-filtrated from the saturated zone.

Mean monthly streamflow is only slightly sensitive to changes in K_{sat} , as illustrated in Figure 6.22, and a change in K_{sat} of 25% results in a change of only 5% in monthly streamflow. As was observed with the ratio of hydraulic to surface gradient parameter, the sensitivity of mean monthly streamflow differs between the dry and wet seasons. Over-estimation of the parameter during the wet season results in higher simulations of monthly streamflow, whilst the opposite trend is observed during the dry season. Increases in K_{sat} result in increases in lateral sub-surface flow and baseflow and these increases are more pronounced during the wet season when saturation deficits are small. However, during the dry season the overall streamflow is reduced when K_{sat} is increased. This phenomenon could be attributed to the more rapid reduction in water table depths due to greater baseflow and lateral flow releases in late summer and autumn. Hence the greater releases prior to commencement of the low flow period result in a lower catchment moisture status and slightly reduced flows during the dry period.

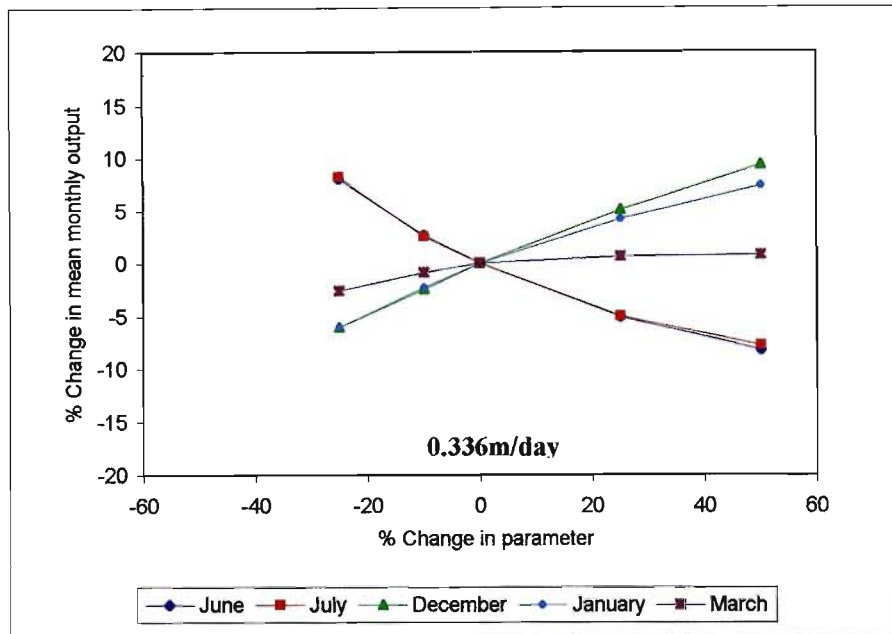


Figure 6.22. Sensitivity of mean monthly streamflow to changes in surface saturated hydraulic conductivity

The saturated hydraulic conductivity of the soil can be measured, although the measurement of K_{sat} , using techniques such as the tension infiltrometer are time consuming and restricted to point measurements. These point measurements then have to be interpolated to generate spatially representative values of K_{sat} for the catchment. Therefore, other soils data that are more easily obtained such as textural data are often used to derive K_{sat} values.

Saturated hydraulic conductivity depth (K_{depth})

The saturated hydraulic conductivity depth parameter represents the depth of the soil profile to solid bedrock. It directly affects the storage capacity of the soil and is also used in the calculation of transmissivity of the soil, which influences the rate of sub-surface or lateral flow.

As illustrated in Figure 6.23, the monthly streamflow displays a moderate sensitivity to changes in the saturated hydraulic conductivity depth parameter during the dry season, whilst during the wet season (summer months) changes in the K_{depth} parameter has little influence on monthly streamflow. During the months of December and January increases and decreases in K_{depth} results in both positive and negative changes in mean monthly streamflow. By increasing K_{depth} the soil transmissivity is increased whereby lateral flow is increased and

streamflow is increased. However, during the wet season it is evident that another more dominant process, affected by the K_{depth} , produces both increases and decreases in streamflow for different years.

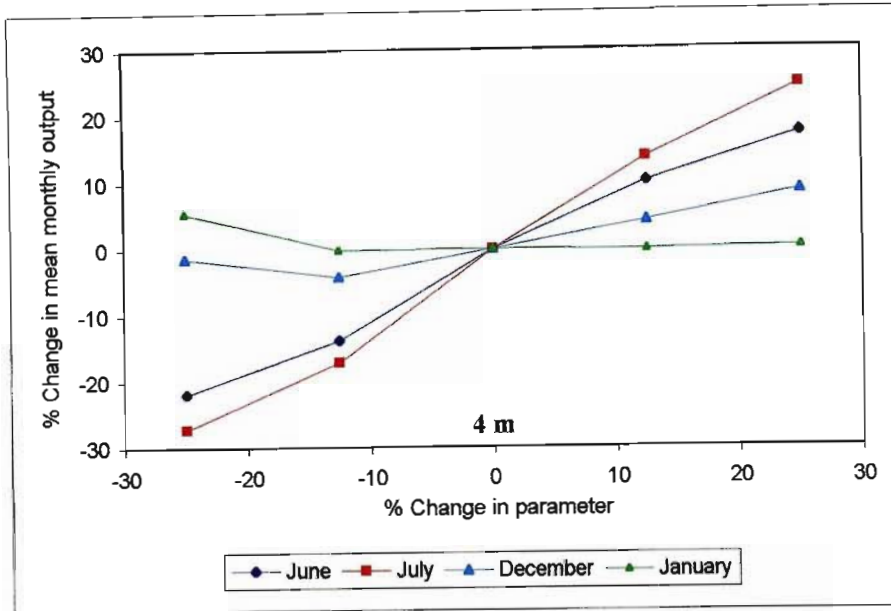


Figure 6.23. Sensitivity of mean monthly streamflow to changes in saturated hydraulic conductivity depth

The depth of the soil to the bedrock information is rarely available and although point measurements could be collected, it is difficult to extrapolate these data to get spatially representative values for large catchments. For both the Witklip and Maritsane catchments no spatial variation in soil depth was assumed, although this assumption was known to be incorrect. However, the absence of suitable soils data makes this assumption unavoidable. The implication of this is that by under-estimating or over-estimating soil depths, the low flows, in particular, are simulated incorrectly.

Saturated volumetric soil water content

By increasing the saturated volumetric soil water content, the maximum volumetric water content of the soil is increased and thereby the water holding capacity of the soil is increased. This greater storage capacity of the soil allows for the greater accumulation of water during the wet season when rainfall inputs are high. During the dry season the store is gradually depleted as a result of evapotranspiration losses and baseflow/subsurface flow releases.

however the soil water deficits remain smaller than would have been the case if the water storage capacity of the soil were lower. As a result the greater θ_{\max} and smaller soil water deficits allow for greater sub-surface lateral flow and baseflow releases during the dry season. From Figure 6.24 it is evident that streamflow is highly sensitive to changes in saturated volumetric soil water content during the dry months.

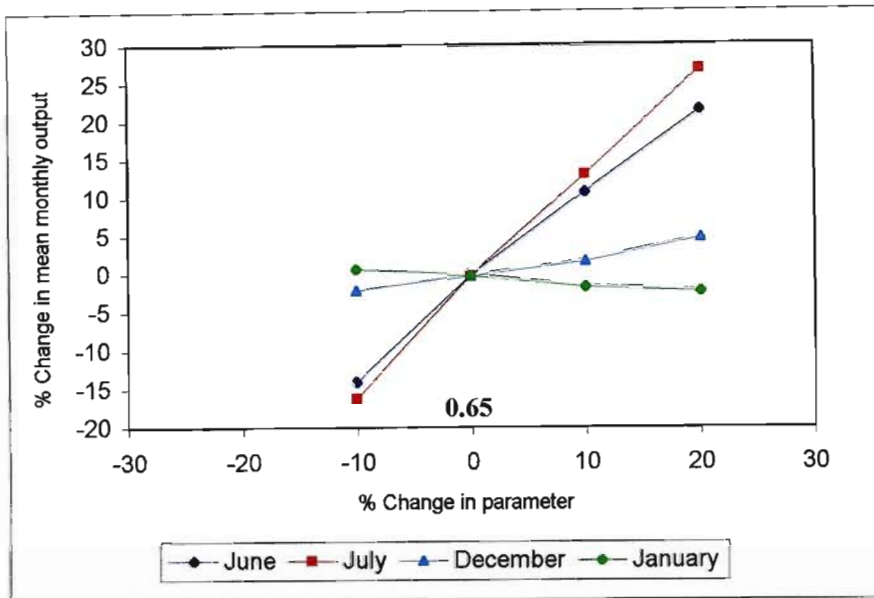


Figure 6.24. Sensitivity of mean monthly streamflow to changes in saturated volumetric soil water content.

During the wet season, an increased or decreased storage capacity of the soil has little effect on the monthly streamflow. This is because soils are typically saturated during the wet months and for small saturation deficits, the increase in water table depth resulting from decreases in saturated volumetric soil water is minimal as shown from a theoretical analysis illustrated in Figure 6.25a. These small changes in water table depth could result in small changes in lateral flow. However, changes in saturated volumetric soil water content during the dry season could result in significant changes in lateral flow. Furthermore, with a small s_{sat} (wet soil), the portion of an ESU that is saturated to the surface does not differ greatly for changes in θ_{\max} as shown in Figure 6.25b. However, when s_{sat} is large as would be the case during the dry season, greater differences in the portion of saturated area are observed with changes in θ_{\max} . As the saturated portion of an ESU increases so does the baseflow which, during the drier periods, is the main contributor to total flow.

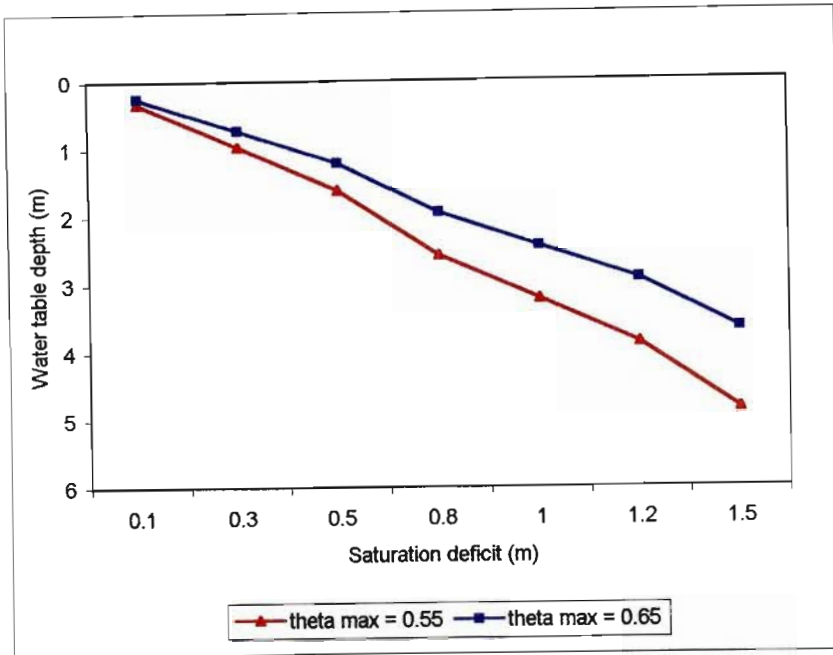


Figure 6.25a. Changes in water table depth with changes in maximum volumetric water content and saturation deficit.

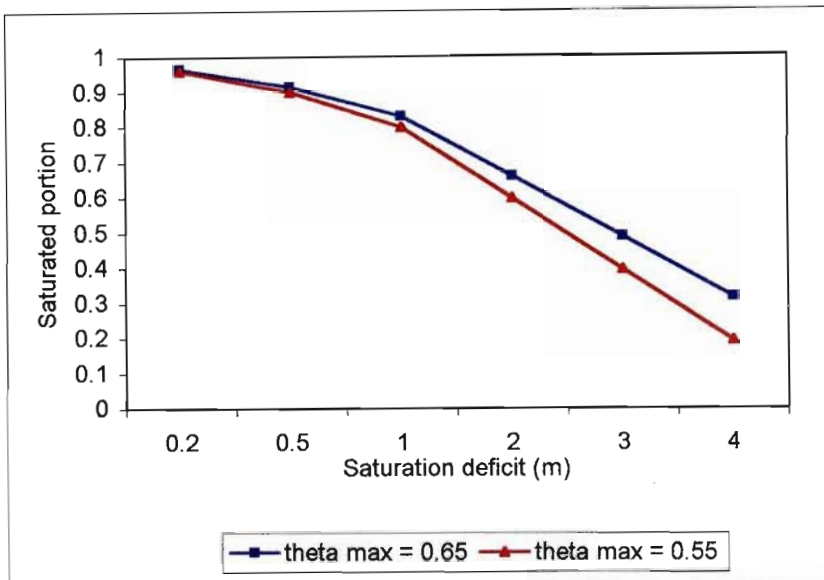


Figure 6.25b. Changes in the saturated portion of an ESU with changes in maximum volumetric water content and saturation deficit.

Maximum leaf conductance ($g_{cl,max}$)

The magnitude of daily canopy transpiration is dependant on the canopy transpiration rate, which in turn is dependant on factors such as net radiation, canopy conductance, and vapour pressure deficit. The transpiration rate calculated using the Penman-Monteith equation is

particularly sensitive to changes in canopy conductance and small increases in canopy conductance result in large increases in the transpiration rate. The canopy conductance parameter used in the Penman-Monteith equation is derived from maximum leaf conductance ($g_{cl,max}$) by multiplying the $g_{cl,max}$ by LAI and the soil/leaf moisture multiplier described in Chapter 5. By increasing $g_{cl,max}$ the transpiration rate is therefore enhanced, with the result that the streamflow is reduced.

The model output displays extreme sensitivity to both over and under estimation of maximum leaf conductance as shown in Figure 6.26. The sensitivity of mean monthly streamflow to changes in $g_{cl,max}$ is slightly higher during the wet summer months. This is attributed to the fact that during the summer it is likely that transpiration is occurring at maximum rates, limited only by maximum leaf conductance. However, during the dry season other factors affecting transpiration rates, such as soil water availability, play a more dominant role in controlling daily transpiration.

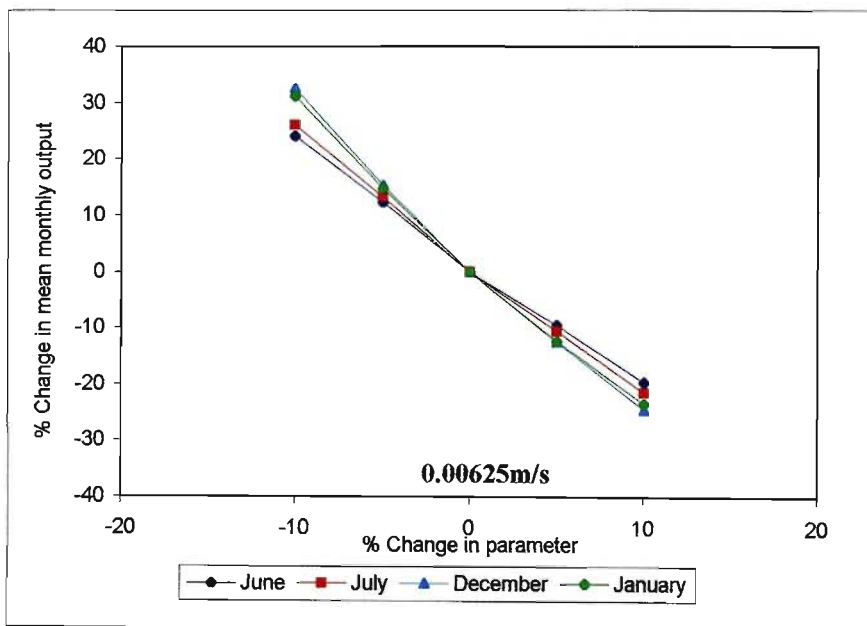


Figure 6.26. Sensitivity of mean monthly streamflow to changes in maximum leaf conductance.

6.4.3 Conclusions

It should be noted that the sensitivity of the parameters described in the previous Section 6.4.2 are specific to the Maritsane catchment and specific to the parameter set, used in this simulation. For a different catchment with a different parameter set the model output may not

display the same sensitivity to changes in input parameters since model parameters do not operate in isolation of one another.

The results of the above sensitivity analysis show that mean monthly streamflow is highly sensitive to a number of soil and vegetation parameters in the Macaque model and more importantly that these parameters have a significant affect on streamflow during the low flow periods. Conversely, the mean annual streamflow is insensitive to changes in the parameters tested, with the exception of maximum leaf conductance. Simulated mean annual streamflow is highly sensitive to changes in maximum leaf conductance and small increases in this parameter result in large increases in mean annual runoff. The other parameters influence the variability of predicted streamflow during the year, but do not affect the predicted annual runoff.

In order to achieve a good simulation during the low flow period, which is critical for the assessment of impacts of forestry, it is necessary that these parameters be accurately known. Although selected parameters are measurable, the question arises whether these parameters are practical to measure at larger spatial scales in order to obtain spatially representative values. Most of these parameters are either difficult to measure or the data is unavailable at the large spatial scale which increases the uncertainty of model results.

* * *

In Chapter 6 the Macaque model application to South African catchments was discussed and some of the difficulties in obtaining input data described. To illustrate the potential effect of poor or inadequate input data on model results a sensitivity analysis was performed. From the results of this analysis it is evident that the Macaque model output is sensitive to a number of input parameters. In the following Chapter the feasibility of applying the Macaque model to South African catchments and the viability of adopting components of the model for incorporation into the *ACRU* model is discussed. Some of the difficulties of applying the Macaque model to South African catchments which were alluded to in Sections 6.2 and 6.3 are further discussed and analysed.

7. DISCUSSION AND CONCLUSIONS

As competition between water users for a share of the water resource increases, water resource managers have the task of optimizing the allocation of water. For this task they require models for use in multi-scenario evaluations. It has been argued that physically based, distributed models are the most suitable in such cases. However, the application of physically based models for water resource management presents a number of problems. Firstly, such models are typically developed by academic institutions and typically a degree of expertise and experience is required when applying them. Secondly, the models have high input demands and the input data are often difficult to obtain and requires considerable processing before use, unless this procedure can be automated. It is these two main problems that are highlighted in the application of the Australian Macaque model in the Witklip and Maritsane catchments in South Africa.

Despite these problems, certain aspects of the Macaque model may warrant further research. In order to determine where the Macaque model could aid development of the *ACRU* model the following issues need to be discussed and analysed:

- Differences in process description between the *ACRU* and Macaque models.
- Identifying the strengths and weaknesses of each model.
- Problems and limitations of applying physically-based models to South African catchments.
- Inclusion of a lateral flow component in the *ACRU* model.

In Table 7.1 a synthesis and comparison of the processes and parameters of the Macaque model and the *ACRU* model is provided. The sensitivity of each model's output to changes in model parameters is also shown in the table. These sensitivities of the *ACRU* model are sourced from the *ACRU* theory manual (Schulze, 1995d), whilst the Macaque model's parameter sensitivity has either been obtained from available literature (Watson, 1999a), or is based on the sensitivity analysis performed as part of this dissertation and discussed in Chapter 6. Both the *ACRU* and Macaque models contain a number of parameters that when changed have a significant effect on model output.

Table 7.1 Comparison of parameters of the *ACRU* model and Macaque model.

Process influenced	<i>ACRU</i> model			Macaque model		
	Parameter/s	Sensitivity ^I	Known or Measurable?	Parameter/s	Sensitivity ^{II}	Known or Measurable?
Spatial variability of rainfall	PPTCOR	H	Yes	MMPI/MLR	H	Yes
Canopy interception	LAI, [Pg] or I ₁	-	Yes	k _{rain} & LAI	M	Yes
Evaporation of intercepted water on forest canopy	LAI	-	Yes	r _{aero,c,ref} (k _{rain} & LAI, [T _{max} , T _{min}])	-	Yes
Stormflow/saturation excess flow	SMDDEP	H	No	p_ratio_surface_hydraulic_gradient	H	No
Sub-surface stormflow/lateral flow	QFRESP	H	No	K _{sat} , K _{min} , K _{shape}	-	Yes
Baseflow generation	COAIM	M	No	p_ratio_surface_hydraulic_gradient	H	No
Baseflow generation				K _{sat}	M	Yes
Transpiration (atmospheric demand)	[E _t]	H	Yes	r _{aero,c,ref} (g _{cl,max} , g _{cl,min} , [T _{max} , T _{min}])	-	Yes
Transpiration (plant physiological control)	LAI/ CAY	H	Yes	g _{cl,max}	H	Yes
Soil water extraction by roots	ROOTA, ROOTB	-	Yes	Z _{c,max} , p _{c,sat}	-	No
Water holding capacity of the soil	ITEXT	L	Yes	θ _{max} , θ _{min}	-	Yes

H = Highly sensitive M = Moderately sensitive L = Low sensitivity

^I Measured as the change in simulated Mean Annual Runoff (MAR) due to a change in input parameter.

^{II} Measured as the change in simulated mean monthly streamflow due to a change in input parameter.

() Parameters that exert indirect control [] Variables (see Chapter 4 and Chapter 5)

Furthermore, certain of these sensitive parameters shown in Table 7.1 are described as being not measurable. The implication of this, is that a degree of subjectivity and uncertainty is introduced to the parameter selection process and in the absence of streamflow records for model calibration, it is possible that small changes in these parameters could produce very poor results. The remaining parameters in the table are listed as being measurable, however, in practice the cost and time needed for measurement and data analyses could make this task impractical at large spatial scales. Furthermore, the scale of the measurement technique available may be less than the scale at which the parameter values are required by the model.

The *ACRU* model and Macaque model are both distributed, process-based models but have adopted different approaches to representing the dominant processes in forested catchments. A similar approach is taken by *ACRU* and Macaque to represent spatial variability of rainfall over the catchment. The Macaque model, however, does not accommodate intra-annual changes in the spatial distribution of rainfall. Conversely, the *ACRU* model can accommodate a change in spatial distribution of rainfall for each month of the year. This latter approach is more realistic since different seasons are often characterised by different rainfall patterns.

The description of runoff processes and transpiration differ considerably between the two models. The Macaque model explicitly represents runoff processes acting in the catchment and uses topography (DEM) to direct water movement downslope and to determine the saturated portion of an ESU. The Macaque model approach to runoff generation is based on the assumption that infiltration excess flow does not occur widely in forested catchments. The simulated streamflow is a combination of saturation excess flow, baseflow from each ESU and sub-surface outflow from the lowest ESU in the hillslope.

The *ACRU* model does not explicitly represent the different runoff generating mechanisms. The model instead uses a set of parameters, namely SMDDEP, QFRESP and COAIM to control the runoff response of a catchment following a rainfall event. The parameter values that are chosen are based on the physical characteristics of the catchment; such as land cover type and climate characteristics. By selecting appropriate parameters the modelled streamflow response reflects the dominant runoff processes acting in the catchment.

The Macaque model requires a large number of parameters that are either used directly or indirectly in the calculation of daily transpiration. These include parameters that characterise

the response of stomatal conductance to environmental factors such as soil moisture content and humidity. The majority of the parameter values used in the Maritsane and Witklip simulation were sourced from literature describing studies undertaken in temperate forests of the Northern latitudes. Without actual transpiration measurements for internal verification there is uncertainty as to whether such detailed representation is justified considering the uncertainty in input parameters. In the *ACRU* model the factors affecting transpiration are usually confined to the LAI or CAY, reference potential evaporation and soil water availability. It became evident when the Macaque model was applied to South African catchments, that the complexity of the model could not be supported by the available data. There is also a degree of incompatibility between model components in Macaque. Whilst the environmental controls on leaf conductance and hence transpiration are represented in great detail, the seasonal changes in spatial distribution of rainfall have been lumped into a single MMPI, despite the fact that the model is more sensitive to precipitation inputs.

The Macaque model spatial template and link to the Tarsier GIS provides an efficient means of inputting spatial data. The potential for input errors is greatly reduced since the user is not required to manually input the data. Temporal variations such as inter-annual and intra-annual temporal variations in parameters, on the other hand, could not adequately be represented in Macaque. For example the model could not accommodate intra-seasonal changes in LAI or rainfall distribution and it was apparent from the Witklip and Maritsane catchments that the model could not readily accommodate the combination of spatial and temporal changes for different parameters. Although the data input process in *ACRU* is more cumbersome, whereby parameters are manually entered for each sub-catchment, the model is more flexible in terms of being able to representing temporal changes for a large number of parameters.

Although the Macaque model inherits many components from other models, it has only been in existence since 1999 and therefore the model has not been extensively applied and tested on catchments other than the Maroondah catchment on which the model was developed. It may be that the complexity and process representations were developed to suit the data availability of the Maroondah research catchment and data availability in research catchments often exceeds that of operational catchments. The South African Maritsane and Witklip catchments differed from the Maroondah catchments in that the type of data available differed, the density of data measurements were less and there was a greater variability in

landuse and landcover. Although the Macaque model could accommodate the changes in LAI with increasing age for forests indigenous to Australia, the model could not readily accommodate the variability in vegetation characteristics of multiple land uses, typical of operational catchments in South Africa, or the variability of tree species and tree age in commercial plantations. The *ACRU* model is better adapted to accommodate different vegetation types, by allowing intra-annual (seasonal) changes in LAI, root mass distribution and other vegetation physical characteristics and inter-annual changes in LAI and vegetation physical characteristics by means of the dynamic land use file.

An advantage of the *ACRU* model is that it has been in existence for over a decade and has been extensively applied to a range of different catchments. As a result the model has undergone continuous development to accommodate new data sources, to incorporate new process knowledge and to overcome the data limitations experienced when applying the model to operational catchments. The *ACRU* model has been developed to function both as a research model and as a predictive model to assist in management decisions. This is evident from the different options in the model. One such example is that, depending on the data availability, the user can opt to use different methods to calculate reference potential evaporation. Furthermore, a comprehensive user manual and detailed model documentation has been compiled. This documentation provides the user with a detailed description of each of the parameters used, a description of processes represented in the model and recommended parameters for different climatic regions and vegetation types. Such a user manual provides that user with a comprehensive understanding of how the model functions and aids the user in selecting appropriate parameters, thereby reducing the possibility of entering incorrect parameters.

A lack of detailed model documentation is a major shortcoming of the Macaque model. Although the initial version of Macaque had been well documented by the model developer, more recent developments and model changes have not yet been documented. As a result the model user does not have a clear definition of the parameters used in the model and has to essentially rely on examining the source code to gain an understanding of model processes and model operations. Without comprehensive documentation the application of such complex, physically based models remains limited to the research institution where the model was developed.

Associated with the need for documentation providing definitions and descriptions of the various model parameters and processes, is the need for documentation of model assumptions and parameter sensitivity as was discussed in Chapter 2. The assumptions used and implications thereof have been very clearly stated in the Macaque model in the form of detailed comments in the source code. These assumptions and indications of where improvements are needed provide the user with an understanding of the model limitations and weaknesses. With the exception of a few processes, most processes within the Macaque have undergone internal verification on the Maroondah catchment. The verification of model components has been part of the model development process and a considerable amount of field measurements have been made, although confined to the Maroondah catchments. The verification of the *ACRU* model for forested application has largely been confined to the verification of simulated streamflow against observed records and very few studies have attempted to assess the performance of the internal processes within *ACRU*. This shortage of internal verification studies can be attributed to the lack of field measurements against which the simulated values can be compared. For example, few catchments in South Africa have measurements been taken of a combination of rainfall, vegetation interception losses, transpiration, soil moisture levels and streamflow. It is this lack of data that prevents the complete verification of models and the ability to analyse where a model is failing.

The runoff representation in the *ACRU* model has been described as too conceptual and it was suggested by Görgens and Lee (1992) that this “black box” approach be replaced by a more physically-based approach, whereby description is given to the processes controlling the response of the catchment. The Macaque model presented a more physically-based description of runoff mechanisms operating in a catchment. These include a lateral sub-surface component based on the Darcy’s Law and the representation of partially saturated areas and saturation excess flow resulting from the rise and fall of the water table. However, based on the above discussion it is apparent that limitations exist in adopting a more physically-based approach to modelling sub-surface flow and runoff processes. From the application of the Macaque model in South African catchments the following limitations were identified:

- The absence of relevant detailed soil input data.
- The model output was very sensitive to changes in subsurface parameters.
- There is limited ability to measure sub-surface processes for model verification.

Therefore, although the model may represent subsurface processes more explicitly, the additional complexity cannot be supported by the available data. For these reasons incorporating the Macaque sub-surface flow component into the *ACRU* model is not a viable solution to the problems identified by Görgens and Lee (1992).

However, despite the uncertainty of input data and parameter values, promising results were obtained from the Macaque simulations on the Witklip and Maritsane catchments. The Macaque model produced better results for the low flow months than the *ACRU* model for both catchments. The *ACRU* model consistently under-simulated low flows in forested catchments and in some cases simulated streamflow ceased altogether during the dry season. Conversely, peak flows during the wet season were typically over-simulated by the *ACRU* model. The runoff process representation by the Macaque model, however, was able to maintain the naturally occurring low flows during the dry season. The reason why Macaque performed better for the low flow period is likely to be the result of the greater storage capacity and deeper soil profile represented by the model. This representation allows for the accumulation of water during the wet season when rainfall inputs are high. Therefore, during the dry season when the rainfall inputs are small the model is able to sustain baseflow releases due to the large accumulated store of water in the soil. Therefore, it is not so much the influence of lateral flow, but the determination of baseflow using hydraulic gradient and topography that provides improved baseflow simulations during the low flow period. Conversely, in the *ACRU* model the over-simulation of peak flows during the wet season suggests that a large amount of water is released from the catchment during a rainfall event. Although the initial abstractions (COAIM and SMDDEP parameters) are set high for forest simulations to allow for greater infiltration and reduced stormflow release, the modelled stormflow response to rainfall input is often greater than observed stormflow. As a result this stormflow that would in reality have been available for the replenishment of the soil water store and baseflow store for release during dry season is “lost”.

Although the Macaque model’s runoff representation may not be ideal due to the data limitations, these results suggest that a topography-based approach to control the timing and volume of runoff releases has the potential for improving the *ACRU* results during the low flow period. However, without extensive internal verification it cannot be concluded that the Macaque model’s runoff representation provides a more accurately representation of runoff generation in forested catchments.

From both the Maritsane and Witklip simulations it was evident that even during the dry season, observed streamflow responded rapidly and significantly to rainfall inputs. The Macaque model, however, could not reflect this catchment response. This type of rapid response indicates the likelihood that other runoff processes, other than saturation excess overland flow and subsurface flow occur in the catchment, which enables the streamflow to rise significantly even though the water table is deep and the saturated portion of the catchment is small. Therefore, the assumption in Macaque that runoff mechanisms are limited to saturation excess flow and sub-surface flow in forested catchments could be an incorrect hypothesis, especially in applications to typical South African situations.

The advantage of the runoff representation in *ACRU* is its simplicity and its universal application to different catchments where different process mechanisms may dominate. However, it is also this simplicity that prevaricates against the accurate simulation of flows during the dry season. The simulation results of the Witklip and Maritsane catchments using the Macaque model have shown that there is potential for incorporating a topography-based approach to model sub-surface flows and baseflows.

Both the *ACRU* model and Macaque model have their strengths and shortcomings and in summary the main findings of this research outlined below:

- The *ACRU* model structure is more suited for predictive modelling on operational catchments, whilst the more complex Macaque model's greatest limitation for application in South African is its high input demands that cannot be supported by the available data.
- Despite the data limitations and data uncertainty the Macaque model's topography-based representation of runoff processes did result in improved low flow simulation compared to the results from the *ACRU* simulations, indicating that lessons can be learned from this modelling approach.
- The Macaque model's link to the Geographical Information System, Tarsier, provided an efficient means to configure the model, input spatial data and view output data. However, it was found that the *ACRU* model is more flexible in terms of being able to accurately represent the spatial and temporal variations in input parameters.

In the following Chapter the recommendations for future research are provided. These suggestions reflect some of the problems experienced during the application and verification of the *ACRU* and Macaque models.

8. RECOMMENDATIONS FOR FUTURE RESEARCH

A number of research needs were identified during the course of this research. The results from the Macaque application described in Chapter 6 indicate that improved low flow simulations were achieved using a topography-based approach. However, the lack of internal process observations and limited input data related to sub-surface processes makes it difficult to conclusively deduce which runoff or sub-surface processes exactly are responsible for the improved low flow simulations compared to that simulated by the *ACRU* model. Therefore, to comprehensively evaluate whether topographically-based models are a more realistic approach for modeling forestry, more emphasis will need to be placed on process measurement in the field to determine the stormflow and baseflow contributions to observed streamflow during the dry and wet months of the year. Furthermore, the inadequate representation of spatial and temporal variation in LAI and other vegetation parameters that influence the transpiration and interception processes would directly impact on sub-surface soil water storage and fluxes. Therefore, the incorporation of LAI curves in Macaque, such as those presented in the *ACRU* model, would improve the accuracy of processes affected directly or indirectly by the representation of LAI.

Although the topographically-based modeling approach could result in improved simulations, it needs to be weighted against the additional model input data required by the models. DEMs are becoming more readily available and the possibility exists to use DEMs as a means of deriving response coefficients to control the subsurface flow and baseflow releases.

The limited range of measurement techniques and the limited range of measurements in time and space indicate a need for the development of new measurement techniques. Furthermore, it is necessary to integrate the field measurement scale with the scale of input required by the model. Paired with the need for new measurement techniques and data collection is the need for process studies and measurements for the internal verification of models. The verification of simulated streamflow against observed streamflow records are widely reported for both the *ACRU* and Macaque models, however few studies have focused on the verification of the models' internal processes. A number of studies have been undertaken in South Africa to measure evapotranspiration from forest stands and from other vegetation types and it would be of benefit if such evapotranspiration measurements could be used in the internal

verification of catchment-scale hydrological models. It is only through the mutual verification of simulated streamflow and a model's internal processes that greater confidence can be placed in the predictions of models used in water resources planning. Both the Macaque and *ACRU* models have undergone a number of changes and developments, yet few of these changes have been extensively verified due to data limitations. Hence, greater emphasis should be placed on the testing of new model components to prevent "our modelling capabilities from surpassing our ability to gather meaningful information for model parameterization and validation" (Vertessy *et al.*, 1993).

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10. APPENDICES

APPENDIX A

Table A1: Macaque model parameters and values used for the Maritsane simulation

Model parameter name	Unit	Fixed value
p_elevation	m	Spatial
p_sin_aspect		Spatial
p_cos_aspect		Spatial
p_slope		Spatial
p_mean_monthly_precipitation_index		Spatial
p_precipitation_scalar		Spatial
p_max_temperature_elevation_lapse_rate	°C/m	0.007
p_min_temperature_elevation_lapse_rate	°C/m	0.004
p_latitude	Degrees	-24.867
p_bristow_and_campbells_a		0.766
p_bristow_and_campbells_b		0.0327
p_bristow_and_campbells_c		1.46
p_total_projected_lai		Spatial
p_canopy_projected_lai		Spatial
p_canopy_height	m	6
p_vegetation_scalar		1
p_leaf_off_day	Julian Day	0
p_leaf_on_day	Julian Day	0
p_rain_interception_coeff(m/LAI)		0.0008
p_slope_leaf_water_potential_vs_rel_water_availability		-80 000
p_minimum_leaf_conductance	m/day	0.0002
p_slope_leaf_cond_vs_cold_temp		0.0002
p_slope_rel_leaf_cond_vs_warm_temp		0.0003
p_canopy_species		Spatial
p_canopy_radiation_extinction_coeff		0.5
p_canopy_reflection_coefficient		0.10
p_canopy_root_depth	m	2
p_canopy_leaf_water_potential_at_stomatal_closure	Pa	-1800000
p_canopy_leaf_water_potential_maximum	Pa	-300000
p_maximum_canopy_leaf_conductance	m/day	0.02
p_slope_canopy_relative_leaf_cond_vs_vpd		-0.0003
p_canopy_leaf_conductance_radiation_threshold		0
p_canopy_reference_aerodynamic_resistance	s/m	5
p_sat_canopy_transpiration_proportion		0.1
p_canopy_leaf_conductance_radiation_parameter		100
p_canopy_leaf_conductance_max_temperature	°C	40
p_canopy_leaf_conductance_optimal_temperature	°C	20
p_canopy_leaf_conductance_min_temperature	°C	0
p_min_temperature_causing_stomatal_closure	°C	-8
p_intercept_canopy_leaf_conductance_vs_lwp	Pa	-2000000
p_shape_canopy_leaf_conductance_vs_lwp	Pa	400000
p_canopy_leaf_conductance_vpd_factor		0.0007
p_canopy_co2_for_max_leaf_conductance		350
p_canopy_co2_causing_stomatal_closure		1050
p_slope_soil_resistance_vs_vwc		-15000
p_soil_resistance_vwc_threshold		0.2
p_atmospheric_co2_concentration		350
p_soil_reflection_coefficient		0.1
p_soil_understorey_aerodynamic_resistance		15
p_evaporation_depth	m	0.003
p_soil_evaporation_tortuosity_factor		2
p_water_reference_aerodynamic_resistance	s/m	75

p_water_reflection_coefficient		0.05
p_surface_saturated_hydraulic_conductivity	m/day	0.336
p_minimum_saturated_hydraulic_conductivity	m/day	0.168
p_saturated_hydraulic_conductivity_shape		1.0
p_saturated_hydraulic_conductivity_depth	m	5
p_ratio_hydraulic_to_surface_gradient		0.98
p_saturated_volumetric_water_content	m(water)/m(soil)	0.55
p_residual_volumetric_water_content	m(water)/m(soil)	0.24
p_clay_fraction		0.15
p_sand_fraction		0.35

APPENDIX B

Table B1: Macaque model parameters and values used for the Witklip simulation

Model parameter name	Unit	Fixed value
p_elevation	m	Spatial
p_sin_aspect		Spatial
p_cos_aspect		Spatial
p_slope		Spatial
p_mean_monthly_precipitation_index		Spatial
p_precipitation_scalar		Spatial
p_max_temperature_elevation_lapse_rate	°C/m	0
p_min_temperature_elevation_lapse_rate	°C/m	0
p_latitude	Degrees	-25.23
p_bristow_and_campbells_a		0.766
p_bristow_and_campbells_b		0.0327
p_bristow_and_campbells_c		1.46
p_total_projected_lai		Spatial
p_canopy_projected_lai		Spatial
p_canopy_height	m	Spatial
p_vegetation_scalar		1
p_leaf_off_day	Julian Day	151
p_leaf_on_day	Julian Day	242
p_rain_interception_coeff		Spatial
p_slope_leaf_water_potential_vs_rel_water_availability		-100 000
p_minimum_leaf_conductance	m/day	0.00625
p_slope_leaf_cond_vs_cold_temp		0.0002
p_slope_rel_leaf_cond_vs_warm_temp		-0.0003
p_canopy_species		Spatial
p_canopy_radiation_extinction_coeff		Spatial
p_canopy_reflection_coefficient		Spatial
p_canopy_root_depth		Spatial
p_canopy_leaf_water_potential_at_stomatal_closure	Pa	-1800000
p_canopy_leaf_water_potential_maximum	Pa	-300000
p_maximum_canopy_leaf_conductance	m/day	0.0002
p_slope_canopy_relative_leaf_cond_vs_vpd		-0.0003
p_canopy_leaf_conductance_radiation_threshold		0
p_canopy_reference_aerodynamic_resistance	s/m	Spatial
p_sat_canopy_transpiration_proportion		0.1
p_canopy_leaf_conductance_radiation_parameter		100
p_canopy_leaf_conductance_max_temperature	°C	40
p_canopy_leaf_conductance_optimal_temperature	°C	20
p_canopy_leaf_conductance_min_temperature	°C	0
p_min_temperature_causing_stomatal_closure	°C	-8
p_intercept_canopy_leaf_conductance_vs_lwp	Pa	-2000000
p_shape_canopy_leaf_conductance_vs_lwp	Pa	400000
p_canopy_leaf_conductance_vpd_factor		0.0007
p_canopy_co2_for_max_leaf_conductance		350
p_canopy_co2_causing_stomatal_closure		1050
p_slope_soil_resistance_vs_vwc		-15000
p_soil_resistance_vwc_threshold		0.2
p_atmospheric_co2_concentration		350
p_soil_reflection_coefficient		0.1
p_evaporation_depth	m	0.003
p_soil_evaporation_tortuosity_factor		2
p_water_reference_aerodynamic_resistance	s/m	75
p_water_reflection_coefficient		0.05
p_surface_saturated_hydraulic_conductivity	m/day	0.336
p_minimum_saturated_hydraulic_conductivity	m/day	0.168

p_saturated_hydraulic_conductivity_shape		1.0
p_saturated_hydraulic_conductivity_depth	m	4
p_ratio_hydraulic_to_surface_gradient		0.90
p_saturated_volumetric_water_content	m(water)/m(soil)	0.60
p_residual_volumetric_water_content	m(water)/m(soil)	0.24
p_clay_fraction		0.2
p_sand_fraction		0.6

Table B2. Witklip, Catchment V soils data.

Witklip	Depth (m)	%Sand	%Silt	%Clay	Porosity (m/m)	Ksat (m/day)
Crest: Site A	0	50	35	15	0.519	0.19
Crest: Site A	0.25	53	20	27	-	-
Crest: Site A	0.5	66	12	22	-	-
Crest: Site A	0.75	55	27	18	0.421	0.08
Crest: Site A	1.0	60	21	19	-	-
Crest: Site A	1.5	70	30	2	0.499	0.08
Crest: Site A	2.0	70	26	4	-	-
Crest: Site A	2.5	75	25	6	0.489	0.09
Midslope: Site C2	0	58	17	25	-	-
Midslope: Site C2	0.25	49	17	34	-	-
Midslope: Site C2	0.5	50	17	33	-	-
Midslope: Site C2	0.75	45	21	34	-	-
Midslope: Site C2	1	45	20	35	0.481	0.14
Midslope: Site C2	1.5	51	18	31	-	-
Midslope: Site C2	2.0	60	19	21	-	-
Midslope: Site C2	2.5	73	27	15	0.433	0.14
Forest site F1	0	62	19	19	0.636	0.39
Forest site F1	0.25	62	12	20	-	-
Forest site F1	0.5	68	9	18	-	-
Forest site F1	0.75	73	22	15	-	-
Forest site F1	1.0	63	25	18	-	-
Forest site F1	1.5	60	23	17	-	-
Riparian site R1	0	70	15	15	0.505	0.19
Riparian site R1	0.25	70	16	14	-	-
Riparian site R1	0.5	65	22	13	-	-
Riparian site R1	0.75	72	14	14	-	-
Riparian site R1	1	70	16	14	0.257	0.02
Riparian site R1	1.25	71	14	15	-	-