Otway Forest Hydrology Project

Impact of Logging Practices on Water Yield and Quality in the Otway Forests

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A report for the Department of Natural Resources & Environment

SINCLAIR KNIGHT MERZ

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Sinclair Knight Merz Pty Limited
ACN 001 024 095
ABN 37 001 024 095
590 Orrong Road
Armadale VIC 3143
PO Box 2500
Malvern VIC 3144
Australia
Telephone: +61 3 9248 3100
Facsimile: +61 3 9248 3364

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Executive Summary

The West Victorian Regional Forest Agreement, signed in March 2000, establishes a framework for best practice management of the Otway forests. In recognition of the importance these forests have as a source of water for urban and commercial consumption, Sinclair Knight Merz was commissioned to investigate the potential impacts timber harvesting may have on water yield and quality.

Water Resource Evaluation

The Otways are situated in one of the highest rainfall zones of Victoria. Streamflow is harvested for supply to a number of regional municipalities, the largest of which is Geelong. The water resource is currently managed by two water authorities, Barwon Water and South West Water. It is estimated that approximately 25% of the 87,600 ML/yr Upper Barwon yield is currently harvested. In the Gellibrand catchment almost 7% of the yield (of 338,000 ML/yr) is currently diverted by South West Water. This particular system is effectively ‘run of river’ because of the small amount of storage available and hence although the current demand is a small proportion of the annual demand the summer demand is a much higher proportion of the yield.

Water Yield Impacts

A modelling framework was developed based upon the observed and modelled response in the Melbourne Water Supply catchments, though it was modified to incorporate an initial increase in yield for a short period immediately following forest disturbance, and to be consistent with the limited information available from the Otway region.

Preliminary modelling was undertaken to estimate the likely average annual yield impacts over the next 100 years. Three water supply catchments were selected – West Barwon (Geelong supply), St George (Lorne supply), and the Arkins catchment (a tributary of the Gellibrand River). Functional yield response curves were developed separately for eucalypt mixed species and Mountain Ash. The response for the Mountain Ash was based upon that from the Melbourne Water supply catchments with consideration of differences in rainfall, forest type and soil depth. The adopted response for the eucalypt mixed species was checked by applying it to the East Barwon catchment

The following three scenarios were simulated for each of the study catchments:

No disturbance. In this scenario no logging, fire or other disturbance occurs and the forest is allowed to age from the age distribution in the year 2000. The yield increase associated with the ageing forest was estimated to be between 7% for the St George catchment up to 28% for the Greater Arkins catchment by 2100.

Logging only. The forest area logged in a particular forest type and age class was provided by the Department of Natural Resources and Environment (DNRE). These areas were calculated using the constraints of: (a) consistent with Regional Forest Agreement (RFA) outcomes; (b) the logging rotations specified in the Otway Forest Management Plan; and (c) the need to maintain a timber yield that varies as little as possible over the period for both Ash and Mixed Species. For the study catchments, the estimated changes (positive and negative) in yield associated with these logging scenarios are less than 5% of the (current) mean annual flow over the next 100 years.

Wildfire only. The third scenario was based upon complete mortality of 50% of the catchment by fire at the beginning of the 100 years, with no logging or other disturbance. The estimated effects of such a wildfire on yield were much greater than that resulting from the previous two scenarios. In each case, yield initially increased significantly followed by a prolonged period of yield reduction, before gradually returning to mature forest conditions by 2100.

Water Quality Impacts

Water quality in the Otway region shows significant variation, related primarily to dominant catchment land use. Given the demand for high quality water from domestic water supply catchments, protection of this water quality is of very high priority. Whilst water quality data for the Otways region are substantial, there is a lack of event-based data, and monitoring dedicated to measuring the impacts of particular land use practices on water quality.

The greatest water quality impacts in the Otways have been related to agricultural land use, particularly those associated with poor management of riparian zones. Nonetheless, a number of studies undertaken throughout Australia and in the Otways have demonstrated that logging activities can have a detrimental impact on water quality. The most substantial impact has been attributed to unsealed and poorly drained stream crossings associated with roads in the Otway Ranges. In recognition of this fact the DNRE and the Colac-Otway Shire have initiated a program to identify, prioritise and upgrade such sites.

The Otway region is susceptible to landslides and depending on the location of the landslides, they can potentially contribute a large sediment load to a stream. The Colac-Otway Shire is currently undertaking a study to review the occurrence of landslides in the Otway Region. There is no conclusive evidence to link the incidence of landslides with timber harvesting activities.

Wildfire

Infrequent severe wildfires have been a feature of the Otway forests and the present fire management plans and strategies are professional, thorough and appropriate. However the probability of an extensive wildfire, although low, will always be present, as will the probability of widespread impacts. Very large areas of future fire-induced forest regeneration cannot be ruled out absolutely. Such an eventuality would have significant short-term impacts on water quality and long-term impacts on water yield.

Best Management Practices (BMPs)

Best Management Practices for the Otway forests are prescribed under the Code of Forest Practices for Timber Production (DNRE, 1996) and regional prescriptions. The measures implemented are generally in line with equivalent soil/water BMPs implemented elsewhere in Australian forests. BMP studies are currently being undertaken by DNRE in the Otways to evaluate their effectiveness.

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1. Introduction

The Otway Ranges are located in south-west Victoria. These forests consist of Mountain ash (*Eucalyptus regnans*) and mixed species, primarily messmate and Otway messmate. The forests are situated in a high rainfall zone and serve as an important source of high quality water for urban and commercial consumption. Water is jointly harvested from the forests by Barwon Water and South West Water and is used to supply over 250, 000 people in the region. At the same time the forest constitutes a commercially important source of timber, is a popular tourist destination and has been recognised for its bio-diversity value.

In March 2000 the State Government signed the *West Victoria Regional Forest Agreement* (RFA), one of five RFAs negotiated for Victoria. This document represents a long term agreement (up to 20 years) between the Victorian State Government and the Commonwealth Government aimed at establishing a framework for best practice management of the forests.

A key component in the process of developing the West Victoria RFA was extensive community consultation. Over 1400 public submissions were received during the RFA consultation for the West and Gippsland regions, demonstrating the breadth and depth of interest the community and other stakeholders have in the continuing management of the State's forests. One of the key concerns raised is the impact timber harvesting will have on the water values of the forested Otway Catchments.

At the signing of West Victoria RFA on 31 March 2000, the government announced the need to conduct further research in the Otways. Sinclair Knight Merz was subsequently commissioned to undertake an investigation of the potential impacts timber harvesting may have on water yield and quality in the Otways. The study involved:

- review of available research and information;
- evaluation of catchment yields and demands;
- modelling water yield impacts;
- qualitative assessment of the water quality impacts;
- review of best management practices; and
- identification of the existing knowledge gaps.

2. Water Yield: Literature Review

2.1 Preface

The following literature review is developed from the review completed by Dr Rob Vertessy as part of the ESFM Project: Water Quality and Quantity for the Upper and Lower North East, Southern RFA Regions (SKM, 1998). Much of this information was subsequently presented in a CRCCH workshop (Forest Management and Water Quality and Quantity, Warburton 1999) in Dr Rob Vertessy's paper; "*The Impacts of Forestry on Streamflows: A Review*". Further information, in particular that relating to the Otway region, was sourced from the Department of Natural Resources and Environment and members of the Otway Forest Hydrology Reference Group.

2.2 Introduction

Water is a limiting and finite resource in the Australian environment, and the requirements of an increasing population and an important agricultural sector ensure that conflicts over water use and water allocation can only increase. Those catchments yielding the most water are not surprisingly located in areas of higher rainfall. These principally occur in elevated areas in relatively close proximity to the sea, the very environment in which we find the most productive native forests. The management of these native forests for timber production can change important characteristics of the forest stand, which in turn may affect critical hydrologic processes within the catchment. While forest clearing for another land use such as agriculture generally produces permanent hydrologic changes, forest management for timber production has dynamic effects on the forest stand, resulting in ever-changing hydrologic outcomes over the entire forest rotation. An understanding of these dynamic processes, and their relative importance, is essential to the successful modelling of water yield in managed forests.

There is now broad scientific agreement that activities associated with forest management have the potential to alter catchment water balances and thus change the amount and timing of catchment streamflows. A large number of studies have been conducted world-wide to ascertain the nature and extent of streamflow changes resulting from forestry activities of different kinds. This review examines the findings of those studies and seeks to develop generalisations about the effects of forest disturbance and management on streamflows. The focus in this review is mainly on water yield or the quantity of streamflow, although attention is also given to streamflow seasonality and flow frequency, and to the magnitude of peak flows.

Our knowledge of the effects of forest management on streamflows in Australia stems mainly from ten major sets of catchment treatment experiments, located in forests in diverse geographic settings (Table 2-1). The earliest studies commenced in the late 1950's, though most were only established in the 1970's. Since the mid 1980's, many of the catchment experiments have been abandoned, primarily due to cost cutting in the State resource agencies running them. Hence Australia is not well endowed with catchment treatment experiments which have been monitored over long time periods. The lengthiest, most complete and most reliable data on the impacts of forestry on streamflows come from the Maroondah, Karuah and Darling Ranges experiments. The brevity of Australian hydrologic records is a general weakness that precludes us from making definitive statements about the long term impacts of forestry on streamflows.

This review is framed around a series of questions pertinent to the impacts of forestry on streamflows, namely:

- What are the principal hydrologic processes which govern streamflow generation?
- What are the fundamental hydrologic differences between forested areas, and areas covered in other forms of vegetation?
- What effect does forest harvesting have on streamflow?
- How do forest regeneration and forest age affect streamflows?
- What affect does forest management have on the streamflow regime?
- Are there other factors which could alter the impacts of forest disturbance and management on streamflows?

A broad geographic focus is adopted in this review, although emphasis is placed on native forest types and climatic conditions prevailing in south-eastern Australia. Particular attention is given to forest areas in southern Victoria and to the streamflow changes likely to arise from harvesting, regeneration, forest ageing, thinning and fire.

2.3 Related Reviews

There are several published reviews which summarise the impacts of forestry on streamflows. These vary greatly in emphasis, with some focussed on water yield, and others adopting a broader approach water values. They cover a wide range of geographic areas, and whilst several fail to consider Australian conditions, they still provide useful insights into the effects of forestry activities on basic hydrological processes. Key reviews are listed in Table 2-1, including a number based on Australian data.

2.4 The Hydrologic Cycle

We begin this review with a brief description of the hydrologic cycle so that the studies we examine later can be properly interpreted. The key hydrologic parameters and processes referred to in this review are defined in Table 2-3.

2.4.1 The Hydrologic Cycle

Streamflow from catchments is generated from precipitation (generally rainfall in Australia) and from discharge of groundwater, although the latter component is usually small. However, not all rainfall is converted into streamflow. Some rainfall is intercepted by vegetation or detained in ground depressions, and evaporates back into the atmosphere. Rain which infiltrates into the soil increases the soil water storage, though this is continually depleted by soil evaporation and the uptake of water by plants through the process of transpiration. In some cases, soil water percolates to deep groundwater systems and may not re-emerge as groundwater discharge within the confines of the particular catchment.

■ **Table 2-1 Listing Of The Main Australian Catchment Treatment Experiments Designed To Ascertain Relationships Between Forestry Activities And Streamflows**

Study	Location	Study managers	Focus	Key references
Maroondah	Near Healesville, VIC Central Highlands	Melbourne Water	Effect of mountain ash forest age and harvesting methods on water yield and quality	Langford 1976; Langford and O'Shaughnessy 1977; Langford and O'Shaughnessy 1980; Moran and O'Shaughnessy 1984; Kuczera 1987; Haydon 1993; Jayasuriya *et al*. 1993; Haydon *et al*. 1996; Watson *et al*. 1998.
Karuah	Near Dungog, NSW lower north east region	NSW State Forests	Effect of forest harvesting methods on water yield and quality	Cornish 1993; Cornish and Vertessy (in press); Cornish (in press).
Yambulla	Near Eden, NSW south east region	NSW State Forests	Effect of forest harvesting methods and fire on water yield and quality	Mackay and Cornish 1982; Moore et al 1986; Mackay and Robinson 1987: Roberts et al 2000.
Tantawangalo	Near Eden, NSW south east region	NSW State Forests	Effect of forest harvesting methods on water yield and quality	Lane and Mackay 2000.
Red Hill	Near Tumut, NSW southern uplands	NSW State Forests	Water yield and quality impact of pasture conversion to pine plantation	Major et al 1998.
Lidsdale	North-west of Sydney, NSW	University of New South Wales and NSW State Forests	Water yield impact of eucalypt forest conversion to pine plantation	Smith *et al*. 1974; Pilgrim *et al*. 1982.
Parwan	North-west of Melbourne, VIC	NRE, Victoria	Water yield impact of eucalypt forest conversion to pasture	Nandakumar and Mein 1993
Stewarts Creek	Near Ballarat, central VIC	NRE, Victoria	Water yield impact of eucalypt forest conversion to pine plantation and pasture	Nandakumar and Mein 1993
Croppers Creek	Near Myrtleford, VIC north-eastern highlands	NRE, Victoria	Water yield and quality impact of eucalypt forest conversion to pine plantation	Bren, *et al*. 1979 Bren and Papworth 1991
Babinda	South of Cairns, QLD	QDPI and James Cook University	Impacts of rainforest clearance on runoff generation	Bonell and Gilmour 1978; Cassells *et al*. 1985
Darling Ranges	South of Perth, south-western WA	CALM, WA, Water Authority, WA and Alcoa	Impacts of clearing jarrah forest on water yields and salinity	Ruprecht and Schofield 1989; Ruprecht and Schofield 1991; Ruprecht and Stoneman 1993; Stoneman 1993

■ Table 2-2 List Of Reviews Focussing On The Hydrologic Impacts Of Forest Disturbance

Published Review	**Details**
Overseas studies	
Bosch and Hewlett 1982	One of the most quoted review papers on water yield, focussing on 94 worldwide studies
Bruijnzeel 1990	The most comprehensive account of water balance and water yield in moist tropical forests
Schofield 1996	A general review of the impacts of forestry on water values, including reference to Australian studies; largely based on other reviews
Stednick 1996	Reviews 95 studies from the United States in a similar manner to Bosch and Hewlett (1982)
Dye 1996	Focuses on water yield changes arising from afforestation of grasslands in South Africa with eucalypts and pines
Australian studies	
Bren 1987	Summarises: Australian and overseas research into the effects of logging with special reference to hydrologic factors. Examines, and criticizes, some literature dealing with the hydrological impacts of forestry in the Otways.
Cornish 1989	Summarises Australian, New Zealand and South African research on water yield from Pine plantations, making reference to grassland and eucalypt forests also
Doeg and Kohen 1990	Examines 36 Australian studies in a case study format, considering a range of water values
Dargavel *et al*. 1995	Summarises the hydrologic impacts of logging, focussing on south-eastern Australia; also addresses related economic and policy issues
Vertessy 1999	Summarises Australian and overseas research on forestry impacts on water yield, and includes reference to recent process and catchment studies

■ Table 2-3 Definition Of Key Hydrologic Terms Referred To In This Review

Process	**Definition**
Rainfall	Precipitated water arriving at the plant canopy
Interception	Detention (and subsequent evaporation) of water stored on plant surfaces
Surface detention	Detention (and subsequent evaporation) of water stored in surface depressions
Stemflow	Movement of water down the stems of plants, to the soil surface
Transpiration	Evaporation of water via the stomata in leaves of plants
Soil evaporation	Evaporation of water from the soil
Evapotranspiration	The sum of interception, surface detention, transpiration and soil evaporation
Infiltration	Movement of water into the soil profile
Percolation	Movement of water through the soil profile
Soil moisture storage	Storage of water in the soil profile
Subsurface flow	Movement of water through the soil
Overland flow	Movement of water over the soil surface
Groundwater recharge	Drainage from the soil profile to a groundwater system
Groundwater discharge	Flux of water from a groundwater system into the soil profile
Streamflow	Sum of overland and subsurface flow, and groundwater discharge, measured in a stream channel

Summarising, the catchment water balance is described by the following equation:

$$Q = P - ET - R + D \pm \Delta S \quad (2.1)$$

where:

- Q = streamflow
- P = precipitation (rainfall or snowfall)
- ET = evapotranspiration
- R = groundwater recharge
- D = groundwater discharge
- ΔS = change in soil moisture storage

2.4.2 The relative importance of water balance parameters

Precipitation is by far the most important factor in streamflow generation, and its great variability in Australia accounts the extremely variable streamflows recorded here. Evapotranspiration is the other major component of the water balance, and can be closely equated with water use by vegetation. Groundwater recharge and discharge are generally quite small in forested environments, but can be relatively larger under other vegetation types. While soil moisture storage can vary markedly from one year to the next, cumulative changes become negligible over a number of years.

Water use by catchment vegetation is therefore the most important water balance term that can be altered by human intervention, and this review concentrates in some detail on this factor.

2.5 Factors Affecting Water Use by Different Types of Vegetation

Forests have higher evapotranspiration (ET) rates (rates of water use) than grasslands or short crops, resulting in generally lower streamflows. This is so because forests have:

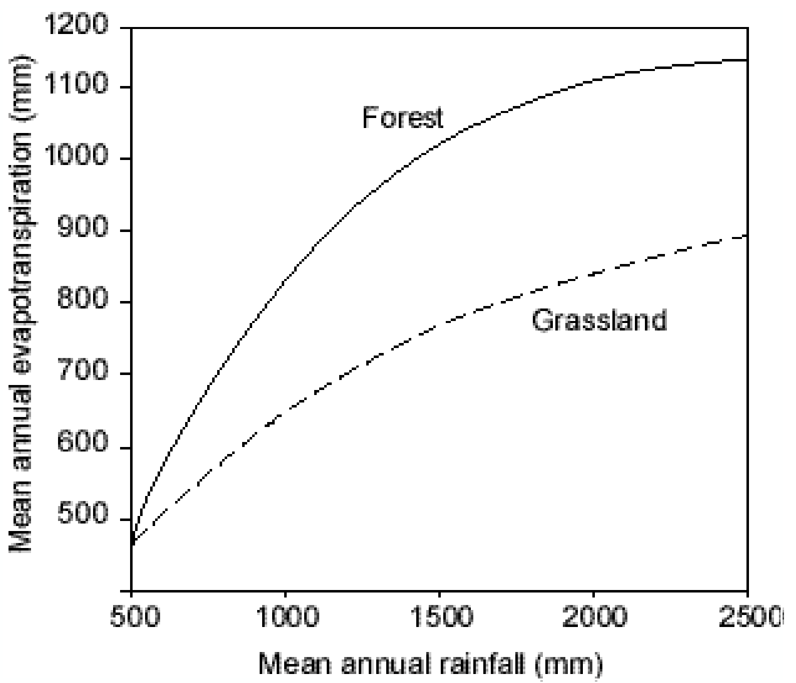
- higher and more persistent leaf area than grasses, leading to greater rainfall interception and greater transpiration;
- greater aerodynamic roughness than grasses, leading to greater rainfall interception and greater transpiration; higher roughness also enhances the likelihood that advected warm, dry air can supplement the radiant energy input (see Calder, 1997);
- deeper rooting than grasses or short crops, enabling better access to soil water, leading to greater transpiration;
- a greater capacity to extract moisture from the soil under dry conditions, leading to greater transpiration;
- lower albedo than grasses, allowing more absorption of radiant energy, in turn leading to greater transpiration.

■ **Table 2-4 Mean Annual Rainfall And Evapotranspiration (Et) For Catchments With Various Vegetation Covers In South-Eastern Australia**

Location	Details	Vegetation	Annual rainfall (mm)	Annual ET (mm)	Reference
Parwan (1-6), VIC	over 6 years of data from each of six catchments - NRE, VIC	native and improved pastures, some grazed	538	491	Nandakumar and Mein 1993
Lidsdale 1, NSW	3 years of data from one catchment - UNSW	grass, weeds, pines seedlings with little LAI	688	567	Pilgrim *et al.* 1982
Mt. Gambier, SA	6 years of data from one catchment - CSIRO Soils	grass	~670	580	Holmes and Sinclair 1986
Kylies Run, Tumut, NSW	7 years of data from one catchment - NSW State Forests	improved pasture, grazed	944	691	NSW State Forests (unpublished data)
Pomaderris, Eden, NSW	7 years of data from one catchment - NSW State Forests	mature, dry eucalypt forest	973	775	NSW State Forests (unpublished data)
Peppermint, Eden, NSW	3 years of data from one catchment - NSW State Forests	mature, dry eucalypt forest	1103	826	NSW State Forests (unpublished data)
Sassafras, Karuah , NSW	7 years of data from one catchment - NSW State Forests	mature, moist eucalypt forest, rainforest	1434	1128	Cornish 1993
Crabapple, Karuah , NSW	7 years of data from one catchment - NSW State Forests	mature, moist eucalypt forest, rainforest	1639	1190	Cornish 1993
Picaninny, Maroondah, VIC	3 years of data from one catchment - Melbourne Water	old growth mountain ash forest, rainforest	1180	848	Vertessy *et al.* 1996
Myrtle 1, Maroondah, VIC	11 years of data from one catchment - Melbourne Water	old growth mountain ash forest, rainforest	1598	882	Vertessy *et al.* 1993
Picaninny, Maroondah, VIC	10 years of data from one catchment - Melbourne Water	regrowth mountain ash forest (age 15-25), rainforest	1245	1061	Melbourne Water (Unpublished data)
Ettercon 3, Maroondah, VIC	11 years of data from one catchment - Melbourne Water	regrowth mountain ash forest (age 33-44), rainforest	1631	1250	Melbourne Water (Unpublished data)
Reefton (1-6), Vic	10 years of data from six catchments - Melbourne Water	mature mountain ash and mixed eucalypt forest	1298	1022	Nandakumar and Mein, 1993

Table 2-4 lists a variety of ET estimates for grassland and forests in south-eastern Australia, determined from catchment treatment experiments. Annual ET from grassland is usually less than 700 mm, whereas annual ET from forests can approach 1300 mm. Streamflow from forests is uncommon in areas with annual rainfall less than 800 mm, unless rainfall is highly concentrated in a particular season. Figure 2-1 shows that there is considerable variation in mean annual ET in forests. This is largely a consequence of varying rainfall amounts, though there are species, productivity and age effects which are also quite important. Such effects are examined in later sections of this review.

Holmes and Sinclair (1986) examined rainfall/runoff relationships for 19 large catchments across Victoria, with mean annual rainfalls ranging between 500 and 2500 mm, and varying mixtures of grass and eucalypt forest cover. They demonstrated that there were clear differences between ET rates for grassland and eucalypt forest catchments and conceptualised this with a pair of curves that emphasised the differences along a rainfall gradient (Figure 2-1). According to these curves, a fully forested eucalypt catchment would evapotranspire 40, 90, 215, 240 and 250 mm more per year than a fully grassed catchment with mean annual rainfalls of 600, 800, 1300, 1500 and 1800 mm, respectively. An analysis by Cornish (1989) on different hydrologic data yielded very similar figures to these [see his Table 4, p.17].



■ **Figure 2-1 Mean Annual Evapotranspiration From Pastures And Forest As A Function Of Mean Annual Rainfall (Adapted From Holmes And Sinclair, 1986).**

The Holmes and Sinclair (1986) relationship (HSR) is a useful generalisation to obtain a quick estimate of ET (and hence streamflow) for large catchments at different isohyets, and has been used by Vertessy and Bessard (1999) to estimate possible water yield changes in the lower Murrumbidgee catchment which could result from extensive plantation establishment on grasslands.

Zhang et al. (1999), using data from over 250 catchments from around the world, developed a model which predicted mean annual ET as a function of vegetation type and cover, mean annual rainfall and mean annual potential evapotranspiration. Their model yielded very similar ET curves to those proposed by Holmes and Sinclair (1986), although they predicted higher ET values where mean annual rainfall exceeded 1200 mm. This study, and the data published by Turner (1991) for 68 Californian catchments, demonstrate that vegetative cover and mean annual rainfall are the critical factors controlling annual water yield.

Additional factors may render the Holmes and Sinclair (1986) relationship inapplicable in some circumstances. For instance, Moran and O'Shaughnessy (1984) reported mean annual ET estimates for 17 catchments in the Maroondah basin, all of which are fully forested with the same eucalypt species (mountain ash). Their ET estimates vary between 740 and 1330 mm per year for catchments with mean annual rainfalls of between 1160 and 1880 mm, and indicate that mean annual ET rates for catchments with similar rainfall vary by as much as 300 mm. Therefore the HSR is only a rough guide to likely ET rates in catchments. Moran and O'Shaughnessy (1984) attribute these ET differences to variations in forest age and stocking rate. These effects will be discussed later in this review.

2.6 Effects of Forest Removal or Forest Establishment on Streamflows

Most catchment water balance studies conducted to date have been designed to determine the immediate effects of forest removal (harvesting) on streamflows. Although the forest may have regenerated after harvesting these studies have not examined the effects of this regeneration, or the effects of a subsequently ageing forest, on streamflows. This subject will be examined in the next section.

Almost all of the studies have shown that streamflow increases as forest cover decreases, and vice versa. They also show that the magnitude of this increase varies as a function of the type of forest treated and the mean annual rainfall of the site. Most of the data reported below is based on catchment treatment experiments where the forest cover is cleared or partially cleared. In some cases, the forest is permitted to regenerate, and only the first few years of data following treatment are used in building relationships. This is problematic for three reasons. Firstly, it takes time for a catchment to adjust it's runoff behaviour. Immediately after clearance, some rainfall is used to replenish the soil water deficit left by the higher ET forest that has been removed. This has the effect of tending to underestimate the effect of forest clearance. Secondly, it is possible that soil disturbance from logging activities can temporarily increase overland flow and change the pattern of streamflow. This has the effect of tending to overestimate the effect of forest clearance, thus off-setting the first effect. Thirdly, because of the short time span used to build the relationship, it is possible that rainfall variability will complicate the catchment response. Depending on the rainfall pattern, this could result in either an underestimate or overestimate of the effects of forest clearance.

On the other hand, much of the data from South Africa and New Zealand is based on afforestation experiments. Results from South African studies tend to be based on long data records, often exceeding 30 years in length.

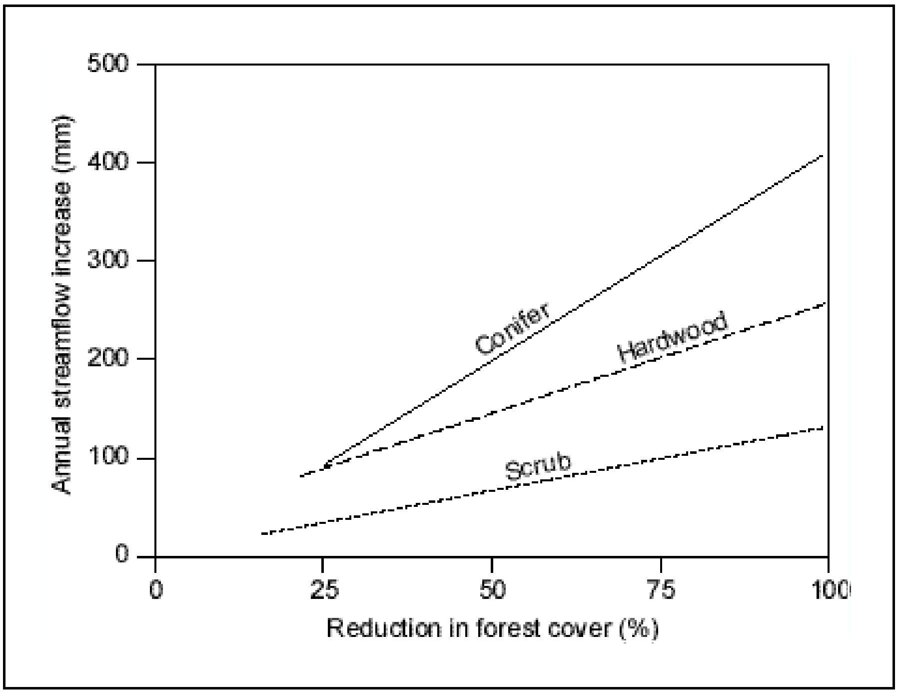
2.6.1 Northern hemisphere studies

Hibbert (1967) summarised the results of 39 catchment experiments carried out in the USA, mainly on deciduous hardwood and conifer forests. He concluded that:

- reduction of forest cover increased water yield;
- afforestation decreased water yield; and,
- the response to treatment was highly variable and unpredictable.

Bosch and Hewlett (1982) added another 55 catchment studies to the Hibbert (1967) data set, including results from Japan, Australia, New Zealand and South Africa. They concluded that water yield increases proportionately with the percent area of forest cleared, but there was a fair degree of scatter in the relationship. Bosch and Hewlett (1982) related much of this scatter to species differences, and Figure 2-2 shows the trend lines they fitted to their data for three different woody species (conifers, deciduous forest and scrub). This indicated that the rate of yield increase is greatest for conifers, and least for scrub. They hypothesised that vegetation growth rate was thus a key control on the likely response of a catchment to disturbance. Figure 2-2 shows that each 10% reduction in forest cover results in an annual yield increase of 40 mm for conifers, 25 mm for deciduous hardwoods, and 10 mm for scrub. Bosch and Hewlett (1982) also concluded that water yield changes could not be detected unless more than 20% of the catchment area was cleared.

More recently, Stednick (1996) reviewed 95 paired catchment studies from various geographic regions in the USA. His estimated yield increases accompanying forest clearance (shown in Table 2-5) were similar to those obtained by Bosch and Hewlett (1982), though his data is very scattered. Stednick (1996) fitted a linear relationship through his entire data set to determine the general rate of increase in water yield as a function of percent area of forest cleared. His line of best fit indicated an increase of about 25 mm in annual yield for each 10% of forest area cleared, though the r2 value for this relationship was only 0.17 and the standard error was 149 mm. Breaking the data down by region, he found that the increase rate varied between 7 and 61 mm per 10% of forest area cleared. Based on his 'all areas' relationship, Stednick (1996) concluded that about 20% of the forest area needed to be cleared before any water yield change could be detected, supporting the earlier finding of Bosch and Hewlett (1982). However, after breaking the data down by region, he determined that the threshold area varied between 15 and 50%.



- **Figure 2-2 General Trend Lines Showing The Relationship Between Cleared Area And Yield Increase For Three Different Types Of Woody Vegetation (Adapted From Bosch And Hewlett, 1982 And Cornish, 1989).**

- **Table 2-5 Estimated Annual Yield Increase Per 10% Of Catchment Area Cleared Of Forest (Ayi/10%) For Various Geographic Regions In The USA (after Stednick, 1996).**

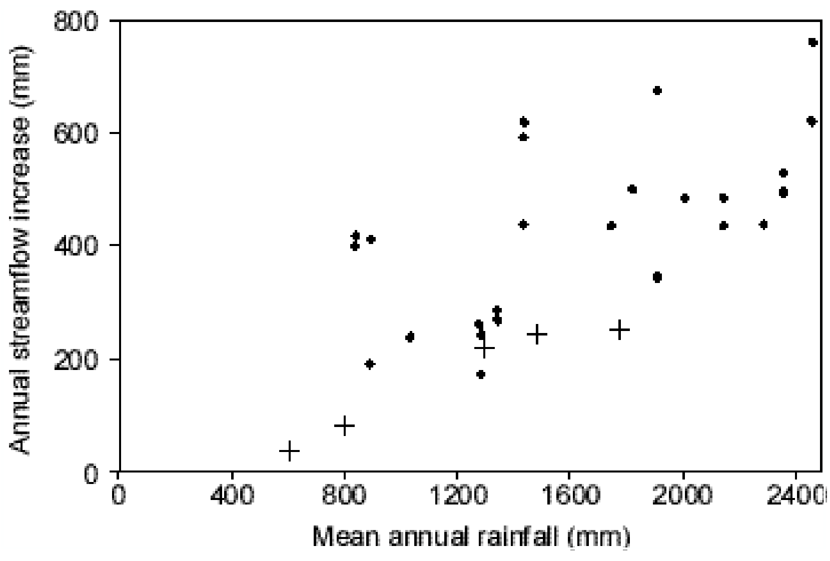
Region	#	AYI/10% (mm)	r^2	SE (mm)	Threshold (%)
All areas	95	25	0.17	149	20
Appalachia	29	28	0.65	75	20
East Coast	7	19	0.02	97	45
Rocky Mountains	35	9	0.01	66	15
Pacific Coast	12	44	0.65	118	25
Central Plains	7	61	0.31	197	50

# denotes the number of studies included in each relationship, SE denotes the standard error, and r^2 denotes the goodness of fit. The 'threshold' is the percent forest area needed to be cleared before a water yield change can be detected

Stednick (1996) fitted a similar relationship to the Bosch and Hewlett (1982) data set shown in Table 2-5, pooled with some English data reported by Calder (1993). That relationship predicted a general increase in annual yield of about 33 mm for every 10% of forest area cleared, and was characterised by an r2 value of 0.50 and a standard error of 89 mm.

Summarising results from a large number of studies in the north-east USA, Hornbeck et al. (1993) determined that the first year increase in streamflows following clearance varied between 12 and 35 mm per 10% of forest area cleared. Hornbeck et al. (1993) stressed the importance of felling configuration and clearance method as a determinant of catchment response. They noted that streamflow increases were greatest in catchments where the forest was cleared from the valley bottom runoff-producing areas.

Bosch and Hewlett (1982) assessed the influence of mean annual rainfall on water yield increases caused by forest clearance. They scaled their data to represent yield increases that would occur if the entire forest area was cleared in the catchments examined. Figure 2-3 shows that water yield gains caused by total removal of conifers and scrub generally increase as a function of mean annual rainfall. Also shown in Figure 2-3 are the yield increases predicted by the Holmes and Sinclair (1986) relationship (HSR) for eucalypt forest conversion to grassland. The yield increases predicted by HSR are similar to the scrub values determined for low rainfall areas (400-700 mm), and are on the low side of the conifer values determined for intermediate rainfall areas (700-1400 mm). For the higher rainfall areas (above 1400 mm), the HSR estimates of yield increase are much lower than predicted by Bosch and Hewlett (1982) for conifers. For instance, according to Figure 2-3, conifer clearance in an area with mean annual rainfall of 1800 mm results in an annual water yield increase of between 400 and 450 mm. HSR estimates that eucalypt forest conversion to grassland in this rainfall regime causes an annual yield increase of 250 mm, while Cornish (1989) predicts an increase of 285 mm (see his Table 4, p. 17).



■ **Figure 2-3 Effect Of Mean Annual Rainfall On Water Yield Increases Caused By Total Clearance Of Conifer And Scrub Vegetation From Catchments, Adapted From Bosch And Hewlett, 1982. Crosses Denote The Expected Water Yield Gains From Conversion Of Eucalypt Forest To Grassland, Based On The Holmes And Sinclair (1986) Relationship.**

2.6.2 Studies in tropical and sub-tropical forests

Bruijnzeel (1990) reviewed some 20 paired catchment studies based in various humid tropical and subtropical countries. He argued that the direction and magnitude of water yield changes accompanying forest clearance in these countries were similar to those reported by Hibbert (1967), Bosch and Hewlett (1982) and Stednick (1996) for temperate forests of the northern hemisphere. Bruijnzeel (1990) noted that light selective harvesting usually had little effect on streamflow, and that the effect increased with the amount of timber removed. He also showed that afforestation of grasslands or croplands with pine or eucalypts caused water yield to decrease and remain at lower levels.

2.6.3 New Zealand studies

Most New Zealand studies focus on the streamflow impacts of converting indigenous, mixed evergreen forest and native tussock grassland to pine plantations. These studies show that pine afforestation causes significant reductions in streamflow, particularly when displacing native tussock grasslands.

Dons (1986) reported a mean annual streamflow decline of 83 mm in a large basin vegetated with scrub that had 28% of its area afforested with pines. This equates to a streamflow decline of 30 mm per 10% of catchment afforested. Faye and Jackson (1997) cite streamflow increases ranging between 22 and 67 mm per 10% of forest area cleared, for the first year after clearance of indigenous forest at Maimai and Big Bush in the South Island of New Zealand (Table 2-6). For the seven catchments they examined, the average streamflow increase was 44 mm per 10% of forest cleared.

■ **Table 2-6 Annual Increases In Streamflow In The First Year After Clearance Of Indigenous Forest At Maimai And Big Bush, New Zealand (after Fahey and Jackson 1997).**

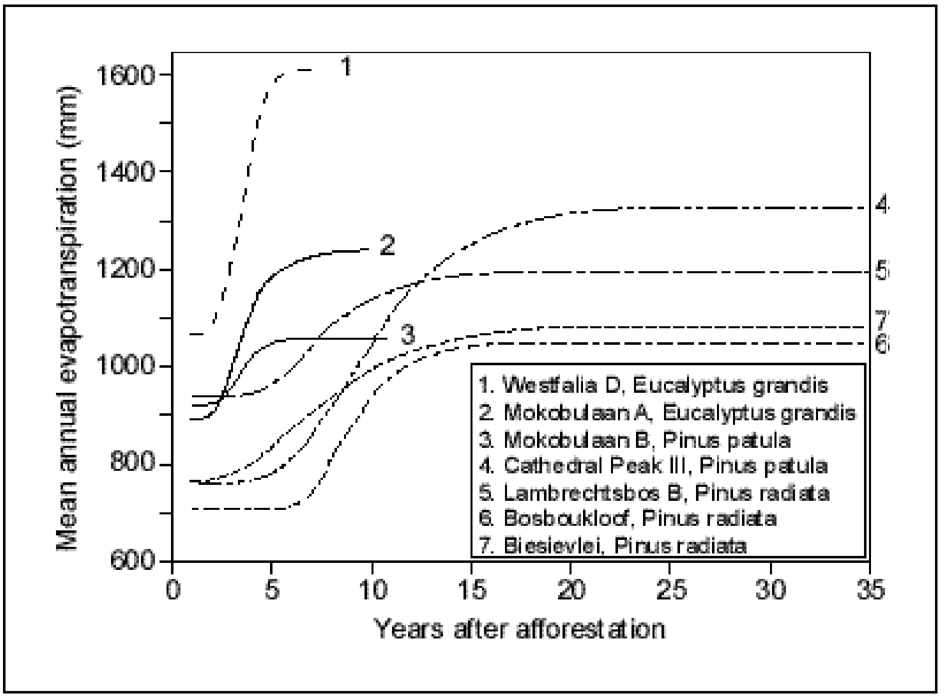
Catchment	Mean annual rainfall (mm)	Streamflow increase per 10% of forest cleared (mm)
Maimai M9	1930	67
Maimai M7	1930	65
Maimai M5	2625	55
Big Bush DC1	1305	37
Big Bush DC4	1305	37
Maimai M8	2827	27
Maimai M13	2625	22

2.6.4 South African Studies

Wood products from plantations contribute about 2% of the GDP in South Africa (Versfeld 1993). According to Dye (1996), the plantations are primarily based on pine (51%) and eucalypt (29%) species, and are confined to a relatively small area of the country which has an annual rainfall of more than 700 mm. These plantations have mostly replaced native scrub and grasslands. There is considerable evidence showing that the plantations have significantly higher ET rates than the original indigenous vegetation, and that streamflows have declined markedly as a result of afforestation. It is worth noting that most of the South African data is based on afforestation studies, rather than forest harvesting experiments. In general, the data sets are of high quality and are long term in nature, with many studies having post-treatment records exceeding 30 years in length.

Bosch (1979) reported that the afforestation of 74% of a catchment under native grassland with P. Patula reduced annual streamflow by about 260 mm over a period of

27 years after planting, equivalent to a decline of 35 mm per 10% of catchment afforested. Van Wyk (1987) reported mean annual streamflow declines of 313, 197 and 171 mm for three grassland catchments afforested with pines in the South Western Cape Province of South Africa. These catchments had plantations established on 98, 57 and 36% of their areas, respectively. Hence, the rates of streamflow decline equate to 32, 35 and 47 mm per 10% of catchment afforested, respectively. The 47 mm value seems to be unusually high when compared to other South African data, suggesting that it may be an artefact of the small area afforested (36%) in that particular experiment.



■ **Figure 2-4 Evapotranspiration Trends In Afforested South African Catchments (After Dye 1996).**

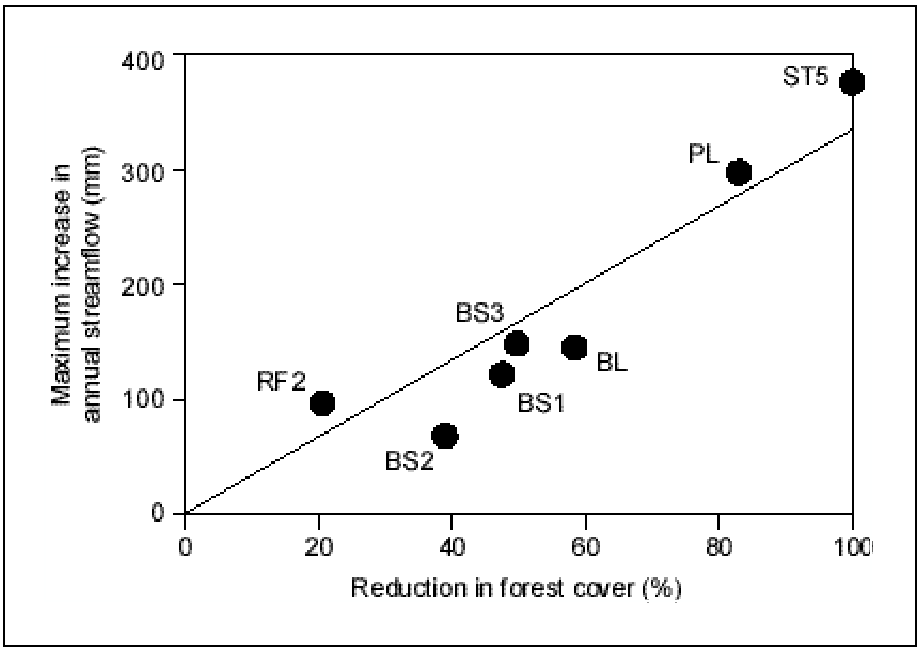
Dye (1996) summarised the results of seven catchment studies carried out by Bosch and von Gadow (1990), Smith and Scott (1992) and Versfeld (1993). He graphed ET changes resulting from afforestation, demonstrating that the rate and magnitude of response varied depending on the type of vegetation change and the mean annual rainfall of the catchment (Figure 2-4). The annual ET in six of the catchments under the indigenous scrub or grassland vegetation ranged between 700 and 950 mm, and increased to between 1050 and 1330 mm after afforestation. The rate of increase was higher for eucalypts than pine, reflecting the relatively rapid initial growth and canopy closure of eucalypt stands in South Africa. Dye (1996) concluded that where the demand for runoff is high, plantations of pines are preferable to eucalypts. This conclusion contradicts Australian findings on comparative rates of ET and streamflow for pine plantations and eucalypt forests.

2.6.5 Australian studies

Pooling data from the Maroondah, Stewarts Creek and Reefton catchments in Victoria, Nandakumar and Mein (1993) quantified the impact of eucalypt forest clearance on streamflows (Figure 2-5). They determined that streamflow increased by

33 mm for each 10% of forest area cleared. The linear regression underpinning this relationship had an r2 value of 0.88 and a standard error of 43 mm, meaning that it is a lot stronger (statistically speaking) than the relationships published by Bosch and Hewlett (1982) and Stednick (1996). The average rainfall of the sites they examined was about 1400 mm. The Holmes and Sinclair (1986) relationship (Figure 2-1) suggests that conversion of forest to grassland at this rainfall isohyet should yield an additional 240 mm of streamflow, whereas Figure 2-5 suggests a value of 330 mm. This difference is partly explained by the fact that the Holmes and Sinclair (1986) value is based on mean annual streamflow increase, whereas the Nandakumar and Mein (1993) value is based on the maximum streamflow increase.

The Nandakumar and Mein (1993) estimate of maximum streamflow increase (33 mm per 10% of forest area cleared) lies between the conifer and deciduous hardwood values reported by Bosch and Hewlett (1982), these being 40 and 25 mm, respectively. This value is very similar to mean values reported by Bosch (1979), Van Wyk (1987) and Dye (1996) for South African scrub and grassland catchments afforested with pines and eucalypts (32-35 mm per 10% area afforested). However, it lies towards the low end of the range determined for cleared indigenous forests in New Zealand (Table 2-6).



■ **Figure 2-5 Relationship Between Maximum Annual Streamflow Increase And Percent Area Of Forest Cleared For Seven Catchments In Victoria (After Nandakumar And Mein 1993).**

Cornish and Vertessy (in press) reported maximum annual streamflow increases of 40 to 50 mm per 10% of forest area cleared in four of the six treated Karuah catchments (Table 2-7). A fifth catchment (Kokata) yielded additional streamflow of 70 mm per 10% forest area cleared, though this was shown to have been heavily compacted, leading to greatly enhanced surface runoff. A sixth catchment (Barrata) did not experience a streamflow increase, though this was only logged over a relatively small area (25%) which is close to the ‘detection limit’ in studies of this kind. The logginginduced streamflow increases reported for Karuah are slightly higher than reported for other Australian studies.

- **Table 2-7 Maximum Annual Increases In Streamflow After Clearance Of Eucalypt Forest At Karuah, NSW (Based On Data From Cornish And Vertessy, In Press).**

Catchment	Mean annual rainfall (mm)	Streamflow increase per 10% of forest cleared (mm)
Kokata	1599	70
Bollygum	1532	56
Coachwood	1472	43
Corkwood	1694	43
Jackwood	1391	37
Barratta	1657	0

Cornish (1991) reported a peak annual streamflow increase of 35 mm per 10% of logged area for the Yambulla 2 catchment, part of the Yambulla group of catchments south of Eden. However, he also showed that the Yambulla 3 catchment experienced an annual streamflow rise of only 20 mm per 10% of forest area cleared. Cornish (1991) attributed the higher rate of increase in Yambulla 2 to the fact that this catchment was burnt after logging, hence reducing surface cover and possibly causing an hydrophobic effect in the soils that would increase surface runoff.

Mackay and Robinson (1987) do not report annual streamflow amounts for the Yambulla catchments but argue that streamflows increased by factors of 2.1, 6.8, 4.9 and 5.5 in the four treated catchments in the three years following forest clearance and/or fire (their Table 9, p. 378). It should be noted that these values represent absolute changes, and are not scaled to percent area of forest treated.

Roberts (per. comm.) analysed streamflow changes ensuing from logging and fire in the Yambulla 5 and 6 catchments. She determined maximum annual streamflow increases of 38 and 16 mm per 10% of forest area treated in Yambulla 5 and 6, respectively.

Lane and Mackay (2000) analysed 11 years of streamflow data from a paired-catchment study conducted in three small catchments in the headwaters of Tantawangalo Creek near Bega, NSW. Patch-cut logging resulted in an initial increase in total streamflow and baseflow in both logged catchments. These streamflow increases varied from 10% to 31% over the first four years. In the fifth year after logging streamflows returned to pre-treatment levels, and then declined a further 20% in one catchment only. These streamflow reductions principally consisted of lowered baseflows. The differences in catchment response were attributed to differences in forest regeneration, with much more vigorous regeneration in the catchment with greater water use.

2.6.6 The persistence and permanence of streamflow increases

The results presented in the preceding sections focussed mainly on (a) conversion of forest to grassland or vice versa, or (b) the few years immediately after forest harvesting In the former case, average changes in annual streamflow were reported, whereas in the latter case, the maximum reported increase in yield was usually reported.

In a forest which is harvested but permitted to regenerate, streamflow increases are both transient and temporary. Streamflow increases normally reach a peak in the second or third year after clearance, then decline to pre-treatment levels over a period of between 5 and 25 years, depending on rainfall, soil factors and forest growth rates. In some cases, streamflows then decline below pre-treatment levels. Such cases will be examined in later sections of this review.

Summarising results from 11 long-term catchments studies in the north-east USA, Hornbeck et al. (1993) noted that post-logging streamflow increases rarely persisted more than 10 years. However, he also noted examples where the streamflow increases had been purposefully maintained by the intermediate cuttings and the periodic application of herbicides to control regrowth. In these cases, streamflow increases over pre-treatment levels were maintained for over 20 years.

Nandakumar and Mein (1993) studied the transience of streamflow changes in the few years following forest clearance and regeneration in six eucalypt forest catchments in Victoria. They determined that peak streamflow increases were attained 2 or 3 years after clearance and that recovery to pre-treatment streamflow levels was reached anywhere between 5 and 25 years later (Table 2-8). As has been noted in north American studies (Swank et al., 1988; Hornbeck et al. 1993), Nandakumar and Mein (1993) found that streamflows declined at a logarithmic rate once the peak increase had been attained. They provide an equation with fitted parameters to predict streamflow recovery rates for the six catchments they studied.

- **Table 2-8 Magnitudes And Time Scales Of Streamflow Increases Following Forest Clearance In Six Victorian Catchments (After Nandakumar And Mein 1993).**

Catchment	Time to reach peak streamflow (years)	Maximum annual streamflow increase (mm)	Time for streamflow to return to pre-treatment level (years)
Stewarts Creek 5	3	296	20
Black Spur 1	2	145	8
Black Spur 2	2	64	5
Black Spur 3	2	137	25
Picaninny	2	388	7
Blue Jacket	2	190	8

Note: The catchments had variable areas of forest cleared. The time to recovery is a predicted value, based on a logarithmic equation.

For the Karuah catchments, Cornish and Vertessy (in press) showed that streamflow increases peaked two or three years after clearance and returned to pre-treatment levels within a period of between four and eight years. They noted that the rate of decline in streamflow after the peak increase was proportional to the stocking rate of the regenerating forest. This finding is consistent with world experience which suggests that streamflow changes are most short lived in forests of high productivity; the quicker the new forest canopy can form and close over, the quicker streamflows will return to pre-treatment levels because of rapid increases in forest water use.

In her analysis of streamflow changes at Yambulla 5 and 6, Roberts (pers. comm.) found that the maximum increase in streamflow in both catchments was only attained six years after treatment. This is much later than observed for most catchments in the international literature, and may be due to the highly variable rainfall pattern in the immediate post-treatment period. She also noted that recovery to pre-treatment streamflows occurred by year 9 in Yambulla 6 and by year 13 in Yambulla 5. This is not surprising given that forest regeneration was allegedly far more vigorous in Yambulla 6.

Differences in forest regeneration were given as the reason for ultimate differences in water yields following logging and regeneration in the Tanatawangalo catchments (Lane and Mackay, 2000), although streamflow in both logged catchments had returned to pre-treatment levels in 5 years.

2.6.7 Comparative effects of uniform thinning and patch cutting

A range of silvicultural treatments in addition to clear-felling are employed in forest management. Various methods of thinning may be carried out on the forest during a rotation, and these are likely to have differing effects on forest water use and streamflow. Unfortunately very few studies, which quantify thinning effects, have been carried out to date.

Jayasuriya et al. (1993) compared streamflow changes ensuing uniform thinning and patch cutting on two of the Black Spur catchments in the Maroondah basin. Both treatments resulted in a removal of about 50% of the forest basal area, and were applied to adjacent catchments with similar soils and rainfall. Initial streamflow increases were similar for both treatments (130-150 mm per year) but were more persistent for the uniform thinning treatment. In the case of the uniformly thinned catchment (Black Spur 3), a 15% streamflow difference was still evident 11 years after treatment, whereas streamflows had returned to pre-treatment levels in the patch cut treatment (Black Spur 1) after 5 years. Nandakumar and Mein (1993) have predicted that it will take 25 years for streamflows in Black Spur 3 (the patch cut treatment) to return to pre-treatment levels.

Following the return of water yields to pre-treatment levels in the two patch-cut Tantawangalo catchments (Lane and Mackay, 2000), the catchments then behaved differently. Water yields in the catchment with the most vigorous regeneration declined further, while yields in the other remained at pre-treatment levels.

2.7 Effects of Forest Age on Streamflow

Australian forest hydrology is distinguished by a unique concern for the effect of forest age on streamflow. Within Australia there is some acceptance of the notion that regrowth forests yield less streamflow than old growth forests, because of differences in ET, yet this trend has only been verified for two native forest areas in Australia (Maroondah and Karuah). Leaf area and ET measurements in a third forest catchment in Australia (Yambulla) suggest that forest age could affect streamflows, though there is no hydrometric evidence to back up this hypothesis. Overseas research has in general not observed that water yield in regenerating forests containing the same species becomes less than that in the previous forest, but this may be a consequence of most logged forests studied being regenerating, as opposed to old-growth, stands. Below, we review the data supporting a link between forest age and streamflow.

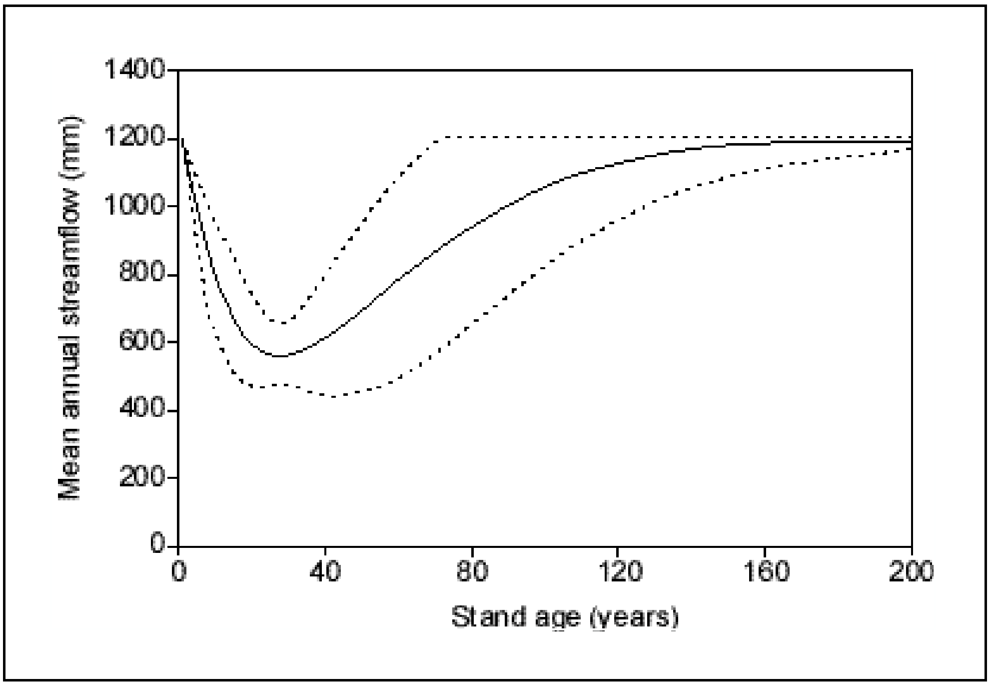
2.7.1 Victorian Mountain ash forests

The most comprehensive understanding of forest age on streamflows in Australia is based on the mountain ash forests of Maroondah catchments in the central highlands of Victoria. Multiple catchment treatment experiments (Langford and O'Shaughnessy, 1977; Langford and O'Shaughnessy, 1980; Watson et al., 1998) have been complemented by a large body of local process studies aimed at elucidating the process of evapotranspiration through the mountain ash forest life cycle (Dunn and Connor, 1993; Jayasuriya et al., 1993; Vertessy et al., 1995; Haydon et al., 1996; Watson and Vertessy, 1996; Vertessy et al., 1997; Vertessy et al., 1998). The Maroondah catchments are fully forested, mainly with the mountain ash (E. Regnans) species, and are characterised by high rainfall (1200-2500 mm per year), and a cool, montane climate. These catchments are renowned for their deep and permeable soils which result in a strong dominance of baseflow in catchment runoff. The ecology of mountain ash forests is very distinctive, in that they regenerate prolifically after severe wildfire which kills the trees and produces a heavy seedfall. These forests are thus usually even-aged and monospecific, and tend to live for several hundreds of years unless they are killed earlier by wildfire. Significantly, the species thins naturally over time, resulting in major changes in forest structure and hydrologic function as stands age (Watson and Vertessy, 1996; Vertessy et al., 1998).

Based on a large body of streamflow data, Langford (1976) developed relationships linking streamflow from mountain ash catchments to forest age. Kuczera (1985) built upon this by developing an idealised curve describing the relationship between mean annual streamflow and forest age for mountain ash forest (Figure 2-6). The curve combines the known hydrologic responses of eight large (14–900 km2) basins to regeneration after fire, and is constructed for the hypothetical case of a pure mountain ash forest catchment. The 'Kuczera curve' is characterised by the following features:

- the mean annual runoff from large catchments covered by pure mountain ash forest in an old-growth state is about 1195 mm;
- after burning and full regeneration of the mountain ash forest with young trees, the mean annual runoff reduces rapidly to 580 mm by age 27 years;
- after age 27 years, mean annual runoff slowly returns to pre-disturbance levels, possibly taking as long as 150 years to recover fully.

The 'Kuczera curve' has three major deficiencies which should be noted. Firstly, the relationship fails to recognise the increase in catchment runoff that occurs for the first 4-6 years after forest clearance (Nandakumar and Mein, 1993; Watson et al., 1998). This increase could not be detected in the large basins examined by Langford (1976) and Kuczera (1985), but has been noted in most small catchment studies in the Maroondah area (Langford and O'Shaughnessy, 1977). Secondly, the curve has wide error bands associated with it, particularly for forests aged between 50 and 120 years (Figure 2-6), so it is difficult to accurately predict when water yields will recover after disturbance. Thirdly, the curve is a generalised one, masking the great deal of variation that exists between ash forest catchments with different site characteristics. For instance, mean annual streamflows from individual catchments of old-growth mountain ash are known to vary between 250 and 1500 mm.



■ **Figure 2-6 Relationship Between Forest Age And Mean Annual Runoff From Mountain Ash Forest Catchments (After Kuczera 1985).). Dotted Line Denotes The 95% Confidence Limits On The Relationship.**

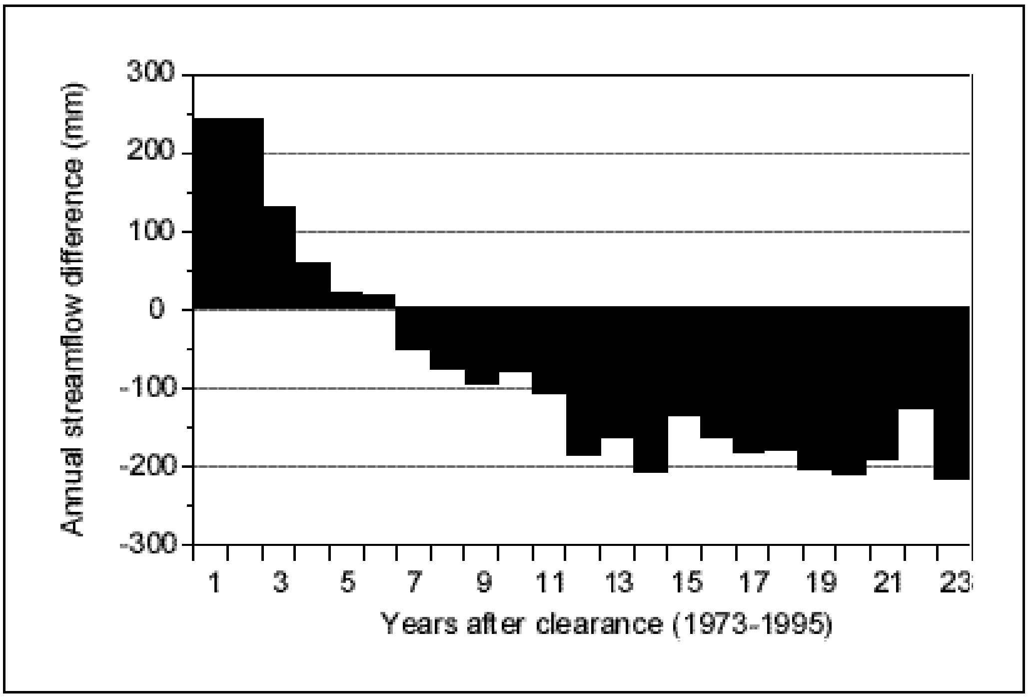
To emphasise this last point, Figure 2-7 shows the streamflow response of the Picaninny catchment to a 78% clearfell in 1972. The shape of the streamflow response is similar to that predicted by the Kuczera curve, though the magnitude of response is significantly less because of the lower mean annual rainfall for this catchment (~1200 mm, rather than the 1950 mm assumed in the Kuczera curve).

Vertessy *et al.* (1998) provided a mechanistic explanation for the 'Kuczera curve' by elucidating leaf area and evapotranspiration dynamics in mountain ash forests of various ages. Their breakdown of the mountain ash water balance for the 1800 mm isohyet, and the manner in which it changes through time, is depicted in Figure 2-8.

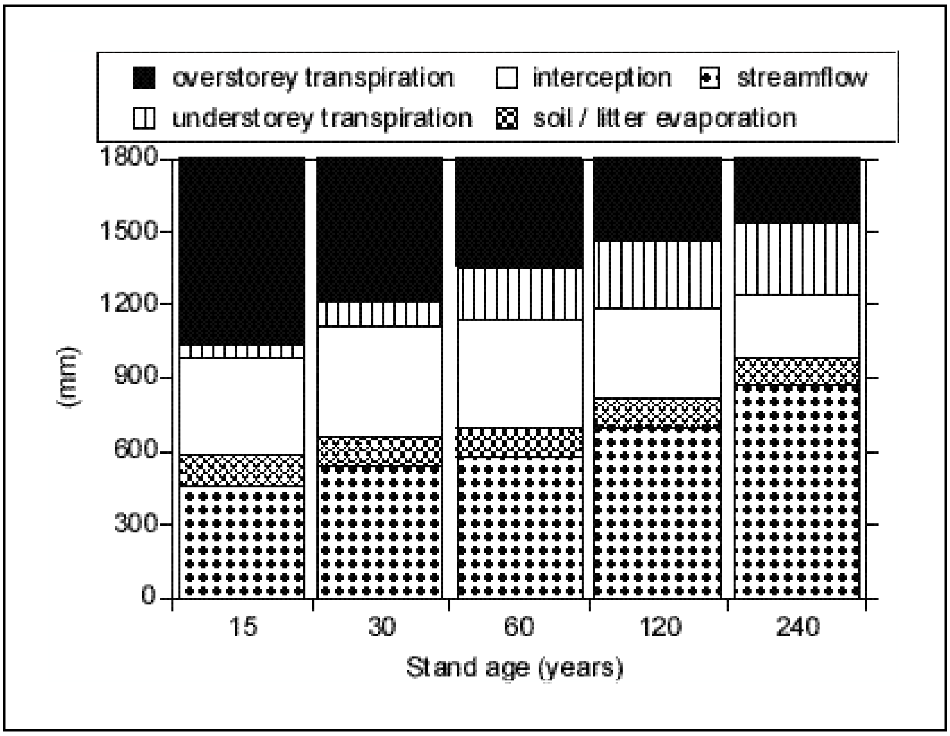
Vertessy *et al.* (1998) noted the following:

- overstorey (mountain ash) leaf area index (LAI) declines from 3.8 at age 15 years to 1.2 at age 240 years;
- understorey LAI increases from 0.4 at age 15 years to 2.4 at age 240 years, thus partially offsetting overstorey LAI declines with age;
- hence, total forest LAI declines from 4.2 at age 15 years to 3.6 at age 240 years;
- however, transpiration per unit area of leaf in the understorey is only 63% of that measured for overstorey;
- annual overstorey transpiration declines from 760 mm at age 15 years to 260 mm at age 240 years;

- annual understorey transpiration increases from 50 mm at age 15 years to 300 mm at age 240 years, off-setting (by half) the reduction on overstorey transpiration over the same period;
- annual rainfall interception peaks at 450 mm at age 30 years and declines to 260 mm at age 240 years, further reducing evapotranspiration;
- overall, there is a 420 mm difference in the annual evapotranspiration of 15 and 240 year old forest, which results in a runoff difference of the same magnitude; and,
- 48% of the change in runoff is attributable to differences in transpiration, 45% is due to rainfall interception differences, and 7% is due to changes in soil/litter evaporation.



- **Figure 2-7 Annual streamflow change from the Picaninny catchment, Victoria, from 1973 to 1995 (after Vertessy et al. 1998). The changes shown are relative to streamflows from the undisturbed Slip Creek catchment, sited adjacent to Picaninny**

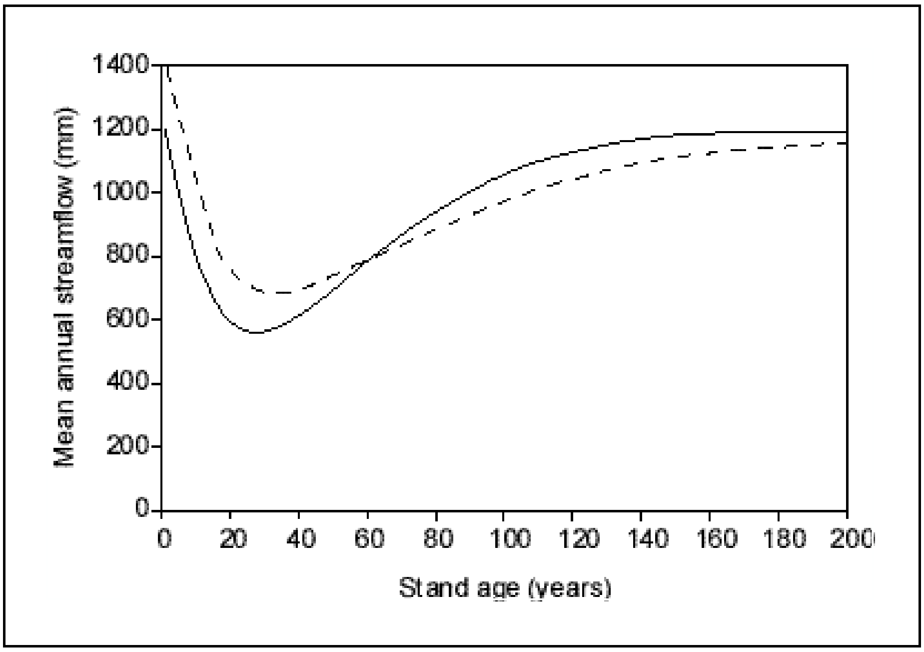


- **Figure 2-8 Water balance for mountain ash forest stands of various ages, assuming annual rainfall of 1800 mm (after Vertessy et al. 1998).**

Using the small catchment experimental data yielded from the Maroondah basin study, Watson *et al.* (1998) developed an alternative forest age-streamflow relationship (Figure 2-9). It differs from the Kuczera curve in the following respects:

- it incorporates *increases* in streamflow which have been observed to occur in the first few years after forest clearance;
- the maximum streamflow reduction is about 100 mm *less* than indicated by the Kuczera curve;
- the rate of streamflow recovery is much more gradual, even though it returns to pre-treatment level in the same length of time;
- it is specific to the Maroondah catchments, rather than generalised for a large region.

Watson *et al.* (1998) point out that their curve is 'fitted by eye', though is arguably just as legitimate as selecting any particular mathematical form to fit through sparse data points. They provide a rather awkward seven parameter equation that describes their alternative relationship, along with parameter values used to produce the curve shown in Figure 2-9.



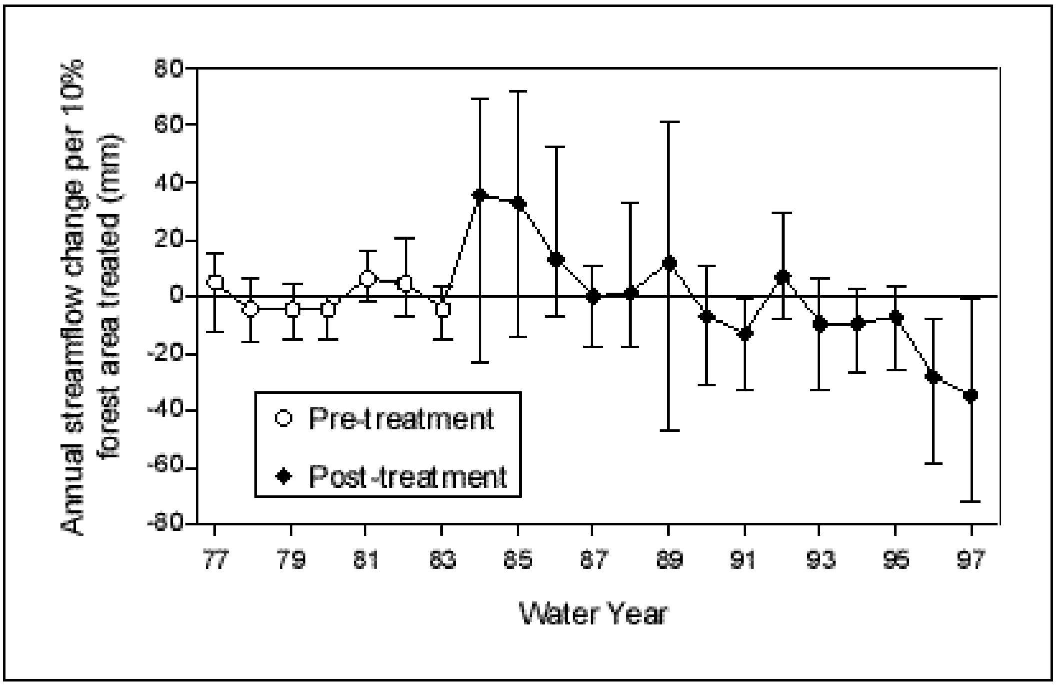
- **Figure 2-9 An alternative 'model' of the streamflow-forest age relationship for mountain ash forests (dashed line) (after Watson et al. 1998). The Kuczera curve (solid line) is shown for reference.**

2.7.2 The Karuah research catchments

The only other Australian studies which have demonstrated a link between streamflow and forest age are those of Cornish (1993), Cornish and Vertessy (1998) and Cornish and Vertessy (in press). These studies are based on streamflow data from the moist eucalypt forests of the Karuah catchments in the lower north east region of NSW. They share some common features with the mountain ash forest catchments in Victoria, notably high annual rainfall (~1600 mm), highly productive, moist eucalypt stands, relatively deep and permeable soils, and strong baseflow. Cornish and Vertessy (in review) analysed streamflow records for six catchments, logged to varying extents (25-79%). Their analysis is restricted to 7 pre-treatment and 14 post-treatment years of streamflow. Hence, the period of record is shorter than that underpinning the mountain ash forest record in Victoria.

Cornish and Vertessy (in review) showed that streamflows declined below pre-treatment levels seven years after logging in three of the six treated catchments, and declined in a regular manner over the next seven years. The other three treated catchments showed an initial decline in streamflows below pre-treatment levels around year 8, followed by a slight increase, then another decrease below pre-treatment levels. Cornish and Vertessy (in press) showed that these three catchments were affected by insect attack, leading to decreased leaf area and ET rates and enhanced streamflows.

Figure 2-10 shows the average and range of streamflow changes caused by forest disturbance in the six treated Karuah catchments. It shows that the maximum decrease in annual streamflow is over 60 mm per 10% of forest area treated, which is similar to the maximum reductions noted for Victorian mountain ash forests. However, some of the Karuah catchments have shown a comparatively modest reduction in streamflow, meaning that the average reduction in streamflow is about 35 mm per year per 10% of forest treated by the end of the post-treatment period of record. It is worth noting, however, that further streamflow reductions are likely in the future as the peak forest growth rate has probably not yet been attained in the Karuah catchments. Also, the catchments which have been affected by insect attack are likely to experience increased growth and further reduce streamflows.



■ **Figure 2-10 Annual streamflow changes amongst the six treated Karuah catchments in the NSW lower north east region (after Cornish and Vertessy, in PRESS). Ends of bars denote maximum and minimum changes, symbols denote mean change.**

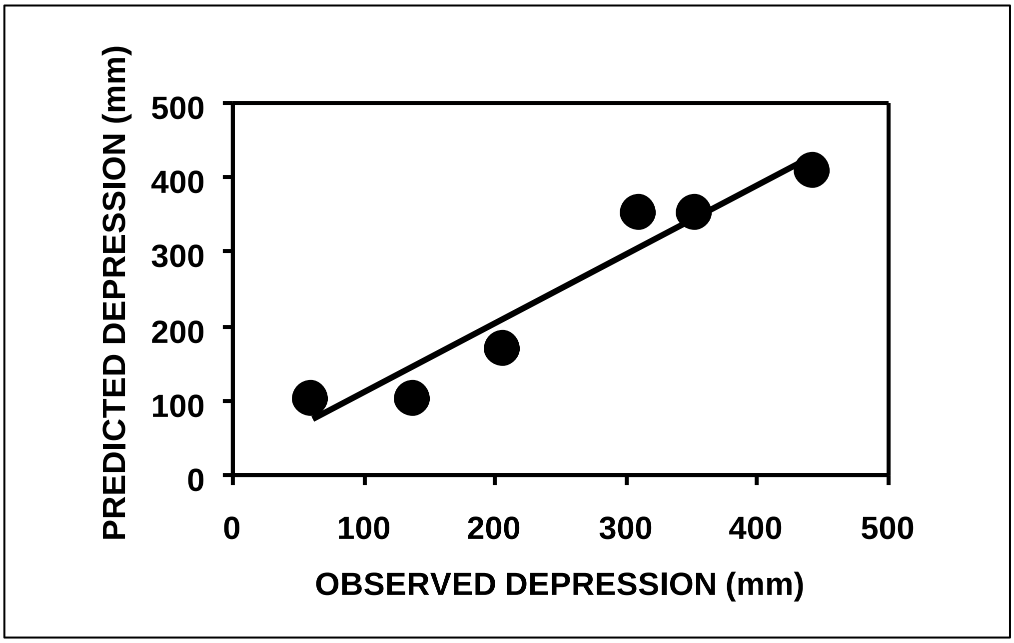
Cornish and Vertessy (in press) related the magnitude of streamflow reductions in the Karuah catchments to mean annual basal area increase (an index of forest growth rate), soil depth (an index of soil water storage) and canopy cover (a crude index of rainfall interception and transpiration rate). They developed the following equation:

$$ASR = -926.0 + 221.94 * SD\ (m) + 17.07 * CC\ (\%) - 1.85 * BAI \qquad (2.2)$$

where:

ASR = reduction in annual streamflow for the final year of record (mm)
SD = soil depth (m)
CC = canopy cover (%)
BAI = mean annual basal area increment

The model fit obtained by Equation 2.2 is shown in Figure 2-11. Equation 2.2 was shown to account for almost 85% of the variation in observed annual streamflow reductions, with soil depth being the most important explanatory variable in the equation. It is noteworthy that the Victorian mountain ash forest catchments have very deep soils.



■ **Figure 2-11 Observed and predicted reductions in annual streamflow for the six treated Karuah catchments (after Cornish and Vertessy, in PRESS). The predictions are based on Equation 2.2 and assume that the entire catchment area was treated.**

2.7.3 The Yambulla research catchments

Roberts *et al*. (2000) compared leaf area index and transpiration rates in *E. Sieberi* forest of different ages in the Yambulla State Forest, nearby but outside of the experimental catchments. Over a two-month period in summer, they determined that mean daily transpiration was 2.2, 1.4 and 0.8 mm for stands aged 14, 45 and 160 years, respectively. These differences were evident in spite of fairly similar leaf area indices amongst the three stands, leading the authors to suggest that 'transpiration per unit leaf area' declines with forest age. Similar trends have been measured but not yet published for mountain ash forests (Vertessy, unpublished data). These findings imply that streamflows should generally *increase* as the *E. Sieberi* forest ages. However the age of maximum growth and productivity in these forests is around age 20 years, and it is possible that *E. Sieberi* stands of this age may have transpiration rates exceeding (comparatively) 2.2 mm/day.

Roberts (pers. comm.) has analysed streamflow data for the Yambulla 1, 5 and 6 catchments. Yambulla 1 is an undisturbed control catchment, whereas Yambulla 5 and 6 are the most heavily treated catchments, having been affected by both logging and wildfire over extensive areas. For the 18 years following disturbance and forest regeneration in these catchments, there is *no evidence that streamflows have reduced to below pre-treatment levels*, despite fairly vigorous regeneration of forest in Yambulla 6. Hence, there is no hydrometric evidence to support the contention of Roberts *et al*. (1998) that forest age affects the water balance of *E. Sieberi* catchments.

2.7.4 The Tantawangalo catchments

Lane and Mackay (2000) reported that water yields subsequently declined below pre-treatment levels in one of two eucalypt catchments logged in 1989. This decline (of 20% over the next four years) was attributed to the more vigorous forest regeneration in that catchment. It is notable that this catchment was patch-cut and only 22% of the basal area removed.

2.7.5 Streamflow Trend Analysis

Sinclair Knight Merz (SKM, 2000) conducted an analysis of time trends in streamflow for a number of streams in Victoria. These streams all had long streamflow records, good rainfall records and their catchments were principally forested. These forests had all been subjected to forest management over many years, leading to general reductions in forest age. All chosen streams were unregulated and unaffected by extractions.

Of the 20 streams investigated, 7 had significant trends in streamflow, and all trends were negative. Annual reductions in streamflow at these sites ranged from 0.4% to 1.76% of mean annual flow. The study concluded that, while these decreases could be attributed to the greater water use of a regenerating forest, no causal link to land use change could be made without in investigation of appropriate historical information.

2.7.6 Applications of the Kuczera curve to other sites

The 'Kuczera curve' and variants thereof, have been used to predict temporal changes in annual streamflows in catchments other than those on which it is based. Moran (1988) used it to predict the water yield consequences of logging mountain ash and mixed eucalypt forests, as well as softwood plantations, in the Otway Ranges of south-western Victoria. She developed a methodology for coping with species other than mountain ash and for adapting the 'Kuczera curve' to different site conditions. Cornish (1997) adapted the 'Kuczera curve' significantly and applied it to the Rocky Creek Dam catchment in the lower north east region of NSW. He used his adapted model to predict streamflow changes ensuing from rotational logging and thinning of small coupes of various ages in a large basin. A major contribution of this study is that Cornish (1997) illustrated the importance of logging rotation cycles and the age of the forest being logged in determining the overall streamflow response of the basin. Generally speaking, the streamflow responses were much lower than might have been anticipated, because (a) streamflow declines following logging were significantly reduced if the logged forest was not old growth, and (b) under rotational forestry, streamflow reductions in regenerating coupes were partly off-set by streamflow increases in freshly logged areas.

Sinclair Knight Merz (SKM, 1998; 1999) adapted this approach in developing a model that has been used to estimated water yield changes in native forests as part of the NSW CRA process. This model adapts the Kuzcera curve for environments in which rainfall is lower and tree growth slower, and estimates water yield changes into the future knowing the year-by-year age class distribution of the forest. The model employs GIS climate layers, streamflow records from gauged catchments and logging information. Catchments were selected where data was available and logging planned, and annual streamflow estimated for the next 120 years. Predicted changes depended on the percentage of the catchment to be logged, and on the present age of the forest. The logging of old-growth forest (and replacement with regrowth) lowered water yields, but the logging of current regrowth only reduced the magnitude of currently increasing yields. This is an important observation, and relevant to native forest harvesting generally. Water use reductions resulting from a regenerating forest will be larger if the relative age of regrowth and pre-logging forests is greater.

2.8 Effects of Forest Management on the Streamflow Regime

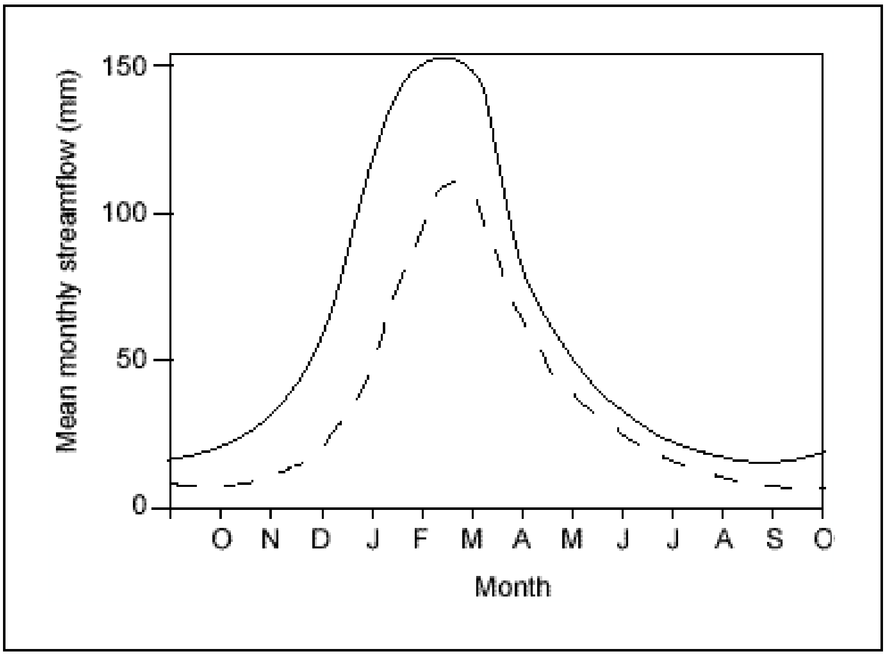
Thus far, this review has focussed on forestry-induced changes in annual streamflows. However, catchment managers also need to know how the streamflow regime might change as a consequence of forestry activities. This is an important issue as the distribution of flows has important consequences for the security of water for downstream enterprises (in the case of low flows), and the safety of dams, roads, culverts and bridges (in the case of high flows). Almost all catchment studies have noted that low, median and high flows decrease as a consequence of afforestation, and increase as a result of forest harvesting and roading (Hewlett and Helvey, 1970; Burt and Swank, 1992; Schofield, 1996; Jones and Grant, 1996). What is unclear is whether low flows and high flows change by the same amount as annual flows, or whether part of the streamflow range is more affected than others.

It is worth pointing out here that the literature on forestry-induced changes to flow regime is fairly confusing. Some workers frame their analyses around monthly streamflow totals, while others focus on instantaneous flow rates or on flow rates of a particular duration (usually hourly or daily). Furthermore, the literature on this topic is riddled with subjective concepts such as ‘low flows’ and ‘high flows’ which appear to be defined in a variety of ways. All of these factors make it difficult to compare findings from various studies. Finally, sub-annual streamflows, particularly instantaneous flows, are much more variable between catchments (because of differences in soils and topography) and in time (because of climate variability) than annual streamflows, meaning that it is difficult to ‘tease out’ the effects of forestry on flow regime.

2.8.1 Effects of afforestation on low flows in South Africa

The most detailed insights into the effects of forestry on flow regime come from catchment afforestation studies undertaken in South Africa. This research has shown that afforestation in South Africa has reduced all flows but that low flow reductions are relatively greater than reductions in annual flows (Bosch, 1979; Bosch and von Gadow, 1990; Smith and Scott, 1992; Scott and Smith, 1997).

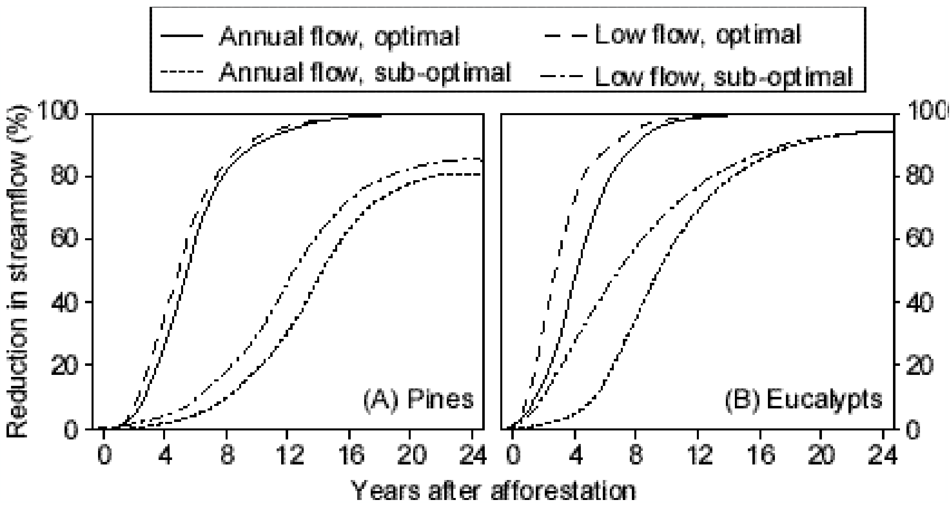
Bosch and von Gadow (1990) compared mean monthly streamflows for the Cathedral Peak catchment in South Africa, prior to and after afforestation of grasslands with pines (Figure 2-12). They demonstrated that absolute reductions in streamflow were greatest during the wet months, but that the reductions were *relatively* greatest during the low flow periods. For example, Figure 2-12 shows the streamflows in March (a wet month) are reduced by about 30% as a consequence of afforestation, but are reduced by over 60% in October (a dry month). They attributed this trend to the fact that grasses are dormant during the low flow periods and thus do not transpire. Bosch and von Gadow (1990) noted that streamflow reductions were relatively uniform throughout the year when pines replaced indigenous scrub vegetation which was also evergreen and thus active during the low flow periods. An interesting feature of Figure 2-12 is that monthly streamflow differences were smallest during the period following peak monthly streamflow (April-July). Presumably, abundant soil moisture is present in the system during this time, enabling the grasses to transpire at relatively high rates. These reserves are probably depleted from September onwards, resulting in diminishing grass ET and thus larger differences between grass and plantation streamflows.



- **Figure 2-12 Mean monthly streamflows in the Cathedral Peak catchment, South Africa, before and after afforestation of the grassland vegetation with pines (after Bosch and von Gadow 1990). Solid (upper) line denotes grassland condition, dashed (lower) line denotes pine plantation condition.**

Scott and Smith (1997) summarised a large amount of streamflow data from five afforestation experiments, comprising nine catchments sited throughout South Africa. They compared reductions in annual streamflows and low flows ensuing from afforestation of grasslands with pines and eucalypts. They also compared results from sites considered to be 'optimal' and 'sub-optimal' for the growth of these two forest types. 'Low flows' were defined as the three driest months of an 'average' year, or more specifically, as those below the 75th percentile of monthly flows. 'Optimal' sites were regarded as those with deep soils and a sub-tropical climate, whereas the 'sub-optimal' sites were regarded as those with shallow soils and cooler mountain climates. Scott and Smith (1997) fitted eight different empirical models of sigmoidal form to the data, seven of which had r^2 values exceeding 0.95. The worst model fit had an r^2 value of 0.89. These eight models were used to predict the percentage reduction in annual and low flows for pines and eucalypts growing under optimal and sub-optimal conditions as a function of time after afforestation (Figure 2-13).

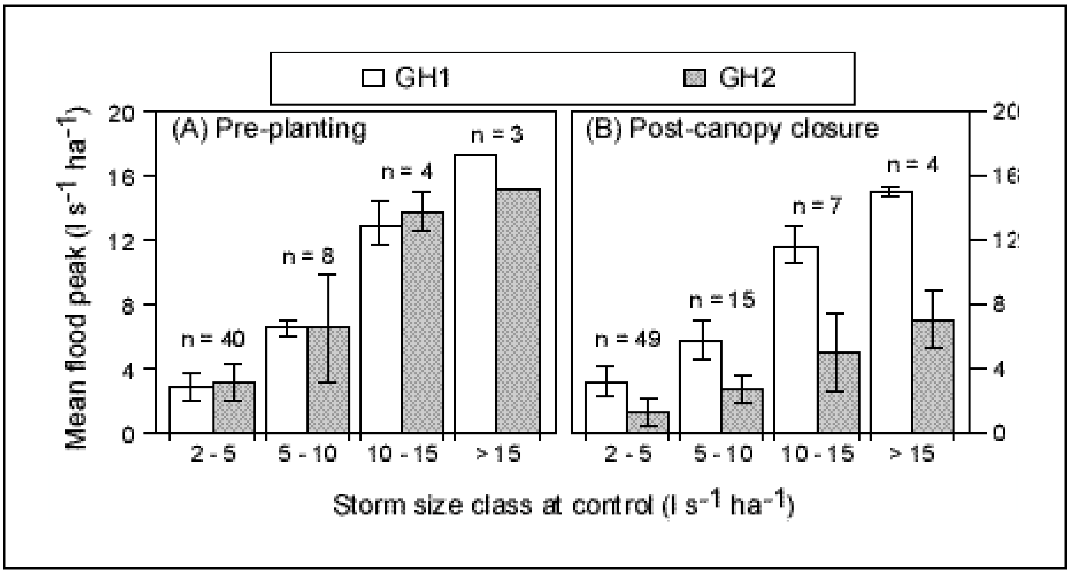
Figure 2-13 shows that the effects of afforestation on annual and low flows were less pronounced for pines than for eucalypts, a finding highlighted earlier in this review. However, flow reductions were far less pronounced for both forest types when 'sub-optimal' sites were afforested. For the 'optimal' sites, streamflows changed from perennial to intermittent after about nine years in the case of eucalypt afforested catchments, and after about 14 years in the pine afforested catchments. However, at the ‘sub-optimal’ sites, annual and low flows persisted through the plantation life-cycle for both forest types, though these were most reduced (~95%) in the case of the eucalypt afforested catchments. Figure 2-13 shows that low flows were reduced relatively more than annual flows, particularly in the case of eucalypts. These differences were amplified in the case of ‘sub-optimal’ sites, particularly in the early life of the eucalypt plantations.



■ **Figure 2-13 Generalised curves for predicting annual and low reductions as a function of age in pine and eucalypt plantations in South Africa (after Scott and Smith 1997). Separate curves are shown for optimal and sub-optimal growing regions for both forest types.**

2.8.2 Effects of afforestation on flood peaks in New Zealand

Fahey and Jackson (1997) showed that the conversion of tussock grasslands to pine plantations in the Glendhu catchments of South Island, New Zealand resulted in uniform decreases in flood peaks across the entire range of streamflows. They plotted the frequency distribution of mean flood peaks for four different size classes of storms, for discrete three-year periods before and after afforestation (Figure 2-14). Figure 2-14 compares the mean flood peak for each storm size class for the control catchment (G1) and the afforested catchment (G2), prior to afforestation and after canopy closure of the plantation. Mean flood peaks for each storm size class were similar in both catchments prior to afforestation. After plantation canopy closure, mean flood peaks were reduced by about 60% in all four storm size classes in the afforested G2 catchment. Fahey and Jackson (1997) also showed that low flows decreased as a result of afforestation in the Glendhu catchments, but they do not provide an analysis to suggest that they changed by a different rate when compared to high flows.

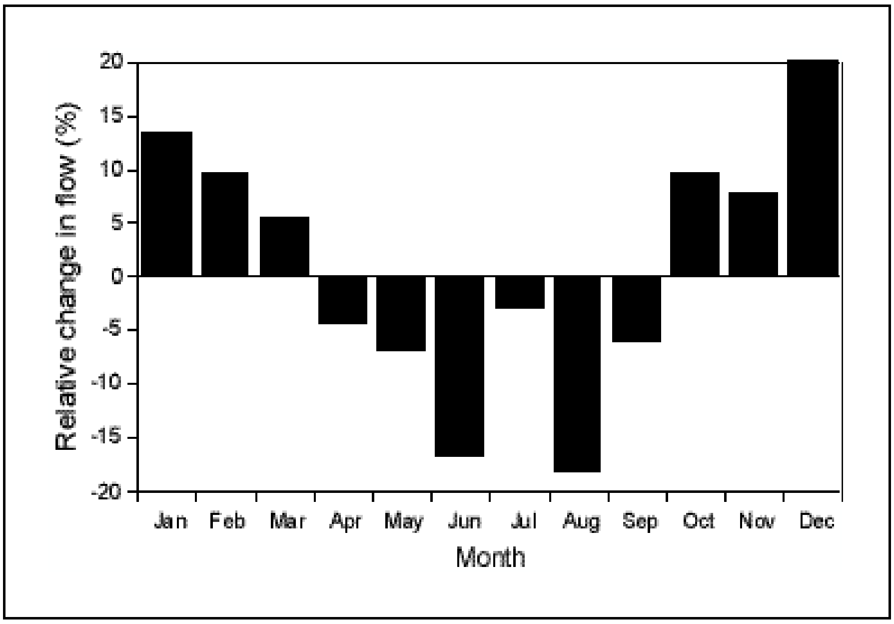


■ **Figure 2-14 Comparison of mean flood peaks for four size classes of storms in the G1 (control) and G2 (treated) catchment, (a) prior to the treatment of G2, and (b) after plantation canopy closure in G2 (after Fahey and Jackson 1997). Bars denote one standard deviation on each mean value, n denotes the number of storms in each class**

1.7.3 Harvesting and regeneration of moist eucalypt forest

Compared to South Africa and New Zealand, there is little evidence to support systematic changes in flow regime as a consequence of forestry activities in Australian catchments. Some relevant Australian data are available from the Victorian mountain ash forest and Karuah studies (Haydon, 1993; Watson *et al.*, 1998; Cornish and Vertessy, in review). These show the usual pattern of increased low, median and high flows in the immediate post-logging period and a recovery to pre-treatment levels once regeneration is established. In cases where old growth forest is replaced by vigorous regrowth, all flows are shown to decrease. There is some evidence for changes in the pattern of flow regime, but this is equivocal.

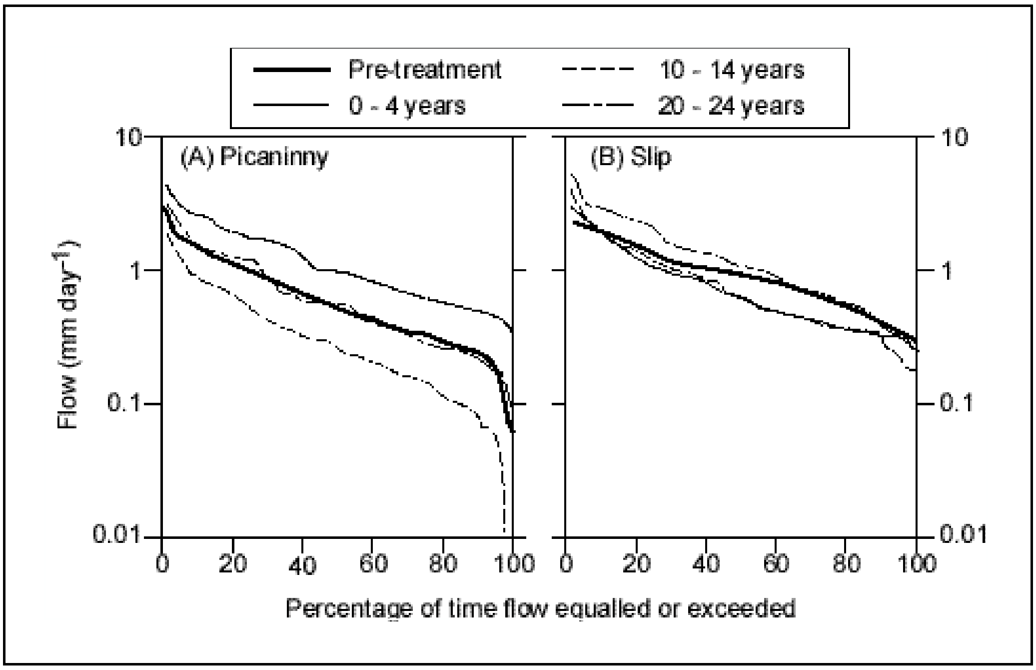
Haydon (1993) examined forest thinning-induced changes in flow regime of the Crotty Creek catchment, a mountain ash forested basin located near the Maroondah group. This 122 ha catchment consisted of 1939 regrowth and was thinned *over a six year period* to 50% of its basal area using a strip thinning pattern. Haydon (1993) compared average monthly flows from a three year pre-treatment period and a four year post-treatment period, commencing in the year after thinning had been completed (1985). As expected, he found that mean annual streamflow increased (by about 290 mm), and that flows of all magnitudes increased. However, he noted that the *rate* of change was not consistent across all months, and that a more uniform flow regime developed (Figure 2-15). Figure 2-15 shows that some of the 'wet' months (June-September) yielded a reduced share of annual streamflow immediately after thinning, whereas some of the 'dry' months (December-February) yielded an increased share. Haydon (1993) did not provide a convincing interpretation for this phenomenon, so further investigation of it is warranted.



- **Figure 2-15 Effect of a 50% basal area thinning on monthly streamflows in the Crotty Creek catchment, Victoria (data from Table 6.1 in Haydon 1993). To de-emphasise the effects of streamflow variability, monthly streamflow is expressed as a percentage of annual streamflow. Positive values denote an enhanced share of streamflow in the post-treatment period, whereas negative values denote a reduced share. The values shown are averages taken over a 3-4 year period.**

Watson *et al.* (1998) provided flow duration curves for three control catchments and five treated catchments in the mountain ash forested Maroondah basin in Victoria. For each catchment, separate curves were provided for the complete pre-treatment period and for multiple post-treatment 'blocks' of five year duration. In the 'wetter' Monda and Myrtle catchments (annual rainfall > 1600 mm), streamflows of all magnitudes increased uniformly immediately after forest clearance, then declined as regeneration commenced, again in unison. However, in the 'drier' Picaninny catchment (annual rainfall < 1200 mm) low flows were more severely reduced than median or high flows, particularly in the later stages of forest regeneration. To illustrate this finding, the flow duration curves for Picaninny, and it's control catchment Slip Creek, are presented in Figure 2-16. When assessing the flow regime changes at Picaninny, as illustrated in Figure 2-16 (a), it is necessary to compare them with rainfall-induced changes evident in the flow duration curves for Slip Creek, shown in Figure 2-16 (b). An important conclusion of the Watson *et al.* (1998) study was that the effects of inter-annual rainfall variability tended to overwhelm the subtle flow regime changes caused by forest clearance and regeneration.

Cornish and Vertessy (in review) showed that flows of all magnitudes increased in the period immediately following logging in the Karuah catchments. During this time, high flows tended to increase most, particularly in catchments that had thinner soils and higher levels of disturbance. By the time vigorous regrowth forest was established in the catchments, all flows had returned to pre-treatment levels, though low flows declined *below* pre-treatment levels in the catchments which experienced the greatest annual streamflow changes. However, no such low flow reductions were evident in the catchments with thin soils. Overall though, Cornish and Vertessy (in review) attributed most annual streamflow changes in the Karuah catchments to changes in baseflows.



■ **Figure 2-16 Flow duration curves for the Picaninny and Slip Creek catchments, Maroondah basin, Victoria (after Watson et al. 1998). Separate curves are shown for the pre-treatment period and discrete five-year blocks in the post-treatment period; only a sub-set of the original data are shown here for clarity. Slip Creek is an undisturbed control of old growth mountain ash forest. Picaninny was converted from old growth to regrowth mountain ash over 78% of its catchment area.**

There is no published evidence of forestry-induced flow regime changes for the Yambulla group of catchments, though Mackay and Cornish (1982) do compare storm hydrographs for undisturbed control catchments and catchments which were burnt and logged. They noted that peak flows and stormflow volumes increased as a consequence of high intensity burning, and were increased further after post-fire salvage logging of timber. They attribute these increases to reductions in ET, but also to reductions in soil infiltration capacity due to soil compaction by logging machinery.

2.9 Additional Factors which may affect Water Use by Trees

There are several additional factors which might influence the water use of trees, and which may result in unexpected hydrologic responses to forest disturbance at the catchment scale. Indeed, these factors help explain the high variability evident in pooled catchment treatment experiment data sets (Bosch and Hewlett, 1982; Stednick, 1996). Some such factors are described briefly below.

2.9.1 Insect predation and defoliation

Bethlahmy (1974) showed that bark beetle epidemics affected long-term streamflow records in two forested basins in Colorado, USA. Mean annual streamflows increased by 24 and 32% over a 25 year period, relative to an undisturbed catchment nearby. Watson *et al.* (1998) reported that insect attack of the forest canopy dramatically altered the streamflows in the Monda group of mountain ash forest catchments in the Maroondah basin. Separate attacks in 1988 and 1996 partly defoliated the forest and reduced ET rates, leading to increases in streamflow. These attacks occurred in the early life of a regenerating forest (the Monda group were clearfelled in 1977), altering expected patterns of forest growth and water balance dynamics in those catchments. Similar insect attacks were reported by Cornish and Vertessy (in review) for the Karuah group of catchments. These attacks reduced ET and significantly changed the expected response of two of the six treated experimental catchments in that group. In both the Karuah and Monda examples referred to here, the effects were transient, lasting between 2 and 5 years.

2.9.2 Variable forest regeneration

Some catchment treatment experiments are confounded by variation in vegetation response to disturbance. The vigour of forest regeneration is affected by soil conditions and the rainfall pattern following the forest disturbance event, so variability between catchments and through time can be expected. Watson *et al.* (1988) noted that poor regeneration of mountain ash forest in the Picaninny catchment delayed growth and altered streamflow response over the first few years following logging. Similarly, Cornish and Vertessy (in review) attributed inter-catchment differences in post-logging streamflow response to differences in the forest regeneration rate amongst the Karuah group of catchments.

2.9.3 Fire

A distinctive feature of Australian eucalypt forests is that their species composition and structure is controlled by fire. Fire frequency and intensity exert important influences on the extent and nature of forest regeneration, resulting in important hydrologic consequences. The Yambulla catchment experiments in the south east region of NSW were initially conceived to assess the impacts of logging. However, after some initial logging treatments, a wildfire occurred and affected most of the catchments. These fires altered the composition of the forest and, to a lesser extent, the soil hydraulic properties, of each catchment in a variable manner as the burn intensity and coverage was variable. Hence it is difficult to isolate the hydrologic effects of logging from fire in the Yambulla experiments.

2.9.4 Interaction with shallow groundwater

The response of catchments to Jarrah forest clearance in south west Australia is often complicated by interactions with shallow groundwater systems. As has been observed elsewhere, forest clearance in these catchments results in reduced ET and thus enhanced streamflow. However, many of these catchments are also coupled to shallow groundwater systems, and several studies have shown that this leads to an altered hydrologic response to forest disturbance (Ruprecht and Schofield, 1989; Ruprecht and Schofield, 1991; Ruprecht and Stoneman, 1993; Stoneman, 1993; Schofield, 1996). These studies show that groundwater recharge increases when ET is reduced and that, under certain circumstances, groundwaters can rise enough to reach the surface in valley bottoms. This has the effect of dramatically increasing the runoff producing area in these catchments, leading to increases in rainfall/runoff ratios which are proportionally greater than reported elsewhere. For instance, Ruprecht and Schofield (1989) showed that annual streamflow increased from 11 to 32% of annual rainfall over a 10 year period in a cleared Jarrah forest catchment. This rise was shown to be coincident with the expansion of the groundwater discharge area from 4 to 20% of the catchment area. Another feature of these catchments is that the time taken for a new hydrologic equilibrium to be reached is longer than normal as groundwater systems tend to adjust slowly (Ruprecht and Schofield, 1989; Stoneman, 1993).

2.10 Water Yield in the Otways

No direct research into water yield changes after harvesting has been carried out in the Otway forests. Moran (1988) considered many of the factors likely to influence water yield in these forests while adapting the Kuzcera (1985) model for local use. The model was applied in two catchments (East Barham and Upper Gellibrand) in which forest age was estimated into the future from forest stand information and planned harvesting regimes. Water yields were then assigned to these ages using a modified Kuzcera (1985) model. As this model was derived for Mountain ash forests around Maroondah, Moran (1988) listed various factors which were somewhat different in the Otways, and which could affect tree water use. Factors related to vegetation and forest characteristics included:-

- The presence of a larger mixed species forest component in the Otway Mountain ash forest association;
- The presence of substantial areas of relatively ‘unstocked’ Mountain ash forest which is likely to consist of a mix of age classes; and
- Slower growth rates in the Otway environment (by about 30% on average)

Moran (1988) analysed the previous streamflow and rainfall record for the two Otway catchments and concluded that they exhibited relatively higher flows than comparable catchments used by Kuzcera (1985) in wet years, and relatively lower flows in dry years. As a consequence the Otway catchments had a greater range of annual flows, consistent with a relatively smaller soil moisture store in the Otway environment. Higher runoff/rainfall ratios were also observed, and Moran (1988) suggested that they also indicated a lower soil moisture store, and lower growth rates with lower water use, in Otway forests. Mean annual long term rainfall was 1444 mm in the East Barham catchment, while it was 1582 mm in East Gellibrand. Equivalent long term streamflow means were 655 mm and 711 mm. Assuming minimal deep seepage losses, long term mean ET values were 789 mm and 871 mm respectively.

Moran (1988) concludes from the modelling exercise in the two catchments, in which 60 and 80 year rotations were incorporated, that future streamflow reductions were likely to be less than 20%, and probably of the order of 5% - 15%. An important observation was that the magnitude of treatment effects on streamflow would be highly dependent on the vegetation composition, and the age-class distribution of the forest, at the time of harvesting.

Brinkman (1988) used the estimates and approach of Moran (1988) to produce water yield/forest age tables for Mountain ash forest, ash/mixed eucalypt forest and softwood plantations in the Otway ranges. Yield estimates for cleared land were also made. The impacts of timber harvesting treatments were shown to be highly dependent on the vegetation composition in the catchment, and on the age class distribution of the various forest species at the time of timber harvest.

O'Shaughnessy et al. (1995) analysed the streamflow and rainfall record for the Lerderderg catchment, in a drier environment north of the Otways, and found no detectable trend in streamflow during a period in which forest harvesting was carried out in 16% of the catchment.

2.11 Conclusions

This review has explored the impacts of forestry on annual streamflows and flow regime, based on published case studies from the USA, South Africa, New Zealand and Australia. The studies reviewed spanned a variety of physical settings and covered a wide range of forestry activities. We conclude by listing the key generalisations which can be made from these case studies. These generalisations are qualitatively defined here as 'well established', 'supported by limited evidence' and 'speculative'.

Well established

- Mean annual evapotranspiration (ET) is higher for forests than for grasslands; the Holmes and Sinclair (1986) relationship provides a guide to comparative mean annual ET rates of grasslands and eucalypt forests for varying mean annual rainfall isohyets in south-eastern Australia.
- In south-eastern Australia, mean annual ET is higher in pine plantations than in native eucalypt forests, primarily due to differences in rainfall interception.
- In South Africa, mean annual ET is higher in eucalypt plantations than in pine plantations, primarily due to differences in growth rate.
- Afforestation of grasslands results in reductions in mean annual streamflow, low flows and high flows.
- Forest clearance results in increases in mean annual streamflow, low flows and high flows; these increases are transient if the forest is permitted to regenerate.
- Streamflow increases following forest logging are linearly proportional to the area of forest logged.
- In forests which are cleared but permitted to regenerate, streamflow increases usually peak in the first three years following treatment; streamflows normally return to pre-treatment levels between 4 and 10 years after disturbance, but may take as long as 25 years to recover.
- The streamflow impacts of forestry activities are amplified by increases in mean annual rainfall; absolute impacts are diminished in drier areas.
- Significant reductions in the leaf area index of forests (such as those resulting from insect predation) can measurably reduce ET and increase streamflow.

Supported by limited evidence

- Forest age affects ET rates, and thus streamflows, in moist eucalypt forests; old growth forests yield significantly more streamflow than regrowth forests of the same species, aged 20-30 years.
- In South Africa, plantation age affects ET rates, and thus streamflows, in pine plantations; streamflow reductions in *P. radiata* plantations tend to diminish after age 27 years.
- In South Africa, afforestation of grasslands reduces low flows relatively more than median and higher flows.
- Forest thinning has similar peak impacts on streamflows as forest clearance, provided that equivalent basal areas are affected; however, streamflow increases tend to persist longer for patch cuts than uniform thinning.
- ET rates in regenerating eucalypt forests are also dependent on moisture supply; therefore rates will be affected by rainfall amount and soil depth (soil moisture store).

Speculative

- Forest age affects ET rates in dry eucalypt forest, though there is no hydrometric evidence to show that streamflows reduce to below pre-harvesting levels.
- Transpiration per unit leaf area of forest declines as eucalypt forests age.

3. Water Quality: Literature Review

3.1 Preface

The following literature review is based upon that undertaken by Dr Jacky Croke and Dr Peter Hairsine as part of the ESFM Project: Water Quality and Quantity for the Upper and Lower North East, Southern RFA Regions (SKM, 1998). Further information, in particular that relating to the Otway region, was sourced from the Department of Natural Resources and Environment and members of the Otway Forest Hydrology Reference Group.

3.2 Introduction

This review restricts itself to the issue of 'water quality', and more specifically to our understanding of the potential impacts of native forest harvesting and roading on catchment water quality.

There are a number of preliminary definitions of scope required before we review the literature. The first pertains to the definition of 'water quality', which is most commonly assessed in terms of physio-chemical indices such as temperature, suspended sediment concentration, dissolved solids, nutrients and bacteria. These indices have traditionally been used to determine water quality for end uses such as domestic consumption. Today, however, there is much interest in preserving the stream ecosystem, and biological indicators are essential to assessing the "health" of ecosystems, and in quantifying impacts from forest management activities. Therefore a much broader definition of 'water quality' is now commonly employed. The National Water Quality Management Strategy (NWQMS) and the Monitoring River Health Program (MRH) were developed, in part, to address the disparity between definitions and definable parameters of water quality. The NWQMS aimed to provide a consistent national framework within which all stakeholders, from government agencies to local communities can contribute to better water quality management. It included a policy and principles document, and detailed sets of guidelines for maintaining and improving water quality. The MRH programs aims to develop a national approach to standards and guidelines and places considerable emphasis on biological and other indicators of water quality and river health. The debate about whether to use biological methods, chemical methods or a combination of both should centre on the objectives of the exercise. If ecological impact is to be minimised, however, it must be accepted that whenever possible, biological indicators should be used (CSIRO, 1992). There are few studies in Australia that use a combined chemical-biological index of water quality so this review is necessarily biased towards studies that use sediment loads and turbidity as a reference for water quality standards, as these two parameters are frequently affected by forestry activities.

The second aspect that requires clarification is the interpretation of literature findings with respect to judgements on the degree of impact. Some studies use terms such as 'significant', 'slight' or 'not perceptible' with respect to the observed impacts. Doeg and Koehn (1990) for example note "some increase in sediment accession to streams following logging, although some of these increases have been small". On the other hand, Cornish (1980, 1981, 1983, 1989a) concludes "in general, forestry operations do not have an adverse impact on stream turbidity levels" (Cornish, 1983). However, the reported values of turbidity in each of these data sets were actually higher in unlogged catchments than those that experienced some degree of logging, making this statement somewhat ambiguous. Quantifying changes in water quality and quantity, as significant, large or small is both subjective and highly dependent upon the accuracy of the measurement techniques employed. Interpretations of 'acceptable' changes to water quality may be quite different depending upon whether the catchment's water resources are for consumptive (eg. drinking water, irrigation and other commercial uses) or non-consumptive (eg. human recreation, maintenance of aquatic life) use. Similarly, the issue of water quantity is, understandably, of higher priority in catchments where streams are used for water supply purposes. Consequently, while some studies describe the effects of timber production on water quality and quantity as "adverse" or "minimal", these are based on a fairly subjective idea of what is acceptable for a given water use. Schofield’s (1996) simple tripartite classification is potentially more useful as a reference point defining the likely impacts of forest practices as;

- Low Impact: small or transient activities with no observable effects (eg tree fall, small patch logging, and low intensity selective logging).
- Medium Impact: transient activities with short to medium term effects but system recovery (eg intensive logging and regeneration, clearfelling and regeneration, slash and burn agriculture).
- High Impact: permanent changes with long term effects (eg partial or complete conversion to agriculture, permanent forest thinning, intensive production forestry).

Where possible we interpret the literature findings within the context of this classification scheme so that the reader can have a qualitative measure of our interpretation of impact across the range of studies.

The potentially adverse impact of forestry activities on soil and water values has been recognised in many Federal and State Government reports (Cameron and Henderson, 1979; DWR, 1989; OCE, 1988; LCC, 1990; RAC, 1992). The Resource Assessment Commission of Australia (RAC, 1992) concluded, however, that there is scant quantitative data available on the relationship between timber harvesting activities and environmental parameters such as soil erosion, water quality, and biota. Doeg and Koehn (1990), and most recently, Dargavel et al., (1995) have made detailed reviews of the literature. The latter provide a very comprehensive review of the issues surrounding ‘logging and water’ specifically for the south eastern seaboard of Australia. In particular, the review includes useful summaries of the major land use regimes (eg logging, silvicultural and transport systems) and the current processes of regional assessment and public policy. Rather than duplicate these reviews, we draw together key experimental and theoretical observations from previous studies and use these data to construct basic generalisations that are either ‘well-supported’ or ‘speculative’ with respect to the literature findings. This approach, used previously by Schofield (1996) and Croke (1999), allows a more realistic assessment of our ability to predict likely impacts for specific catchments or regions. This style of predictive management is now regarded as essential with increasing public awareness of the environmental issues surrounding timber harvesting, particularly native forests, and major land use impacts on water values.

3.3 Traditional Approaches to the Issue

Our current understanding of the impacts of forest harvesting practices on water quality stem largely from two research approaches which have been used to address this issue for over 30 years. The first approach centres on in-stream measurements of sediment concentrations and turbidity at the catchment outlet, usually using a paired-catchment approach monitored before and after the period of major disturbance. The second approach has centred on quantifying erosion rates on specific land elements such as roads, tracks and general harvesting areas (GHA) and using these data to construct sediment budgets and scaling approaches to assess changes in catchment water quality due to forest harvest disturbances.

3.3.1 Problems with these approaches

Both approaches have inherent problems, many of which are commonly overlooked in our interpretation of the research findings. The former catchment monitoring approach is commonly referred to as a 'black-box' (Walling, 1983) where data at the outlet is interpreted without any understanding of the relative contribution of sediment sources throughout the catchment. The main problem with this approach is the difficulty of 'finger-printing' the sediment, and determining whether the material measured at the outlet is in fact derived from forest harvesting activities, or the remobilisation of secondary sources and/or in-channel deposits. Increased turbidity at the catchment outlet may be related to increased channel erosion due to changes in the streams hydrograph and storm flow response after harvesting, and may not necessarily relate to increased hillslope erosion or delivery rates. The problems are particularly complex in mixed land use catchments where it is impossible to differentiate the relative contribution of material from the range of sediment sources. There is also the secondary problem of the accuracy of the suspended sediment data due to problems with event sampling, technical equipment failures and matching the rainfall record in the pre-logged and post-logging measurement periods. Bren (1990) suggests that both Australian and overseas catchment turbidity studies do not provide a clearly transferable message on sediment loads and forestry operations. This is reflected in the study of Olive and Rieger (1987) where an analysis of suspended sediment response patterns in the Yambulla catchments near Eden, NSW, showed "marked variability".

Doeg and Koehn (1990) also draw attention to the accuracy and reliability of reported values of suspended solids and turbidity in the literature. They suggest that poor sampling frequency means that suspended sediment peaks and total loads have probably been underestimated, bed-load levels have not been investigated, and deposited sediments have been ignored. Within the Australian context, the length of time over which studies need to be conducted in order to assess the effects of forestry operations on water quality is further cause for concern. Some overseas studies have shown that elevated levels of stream sediment and woody debris were still present 20 and 50 years after logging (Beschta, 1978; Webster et al., 1987; Andrus et al., 1988; Platts et al., 1989). Only one Australian experimental catchment has been studied for more than 30 years (MMBW, Coranderrk Experiments) with the majority investigated for periods of less than 15 years (Doeg and Koehn, 1990). Overall, short-term catchment monitoring studies are of limited value in understanding the magnitude of the disturbance or in pinpointing best options for remedial or preventative practices within the catchment.

The problems with the second, on-site erosion approach stems largely from the scale of analysis. Not all of the sediment eroded from a particular hillslope will be delivered to the stream. Many plots are too small to measure or quantify redistribution and storage processes. Quantifying this delivery ratio remains difficult because of the spatial and temporal variability in some of the controlling factors such as hillslope shape, soil type, vegetation, rainfall intensity and the degree of disturbance.

There have been recent advances, however, in the application of both approaches. The former has benefited considerably from the introduction of conservative sediment tracers to finger print the sources of sediment within a catchment and construct a sediment budget using tracer data. The second approach has been advanced by larger plot-scale studies where the physical processes of sediment storage and redistribution have been quantified, allowing a more accurate assessment of delivery rates and ultimately hillslope contributions. More importantly, recent advances in our understanding of sediment and water quality have led to the recognition of the data that are essential to more fully understand the issue of forest management impacts on water quality.

3.3.2 Essential information

In order to fully understand and quantify the impacts of forest practices on water quality we require a fundamental understanding of:

- The nature of sediment sources and their spatial distribution with respect to streams.

  Justification: not all parts of the disturbed forest generate sediment equally: it is essential therefore, to quantify the relative contributions from a range of sources and map their location with respect to streams.

- The nature of the delivery pattern from source to stream and potential for storage both on the hillslope, in erosion control structures and in near-stream areas.

  Justification: not all sediment eroded from a particular source will be delivered to the stream or catchment outlet. It is important to understand the nature of the delivery and storage patterns and determine the most potentially damaging delivery pathways with respect to in-stream water quality.

- The effectiveness of best management practices with respect to sediment production and delivery.

  Justification: the potential impact of forest harvesting practices on water quality may be reduced, and may become transient if best management practices are employed, and these are shown to be successful.

Data on these three key factors allow us to construct a more meaningful and accurate assessment of any likely impacts. The problem lies in acquiring data for a given region or catchment. Few studies in the literature address all of these issues. Taken as a whole, the studies are conducted in a range of forest environments of varying topographic and soil characteristics and predictive management relies heavily upon extrapolation of data from different scales and regions and the use of models to predict likely impacts. The following section reviews our understanding of these key variables and is followed by a brief discussion of predictive management. The review concludes by identifying areas where scientific uncertainty remain, broadly generalised knowledge is summarised and some suggestions are made for the future management of forested catchments.

3.4 Sediment Sources in Forests

3.4.1 Sediment sources

Many studies describe the effects of particular land uses on sediment yields without differentiating between sediment sources. Of all forestry operations, however, the literature both in Australia and overseas points consistently to road infrastructure as the most damaging effect on sediment accession to streams. Langford and O'Shaughnessy (1980) note that the prime source of sediment accession in their experimental coupes was roads and tracks. Langford et al., (1982) argue that the "small" increase in sediment concentrations in the Coranderrk catchments were due to "poor stream crossings" rather than direct logging activities per se. Kreik and O’Shaughnessy (1975) also detected the effects of pre-logging road construction on suspended solids and turbidity. Davies and Nelson (1993), investigating the impacts of sediment accession on fish and macro invertebrates, suggested that “sediment input from uncontrolled road crossings is considerable”. These roads, draining 6% of the catchment, were constructed some 30-50 years prior to the study but remain active sources of sediment through recreation use.

Reid and Dunne (1984) also found that the major source of sediment on logging roads was the pounding of the surface by heavy log trucks but once traffic stopped, unsealed forest roads stabilised and sediment loss from the road surface "greatly diminished". Sediment yields from logging roads have been well documented in studies from the USA, and in general, show a 2- to 50-fold increase over background levels (Reid, 1993). Sediment yields decrease rapidly after road use is discontinued and logged areas regenerate so yields measured more than 5 years after logging are usually less than five times background rates (Reid, 1993). Cornish (1999) and Cornish (in press) found that turbidity levels only increased after harvesting in those Karuah catchments with permanent roads. Only one of the three roaded catchments was still exhibiting elevated turbidity levels six years after harvesting.

Recent field experiments in selected forest management areas in southeastern Australia also highlighted the importance of temporary roads or forest snig tracks as important sources of surface sediment (Croke et al., 1997). Snig tracks in the highly erodible granite soils around Bombala, NSW, yielded sediment erosion rates of the order of 12 t/ha of track surface for a 100 y storm of 30-minute duration. These yields were almost a magnitude lower than those reported for more stable soils on the Ordovician soil types around the coastal forests near Bermagui where sediment transport processes were predominantly sheet flow (Croke et al., 1997; Croke et al., 1999a). Wilson (in press) also reports similar erosion rates on forest snig tracks on highly erodible sandy soils in Tasmania. Radionuclide tracers also confirmed snig tracks as net erosion areas with estimated yields in the order of 70 t/ha per year (Wallbrink et al., 1997). Similar experiments on a range of secondary access and log access tracks in Bermagui on Ordovician Metasediment soils produced yields of ~ 8t/ha for a 100 y storm of 30-minute duration on a stretch of forest road. The road is well used by heavy logging trucks and private access vehicles (Croke et al., unpublished data; Appendix 1).

Unsealed forest roads can represent between 1-5% of a managed forest, with tracks and temporary roads occupying larger areas between 5-10% of the catchment or logging compartment. In the eucalypt forests of southeastern Australia, for example, Mathews and Croke (1998) estimated that temporary roads and tracks formed 12-15 % of the 160 ha forest compartment. In plantation forests, the density of roading is considerably greater and the effects of this have been reflected in the magnitude of sediment production reported in some studies. Increased streamwater suspended sediment and turbidity have been associated in particular with the establishment and management of P. radiata plantations (Cornish, 1989b). Factors responsible for this include a high roading density, the frequent occurrence of streamside roading networks and a reliance on downhill snigging (Cornish, 1989b).

Grayson et al., (1993) attempted to separate the effects of road use and maintenance on water quality from logging operations. They found that annual sediment production from forest roads was in the range of 50-90 tonnes of sediment per hectare of road surface per year. The use of gravel and higher levels of road maintenance reduced sediment production. Conversely, when road maintenance was not increased, sediment production increased by approximately 40%. In discussing sediment generation on forest roads, Haydon et al. (1991) showed that unsealed forest roading for recreation can increase sediment generation by two orders of magnitude over undisturbed catchments (increasing from 0.3t/ha/yr to 30t/ha/yr). The intensity and duration of road use and the level of road maintenance influenced these effects. Numerous strategies have been developed to limit sediment production from forest roads and tracks, including revegetation, gravelling and regular maintenance (Haupt, 1959; Diseker and Richardson, 1962; Kidd, 1963; Dryness, 1970, 1975; Carr and Bullard, 1980; Cook and King, 1983; Burroughs et al., 1984; Kochenderfer and Helvey, 1987; Burroughs and King, 1989; Heede and King, 1990).

In contrast to the wealth of studies, mainly from the USA, reporting sediment production rates from forest roads, there are relatively few studies that compare the magnitude of sediment generation from other sources. Sediment production rates from forest snig tracks were compared with those from the general harvesting area for 13 sites in the Eden and East Gippsland Forest Management Areas (Croke et al., 1997). Field experiments revealed that sediment and runoff production on the GHA were several orders of magnitude lower than the snig tracks or forest roads (Croke et al., 1997; Croke et al., 1999b). Although partially disturbed during harvesting, the retention of a high degree of forest vegetation contributes to the lack of sediment transport in these areas. Lacey (2000) reported that 10 metre buffers of undisturbed forest were sufficient to remove 80-90% of runoff and over 95% of sediment produced on logging snig tracks in eastern NSW. Channelised flow is rarely present on the GHA limiting the ability of runoff to transport large amounts of sediment. Runoff production appears to be dominated by the bare or more disturbed areas, however, suggesting that if severe broad-scale disturbance occurred, sediment transport rates and soil losses on these areas would be significantly increased (Croke et al., 1997). Values of sediment movement reported in this study, which utilised large rainfall simulators, appear consistent with erosion rates reported for disturbed forests elsewhere (Dissmeyer and Stump, 1978; Ziegler and Giambelluca, 1997; Wilson, in press). It should be noted, however, that sediment yield estimates are more commonly reported in terms of annual averages, as compared with the event base values produced using the simulator studies. Rapid revegetation of disturbed areas is a feature of many Australian native forests, and this new surface cover can provide substantial protection from erosion. Cornish (1999) and Cornish (in press) report significantly reduced turbidity levels in rapidly revegetating catchments at Karuah in comparison with the previous old-growth forest. This was attributed to better surface cover in the regenerating forest, and the presence of high-energy gravity raindrops in the pre-treatment forest. A similar reduction was noted by Cornish and Binns (1987) in the Yambulla catchments after logging and wildfire, but the effect there was delayed by two years while the revegetation established.

The persistence of roads and tracks as significant sources has also been investigated in a number of studies (Megahan, 1974; Beschta, 1978; Megahan et al., 1983). The rainfall simulator experiments in the Eden and East Gippsland FMA (Croke et al., 1997) were conducted on snig tracks of varying age since logging. The results indicated that there is a temporal recovery both in terms of runoff and sediment production over a period of 5 years after logging where sediment production levels had declined on the snig tracks to levels comparable to a lightly disturbed GHA (Croke et al., 1997). The discontinued use of tracks between cutting cycles is seen as a significant factor in limiting sediment supply for transport. Similar periods of recovery have been reported on highly erodible forest roads (Megahan, 1974). Recovery was also observed on the GHA over this time frame of 5 years, though the degree of recovery was lesser, as the potential changes in soil and vegetation properties was much lesser. On roads the intensity of traffic-usage is also seen as a key factor in the persistence of these areas as a sediment source. Roads that are used infrequently but remain open to the public for recreation had generation rates almost one order of magnitude lower than those sections of road used frequently by both logging trucks and private traffic (Croke et al., unpublished data).

Compaction of the surface soil is commonly noted to be a persistent feature of soil disturbance during logging, particularly on snig tracks and landings (Rab, 1994, 1996). Some studies have reported that impacts persist for up to 30-50 years after logging (Greacen and Sands, 1980; Increate et al., 1987). In terms of sediment production, however, recovery times appear to be significantly shorter (of the order of 5 years) (Megahan, 1974; Reid, 1993; Croke et al., 1997). Factors that are likely to affect the rate of recovery are soil type, regeneration rates and the persistence of rills or gullies. Once formed rills and gullies are difficult to remediate and represent the continued persistence of concentrated flow paths over time.

3.4.2 Comparison with other land uses

Patric et al., (1984) contrasted sediment yields from forested watersheds (<5 km2) to sediment yields from watersheds with other land uses (primarily agriculture). In the eastern U.S. the average annual sediment yield for managed and unmanaged forested watersheds was about 0.15 mg/ha as compared to 0.35 mg/ha for the other watersheds. In the western, forested watersheds had a mean sediment yield of 0.15 mg ha-1 yr-1 as compared to 0.42 mg ha-1 yr-1 for other land uses. Patric et al., (1984) also compared the concentrations of suspended sediment in major rivers draining mostly forested areas with rivers draining areas with other land uses. The concentration of suspended sediment was about 10 times greater in rivers draining non-forested areas. Yoho (1980) compiled erosion rates for a variety of land uses on small watersheds in the south of the USA. Intact pine forests yielded the lowest quantities of sediment (from 0 to 0.2 mg ha-1 yr-1), with carefully clearcut forests (applying BMPs) yielding only moderately more (0.1 to 0.4 mg ha-1 yr-1) over one to several years. Pastures produced 0.9-4.5 mg ha-1 yr-1, and carefully cultivated agricultural fields were found to yield from 0.9 to 16.8 mg ha-1 yr-1. Harvesting and site preparation without use of BMPs yielded between 3 and 14 mg ha-1 yr-1 for one to several years, temporally matching the erosion rate from carefully cultivated fields.

3.4.3 Broad generalisations

Well Established

- Unsealed forest roads are the major sources of sediment in managed forests.
- Road usage is a critical factor in explaining sediment production rates on roads.
- Sediment yields from forested (managed and un-managed) watersheds are considerably lower than those from other land uses, particularly agriculture.
- Sediment production rates on roads and tracks decline within the time frame of 2- to 5 years.

Limited Evidence:

- The GHA is not a significant source of sediment due to limited sediment availability, high retention of vegetation cover and spatially variable infiltration rates.
- Rapid and extensive revegetation of disturbed areas after logging can limit, or entirely eliminate, the GHA as a sediment source.

Speculative:

- Hillslope disturbances during logging result in significant post-logging changes in stream turbidity at the catchment outlet.

3.5 Nutrient Sources in Forests

A detailed review of nutrient production and potential losses due to such factors as fire, soil disturbance and erosion is presented in Attiwill and Leeper (1987). In general terms, nutrients in the forest ecosystem are contained within the living biomass of plant and animal life, within dead organic matter and the soil, and are cycled through this soil-plant-litter subsystem. Inputs to the nutrient pool come from weathering of the soil parent material or from the atmosphere. Atmospheric inputs include fixation of atmospheric gases (eg nitrogen fixation), aerosols (particles suspended on air) or in rain or other precipitation. Inputs from parent material can be considered to be very slow. The two most dominant macronutrients in our soils are nitrogen and phosphorous. Most of the nitrogen in surface soils is associated with organic matter. The amount of nitrogen in the form of soluble ammonium and nitrate compounds is seldom more than 1-2% of the total present, except where large applications of inorganic fertilisers have been made. Organic nitrogen is largely protected from loss but largely unavailable to higher plants. This process of tying up nitrogen in organic forms is called immobilisation; its slow release, specifically organic to inorganic conversion, is called mineralisation. Only a small proportion of total soil N (<5%) is readily mineralisable, and this is recycled through litter decomposition with turn over rates of between 3-5 years (Attiwill and Leeper, 1987). However a much larger proportion of total soil N, with slow turn over rates, is available for plant uptake over several rotations. These long term reserves of N are estimated at 40% of total soil N (Hopmans et al., 1993). Inorganic nitrogen is most susceptible to loss from soils by leaching and volatilization. Phosphorous is also mostly present in organic forms in forest soils (Kelly and Turner, 1978) or is temporally converted to organic forms in the early stages of litter decomposition by microbial activity (Harrison, 1988). There are two important pathways for the export of nutrients from areas of forestry operations: the solution pathway is where nutrients moved dissolved in water often dominantly by sub-surface water flow, and the sorbed pathway is where nutrients are attached to sediment. This second pathway becomes significant where soil erosion is present.

The main factor affecting nutrient depletion is the overall distribution of nutrients within the soil profile. Soils with large reserves of nutrients held in lower layers (eg basaltic soils) are much less susceptible to nutrient depletion than soils with poor reserves of subsoil nutrients (eg soils formed on quartzite) (Turner and Lambert, 1986). With forest ecosystems highly dependent upon litter fall processes, it is not surprising that most available nutrients are concentrated on the soil surface and can be directly related to measures of organic matter content.

Following the site exposure that accompanies logging, some forest soils experience an accelerated release of certain ions into the solution pathway from the mineralisation of organic matter and from mineral weathering (Aubertin and Patric, 1974). The extent to which those ions released by exposure are removed from the site by leaching to streams is a function of the uptake of those ions by vegetation and the ion exchange properties of the soil. A number of studies have observed increased leaching of dissolved nutrients from logging slash (Likens et al., 1970; Hewlett et al., 1984; Meyer and Tate, 1983). Hopmans et al., (1987) found that export of nutrients and suspended solids were significantly higher because of increased discharge following clearing as a consequence of increased water exported from the catchment. The impact of harvesting practices on nutrient losses have focussed primarily on the relationship with prescribed fires and regeneration burns. Increased concentrations of nutrients in streams draining logged catchments subject to prescribed fires are primarily the result of direct precipitation of ash into the stream (Spencer and Hauer, 1991); overland flow that has been in contact with ash (Grier and Cole, 1971); and nutrient transport from groundwaters after leaching (Grier and Cole, 1971).

There are few studies that investigate the level of nutrient losses associated specifically with surface erosion by overland flow. Croke et al., (1997) examined representative concentrations of total nitrogen (TN) and phosphorous (TP), and their associated forms such as nitrite (N02), nitrate (N03) and ammonia (NH4) and reactive (P04 2) and dissolved phosphorous, transported via overland flow from the nine disturbed sites of vary age and soil type. Nutrient concentrations on the sites were relatively low with mean concentrations in the order of 2.7 mg/L and 0.42 mg/L for TN and TP respectively. Nutrient loads varied according to the degree of site disturbance and were significantly higher on the snig track elements compared to the GHA. Net export of TN from the plots averaged about 75, 32 and 9 mg/m2 for sites of varying age (0, 1 and 5yrs post burning) using a series of rainfall simulator events as described in Croke et al., (1997). This value represents total nitrogen loads leaving a 300 m2 hillslope plot and reflects the effects of redistribution and storage processes over this area.

Forest fertilisation is commonly regarded as a significant factor in explaining increased nutrient concentrations in streams draining managed forests. Most studies in the USA have shown that these increases are too small to degrade water quality (Binkley and Brown, 1993). A few exceptional cases have been reported where nitrate concentrations in excess of 10 mg-N/L were recorded several months after careful application of fertilisers in the Fernow Experimental Forest in West Virginia (Kochenderfer and Aubertin, 1975; Helvey et al., 1989, Edwards, 1991). Fertilisation is a routine practice on many intensively managed pine forests in the Southeast of the United States of America but few studies have examined fertilisation effects on water quality (Shepard, 1994). In the pacific North West several dozen forest fertilisation studies (Fredriksen et al., 1975; Meehan et al., 1975, Tiedemann et al., 1978, Bisson 1982, 1988, Hetherington, 1985, Bisson et al., 1992a) found nitrate concentrations well below the drinking water standard. A few harvesting studies have shown slight increases in phosphate concentrations after logging (e.g. Salminen and Beschta, 1991) but these increases were far too small to degrade water quality, “although some increase in stream productivity may have resulted”. This appears to summarise the potential effects of increased nutrients on fish populations and the literature in the USA contains no examples of damage to fish populations from nutrient concentrations following harvesting or fertilisation. The effects of fertiliser application in Australian pine plantations have not been reported. Recent work initiated in the Croppers Creek catchment in Victoria aim to address this issue although not quantitative data has emerged from the project to date.

Some mention to the difficulties of measuring nutrient concentrations in water draining forested catchments has also been made. The measurement of dissolved solids and nutrients may be plagued with similar problems to the measurement of sediment concentrations, particularly with regard to the relationship between concentration and discharge. A water-sampling program intended to investigate suspended sediment or nutrient loads and based on regular monthly or weekly samples may be ineffective in detecting changes in nutrient concentration and load due to forestry operations (Hart, 1982; Campbell, 1982, 1986). In addition, there is a growing awareness that the traditional total phosphorus and soluble reactive phosphorus parameters incorrectly characterise nutrient flux in streams and as such, are poor estimators of bio-available phosphorus in flowing water systems (Hart, 1982).

3.5.1 Comparison with other land uses

Omernik (1977) estimated the concentrations of total N and P in streams draining large areas of differing land use. Streams draining forested areas had concentrations of N (0.6 mg/L) and P (0.02 mg/L) that were an order of magnitude lower than streams draining agricultural areas (5.4 mg N/L and 0.2 mg P/L). Nitrate-N was also lowest in streams draining forests. A summary of water quality from the USGS’s Hydrologic Bench Mark Streams showed that “natural” watersheds (primarily managed and unmanaged forest) averaged 0.06 mg N03-N/L compared with 0.3 mg-N03-N/L for other streams (Biesecker and Leifeste, 1975). Most forest harvesting studies in the United States have documented increased concentrations of nitrate following harvesting (Binkley and Brown, 1993), but in almost all cases these increases have remained well below the 10 mg-N/L drinking water standard. Two notable exceptions include the high concentrations (average 5 mg-N/L) of nitrate observed in waters draining from high elevation forests in the Southern Appalachian Mountains (Silsbee and Larson, 1982). Factors contributing to elevated nitrate concentrations may include high rates of atmospheric nitrogen deposition and low rates of nitrogen uptake by the forest which may be affected by harvesting and by changes in vigour that have been attributed to stand maturation or regional air pollution.

3.5.2 Broad generalisations

Well Established

- Nutrient concentrations in streams draining forested catchments are considerably lower than those reported for other land uses, primarily agriculture.
- The dominant cause of increased nutrients in forest streams, if observed, is due to the effects of prescribed burning and wildfire.
- Observed impacts are short-lived and transient with no long-term effect.

Limited Evidence

- Fertiliser applications cause no increase in nutrient concentrations of streams draining managed forests.
- Fertiliser applications have no impacts on other values such as stream productivity, fish populations.

Speculative

- Timber harvesting activities significantly affect nutrient exports measured using a discharge–concentration relationship at the catchment outlet.

3.6 Sediment Delivery Patterns and the Potential for Storage Within the Catchment

3.6.1 Pattern and process in managed forests

Once sediment is generated, the portion of sediment delivered to the stream is the key variable that is influenced by both environmental attributes and management inputs. Although surface erosion rates are widely measured, much less is known of sediment delivery rates to channel networks. A common, but erroneous, perception is that all sediment that is eroded is delivered to the stream. Many predictive approaches have combined empirically-derived or plot erosion rates with arbitrary sediment delivery ratios, resulting in over-predictions of likely in-stream responses.

Forest environments are relatively unique in their ability to store sediment due to the retention of a relatively high percentage of vegetation. Even in a disturbed state there is a high degree of material such as trash, litter, stones and fallen logs that remain on the ground surface providing a cover for both the removal of soil particles from the exposed soil and their delivery downslope. This fact has been formally recognised by the inclusion of a cover-management factor in the Universal Soil Loss Equation (USLE) as applied in forestry environments (Dissmeyer and Foster, 1980). Williams (1975) also incorporated a sediment delivery term into the USLE and Tollner et al., (1976) developed an equation for estimating deposition from sheet flow as a function of flow character, vegetation character and transport distance through vegetation. Heede et al., (1988) examined the role of vegetation recovery after a chaparral fire in controlling the timing and rates of sediment delivery to streams and thus, in controlling the timing and location of channel adjustments.

Each sediment source is likely to have its own very specific delivery pattern dependent upon its spatial distribution within the catchment and the management practices employed. Forest roads for example have a very specific delivery pattern determined by the arrangement and location of drainage structures such as culverts and mitre drains within the catchment. Runoff from road surfaces is commonly discharged at concentrated outlets onto the adjacent hillslope via a network of culverts or drains. In contrast, drainage of temporary roads or forest snig tracks is a more dispersive pattern resulting in the redistribution of runoff and sediment via a network of cross drains. A number of studies have now illustrated the potential significance of concentrated paths at road outlets with respect to channel initiation and severe gullying. Montgommery (1994) for example estimated that gullying along ridge-top roads in a catchment in Oregon resulted in a 23% increase in the natural drainage density. Wemple et al., (1996) also observed a significant increase in the natural drainage density due to road construction.

Gully initiation at road outlets has also been recorded in the Cuttagee Creek catchment around Bermagui in the south eastern part of NSW (Mockler and Croke, 1999). In this study, gully initiation resulted in a 6% increase in the natural drainage density of the catchment over a period of approximately 30 years. The study also indicated that the contributing area to the drain outlet along with the slope of the hillslope at the discharge point were significant factors in explaining gullied pathways. The threshold relationship identified between these two variables provides a robust method of preventing gully initiation at road outlets. The study also recognised a clear relationship between gully initiation and the type of drainage structure used (eg mitre drain or culvert) with most gullying associated with culvert pipes on steep hillsides. Road culverts have a restricted use in forest roads in that they are only used where and when other drainage structures cannot be used- that is to drain a road cut into the hillside. Significant gully initiation was also observed to result from soil disturbance during pine (Pinus Radiata) plantation harvesting in Bombala where the placement of windrows was also recognised to concentrate runoff into certain parts of the landscape increasing the likelihood of channel initiation and gullying (Prosser and Soufi, in press). Channelised flow paths form a very efficient conduit for the delivery of sediment and nutrient to streams- once gullies are formed they are also very difficult to remediate so prevention is the best option.

Davies and Nelson (1993) recently illustrated that fine sediment input to these ephemeral, first order streams, such as those formed due to gullying at road outlets, is significantly enhanced by logging on steep slopes, by factors of two to three times the median values for unlogged streams. These small ephemeral streams are often treated in much the same way as the rest of the coupe, but have a more significant role in the export of fine sediment to downstream sites (Davies and Nelson, 1993). Recent changes to the Victorian Code of Practice have afforded these areas greater protection with respect to sediment delivery. The steeper slopes and potentially larger contributing areas of first order streams means that they play a significant role in the delivery of sediment and nutrient downstream. Duncan et al., (1987) for example found that over 50% of the fine-grained material delivered to steep first order streams reached sites downstream. This summarises the relationship between the size of the transported material and the delivery ratio. Many studies report an inverse relationship between the percent delivered and the size of entrained sediment, reflecting processes of preferential erosion and deposition at various spatial and temporal scales throughout the catchment (Walling, 1983). Croke et al., (1997) also report a relationship between the size of the eroded material and sediment delivery at cross banks draining forest snig tracks in southeastern NSW. Similar conclusions have been identified in a series of rainfall simulator field experiments on the effectiveness of riparian buffer strips in trapping sediment of varying size (Pearce et al., 1998).

The ability of the hillslope to absorb sediment and runoff and thereby control sediment movement downslope is dependent upon the specific topographic, soil and vegetation characteristics of the hillslope. Hillslopes that are disturbed during harvesting may represent an additional sediment source, potentially limiting the ability of the area to store sediment. The period that the GHA is an additional sediment source will depend on the rate and extent of revegetation (Cornish, 1999; in press). Much of the recent literature regarding sediment delivery from hillslopes centres around the effectiveness of the riparian or buffer vegetation in trapping sediment prior to entering the stream and this is discussed in more detail in the following sections.

Sediment delivery in large watersheds has also been correlated with morphological factors. Roehl (1962) for example found that the proportion of sediment eroded on the hillslopes that arrives at a watershed outlet decreases with increasing watershed area and channel length, and increases with increasing relief ratio. This relation implies that some sediment is lost in transport and may reflect lowland aggradation or chemical dissolution during transport and storage. Khanbilvardi and Rogowski (1984) and Novotny and Chesters (1989) reviewed methods of estimating delivery ratios on the scale of plots and hillslopes.

3.6.2 Broad Generalisations

Well Established

- Channelised pathways forming at road drainage outlets form the most efficient conduit for sediment and nutrients.
- Sediment delivery ratios are closely associated with the size composition of the in-situ and eroded soil.

Limited Evidence

- The interaction between factors such as slope, runoff, and morphological factors determines sediment delivery ratios for both the hillslope and catchment.

Speculative

- Data sets constructed from empirical relationships or plot erosion data that do not accommodate for processes of deposition and storage within a land element. These data are likely to over estimate the hillslope contribution and consequently the magnitude of catchment response

3.7 Effectiveness of Best Management Practices in Protecting Water Quality

3.7.1 Best Management Practices Employed In Forests

Several methods are used in forestry operations to mitigate the impact of logging on streams. These include the use of riparian buffer strips of varying widths, patch harvesting, siting and design of roads and road crossings to minimise sediment inputs, and restrictions to logging activities in relation to coupe slope and soil type. There are a number of studies in the literature which investigate the effect of some or all of these prescriptions in protecting soil and water values. An early, but relevant, example of this type of study is that of Hornbeck and Reinhart (1964), who examined the effects of prescriptive measures (eg bars across snig tracks, a ban on stream crossing and the location of roads away from streams) on sediment concentration in the United States. Suspended concentration varied from 56,000 mg/L where no prescriptions were employed to 15 mg/L when all of the above were imposed. Lynch et al., (1975, 1985) also report the effectiveness of BMPs in minimising impacts on sediment concentrations in Pennsylvania where only half of the watershed was harvested, 30-m buffer strips were retained along streams, the locations of roads and tracks were determined in advance and all roads and trails were rehabilitated after logging. Sediment concentrations in the first year after harvest averaged 1.7 mg/L for the control watershed and 5.9 mg/L for the harvested. Sediment concentrations remained slightly elevated above the control watershed for about 10 years and the researchers concluded that while the implementation of BMPs did not completely prevent impacts, the impacts were relatively small and of no direct concern to water quality standards (Lynch et al., 1975; 1985).

Grayson et al. (1993) found that applying a strict enforcement of Code prescriptions (eg suspension of logging during wet weather, protection of runoff producing areas with buffer strips, and the management of runoff from roads, snig tracks and log landings) eliminated intrusion of contaminated runoff into the streams, thereby avoiding the adverse effects of logging. Karr and Schlosser, (1978) also illustrated that while unmitigated clearcutting over a period of years doubled suspended sediment concentrations in runoff and increased nitrate levels by a factor of four, clearcutting with the retention of buffer strips caused only a 50% increase in suspended sediments and had no effect on nitrate levels.

Recent field experiments on snig tracks in the Eden and East Gippsland Forest Management Areas also highlighted the importance of track rehabilitation and drainage after logging. The field experiments suggested that around 50-60% of eroded material is stored in cross banks draining forest snig tracks. The deposition rate was inversely related to the percentage fine material eroded from the snig track, highlighting the relative ineffectiveness of these features in trapping fine material (Croke et al., 1997; Croke et al., 1999a). Given our concern with fine-grained material and their ability to transport adsorbed nutrients, great care should be taken to avoid exposing highly dispersive clay subsoils during cross bank construction. The predominant purpose of these features is to reduce the local catchment area so that the discharge plume at the outlet of the cross bank is minimised. Croke et al., (1999b) also reported that the practice of redistributing runoff at cross bank outlets after logging was a successful method of reducing the potential contribution of water and sediment to streams, particularly during small to medium rainfall events (2-10 years recurrence intervals). During the more extreme events of a 100 y storm, both the hillslope and snig track are generating runoff and the ability of the hillslope to absorb excess runoff is reduced. The spacing of cross banks becomes critical under these conditions, where the length of the discharge plume must not exceed the designated cross bank spacing. Field surveys on forest roads also highlight the importance of adequate drainage and drain installation (Mockler and Croke, 1999). Smith and O'Shaughnessy (1998) also report the effectiveness of obstacles or flow divergence structures at road drainage outlets in significantly reducing sediment delivery to streams.

Because of the importance of buffer strips as a prescriptive measure in stream protection, the following section provides a more detailed review of the literature relating to their role, width and placement in forest management strategies.

3.7.2 Buffer Strips

The use of vegetated buffer strips as a method of controlling sediment accession to streams has been recognised and accepted in Australian forest operations for more than two decades. Vegetation in these buffer zones generally comprises that existing prior to logging operations and is retained with the objective of protecting drainage lines and streams. Clinnick (1985) reviewed 'Buffer strip management in forest operations', paying particular attention to their effectiveness as a physical barrier to the transport of displaced soil from roads and forest harvesting areas. Vol Norris's (1993) review of buffer strips placed greater emphasis on their potential for removing pollutants from surface runoff in both forested and agricultural environments. The most recent review of buffer strip management in controlling waterway pollution has been undertaken by Barling and Moore (1992, 1994) as part of the Land and Water Resources Research and Development Corporation's program initiative for the study of riparian lands.

The effectiveness of buffer strips in protecting water quality is outlined in the case-study results of Karr and Schlosser (1978), Lynch et al. (1985), Aubertin and Patric (1974); Martin and Pierce (1980) and Borg et al. (1988). Further examples are provided, mostly from overseas work, in the reviews of Clinnick (1985), Vol Norris (1993) and Barling and Moore (1994). Cornish (in press) reports that turbidity levels in logged, but unroaded, Karuah catchments did not increase after harvesting because of the effectiveness of BMPs, including 20m buffer strips that remained unburnt. Cornish (1989b) suggests, however, that there are numerous problems with the provision of streamside buffer strips in P. radiata plantations including the problems of wind throw in the later years of the rotation and management problems on what type of vegetation to encourage in the strip in the second rotation. In addition, it is not always possible to protect buffer strips from being burnt in these plantations where fire is commonly used as a tool in the establishment and re-establishment of P. radiata.

There is still considerable confusion and ambiguity regarding both the placement and width of buffer strips in catchments. For forested systems, there are two possible approaches for locating buffer strips; one based on determining appropriate sediment transport distances through the buffer strip and the other that attempts to protect runoff-generating areas in the landscape. In the case of the former, a 30 m buffer is typically regarded as effective in trapping most of the sediment from cleared areas, although absolute width is dependent upon specific site conditions (Clinnick, 1985; Barling and Moore, 1994). All the available literature on adequate buffer strip widths are, nonetheless, site-specific (Trimble and Sartz, 1957; Packer, 1967; van

Groenewoud, 1977; Corbett et al. 1978; Cameron and Henderson, 1979; Chalmers, 1979; Graynoth, 1979; Bren and Turner, 1980; Borg et al. 1988). There is little or no data, which can be used to predict acceptable buffer widths under variable catchment characteristics. In one of the few Australian studies that examined the question of buffer widths, Borg et al. (1988) found that halving the buffer strip widths from 200 m to 100 m and from 100 m to 50 m had little if any detrimental effect on water quality. Their complete removal, however, led to changes in the stream channel profile and to algal blooms. In a recent study, Davies and Nelson (1994) examined the impacts of forest logging on in-stream habitat, fish and macro invertebrate populations in Tasmania and related the observed impacts to the width of the riparian buffer strip at each site. Their results are among the first in Australia to quantify changes in water quality due to altered buffer strip widths. Their conclusions state clearly that "all impacts of logging were significant only at buffer widths of less than 30 m".

In an attempt to construct a more formal method of determining extensions to the minimum streamside reserve width, Doeg and Koehn (1994) formulated a 'Decision Support System' to determine buffer width extensions at the coupe level. This system requires the user to input certain data related to the physical characteristics of the forest coupe to be harvested and is designed for simple computerisation. Unfortunately, this system of checks and questions suffers from a lack of scientific data to validate the recommended extensions and is clearly too detailed to be included directly into a Code of forest practices. Herron and Hairsine (1998) also examined a scheme for evaluating the effectiveness of riparian zones in reducing overland flow to streams and suggested that a riparian zone width not exceeding 20% of the hillslope length is a practical management option. They also concluded that buffer zones need to be distributed around the stream network where upslope sediment sources exist, if riparian buffers of realistic widths are to be effective. Some states, such as New South Wales, have constructed 'windows' outlining a range of buffer widths for varying soil, slope and erodibility classes.

In relation to protection based on runoff generating area, Cameron and Henderson (1979) recommended that buffer strips be required where the catchment area exceeds 100 ha and should extend along the entire length of the stream. In general, it is agreed that buffer strips should extend to the springhead or runoff confluence point of any sub-catchment and should be well upstream of any existing channel or streambed, since flow will occur at a higher point in the catchment once the forest has been cleared (Bren and Turner, 1980; O'Loughlin et al., 1989; Finlayson and Wong, 1982). The factors that affect the location of variable source areas of runoff generation include: soil characteristics; topography; vegetation and weather (Moore et al., 1991). Saturated source areas exist whenever the accumulated drainage flux from upslope exceeds the product of the soil transmissivity and the local slope (O'Loughlin, 1981). It is crucial when defining buffer strips in the field that all sources of runoff generation are included within the buffer strip zone. It is essential to incorporate the 'saturated zone', which is the area along the stream or drainage line that is permanently saturated (eg swampy ground) or becomes saturated (eg seepage area) with the onset of rain (O'Loughlin, 1981). Likewise, for buffer strips to provide protection during peak flow situations, Cornish (1975) proposed that catchment area, not stream permanence, be used as the criterion for defining the provision of buffer strips.

A number of equations have been formulated to determine where this condition applies in complex landscapes (Beven and Kirkby, 1977, 1979; O'Loughlin 1986;

O'Loughlin et al., 1989; Moore et al., 1993). The equation of Moore et al., (1993), essentially a refinement of previous equations, is designed to account for spatially variable evapotranspiration, deep drainage and vegetation characteristics. There are no empirical studies that have applied these equations to examining the impact of forestry operations on water quality. Bren (1998; 2000) defined a buffer loading for any stream reach as the contributing watershed area per unit area of buffer, and argued that to achieve a constant buffer loading throughout a catchment buffers needed to be wider in up-stream convergent areas and narrower in downstream divergent areas. The current system of constant buffer width was shown to under-protect lower order streams and over-protect higher order streams. Bren (2000) shows that the adoption of rigorous rules to compute constant loading buffer widths is unlikely to be acceptable in practice, and suggests that a modification of the fixed buffer width system that increases stream protection in converging areas and decreases protection in divergences, is the best method of assigning buffer widths.

Increasing the width of buffers has a marked impact on the area of forest available for harvest. Bren (1995) showed that in the Tarago catchment in eastern Victoria the area of land occupied by buffers increased substantially with increasing width of buffer. Buffers of 100m width entrapped large areas of forest that became effectively inaccessible, while 10m buffers produced a very complex pattern of buffer boundaries. Bren (1997) extended this study with the inclusion of forest value lost in buffers. Typically 50% of the loggable land was removed by a 90m buffer, while 50% of the commercial value was lost by an 85m buffer. In general the coupe value reduced at a slightly greater rate than available resource area as buffer width increased, reflecting the location of higher value forest near streams.

3.7.3 Broad Generalisations

Well Established

- BMPs play a significant role in the reduction of adverse effects in forested catchments.
- Forest buffer strips are an effective measure in reducing the volume of surface runoff and the quantity of sediment/nutrients delivered to a stream.

Limited Evidence

- The best location and design of buffer strips in forested catchments is known for a range of topographies and land uses.
- The specific role and effectiveness of hillslope, versus near-stream, BMPs are understood.

3.8 Predictive Management

Many forest managers and environmental protection agencies now perceive the need for a quantitative method of predicting the likely impact of major land use changes on water quality. There are essentially two approaches to predictive management in common use, extrapolation and modelling (Schofield, 1997). Both approaches are poorly developed in managed forests. The most commonly used method of erosion prediction in both agricultural and forestry environments has been the Universal Soil Loss Equation (USLE) or modified and hybrid forms of the relationship (see Hairsine and Hooke, 1993 for review). Despite modifications to the original formulae for forestry environments (Dissmeyer and Foster, 1980; Rosewell, 1993) there are some inherent problems with the USLE and its use as an erosion hazard prediction tool in these environments. The main problems lie in the exclusion of gullying or mass movement processes; deposition and redistribution processes and the fact that the slope length and steepness factors must be determined only on the area that is contributing runoff. The major limitations of all existing methods of erosion hazard prediction were summarised by Hairsine and Hooke (1993). These include the fact that:

- None of the methods assess the relative importance of different forms of erosion and the user is left to decide what the relative significance of gully, sheet or rill erosion is within a particular land area or region.
- Thresholds are not usually considered.
- There is a lack of dynamic components, including relative capacity to regenerate vegetation and redevelop soil properties.
- The quality of the input data limits the quality of all predictions. It is the reproduction of erosion processes represented by the method that largely determines its utility in predicting erosion, and in developing appropriate management strategies. In the absence of a good thorough understanding of the physical processes of sediment transport and delivery, the proliferation of interpretive software such as geographical information systems (GIS) will not improve the quality of predictions.

Ryan et al. (1998) have reviewed the approaches taken throughout Australia to the assessment of soil erosion hazard in forests. Victoria has developed an Erosion Hazard Assessment which is carried out in conjunction with the application of the legislated Code of Forest practice (CNR, 1995a; 1995b; NRE, 1996). The assessment scheme has two components:

- A broadscale categorisation of inherent soil erosion hazard
- An assessment of soil erosion hazard at the local (coupe) scale

The broadscale categorisation of inherent hazard is inferred from Land Systems mapping and associated surveys designed to provide a broad initial coverage of the State of Victoria. The local scale assessment is carried out on each coupe and is the product of two processes, an assessment of inherent soil properties (soil erodibility and infiltration and drainage characteristics) and soil erosion sit factor rating (incorporating sediment mobilisation and transport). This methodology is new, only partially field tested and not yet in widespread use. Nevertheless it is expected to be in general use with a standard methodology within the next year. Assessments will feed into the development of locally applicable forest management prescriptions (locally formulated BMPs).

The lack of a clearly structured framework within which to assess the potential impacts of forest harvesting practices on water quality has seriously limited the usefulness of predictive management in the past. In the USA, there is a greater emphasis on what it is the community wants to protect, and what must be done to ensure protection. Examples of community driven issues are channel destabilisation, increased peak flows, channel sedimentation and removal of stream-side vegetation. Many studies in the United States are specifically designed to examine potential impacts on fish species and ecological habitats (eg many rivers in the Pacific North west of the United States are of high ecological value with respect to salmon breeding or endangered fish species). In Australia, aside from water-drinking standards, there has been little emphasis on thresholds of concern (TOC) or more fundamental questions such as:

- what are the key areas requiring protection ?
- what is the capacity of the system to withstand change ?

In the absence of a clearly defined framework for water quality protection, there has been much emphasis (perhaps over emphasis) on sediment production, which in turn is commonly used as a surrogate measure for stream suspended sediment levels, turbidity and ultimately in-stream water quality. This occurs in spite of extensive evidence from the literature that the waters draining forested catchments (Harvested or not) is of a higher standard than those in any other catchment land use (NSCAI, 1994).

3.9 Forestry and Water Quality in the Otways

3.9.1 Relevant Studies and Reports

Because of the importance of the Otways to the supply of water for domestic consumption, there has been concern over the potential effects that forestry activities may have on water quality for many years. A report examining the potential impacts of pulpwood harvesting in the Otways (VIC, 1982) indicated that at the time turbidity levels in some streams such as the Gellibrand River were high, and that E.coli counts were high in all Otway water supply offtakes. This report suggested that some deterioration in water quality would follow an expanded harvesting regime, but that this deterioration would be minimal if logging areas were not increased and appropriate management prescriptions were employed.

Farrell and Novotny (1985) investigated water quality in five sub-catchments of varying size in the west Barham catchment between November 1982 and April 1984. In streamwater from harvested sub-catchments, turbidity exceeded 5 NTU in 24% of the weekly samples compared with 9% of samples from undisturbed sub-catchments. Poor drainage from compacted areas (roads, snig tracks and log landings) was identified as the key factor responsible for these effects. Other water quality parameters appeared unaffected by logging. A storm with an average recurrence interval of 10-17 years initiated landslips in areas disturbed by forestry activities. Exceptionally high streamflows were responsible for numerous streambank slumps. Harvesting in a sub-catchment in which more stringent BMPs were employed resulted in no effect on water quality. Farrell and Novotny (1985) identified roads as a significant water quality hazard, and recommended that more attention be given the drainage, design and construction of stream crossings.

Farrell and Novotny (1986) carried out a similar water quality study in five small and two large sub-catchments of the West Barwon catchments, also between November 1982 and April 1984. Logging had a moderate effect on turbidity levels in the smaller sub-catchments but no effect in the larger sub-catchment. In streamwater from harvested sub-catchments, turbidity exceeded 5 NTU in 16% of the weekly samples compared with 8% of samples from undisturbed sub-catchments. Again poor drainage from compacted areas was considered to be responsible for these effects. Roads were not considered to be a major sediment source here because of their location on ridges.

Other water quality parameters appeared unaffected by logging, being related to intrinsic catchment characteristics.

Clinnick (1985b) carried out a short water sampling program in tributaries of the Upper Barwon catchment in November 1985. From this very limited data set Clinnick (1985b) considered that forest roads were a source of sediment, particularly at stream crossings. Unstable and erodible streambanks were a feature of the catchment, and the dispersible clay in catchment subsoils were a potential erosion concern. Water quality was considered to be poor, as a consequence of soil type, past land use and current agricultural/pastoral use. Landslips were observed to be common in the catchment.

Five small catchments were instrumented in the Barham East catchment in 1990 to investigate harvesting impacts on water as part of the Silvicultural Systems Project (SSP). Meteorological, hydrological and water quality data for the period 1990-1994 was examined by Raveenthiran and Papworth (1995?). This report found that much of the hydrological data was either unprocessed or missing, and no data was presented. Summary information from a weekly grab sampling program for water quality showed that turbidity and suspended sediment levels in these unlogged catchment streams were low. A concurrent storm sampling program suffered from methodology problems, and comparative data was rarely obtained. This component of the SSP was discontinued after 1995.

A demonstration site has been set up by the DNRE on Sayers Track in Otway State Forest, to compare methods of road drainage and to assess the effectiveness of a range of structures in improving water quality (Sheridan and Lane, 2000a). This study, which employs a modelling of erosion processes on the road, aims to facilitate the design of optimal BMPs along Sayers track. The installation of improved BMPs at the Asplin Creek crossing is predicted to reduce sediment delivery to the stream by a factor of 10.

Sheridan and Lane (2000b) have determined erosion parameters for three Otway soils using a rainfall simulator as part of an Erosion Hazard Assessment. All three soils, from different sites, had a clay A horizon, a dispersible B horizon and a sand C horizon. Conclusions were that the erodibility values were very high for both rill and interill erodibility in the dispersible B horizon, high in the sand and very low in the clay. This indicates a high susceptibility to erosion when the B horizon is exposed.

A substantial landslide occurred downslope of a forest road in the Western Otways in November 1995 following 280 mm of rain in three days. Nielsen (1996) has reported on the nature of this landslide, which caused damage to the road and resulted in the slumping of a substantial quantity of material rich in clay. Nielsen (1996) considered the slumped material to be a silty clay with low shear strength, leading to instability when very wet. As wet conditions are a frequent feature of the Otways, landslips appear to be an ongoing hazard in the area.

3.9.2 Generalised Inferences Applicable to The Otways

Some generalizations for the Otways can be drawn from the above:-

Well Established

- Unsealed roads, and particularly road crossings of streams, are a potential source of sediment.
- The subsoils of the Otways are dispersive and highly susceptible to rill and interill erosion.
- The Otway forests are very susceptible to landslips during prolonged rainfalls.

Limited Evidence:

- Forest BMPs reduce or eliminate the effects of forest harvesting on water quality.
- Other land uses in the Otways have more effect on water quality than forestry activities.

3.10 Conclusions

During this review of research into the impacts on water quality due to timber harvesting and associated activities, two important issues appeared to be inadequately researched. There was a clear lack of studies examining water quality issues at a range of temporal and spatial scales. Short-term studies fail to consider the dynamic nature of forests and forest management and cannot present a total picture of forestry impacts on water quality over a rotation. Likewise most research was of a small-scale or local nature, and there is little published information on forestry effects at a large catchment or regional scale. Sediment export is a natural (and extremely variable) phenomenon in all catchments, including those containing pristine forest, and reflects the dynamic processes of sediment supply, sediment transport and channel adjustment.

The principal generalised conclusions drawn in earlier sections include:-

Well Established

- Unsealed forest roads are the major sources of sediment in managed forests.
- Road usage is a critical factor in explaining sediment production rates on roads.
- Sediment yields from forested (managed and un-managed) watersheds are considerably lower than those from other land uses, particularly agriculture.
- Sediment production rates on roads and tracks decline within the time frame of 2- to 5 years.
- Nutrient concentrations in streams draining forested catchments are considerably lower than those reported for other land uses, primarily agriculture.
- The dominant cause of increased nutrients in forest streams, if observed, is due to the effects of prescribed burning and wildfire.
- Observed impacts are short-lived and transient with no long-term effect.
- Channelised pathways forming at road drainage outlets form the most efficient conduit for sediment and nutrients.
- Sediment delivery ratios are closely associated with the size composition of the in-situ and eroded soil.
- BMPs play a significant role in the reduction of adverse effects in forested catchments.
- Forest buffer strips are an effective measure in reducing the volume of surface runoff and the quantity of sediment/nutrients delivered to a stream.
- The subsoils of the Otways are dispersive and highly susceptible to rill and interill erosion.
- The Otway forests are very susceptible to landslips during prolonged rainfalls.

Limited Evidence:

- The GHA is not a significant source of sediment due to limited sediment availability, high retention of vegetation cover and spatially variable infiltration rates.
- Rapid and extensive revegetation of disturbed areas after logging can limit, or entirely eliminate, the GHA as a sediment source.
- Fertiliser applications cause no increase in nutrient concentrations of streams draining managed forests.
- Fertiliser applications have no impacts on other values such as stream productivity, fish populations.
- The interaction between factors such as slope, runoff, and morphological factors determines sediment delivery ratios for both the hillslope and catchment.

Speculative:

- Hillslope disturbances during logging result in significant post-logging changes in stream turbidity at the catchment outlet.
- Timber harvesting activities significantly affect nutrient exports measured using a discharge–concentration relationship at the catchment outlet.
- Data sets constructed from empirical relationships or plot erosion data that do not accommodate for processes of deposition and storage within a land element. These data are likely to over estimate the hillslope contribution and consequently the magnitude of catchment response
- The best location and design of buffer strips in forested catchments is known for a range of topographies and land uses.
- The specific role and effectiveness of hillslope, versus near-stream, BMPs are understood.

It is clear from this, and other reviews (eg Dargavel et al., 1995), that the most important issues in relation to water quality relate to the standard of forest management practices. Improvements in the standards and guidelines for harvesting operations can only improve the level of protection afforded to water quality in these catchments. A number of issues emerge from the review that could be incorporated directly into current practices.

- Road use is a critical variable in explaining enhanced sediment production rates. If roads are not being used within a catchment then there should be serious consideration given to the rehabilitation of these surfaces through revegetation and drainage.
- The most persistent and threatening impact of sediment delivery due to roading relates specifically to gullying and channelisation at road drainage outlets. Once gullies have formed there is limited, if any opportunity, to remediate these features so that they remain a persistent threat to in-stream water quality. Every opportunity should be taken to ensure that gullying does not occur through appropriate spacing of drainage structures.
- Finally, it should be recognised that the most dramatic impacts on water quality will come about in response to the most dramatic land use changes. For example the permanent removal of trees increases the probability of accelerated mass erosion (Ziemer and Swanston, 1977; Schofield, 1996). The development and refinement of BMPs within the forest industry over the past 10 years has significantly improved methods of timber harvesting and reduced impacts. The development of environmentally sound methods of timber harvesting and transport must continue.