FOREST MANAGEMENT FOR WATER QUALITY AND QUANTITY

Proceedings of the Second Forest Erosion Workshop

May 1999

J. Croke
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Report 99/6

COOPERATIVE RESEARCH CENTRE FOR
CATCHMENT HYDROLOGY

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PREFACE

This report contains the Proceedings of the Second Erosion in Forests Workshop held in Warburton in May 1999. This volume of short papers and abstracts reflects the wide range of research approaches and tools currently used to measure and model the impacts of timber harvesting activities, including road construction and vegetation changes, on water quality and quantity. The workshop follows on from the first meeting held in Bermagui in March 1997 where over 70 people gathered to discuss various aspects of forest research on in-stream water quality and forest management. We are happy to have a similar number of people attend the workshop in Warburton where the 24 presentations and one day field trip* into the neighbouring Mountain Ash forests around Noojee will provide the focus of our discussions.

The Cooperative Research Centre for Catchment Hydrology's Forest Hydrology Program draws to an end in June 1999 and this event represents a very timely opportunity to examine issues for future research and to discuss appropriate means to transfer the findings of this research to the industry and wider community. We place considerable value in our ability to deliver quality products and tools to the relevant stakeholders and workshops such as this one provide a very important forum in achieving this end. I hope you enjoy the meeting and I look forward to hearing your views on the way forward in technology transfer for Forest Hydrology Research.

Dr Robert Vertessy
Deputy Director,
Forest Hydrology Program Leader
Cooperative Research Centre for Catchment Hydrology.

* *Please note: A video of the field day associated with this report is available from the Centre Office, ph 03 9905 2704*

TABLE OF CONTENTS

ABSTRACT

REVIEW PAPERS

INTRODUCTION

Welcome to the Cooperative Research Centre for Catchment Hydrology's Second Erosion in Forests Workshop in the Warburton- Noojee District of Victoria. Following on from the first workshop two years ago in Bermagui NSW, we are availing of this opportunity to expose new aspects of our research and through interaction with the varied participants, critically review our impact on the management of forest environments in Australia. The workshop represents an excellent opportunity for agencies and industry to constructively contribute to research initiatives and opportunities for technology transfer. The healthy registration list of people reflects a willingness to participate on the part of many, and a genuine concern and interest people have in managing our forest systems.

The collection of papers in this volume illustrates that indeed new research is progressing, and that certain philosophies and approaches are changing in response to this research and improved technology. This can only contribute in a very positive way to an improved understanding of the processes and complexities of measuring and modelling these disturbed landscapes. Pat Lane, Tanya Jacobson and I have 'hassled' and harangued many to present on as wide a range of issues as possible and I hope that the program fairly represents the extent of research in this area and its current relevance to the effective management of timber harvesting in Australia. Whilst we have maintained the general theme of erosion in forests and its associated impacts on water quality, we have also sought to include impacts on water quantity and in-stream values such as macroinvertebrate diversity, reflecting the general trend of a more holistic approach in reviewing the impacts of a particular land use on environmental protection.

This volume also contains two review papers on the state of our knowledge in relation to the effects of timber harvesting on water quantity and quality which were conducted as part of the Federal Government's initiative for Ecological Sustainable Forest Indicators (conducted by Department of Urban Affairs and Planning with sub contracts to Sinclair Knight Merz). These are used as a framework for discussing future directions and issues that may not be well developed in current forest research or management in Australia.

As the Forest Hydrology Project F01 'Sediment movement in forestry environments' comes to the end of its 3-year term in June 1999, I would like to acknowledge the very positive contribution of a great many people. I owe a great deal of gratitude to the team of Jim Brophy, Peter Fogarty, Peter Hairsine, Danny Hunt, Simon Mockler, Bob McCormack, Matt Nethery, and Peter Wallbrink who have worked along side me over this period to reach our agreed targets and milestones. Our collaborative interactions with NSW DLWC, NSW State Forests, CSIRO Forests and Forest Products, DNRE, and CFTT Victoria have been very rewarding and my sincere thanks to the agencies and organisations for making people and resources available. In particular, Peter Fogarty, Steve Lacey, Andrew Loughhead, Phil Ryan, Pat Lane, Paul Dignan, and Abdur Rab for their participation and contributions. The research could not have been undertaken without the substantial financial and logistical support of the CRC for Catchment Hydrology and CSIRO Land and Water in Canberra, including the great support network and assistance of Tanya Jacobson, David Perry and Daniel Figucio. The project has also benefited considerably from the support and enthusiasm of the Forest Hydrology Program Leader, Rob Vertessy.

My thanks to all and I hope you find the workshop both interesting and relevant. We look forward to hearing your comments and please feel free to contribute to the full range of discussions throughout the presentation and field days.

Jacky Croke
Project Leader
CRC for Catchment Hydrology
CSIRO Land and Water.

NOTE: The views and opinions expressed in the following abstracts are those of the author(s) and abstracts have not been subject to external review.

Road Drainage Structures to Convey Water Runoff from Internal Forest Roads Suitable for Log Haulage

C. BELL, Department of Land and Water Conservation, PO Box 118, Bega, 2550

Summary: Natural surface forest roads are perceived to be a major source of sediment contribution to forest streams. Construction of frequent, effective and trafficable road drainage structures is essential to prevent sediment movement into streams whilst allowing timber haulage to occur. This abstract briefly describes the structures being used by NSW State Forests to meet this goal.

1. INTRODUCTION

NSW State Forests is issued a Pollution Control Licence annually by the NSW Environment Protection Authority (EPA). The licence coverage is for areas or Compartments to be harvested or for other operations such as major road construction. Each licence specifies the maximum allowable water flow distance on roads and the minimum effective height of structures to convey water runoff from the road surface. The licence also specifies that the conditions must be applied upon commencement of operations. It is the later requirement that has been a challenge for State Forests to comply with as log haulage difficulties arise once road grades exceed 5 degrees.

2. SPACING OF ROAD DRAINAGE STRUCTURES

The spacing of road drainage structures should aim to minimise the length of water flow along the road surface thereby minimising the sediment loss and maintaining the integrity of the road surface. The two main determining factors are soil erodibilty and road grade.

The maximum distance of water flow or potential water flow is specified in the Pollution Control Licence (Table. 1). These spacings were initially formulated by a joint taskforce including representatives from NSW State Forests and NSW Soil Conservation Service (incorporated in the Department of Land and Water Conservation).

Table 1: Maximum spacing of road structures

Road grade (degrees)	Maximum Spacing (metres
1	200
2	175
3	150
4	125
5	100
6	90
7	80
8	70
9	65
10	60
11	55
12	50
13	45
14	40
15	40

3. TYPES OF ROAD DRAINAGE STRUCTURES

To meet the challenge of complying with the current licence conditions a number of conventional and innovative structures and road construction techniques are currently being used. The description and recommended use of the different road drainage structures and road construction techniques is detailed in Table 2.

NSW State Forests have generally found that the use of conventional, maintenance free, road drainage structures, such as rollover crossbanks, is limited to roads grades of less six (6) degrees, whilst other structures that rely on outfall are constructed on steeper roads.

4. CONCLUDING COMMENTS

Generally construction of road drainage structures capable of allowing safe log haulage on grade of less than six (6) degrees can be achieved. For steeper grade roads drainage that complies with the current regulations and also safe for log haulage is posing a challenge for managers of State Forests in NSW.

This challenge is being met with the trialing of new drainage structure and techniques.

5. ACKNOWLEDGMENTS

The author acknowledges the work of Steve Hunt and Graham Riches, State Forests Narooma, for their trialing of Rubber Drains, and State Forest staff , South East Region for there perseverance with the new road design and construction techniques.

Table 2: Types and assessment of road drainage structures

New or Existing Roads

Type	Description	Suitability	Cost of installation	Comments
Rollover Crossbank	A crossbank constructed with a smooth cross-section and gentle batters, which is well compacted to allow permanent vehicular traffic.	Internal forest roads with grades of <6°	Low to moderate depending on road grade and trafficability	Effective maintenance free structure if adequately constructed
Spoon Drain	A drain with semi-circular cross-section which has no associated ridge of soil, and which relies on crossfall to convey runoff from the road surface. The capacity being solely determined by the excavated channel dimensions.	Internal forest roads with grades of < 8°.	Low cost	Regular maintenance required to maintain crossfall during dusty or wet conditions
Finger Drains	A drain with a broad excavated channel and sharp uplift, which has no or little associated ridge of soil, and which relies on crossfall to convey water runoff from the road surface.	Internal forest roads with grades of <10°	Low Cost	Regular maintenance required to maintain crossfall during dusty or wet conditions. Inconvenient for log haulage due to speed reduction
Mitre Drain	A drain used to convey water runoff from the road shoulder, and is often the extension of the road tabledrain.	All internal forest roads.	Low cost	On non-crowned roads frequent maintenance is required and often ineffective .
Gravel Rollover or Finger Drain	A drain, as described above, which is constructed of imported gravel material.	As for Rollover or Finger Drain	High cost due to importing of material.	As for Rollover and Finger Drain.
Rubber Drain	A strip of rubber conveyor belting buried in the road surface to convey water runoff from the road.	All internal forest roads.	Moderate to high cost depending on the cost of installation.	Effective maintenance free structure currently being trialed.

New Road Construction

Level section	A section of road constructed at, or close to, 0° grade which relies on outfall to convey water runoff from the road surface.	Most internal forest roads of any grade.	Low additional cost of road construction	Not suitable for ridge top road construction.
Reduction of road grade	Constructing steeper grade roads with short sections of lower grade to allow installation of effective road drainage structures.	All internal forest roads.	Low additional cost of road construction.	Curving of road can effect log haulage on steeper grades in unstable soil types.

Aspects of the Geometry of Buffer Strip Design in Mountain Country

L.J. BREN, Department of Forestry, University of Melbourne

Summary: Three methods of buffer design were applied to streams in a mountainous catchment and the resulting stream buffer geometry reviewed. To facilitate reporting the methods have been applied to a catchment of regular shapes. The simplest method uses a fixed width from streams. Compared to more rigorous methods this under-protects the stream head, but overprotects divergent areas downstream. A method based on a constant ratio of upslope contributing area to buffer area gave the widest buffers at the stream head and buffers of diminishing width as one moved downstream. A method based on absolute convergence tended to put most buffer protection into convergent stream heads. The more rigorous methods gave buffers which were non-symmetric, non-intuitive, and strongly influenced by upslope geometry.

1. INTRODUCTION

The concept of a forestry stream buffer is that a proportion of a watershed is retained from logging. The question of just which portion of the catchment should be retained or how the buffer should be designed has received little attention in the literature. This paper gives an overview of work examining aspects of the geometry which results from stream buffer strip design methods. As reported here the work centres around a "model" first-order catchment (Figure 1) which can be regarded as a "building block" of larger watersheds. This catchment is based on regular geometric figures; however this is an abstraction to simplify presentation. In fact most of the work was done using flow-netting on a 67 km^2 Tarago catchment. Fuller details of the work are presented in Bren (1995, 1997, 1998) and papers under review. The catchment can be viewed as consisting of a convergent bowl-shaped head, parallel flanks, and divergent spur ends. The bowl-shaped head has square flanks, giving it an appearance somewhat like a portion of a bath. This abstraction encompasses much of the geometry encountered in complex combinations in the Tarago catchment.

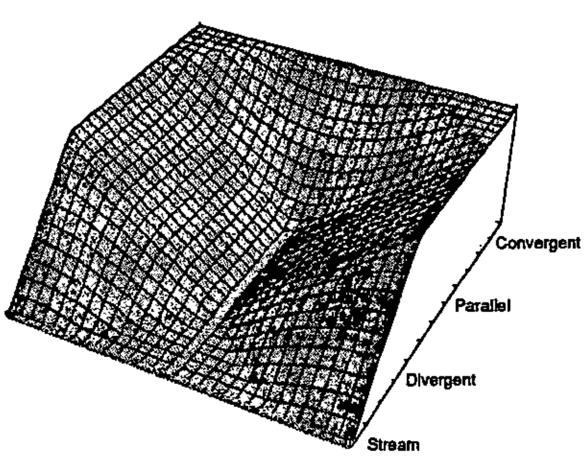

Figure 1: Oblique view of the "model catchment.

2. CONSTANT BUFFER WIDTHS

Figure 2(a) shows this concept applied to the streams in our model catchment. The approach is as simple as it comes in that the stream is defined and a buffer marked out a fixed distance (commonly 20 m in Victoria) from the stream. Usually this distance does not vary with catchment slope or contributing area. When applied to the Tarago catchment the results showed a clear relation between percentage of the area taken by buffers and the defined width of buffer:

$$P = -0.489 + 0.65\,w - 0.001\,w^2$$

where P is the percentage of catchment occupied by buffers, and w is the width of buffer from stream centre-line, m.

Tarboton *et al.* (1988) showed that stream networks are essentially fractal in nature, and that two distinct fractal dimensions can be defined. The first is the fractal dimension of the stream network, while the second is the fractal dimension of the individual segments of the stream. Thus, the resulting buffer also has a similar level of fractal dimension at two levels. The first can be viewed as the complexity of the boundary segments, which reaches a maximum at 10 m buffer width. The second is the complexity of the network of "in" and "out" areas - as the buffer width is increased buffers from neighbouring streams will often coalesce, forming areas which can not be accessed except by passing through other buffer areas. Whether this is acceptable or not is controversial.

Constant width buffers have a simplicity of definition which is attractive but they do not take into account complexities of hydrologic loading, and the point at the stream head at which the stream is deemed to "start" is often difficult to define. This leads to many disputes between pro and anti-logging parties.

3. CONSTANT BUFFER LOADINGS

In this concept the ratio of area upslope of each segment of buffer to the area of buffer is held constant. Thus each element of the buffer has exactly the same loading on an area basis. Figure 2(b) shows our catchment with 20% and 40% of the logging coupe being given over to

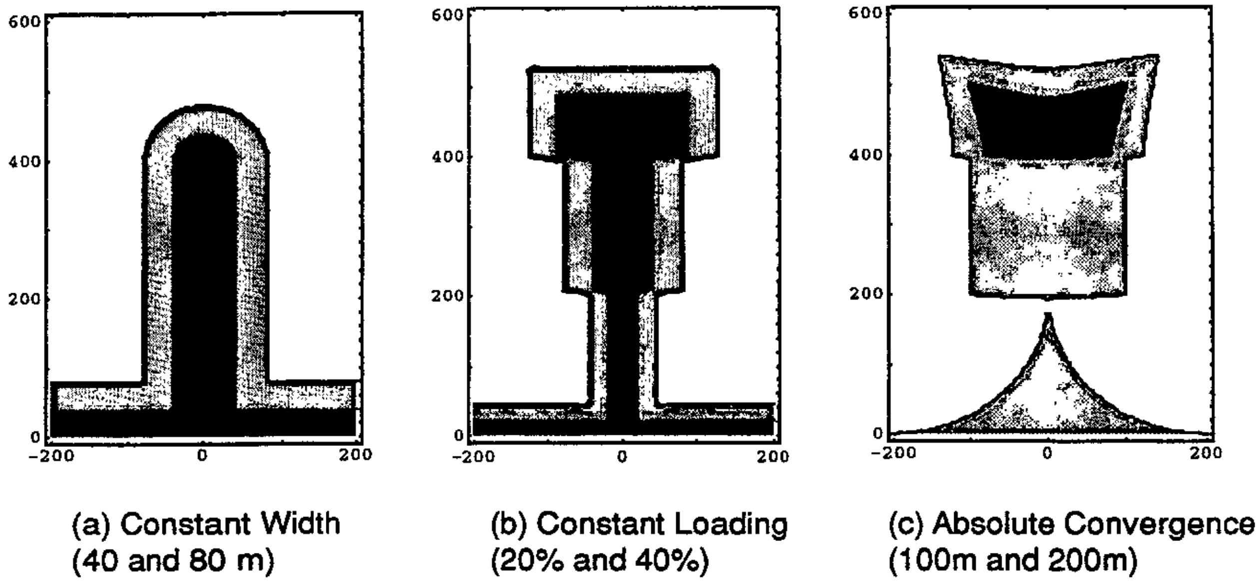

Figure 2: Computed variations in the buffer shape for the three methods of defining buffer geometry. The buffers are unusually wide to illustrate the geometry. The top third (400-600 m) is the "convergent" zone, the middle (200-400 m) is the parallel zone, and the bottom zone (0-200 m) is the divergent portion.

buffer. It can be seen that, for a constant loading, there are three separate buffer widths for the convergent, parallel, and divergent zone. There are discontinuities where these zones meet, but the main effect is that to obtain a constant buffer loading an increased buffer width is needed in convergent zones, and a decreased buffer width in divergent zones. Additionally the square shape of the edges of the bowl-shaped area are reflected in the shape of the computed buffer boundaries in this zone, although our model catchment is actually bowl-shaped in the vicinity of the buffer.

When applied to our Tarago catchment all these effects were evident. However the buffers computed were also unsymmetrical and widely varying along reaches of streams. Also of interest was that as one moved downstream in the natural catchment, the areas contributing directly to the higher order (larger) streams tended to be divergent (and in fact, most water enters the stream from contributions to tributaries in convergent areas, with very little being contributed directly to the larger streams). Thus, *de facto*, the buffers computed for higher order streams tended to be smaller than those for the smaller headwater streams. This makes sense hydrologically but is probably politically unacceptable and appears counter-intuitive.

4. METHODS OF ABSOLUTE CONVERGENCE

A number of geomorphologists (eg Montgomery 1994) have suggested that when there is enough flow convergence the soil is carried away and hence gullies originate. It follows that a method of buffer design could use a measure of the absolute flow convergence. The "natural" erosion limit could be obtained by field measurement of gully heads. Any areas which exceed this limit would be placed into buffer, while areas with less convergence could be subject to forestry operations. The concept is embodied in a number of popular hydrologic spatially distributed models (eg "TOPMODEL" of Bevan, 1997, "TOPOG" of Vertessy *et al.* 1994). Such a method does not rely on the area being already defined as a "stream" and hence can be used to help define whether "dry gullies" should be given buffer protection.

The measure of absolute convergence used in this study was the ratio of contributing upslope area per unit length of contour line and has the units of meters. We have applied this concept to our hypothetical area (Figure 2(c)) and achieved a rather interesting result. If we settled on a cut-off of 200 m then all buffer would be allocated by the method to the convergent head. If we went for the lower value of 100 then we may get a narrow portion in the parallel zone and a small, cusp-shaped buffer at the areas of heaviest loading in the divergent zone. Again, in a similar way to the method of constant loading we get discontinuities in the geometry. The Tarago work showed that a cutoff value of 400 m would give approximately the same overall percentage of buffer as the current 20 m fixed width buffer. However the location of this would be entirely in convergent areas.

When applied to our Tarago catchment the results also gave asymmetric and counter-intuitive buffers. A study of the levels of convergence in gully heads and natural slopes of the Tarago catchment gave a fairly wide variation in numbers. A suitable natural limit was 400 m. When this "absolute convergence limit" was applied to a series of hypothetical logging coupes the results were interesting. Thus many areas which had only small convergences or were basically parallel or divergent were allocated no buffering, while some areas basically attracted all the buffers. The strong characteristic of this method is that it allocates buffers to convergences, and that only at relatively low levels of convergence do buffers begin to appear in parallel and divergent areas, reflecting their relatively low hydrologic loading. As before, the shape of the catchment upslope dominates the shape of the buffer; thus in the convergent zone this results in a more or less square buffer although the catchment in this area is bowl-shaped. Again this result makes sense at a hydrologic level, but appears counter-intuitive or puzzling to people in the field. The method can also allocate areas that have a high convergence but are not connected to the stream as buffer, and so can have advantages in more clearly distinguishing between "drainage lines" and streams.

5. SOME GENERALISATIONS

Bren (1998) showed that the "constant width" buffer design method gave a highly varying buffer loading and probably underprotects buffers at the stream head and overprotects as one moves downstream, (relative to the mean level of protection).

The results of the "constant buffer loading" and "absolute convergence" methods are of particular interest because they were rigorous methods which meet criteria of "hydrologic respectability." To varying degrees they suggest that the convergent areas generally require more buffering that is usually given, but that parallel areas and divergent areas require little or no buffering because of their low hydrologic loading. Discussion with colleagues indicates unease about such results, with the suggestions of "other factors" being introduced. The "design" methods also give asymmetric buffers (ie the buffers on the left-hand side of a stream differ from those of the right hand side), and introduce anomalies such as buffer widths becoming very small at stream junctions because the top of the spur is close to the stream at such places. Particularly puzzling to field observers is that the buffer width reflect buffer loadings which are not visible in the field. Thus, in an otherwise uniform area of forest, the buffer will change in distance from the stream because of changes in upslope characteristics.

A variant of the absolute convergence method ("slope index") used an additional factor - the slope at the contour line - and this factor introduced another level of variation that tended to make results even more counter-intuitive. Again many colleagues suggest that factors such as soil erosivity or depth of soil should be introduced to buffer design. Our hypothesis is that each additional parameter introduced into a formal buffer design method will increase the complexity of the resultant buffer, and make the result less intuitive, more complex, and less satisfactory. I would presume that introduction of additional criteria based on factors such as animal conservation or aspect would also further increase complexity.

Such methods also have the disadvantage of requiring more and more information, and even in "advanced" countries this level of information is generally not available. Thus it is concluded that although simple methods such as "constant distance from streams" may not meet many (or even any) hydrologic criteria very well but they probably do meets manager's expectations fairly well, and conflict is isolated to the traditional questions of "where does the stream start." Perhaps an outgrowth of such studies should be some method of widening the buffers in convergent areas, and a reduction in divergent areas.

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The Effects of Roading, Harvesting and Forest Regeneration on Streamwater Turbidity Levels in a Moist Eucalypt Forest

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Summary: Turbidity levels were measured in streamwater leaving eight small eucalypt catchments near Dungog, NSW over seven pre-treatment and eight post-treatment years. Permanent roads were constructed in four of these catchments prior to logging in 1983. Turbidity increased after harvesting during storms, but the effectiveness of the BMPs employed, and natural variability, ensured that levels were no greater than those measured in unlogged catchments. Rapid and extensive revegetation of logged catchments appeared to provide better protection to the catchment surface than the previous forest, resulting in reduced post-treatment levels at moderate-low discharges in catchments without permanent roads.

1. INTRODUCTION

Forest harvesting and associated roading operations have been shown to affect the quality of streamwater in many countries, principally through additional sediment supply to drainage systems (Reid and Dunne 1984). Effects can be reduced by the adoption of a range of Best Management Practices (BMPs) during forestry operations. This paper presents the results of a 15 year study into the effects of roading and harvesting (with BMPs) on streamwater turbidity levels in moist eucalypt forest in eastern New South Wales.

2 FOREST TREATMENTS

The Karuah Hydrology Research Area comprises eight small and relatively steep catchments which were instrumented in 1975. Permanent access roads were constructed through four catchments in 1982 and six catchments were harvested during 1983. The old-growth pre-treatment forest and the imposed harvesting treatments have been described (Cornish 1993). In summary these were:-

1. Unharvested Control (C).
2. Harvesting without a regeneration burn (L-).
3. Harvesting with a regeneration burn (L+).
4. Eucalypt plantation establishment after clearing and burning (P).

Each treatment was split with one catchment permanently roaded and its duplicate left unroaded. This design was imposed on the study by the practical need for timber extraction access.

The extent of harvesting, and the effects of harvesting and burning on the catchment surface, were quantified by aerial photography and ground survey in 1984. Similar assessments in 1986 showed extensive catchment revegetation and surface recovery (Cornish 1991).

3 MATERIALS AND METHODS

Each catchment has a V-notch weir, stilling well and stream height recorder. There is a good network of storage rain gauges and pluviographs. From 1979 onwards catchments were progressively instrumented with GAMET automatic water samplers set to collect samples over storms. Stream height (and therefore stream discharge) was determined for each sample collected. This storm data set continued to 1989 but there were many instances where samples were not obtained from all catchments during a particular storm, generally because of equipment malfunction. The period 1980-1982 was relatively dry and far fewer storms were sampled pre-treatment than post-treatment.

A weekly grab sampling program resulted in another data set which extended from 1976 to 1991. While this program provided essentially lower flow data, many samples were obtained during high discharges. Grab samples were always collected on the same day in all eight streams, and generally within a three hour period. They therefore form a lengthy set of paired samples with known collection discharges. The range of sampled discharges included many higher values, and only the top 1.5% of study period discharges remained unsampled. Ten percent of the samples were associated with the top seven percent of study period discharges. Samples were analysed as soon as possible after collection.

4 RESULTS

Storm Data

Because of equipment failure the number of storms sampled varied between catchments in both the pre- and post-treatment periods, making direct comparisons of data difficult. Turbidity values were not normally distributed, but became generally so after a log transformation. An analysis of variance (GLM) performed on log-transformed values indicated that means were significantly higher in the post-treatment period in all catchments except Sassafras (Control, no road) and Jackwood (L+, no road). Means were all less than 50 NTU, and mean differences between periods were less than 20 NTU.

A better comparison between catchments in the post-treatment period was made by selecting a sub-set of storms during which treated and control catchments were sampled in an equivalent manner. Data obtained during these matched storms were then log-transformed and means statistically compared using GLM (Table 1). In Bollygum (L-, no road) turbidity values were significantly greater than in either control catchment, while Kokata (P, no road) had significantly

Table 1. Mean turbidity values (NTU) of samples obtained during storms in the post-logging period. Comparisons were only made for storms matched between logged catchment and both controls. Means of log-transformed data, then back transformed.

MATCH		Crabapple. Control +Road	Barratta Log, No Burn, +Road	Bollygum Log, No Burn, No Road	Corkwood Log, Burn, +Road	Jackwood Log, Burn, No Road	Kokata Plant.,. +Road	Coachwood Plant., No Road
		CATCHMENT						
With Sassafras Control (NO ROAD)	Catchment Mean (NTU)	26.5	37.1	48.5	40.4	24.7	49.5	35.3
	Control Mean (NTU)	25.2	26.7	30.3	28.8	27.8	30.4	30.0
With Crabapple Control (ROAD)	Catchment Mean (NTU)	-	23.0	49.5	34.2	21.4	44.3	28.2
	Control Mean (NTU)	-	21.3	27.7	33.4	25.6	32.7	27.0

Shading indicates a significant difference at the 5% level between the catchment mean and the control mean.

greater turbidity values than Sassafras. No other catchments had post-treatment values significantly different from those in control catchments. Mean turbidity values did not exceed 50 NTU, and differences between treated and control catchment means were 22 NTU or less. There were an insufficient number of sampled and matched storms to permit this analysis to be performed on pre-treatment data.

A further analysis of post-treatment data obtained from matched storms was carried out by calculating the mean discharge-weighted turbidity values for treated and control catchments. These were generally higher in harvested catchments, and higher than equivalent means obtained from log-transformed data. Catchment relativities were similar to those resulting from the comparison of log-transformed means, but the weighting procedure did not allow statistical comparisons to be made.

Grab Sample Data

An analysis of variance performed on log-transformed turbidity values obtained before and after treatment indicated that means were low (<4 NTU), and that they differed significantly between catchments in both periods. One control catchment (Sassafras) had a significantly greater mean value in the post-treatment period, as did Corkwood (L+, road) and Coachwood (P, no road). Two catchments, Bollygum (L-, no road) and Jackwood (L+, no road) had significantly lower means in the post-treatment period. Bollygum means were significantly greater than all other catchment means in both periods. No catchment after treatment had a mean value as high as that exhibited by Bollygum before disturbance.

Paired data obtained in the pre-treatment period was used to determine regression relationships between turbidity values in to-be-treated catchments and corresponding values in the two control catchments. These regressions were improved by including stream discharge as a covariate, and were then applied to control catchment turbidity data, and discharge values in to-be-treated catchments, in order to compute predicted values of turbidity for each paired sampling date in the entire record. Two sets of residual (observed-expected) values were then computed, and post-treatment mean values calculated, one obtained from each set of control catchment data.

These two mean post-treatment residuals for a particular catchment agreed well. Only Corkwood catchment (L+, road) had mean residual values significantly greater than zero, while all three logged catchments without roads (Bollygum, L-; Jackwood, L+; and Coachwood, P) had mean residuals significantly less than zero. Time series plots of these residual values showed rapid and significant turbidity reductions taking place after logging in catchments without roads.

The different behavior of catchments without roads and catchments with roads is graphically illustrated by the turbidity double-mass plots in Figure 1. Cumulative turbidity values from two logged catchments (Jackwood, L+, no road, and Corkwood, L+, roaded) are plotted against cumulative values for Sassafras catchment (control, no road). Only data from paired samples has been employed. Values for Corkwood deviate upward (relative to the pre-treatment trend) at the time of road construction, and the slope remains elevated thereafter.

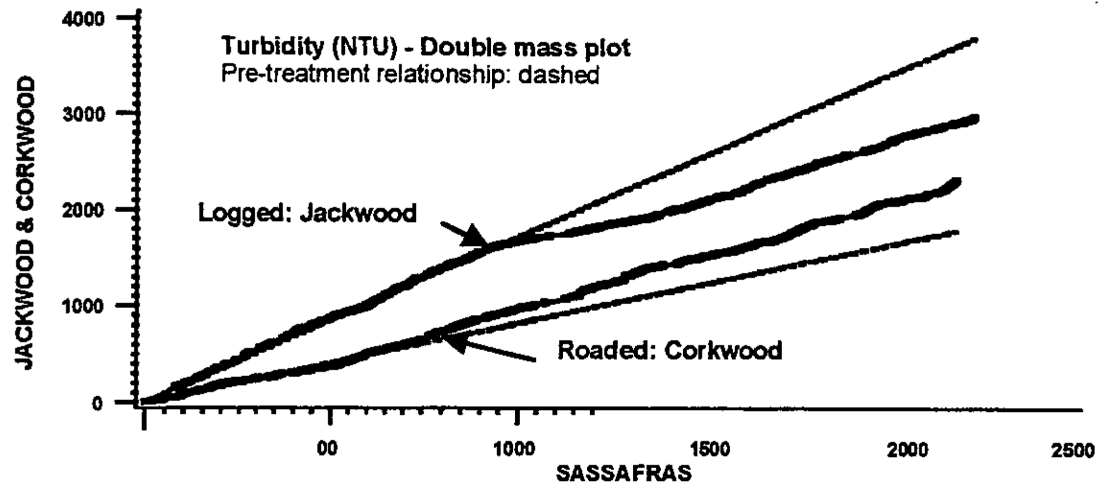

Figure 1. Turbidity double mass plots for Corkwood (roaded, L+) and Jackwood (unroaded, L+) catchments versus Sassafras (unroaded control catchment). Data extend from 1976 to 1991, and the times for initiation of roading and logging are indicated.

In contrast values for Jackwood deviate downward soon after logging begins, and the slope remains lower to the end of the study. Clearly pioneer vegetative cover has afforded better catchment protection than the pre-treatment vegetation, and lowered turbidity values relative to those in the control catchment. On the other hand roading has resulted in an immediate and permanent increase in turbidity values relative to the control catchment. Similar, but less striking, revegetation and roading effects were observed in the other catchments.

All Data

Percentile-percentile plots of ranked turbidity values versus ranked discharge values were constructed pre- and post treatment for each catchment using all data from both sampling programs. All catchments demonstrated higher post-treatment turbidity levels at higher flows apart from Sassafras (C, no road) and Jackwood (L+, no road). Corkwood (L+, road) had elevated turbidity levels over the entire flow range. All harvested catchments without roads showed lower post-treatment turbidity levels at moderate to low flows. This observation was consistent with the sign of the post-treatment residuals determined from pre-treatment regressions.

5. DISCUSSION

Turbidity values were greater during storms in the post-treatment period in most catchments, but these values were significantly greater than those in control catchments only in Bollygum (L-, no road) and Kokata (P, road). Turbidity levels declined post-treatment in all treated catchments without roads at moderate to low discharges. In catchments with roads values either increased (Corkwood, L+, road) or remained relatively unchanged.

All harvested catchments exhibited a rapid and extensive revegetation of disturbed areas in the immediate post-logging period. This colonisation by eucalypts and pioneer species was particularly widespread in those catchments burnt after logging, resulting in a complete surface cover of low vegetation within two years. This contrasts with the variable surface cover afforded by the overstorey and understorey trees in the undisturbed forest, where the canopy is dense and prevents the establishment of short ground cover vegetation. Leaf and bark litter, which tends to be seasonal in production, provides the only surface protection from raindrops increased in size by the canopy. The soil surface therefore may be better protected from raindrop impact by the regenerating forest, resulting in the small, but statistically significant, reductions in stream turbidity levels demonstrated in this study. Reductions did not occur in catchments with roads, suggesting that increased sediment from roads was either negating or, in the case of Corkwood catchment, exceeding these surface effects.

6. CONCLUSIONS

Although streamwater turbidity levels during storms were higher following harvesting, these values were for the most part not significantly greater than values in unlogged catchments. Post-treatment values were not of a magnitude generally associated with accelerated erosion, indicating that the suite of BMPs applied (including retention of stream buffers and drainage of snig tracks) were effective.

Roads appeared to supply small quantities of additional sediment to streams at moderate to low discharges, negating or exceeding a demonstrated reduction in turbidity levels in non-roaded catchments. This demonstrated reduction may be due to better protection of the catchment surface from raindrop impact by the rapidly regenerating vegetation.

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Forest Roads: Can we tar them all with the same brush?

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Summary: Runoff and sediment generation rates were investigated on 7 sites in the Cuttagee Creek catchment near Bermagui in NSW. Generation rates and total sediment yields varied significantly with road class (Secondary Access-Dump roads) and with sediment availability, reflecting the effects of road usage and traffic intensity, for each 30-min simulation. The road surface provided the dominant source of sediment at each site with limited contributions from the road ditch, batter or upslope contributing hillslope. In terms of sediment delivery down the adjacent hillslope, sediment concentrations were significantly higher in channelised flow paths than in dispersive well-vegetated pathways. In terms of water pollution protection, prevention of direct connectivity via gully erosion at road drainage structures is essential to limit the delivery of sediment from forest roads to streams.

1. INTRODUCTION

The interaction between forest roads and water lies at the heart of several key issues surrounding the environmental protection of water resources. At the scale of individual road segments, roads can have a significant effect on hillslope hydrology (Wemple *et al.*, 1996; Dawes and Croke, this vol). At the larger scale, roads may also influence the timing and magnitude of streamflows from the catchment with possible consequences for downstream channels and aquatic ecosystems. The location of unsealed roads in areas proximal to natural drainage lines or on steep slopes with high delivery potential, means that they can also represent a significant threat to in-stream water quality.

While the erosion potential of unsealed roads is well established in the literature, their contribution to water pollution is less clearly defined. For water pollution to occur, sediment from these surfaces must be delivered directly to the channel. Sediment delivery varies not only with source strength, as determined largely by generation rates on the road surface, but also with the nature of the flow path, the gradient of the discharge hillslope and distance to the stream (see Mockler and Croke, 1999, this vol; Hairsine, this vol). These factors vary significantly with catchment terrain, road location and usage, and management regime so that within any given catchment, there is considerable spatial and temporal variability in sediment generation and delivery rates.

Here we report the results of a series of field rainfall simulator experiments on unsealed forest roads in the south east forests around Bermagui NSW. This paper specifically addresses sediment generation and delivery and the role of factors such as road class, position in the landscape, road drainage type and the nature of the delivery pathway. Other papers within this volume examine aspects of road to stream connectivity and the relative importance of discharge, slope and soil erodibility (Mockler and Croke, this vol; Hairsine, this vol) and modelling the interaction between road surfaces and local hillslope hydrology (Dawes and Croke this vol).

2. STUDY CATCHMENT

Cuttagee Creek is a 57km^2, predominantly forested catchment in the coastal lowlands of the Eden Forest Management Area (EFMA) of south eastern NSW. The catchment is serviced by 75.23km of roads, representing approximately 1% of the catchment area, with a mean density of ~ 2 km per km^2 (Mockler and Croke, 1999). Most (85%) of the roading network was constructed between 1964 and 1976. Road construction during the period post-1976 was confined predominantly to access roads servicing log dumps used during specific harvesting cycles. There are two types of road within the catchment, a cut-and-fill road which is cut into the hillslope and captures runoff as a result of surface and subsurface flow from upslope contributing areas (UCA) and a ridge-top road where the road surface is the only source of water to the drainage outlet (Fig 1). Road maintenance standards range from well-graded, crowned roads to those that are not well maintained and are abandoned for large parts of the year (Table 1). Road surfaces are drained predominantly by simple mitre or push-out structures that redirect runoff and sediment onto the adjacent hillsides and reflect the predominant ridge-top characteristics of the study area (Fig 1). On cut-and-fill type roads, the dominant drainage structure is a box culvert which redirects runoff and sediment under the road surface to the adjacent hillslope. Mockler and Croke (1999) report that ~ 5 km of roads within the catchment are fully linked to the natural drainage line and a further 0.67 km are partially linked by concentrated pathways. Most of this connectivity occurs at culverts draining road surfaces with large contributing areas and discharging onto areas with steep hillside gradients (see Mockler and Croke, this vol).

3. METHODS

A total of 7 sites were selected for field rainfall simulations within the catchment representing sediment generation rates on sites of varying

- *road class*
- *road type*

- *road drainage structure* and
- *delivery path* (Table 1).

The contributing length of each road segment was fixed to 40 m, which is representative of the median value of inter-drain spacing within the catchment. Road surface gradient averages about 5 degrees throughout the catchment.

Table 1. Summary of road class, type and road drainage characteristics at each site.

Site No.	Road Class	Road Type	Road Drainage Type	Delivery Path
1	Feeder Access	Cut-and-Fill	Mitre	Channel
2	Feeder Access	Cut-and-Fill	Culvert	Channel
3	Secondary Access	Ridge-Top	Mitre	Channel
4	Feeder Access	Ridge-Top	Mitre	Dispersive
5	Secondary Access	Cut-and-Fill	Culvert	Channel
6	Secondary Access	Cut-and-Fill	Mitre	Channel
7	Dump	Cut-and Fill	Outfall	Dispersive

Rainfall-runoff was generated on the selected road segments using a modified layout of the large simulator described by Croke *et al.*, (1997) in previous runoff experiments on forest snig tracks. The area covered by the simulator ranged from 300 to 600 m^2 and varied according to whether the road segment was ridge top or cut-and-fill respectively (Fig. 1). Two rainfall events of 30-min duration with intensities of 75 and 110 mm hr^{-1} were applied at each site. These intensities correspond to the average 10-y and 100-y rainfall events for this region. Events of this magnitude and short duration were required to generate sufficient runoff volume to exceed the storage capacity of the mitre drains and deliver runoff and sediment discharge downslope. Cut banks within the catchment are composed predominantly of weathered bedrock and are relatively consolidated with minor rock fractures and fissures. The overlying soil layer is thin (<0.5 m) and there was limited evidence of root macropores or major subsurface pathways. Contributing road width ranged from 2.5 m to 5 m and varied with road class and the presence of a well-maintained crown which sheds water either side of the road centre line and (Fig 1).

There were 4 measuring points within each of the plots (Fig 1). Runoff and sediment from the road surface was measured within a 10-20m^2 plot (A in Fig 1) where sediment discharge samples were taken at regular 3-minute intervals using a 2L sample bottle attached to a vacuum driven sediment sampler. The cut batter sample location (B in Fig 1) collected runoff and sediment from the upslope contributing area and batter surface using galvanised metal guttering which directed runoff to a sample collection point. Total runoff and sediment from the entire road segment and routed to the entrance of the drainage point (C in Fig 1), was recorded using a RBC flume, which was used to record discharge using a depth-discharge relationship. Manual total load sediment samples were also taken at regular intervals. Runoff and sediment samples were also collected at selected distances down the discharge hillslope (D in Fig 1) using a vacuum sediment sampler.

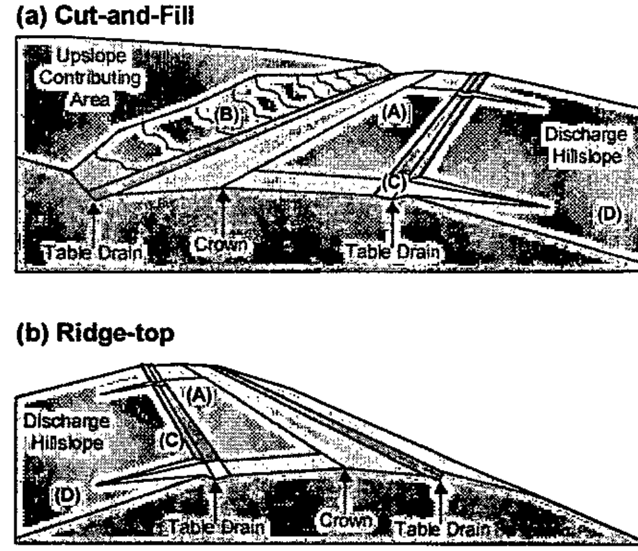

Figure 1. Types of road construction in the catchment (a) Cut-and-Fill with upslope contributing area and (b) ridge top with the location of the 4 sample locations (A-D) used in the experiments.

4. RUNOFF AND SEDIMENT GENERATION

Surface runoff developed on the road surfaces very quickly, reflecting the highly compacted nature of these surfaces and their associated low infiltration rates. The average infiltration capacity for the road surface across all sites was 12 mm hr^{-1}, with rates as low as 0.5 mm hr^{-1} on some segments. Infiltration capacity did not vary significantly with road class, ie. secondary, feeder and dump access. Steady-state discharges ranged between 1-5 L s^{-1} across the sites and were attained within the first 5 minutes of each run. A representative hydrograph is presented in Dawes and Croke (this vol). Runoff coefficients, which ranged from 54 % to almost 99% across the sites, also reflected the fact that most rainfall was converted to runoff. Runoff depth per unit area was higher, but not statistically different, on the cut-and-fill roads, reflecting a small contribution of surface overland flow from the upslope contributing areas (Table 2).

These upslope hillslope elements generated patchy runoff during the 10-y event of 75 mm hr^{-1} but under the higher rainfall intensity of 110 mm hr^{-1}, shallow sheet flow developed and this contributed to the total runoff volume recorded at the drainage outlet. We observed very little subsurface flow discharging from the thinly fractured rocks composing the cut batters on these road types.

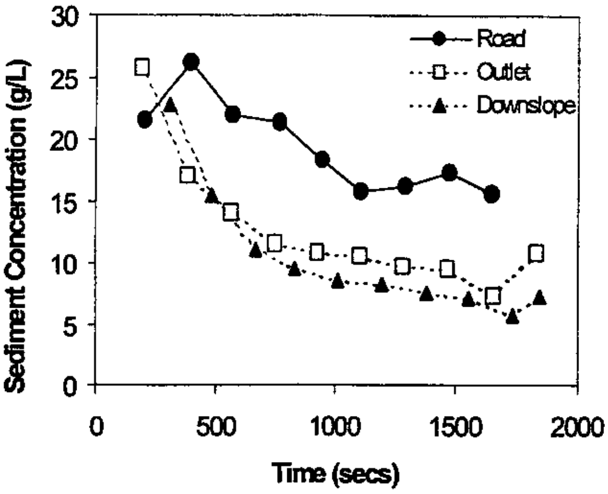

Figure 2. Representative plot of changes in sediment concentration for a secondary access road for a 30-min simulation. Sample points represent sediment concentrations from the road surface, road discharge outlet and downslope of the outlet.

Sediment concentrations varied significantly with road class and were between five and eight times higher on the well-used, secondary access roads than the abandoned feeder and dump access sites (Table 2). The results of a multiple stepwise regression model suggested that 75% of the variance in sediment concentrations across the sites is explained by a combination of road type, and the rate of sediment exhaustion. Sediment exhaustion was measured as the rate of change in sediment concentration per unit time or sample number. Sediment concentrations displayed a very consistent pattern over each 30-min simulation consisting of an initial flush of sediment early in the rain event which declined over time to a supply-limited rate (Fig 2). Overall, sediment concentration showed no significant relationship with rainfall intensity (Table 2). We interpret these data to suggest a critical relationship between sediment generation and the nature of road sediment storage and strength. Roads that have high intensity traffic have the greatest store of surface material available for transport whereas roads used infrequently have limited sediment availability and this was clearly reflected by very marked declines in sediment concentration over the event. Sediment accumulation on the road surface as a result of traffic intensity is a critical factor in estimating sediment generation rates on these surfaces. In the Cuttagee Creek catchment we observed large stores of fine, powdery material (bulldust) which accumulates during dry weather conditions and periods of high traffic intensity. The nature and magnitude of rainfall events immediately after this period of intensive road usage will be critical in determining whether this material is stored within the drains or delivered directly to stream channels.

Table 2. Mean sediment concentrations and total yields for Road surfaces (R_s) and Road discharge outlets (R_o) for the 7 sites for the 10 and 100-y rainfall events.

Site No.	Road Type	Rainfall Intensity mm hr^{-1}	Runoff depth mm	Sediment Con g L^{-1}		Total Sediment Yield t ha^{-1}	
				R_s	R_o	R_s	R_o
1	Feeder	75	21.98	5.59	2.62	0.65	0.79
		110	51.31	5.54	1.80	1.04	0.74
2	Feeder	75	20.85	10.43	2.89	6.06	0.71
		110	54.01	4.61	3.01	0.95	0.76
3	Sec Acc	75	23.11	33.31	14.38	13.91	5.74
		110	38.80	18.62	15.27	8.69	7.46
4	Feeder	75	19.21	4.23	2.56	1.57	1.32
		110	49.28	2.56	2.06	1.14	0.91
5	Sec Acc	75	28.79	19.38	12.75	6.93	3.72
		110	55.27	19.89	14.24	11.38	4.13
6	Sec Acc	75	27.86	24.69	6.01	9.60	2.00
		110	60.09	18.34	5.75	10.93	2.35
7	Dump	75	26.37	2.58	4.04	0.18	1.08
		110	62.60	1.80	3.11	0.40	1.11

Maximum sediment yields were recorded from the road surface reflecting their dominance as the major source of sediment. Total sediment yields for the entire road segment (road outlet) were generally lower than those from the road surface alone (Table 2). We observed material being trapped in litter dams and in depressions along the rough surfaces of the table drain. Estimates of sediment yield from the road surfaces are almost one order of magnitude higher than those for recently logged snig tracks (6 months) in the same catchment and for equivalent rainfall intensities (Croke *et al.*, 1997). This indicates the importance of unsealed road surfaces as a dominant source of sediment in this catchment.

5. SEDIMENT DELIVERY

Changes in sediment concentration down the discharge hillslope were examined for two dominant types of flow paths, channelised/gully and dispersive. The former are characterised by distinct morphological evidence of flow concentration whereas the latter show no evidence of runoff concentration and are characterised by relatively planar slopes. The relationship between sediment concentration at the road discharge outlet ($Conc_O$) and the furthest measurement point down slope ($Conc_H$) is presented in Figure 3. Points which plot on, or close to, the 1:1 line represent little change in sediment concentration between the two measurement points thus indicating a linear relationship between the sediment source and delivery point. In general, points within this area of the plot reflect sediment delivery in channelised flow paths or alternatively, in dispersive pathways with very short flow path lengths where there is limited opportunity for sediment deposition before runoff infiltrates. Many of the gully flow paths extended downslope for up to 50m and represented very efficient conduits for runoff and sediment. In contrast, flow on the dispersive slopes was evident for distances of <10m before infiltrating. Points lying below the 1:1 line indicate reductions in sediment concentration along the flow path and hence net deposition of material. Finally, points above the 1:1 line indicate an increase in sediment concentration along the flow path and indicate that the hillslope is a source of material. For the 2 points illustrated here this was due to the presence of very loose ash and burnt material on the discharge hillslope which moved in debris flows during the high intensity rainfall events.

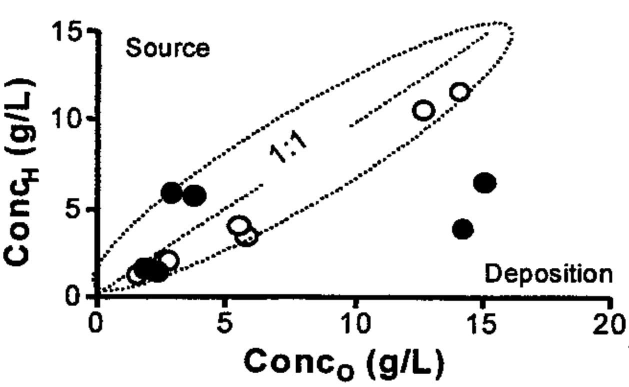

Figure 3. Relationship between sediment concentration at the road outlet ($Conc_O$) and sediment concentration down the hillslope ($Conc_H$) for each rainfall application at the 7 sites. Open circles represent channelised or gully pathways whereas closed circles represent dispersive pathways.

We related the rate of change of sediment concentration, c with distance down the discharge hillslope, x by the expression

$$dc/dx = -kc$$

where k is a fitted constant, (Table 3). Results suggest that sediment concentrations in flow on dispersive pathways are substantially reduced for both total and suspended (< 63 μm) sediment concentrations. The negative sign for the culvert discharge outlet indicates an increase in the suspended sediment concentration as flow travelled downslope due to the storage of material within the gully itself.

Table 3. Percentage reduction in sediment concentration per 10 metres of hillslope with estimated standard error for mitre and culvert drains and for dispersive and channelised pathways.

	Mitre Drain	Culvert	Dispersive	Gully
Suspended (< 63 μm)	10± 9.1	-30.1± 11.2	37 ± 8.3	3 ± 12.1
Total Sed Conc	8.4 ± 7.1	5.2 ± 9.9	34.3 ± 5.8	15.6 ± 11.1

6. CATCHMENT SEDIMENT BUDGET

Infiltration rates, road contributing areas and mean sediment concentrations at the outlet for each road type were used to construct a sediment budget for road-derived material for Cuttagee Creek catchment. Total road lengths for each road class and the spatial distribution of drainage types were determined from GIS data stored within ArcView (see Mockler and Croke, this vol). We estimate that for a 100-y storm event of 30-min duration, a total of 39 tonnes of road derived sediment is generated under the conditions of road usage as represented by the sites at the time of the experiments. Using the degree of connectivity between road drainage outlets and streams outlined in Mockler and Croke (1999), we suggest that approximately 7 tonnes, or 17% of this road-derived material would be delivered directly to first order streams in this catchment. Gully development at road discharge outlets, therefore, plays a critical role in the delivery of road-derived material to streams. As suggested previously, the most severe gully erosion occurs on roads that are drained by culverts with large contributing areas and steep discharge gradients (Mockler and Croke, 1999). There is also a notable spatial and temporal coincidence between gully erosion at the road discharge outlets and roads that had been constructed immediately prior to the extremely large-scale rainfall event of March 1975, which delivered 360 mm of rainfall in 24 hours (see Dawes and Croke, this vol). Our observations of very limited storage of fine-grained material within the creeks draining this catchment suggest that much of this material is likely to be stored within the downstream floodplains and the estuarine Lake Cuttagee. The large scale floodplain sedimentation reported for the neighbouring Dry River (Murrah River) has also been attributed to this event (see Wallbrink *et al.*, this vol).

7. SUMMARY

This study quantified sediment generation rates on a range of forest roads in the Cuttagee Creek catchment of south eastern NSW. Generation rates were significantly correlated to road usage and the availability of sediment on the road surface, itself a function of traffic intensity and the sequence of rainfall events immediately following periods of intense road usage. Sediment delivery of this material to the stream network varied according to the type of drainage structure and the nature of the flow path on the discharge hillslope. Concentrated flow paths caused by gully erosion at drainage outlets significantly increase the percentage suspended material delivered downslope. The prevention of gully erosion through appropriate spacing of structures is clearly a matter of considerable significance for forest management (see Mockler and Croke, this vol). In constructing a total road-derived sediment budget for the catchment, we determined that ~ 17 % of mobilised sediment was delivered directly to first order streams for a high intensity rainfall event exceeding 75 mm hr^{-1}.

8. ACKNOWLDEGMENTS

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Filter Strip Effectiveness: Changes due to Vegetation Disturbance and Antecedent Moisture Conditions

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Summary: We examined the effects of surface soil disturbance and antecedent moisture on vegetated filter strip sites in northern NSW using rainfall simulator applications and in-situ TDR measurements. Treatments included a 10m filter strip disturbed by machinery entry, 10m and 15m undisturbed filter strips, each receiving sediment from a 100 m^2 area of bare, freshly disturbed snig track upslope. Data from the two undisturbed filter strips confirm the overall effectiveness of these areas in reducing total sediment loads, with trapping efficiencies ranging between 77% and 99%. On the disturbed filter strip, concentrated flow paths developed in the vicinity of the bulldozer tyre tracks, producing a *source* of sediment. Under the experimental conditions examined here, decreasing the water holding capacity of the surface soil through successive rainfall applications did not limit the overall effectiveness of these areas in trapping sediment. The importance of surface roughness and vegetation density in reducing flow velocities, inducing sediment deposition and storage were observed.

1. INTRODUCTION

Under the current NSW Pollution Control Licence (EPA, 1988) approximately 15-20% of State Forests harvestable timber reserved in streamside buffers (State Forests, *pers comm.*). These buffer or filter strips (the term varies on a statewide basis and will be referred to here generically as filter strips) serve several important functions for water pollution and ecological protection (see summary in Hairsine 1998). In terms of water pollution protection, increased hydraulic roughness within the filter strip, largely determined by the density of vegetation and obstacles such as logs, fallen debris and leaf litter, act to slow surface flow velocities and induce sediment deposition (see Barling and Moore, 1994; Hairsine, 1998). Secondly, high hydraulic conductivities within these vegetated areas promote increased surface water infiltration, limiting the delivery of overland flow and pollutants to streams.

Overall, the literature confirms that filter strips perform well in relation to both these functions (see reviews in Clinnick, 1984; Norris, 1993; Barling and Moore, 1994). However, certain conditions result in reduced effectiveness and it is essential that we can identify these factors to best plan the design and location of filter strips in catchments. Sediment trapping within the filter strip may be reduced, for example, if flow is concentrated entering, or passing through, the filter strip (Fitzpatrick, 1984). In terms of management implications, this may effect the minimum distance required between concentrated discharge points (eg from roads or tracks) and the nearest drainage line. The prevailing runoff generating mechanisms and soil hydraulic properties will also effect the ability of the filter strip to reduce the volume of overland flow. Streamside filter strips may act as runoff *sources* due to rising groundwater levels in 'wet areas' immediately adjacent to the stream (Hill, 1996). The trapping of very fine-grained material is likely to be highly dependent upon runoff infiltration mechanisms within the filter strip.

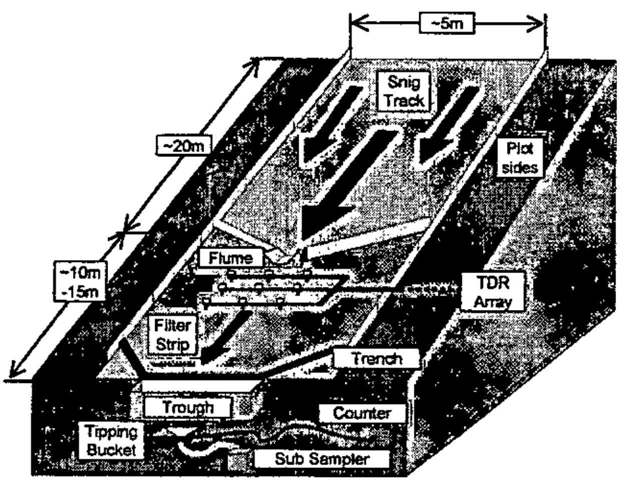
Here we report the results of rainfall simulator experiments on selected filter strips in Coffs Harbour, NSW. The primary objective of the experiments was to investigate changes in filter strip effectiveness due to soil cover disturbance, specifically machinery entry, and increasing soil moisture status.

2. STUDY AREA

Three sites were selected in the Orara West State Forest in Urunga Management Area of Coffs Harbour. Runoff and sediment generation data were previously reported for these sites under natural rainfall events by Lacey (1998). Erosion data for three artificial rainfall applications are also summarised in this volume (Lacey *et al.*, this volume). The sites had an upslope contributing area of bare snig track ~ 100 m^2 in size on steep 22 degree slopes, which delivered sediment into a vegetated filter strip, ~ 50 to 100 m^2 in area, on similar slopes. The general layout of each plot is summarised in Figure 1. The three sites represented conditions reflecting;

- a 10m filter strip disturbed by machinery entry (Site 1);
- a 10m undisturbed filter but with patchy understorey and reduced roughness elements (Site 2) and
- a 15m undisturbed filter with high roughness elements as a result of a dense under-and overstorey and extensive ground cover protection (Site 3).

Several passes of a bulldozer freshly disturbed each snig track prior to the simulations. Site 3 was an undisturbed control site in previous monitoring experiments (Lacey, 1998) and the snig track was created specifically for these experiments. Two artificial rainstorms equivalent to the 3-y and 50-y rainfall event, with average rainfall intensities of 75 and 130mm hr^{-1} were applied at each site. Sediment discharge measurements were taken at 3-minute intervals at the exit of both the snig track and the filter strip using 2 RBC flumes and manual sampling (Fig. 1). Additional measurements were taken within the filter strip at selected distances using a vacuum sediment sampler. Changes in volumetric soil moisture content within the filter strips were determined using a multi-probe, Time Domain Reflectrometer (TDR). Sediment concentration and flux were determined from measurements of total sediment weight and discharge. A representative sediment sample from each rainfall event was analysed for aggregate particle size using the procedure outlined in detail in Croke *et al.*, (1997).



3. RUNOFF GENERATION

Undisturbed filter strips (Sites 2 & 3), without any incoming source of runoff from the snig tracks, did not generate significant runoff. Runoff was patchy and widespread infiltration occurred over much of the filter strip area. On Site 3, for example, it took 20 min for continuous overland flow to reach the bottom of the filter strip area and much of the runoff was generated in a small contributing area towards the base of the plot. Apparent infiltration capacities determined from the steady state discharge and rainfall rate, ranged from 60 to 119 mm hr^{-1} on the filter strips. These values assume a spatially averaged infiltration rate across the entire filter strip area but we observed a high degree of variability in response time and pattern of runoff generation (see Section 6 below). The dominant source of runoff was water entering the filter strips from the snig track upslope. Under these conditions, runoff coefficients ranged from 5% to 25% on the undisturbed filter strips for the first rainfall event of 75 mm hr^{-1} and increased with successive rainfall applications (Table 1). Under the more extreme rainfall intensities at Site 2, significant increases in the volume of overland flow from the snig track produced a three-fold increase in the runoff coefficient (Table 1).

On the disturbed filter strip (Site 1), runoff discharging onto the area from the snig track upslope created two distinct rills ~ 0.5 m wide in the vicinity of the bulldozer tyre tracks within 10 minutes of the first simulation. These rills concentrated flow and the majority of runoff from the snig track was delivered to the base of the plot in these channels, as reflected in the runoff coefficient of 97%. Under higher rainfall intensities flow width increased to ~ 5 m and shallow sheet flow was observed over a much larger area of the filter strip and the runoff coefficient reduced to 73% reflecting infiltration in the less disturbed parts of the filter strip.

Table 1. Runoff characteristics for the filter strip areas at each site.

Site	Run	Site Description	Apparent Infiltration mm hr^{-1}	Runoff Coefficient %
1	1	Disturbed 10m	46	97
1	2	Disturbed 10m	77	73
2	1	Undisturbed 10m	65	25
2	2	Undisturbed 10m	69	87
2	3	Undisturbed 10m	71	97
3	1	Undisturbed 15m	73	5
3	2	Undisturbed 15m	119	29

4. SEDIMENT TRAPPING

Mean sediment concentrations and total yields entering, and leaving the filter strips are summarised for each site in Table 2. A single low intensity rainfall application on the undisturbed filter strip at Site 3, with no incoming source of sediment from the snig track, indicates a 'background' sediment concentration of ~0.5 g/l for these areas. On the two undisturbed filter strip sites (Sites 2 and 3), mean sediment concentrations in the runoff leaving the filter strip were significantly lower than sediment concentrations entering the area from the snig tracks upslope (Table 2). We observed runoff pathways infiltrating under dense leaf litter and considerable material remained stored in litter dams and surface depressions. Sediment trapping ranged from 77% to 99%, confirming their overall effectiveness in reducing total sediment loads (Table 2). Trapping efficiency declined from 95% to 77% between runs 2 and 3 and increased again to 95% during the last rainfall application on Site 2. The reduction in sediment trapping during run 2 is largely explained by remobilisation and transport of deposited sediment. We observed sediment dams mobilise and release an additional supply of material for transport. Roughness elements and cover density are critical factors in maintaining the trapping efficiency of the filter strip, especially under high intensity rainfall and on sites with steep slopes. Sediment storage within the vegetation has a finite limit, however, and with successive high intensity rainfall events, re-entrainment of deposited material is likely to occur. The density of ground cover vegetation and roughness elements at Site 2 was lower than that at Site 3 where we did not observe remobilisation of stored material.

A mass balance constructed from sediment flux data on individual particle size classes indicated that while maximum trapping efficiency occurred within the coarse to medium particle size range, on average, 50% of the very-fine grained (<63 μm) material was also trapped within the undisturbed filter strips.

Table 2. Summary of the filter strip trapping efficiency for total sediment, and coarse (>63μm) and fine (<63μm) particle sizes.

Site	Run	Sed Con g/l IN	Sed Con g/l OUT	Tot Sed. (kg) IN	Tot Sed. (kg) OUT	% Tot Sed. Trap.
1	1	20	43	38	139	+46
1	2	19	35	88	182	+25
2	1	91	17	139	7	95
2	2	59	15	182	41	77
2	3	31	1.6	115	6	95
3	1	200	0.09	249	0.06	99
3	2	102	1.5	209	0.85	99
3	FS only		0.5			

+= Net erosion and sediment transport.

5. EFFECT OF SOIL DISTURBANCE.

In contrast to the undisturbed filter strip sites, sediment discharge data from the disturbed site (Site 1) indicate that surface erosion occurred *within* the filter strip (Table 1). Mean sediment concentrations leaving the filter strip were at least twice as high as those in runoff entering the area from the upslope snig track area (Table 2). Total sediment loads from the filter strip increased by 50% and 25% for the two rainfall events respectively. Evidence of erosion within the filter strip was also confirmed by the particle size data. We measured a significant increase in the percentage coarse material delivered to the plot exit during this simulation, attributable to surface soil erosion during rill initiation in the tyre tracks. The percentage coarse material transported through the filter strip during this second event decreased.

Disturbance of the understorey vegetation by machinery entry clearly effected soil cohesion to the point that it had a significant impact upon sediment transport rates and conversely the sediment trapping ability of this area. Results from other field studies confirm that where surface soils and vegetation density are reduced through disturbances such as stock trampling, the resistance of the soil to erosion is reduced thereby inducing flow concentration and increased sediment transport capacities (Prosser *et al.*, 1995).

6. EFFECTS OF ANTECEDENT MOISTURE

The effect of antecedent moisture conditions on sediment trapping efficiency are best observed at Site 2 due to a combination of compounding factors that limit result interpretation in the remaining two sites. Any effect of soil moisture at Site 1, for example, would have been overridden by the filter strip disturbance, rill development, deposition and re-entrainment of sediment within the filter strip. Likewise, despite two wetting up events at Site 3, soil moisture contents for run 2 displayed considerable spatial variation as shown by individual TDR probe values (Fig. 2). Soil heterogeneity appears to have prevented filter strip saturation across the entire area and runoff coefficients for this area remained low even during rainfall intensities of 110 mm hr^{-1}.

The spatially averaged TDR record for runs 2 and 3 for Site 2 indicate volumetric soil moisture contents (θ) increasing sequentially with rainfall application (Fig 3). Although no soil moisture data exists for run 1 due to equipment malfunction, a pre-run observation indicated moisture contents to be 0.17 cm^3 cm^{-3}. Hydrographs reflect these conditions, with infiltration decreasing as soil moisture rises.

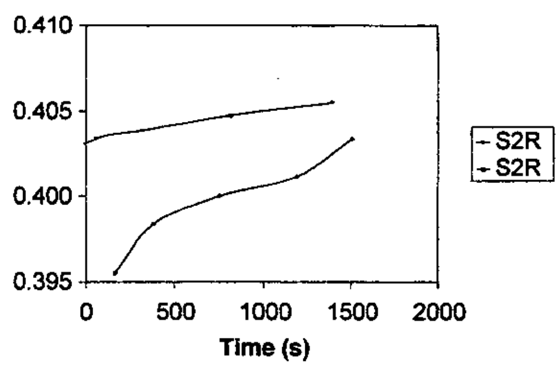


Figure 2. Spatial variation in soil moisture content, Site 3 run 2.

Trapping efficiencies for runs 2 and 3 at Site 2 ranged from 77% to 99% but as suggested previously, the reduction in trapping rates during run 2 was largely explained remobilisation of previously deposited material as litter dams and vegetation structure weakened. All three runs displayed a very similar fine particle size distribution of material leaving the filter strip suggesting limited effect of antecedent moisture conditions on fine-grained sediment deposition.

These results indicate that roughness elements in the form of ground vegetation and in particular leaf litter are very important in trapping sediment. Under the experimental conditions investigated here, these factors would appear to equal the importance of runoff infiltration into unsaturated soil in trapping total sediment loads.

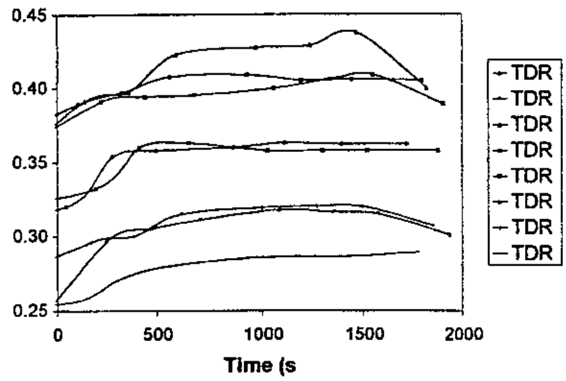


Figure 3. Spatially averaged soil moisture content at Site 2 runs 2 and 3.

Development of saturated overland flow as expected in near-stream areas, would produce a greater depth of runoff than the patchy Hortonian flow generated on this experimental hillslope, especially under such steep slopes. The runoff generating mechanism may thus be very important to the overall effectiveness of the filter strip

7. CONCLUSIONS

These experiments examined the effects of surface soil disturbance and changing soil moisture on three forest filter strips in the area around Coffs Harbour NSW. Sediment and runoff data from the two undisturbed filter strips confirm the overall effectiveness of vegetated areas in trapping sediment and reducing overland flow volumes. Weakening of the vegetation density and surface soils by machinery entry into the filter strip had a significant effect on the ability of this area to trap sediment. Our observations of relatively rapid rill initiation suggest that the effectiveness of these areas is very sensitive to flow concentration and the associated effects on flow velocities and sediment transport capacities. Concentrated flow paths have the potential to limit the effectiveness of these vegetated areas. The effects of increasing soil moisture status on sediment trapping and infiltration appear to be largely overridden by the trapping efficiency of roughness elements such as leaf litter and surface obstacles. Our observations of runoff generation mechanisms within the filter strips confirm the existence of limited Hortonian runoff generation in these areas. The high degree of spatial variability in moisture capacity, however, supports the large spatial variability in soil properties and hydraulic conductivities characteristic of forest soils (Bonnell, 1993).

These experiments could not address the potential effects of rising groundwater and the development of wet areas immediately adjacent to the stream on filter strip effectiveness. Result interpretation is also limited by the small number of field sites and by the lack of site replication. A more detailed assessment of the full range of potential disturbances, such as tree felling and dragging, and their impact on filter strip effectiveness is required.

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Modelling Runoff Generation and Delivery from Roads with TOPOG : What We Can Do and When We Should Do It

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Summary: TOPOG is a small-catchment, hydrologic model developed by the CRCCH since 1988. This contour-based model has been modified to allow non-topographic features, such as roads, to affect water flow. These modifications are ideally suited to modelling the impact of roads on flow accumulation and sediment transport on a storm basis. Model output is compared with field data from a series of rainfall simulator experiments in a 57 km^2 catchment in south eastern NSW.. A further application, of a major event that occurred soon after forestry operations and caused major gullying, is used to explore the sensitivity of the water and sediment balance to the presence of stream channels.

1. INTRODUCTION

The degree to which forest roads interact with, and alter, the hydrology and sediment delivery pattern of a catchment is an issue of considerable concern in environmental management. Many researchers are now exploring the possibilities of physically-modelling the interaction between these impermeable surfaces and the natural drainage system, both to advance our understanding of the catchment-scale impacts and to use this information in the design and planning of new roads in the landscape. Existing methods include the Water Erosion Prediction Project (WEPP) model and a number of GIS based flow-routing approaches, but these are not well-equipped to investigate the potential impact of road surfaces on hillslope or catchment hydrology.

The TOPOG modelling tools (Dawes and Hatton 1993, Dawes and Short 1994) have been used in many studies of water balance (e.g. Vertessy et al. 1993, Dawes et al. 1997), and water balance in conjunction with plant growth (Vertessy et al. 1996, Zhang et al. 1999). These applications use only the natural topographic variation to determine water flow directions. In catchments where roads and forestry operations are present, surface water flows are modified according to the layout of roads and their drainage outlets. To properly model these effects, a transient TOPOG model (_dynamic V9.11) was modified to allow an arbitrary redirection of flow that did not push water uphill. The model was further modified to allow the accumulation, but not routing, of water and sediment into streams within land elements. The model defines the presence or absence of streams within each specific land element.

A series of rainfall simulator experiments on a range of forest roads in the Cuttagee Creek catchment near Bermagui, NSW, provided the opportunity to test the performance of the model with field data (see Croke *et al.*, this vol). Our immediate tasks were to

(i) model runoff generation on a road surface and compare the predicted hydrograph with that observed from the rainfall simulator experiments, and

(ii) model runoff generation and delivery within a sub-section of the catchment which contains cut-fill type roads that have caused significant gully development (see Mockler and Croke, this vol). In this scenario, we ran the model with and without road redirection, and with and without stream contributions.

The first task gives us confidence the model actually works and is simulating the processes observed in the field, while the second allows us to examine the geomorphic response of a hillslope to runoff delivery from roads, in particular, in relation to gully development and stream connectivity. The results of the sensitivity work tell us when we should use TOPOG and when other methods or models are more appropriate.

2. STUDY CATCHMENT

Cuttagee Creek is a predominantly forested catchment covering 57 km^2 of the coastal lowlands in the Eden Forest Management Area (EFMA) of south eastern NSW. The catchment is serviced by 75 km of roads, representing approximately 1% of the catchment area, with a mean density of ~2 km/km^2 (Mockler and Croke, 1999). Twenty-five percent of the road network was constructed in 1964, five years before logging first commenced within the catchment. A further 70% of the road network was constructed between 1971 and 1976, with relatively minor road network expansion in later decades. Road construction during the period post-1976 was confined predominantly to access roads servicing log dumps used during specific harvesting cycles. Road maintenance standards vary according to the road class which include secondary access (Class 4), feeder access and dump access (Class 6). Class 4 roads represent well graded and crowned roads whereas Class 6 roads are used sporadically between cutting cycles and abandoned for large parts of the year. Road surfaces are drained by simple mitre or push-out structures that redirect runoff and sediment onto the adjacent hillsides and reflect the predominant ridge-top characteristics of the study area. Croke *et al.* (this vol) report runoff and sediment generation rates for 7 road segments within the catchment. For the same catchment, Mockler and Croke (1999) report on the degree of road to stream connection and highlight the extent of fully-connected pathways between road drainage outlets and the natural stream network.

3. RAINFALL SIMULATOR HYDROGRAPHS

Two hydrographs were selected at random from the field experiments reported by Croke *et al.* (this vol). The depth-discharge relationship used to infer flux was translated into TOPOG and reduced to a second-order polynomial. The road layout was digitised into ARC/INFO, projected onto a topographic analysis of the hillslope, and generalised to the scale of the land elements. TOPOG was run using the two rainfall rates applied in the experiment, and hydrographs of water were produced at the drain outlet of the road. Figure 1 shows the observed (solid line) and modelled (dashed line) hydrograph for a rainfall rate of 72 mm/hour, and using exactly the same conditions, Figure 2 shows the hydrographs for a rainfall rate of 106 mm/hour, for Site 4 on Road 7.

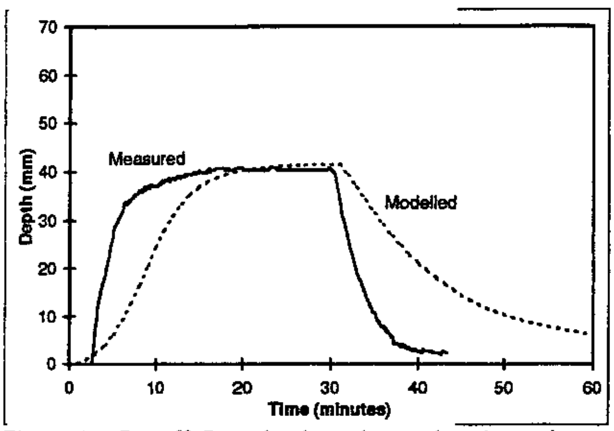

Figure 1 : Runoff flow depth at the road outlet, using a rainfall simulator at a rate of 72 mm/hr, Site 4 on Road 7.

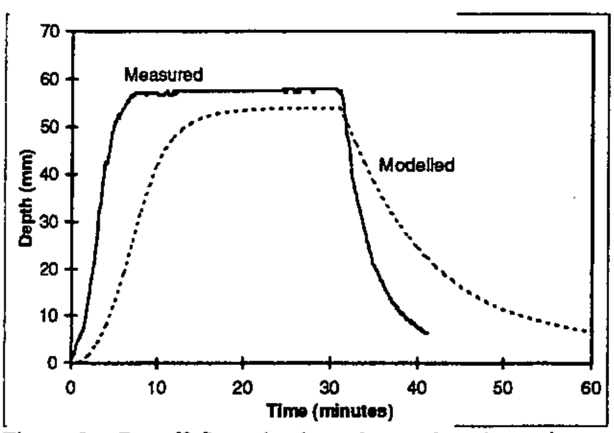

Figure 2 : Runoff flow depth at the road outlet, using a rainfall simulator at a rate of 106 mm/hr, Site 4 on Road 7.

The consistency of the modelled traces provides confidence that the model is reproducing the processes occurring in the field on this forest road site.

4. CUT-FILL ROAD SIMULATIONS

The rainfall simulator data shown in Figures 1 and 2 were obtained from a ridge-top road, ie. where the road surface provides the only source of runoff to the drainage outlet. Cut-fill roads on the other hand, receive runoff contributions from the road surface, from the upslope contributing area, and any sub-surface flow contributions intercepted by the road cut (see Croke *et al.*, this vol). Figure 3 shows the steady-state natural flow accumulation pattern for the selected 16 ha hillslope along Nutleys Road within Cuttagee Creek catchment. Natural streams show up clearly as areas of high flow accumulation, or soil wetness in this presentation

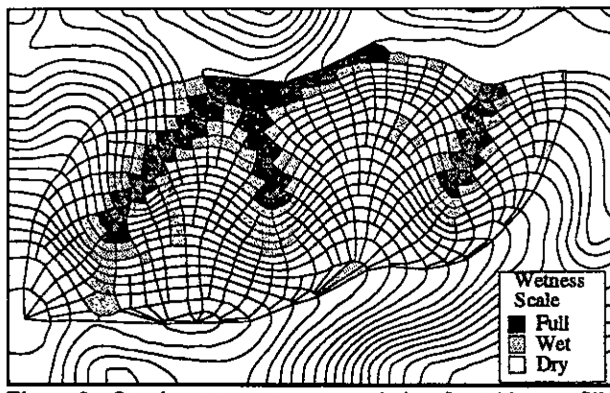

Figure 3 : Steady-state water accumulation for 16 ha cut-fill road section along Nutleys Road using only natural topography.

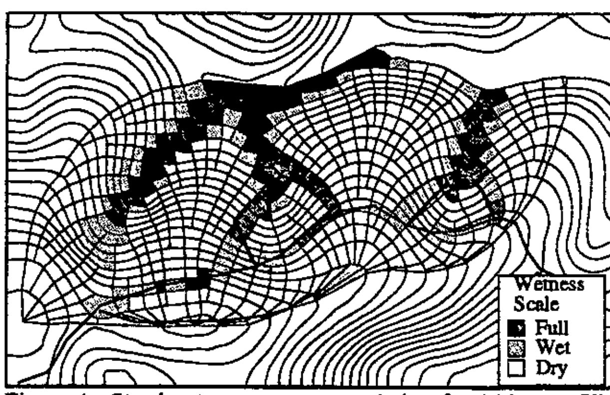

Figure 4 : Steady-state water accumulation for 16 ha cut-fill section of Nutleys Road using road redirections, showing road location (dark lines) and eroded gullies (thick lines).

In contrast, Figure 4 shows how the natural accumulation patterns are altered by the presence of a forestry road, shown as a dark line. The four drain outlets occur at the top of the new areas of high accumulation, and coincide almost exactly with the mapped gullies, shown as thick lines (see Mockler and Croke, 1999). The western-most gully in Figure 4 appears to have been incorrectly surveyed, or there is some other man-made or vegetation structure causing the gully to have eroded almost parallel to a contour on a slope of approximately 30°.

The contributing area, road length and slope of each element at a drain outlet were extracted from TOPOG, and these values along with the gully connection threshold value of Mockler and Croke (1999) are shown in Table 1; note drain outlets are numbered from left to right on Figure 4.

It is clear that consideration of the gully threshold value alone would indicate a gully hazard without any water or sediment balance modelling. The Contributing Area Ratio, ie. the contributing area including roads divided by the contributing area of natural flow patterns only, shows how dramatically the roads alter runoff accumulation and could cause gully initiation (see Mockler and Croke, this vol).

Table 1 : Observed and critical values of the Mockler and Croke (1999) gully connection threshold, and the increase in contributing area via road diversions, for four road drain outlets along Nutleys Road.

Outlet Number	Contributing Area Ratio	Road Length	Mockler-Croke Threshold
1	9.3	180	56
2	17.4	160	66
3	13.4	90	62
4	20.4	140	83

Following a logging operation late in 1974, an extreme rainfall event occurred in March 1975 onto the relatively bare surface. Between the 9th and 13th of March 638 mm of rain fell, including 360 mm on the 12th of March alone. This latter storm was transiently modelled with _dynamic as a single event lasting 3 hours, because details on the actual storm duration were not available and the implied rate matches the highest intensity from the rainfall simulator experiments reported earlier.

Five scenarios were modelled to examine the sensitivity of the hillslope to changes in hydrology caused by road runoff and redirection

- topographic flow only without roads or streams,
- road modified flows without any streams,
- topographic flow only with original streams,
- road modified flows with original streams, and
- road modified flows with streams and mapped gullies.

With the last three scenarios, contributions of water and sediment fluxes are not routed within the stream but simply accumulated. Table 2 lists predicted water and sediment fluxes expressed as ratios of the values obtained for Case 3, topographic flows with original streams.

One important point to note about the sediment results in Table 2 is that the sediment transport algorithm used in TOPOG assumes that sediment is always available and that the transport capacity of the water is the limiting factor. Sediment was allowed to be picked up from road surfaces, and since these carry a majority of flow, their contribution will be overstated. Data from rainfall simulators shows that there is an initial flush of sediment and a rapid recession in sediment concentration (Croke *et al.* this vol). The modelled values from TOPOG are therefore likely to overestimate sediment yields. However the result that loads are likely to increase with the presence of roads is consistent with observations.

Table 2 : Water and sediment delivery amounts for cases involving streams, gullies and the redirection of flow by roads, expressed relative to control Case 3.

Case	1	2	3	4	5
Sediment Mobilised	1.6	32.2	1.0	31.6	30.8
Sediment to Stream	0.0	0.0	1.0	1.0	1.2
Total Sediment Export	0.7	0.8	1.0	1.0	1.2
Water to Stream	0.0	0.0	1.0	1.1	1.2
Total Water Exported	0.7	0.8	1.0	1.1	1.1

This also has implications for the hydrographs in Figures 1 and 2. Although the shape of the two hydrographs is reasonably matched between the observed and predicted, in both cases there is poor correspondence with the rising and recession limbs of the hydrographs. If the situation was that sediment was freely available and not limiting, then all points on the water hydrograph are important for determining the amount of sediment mobilised and transported. If however sediment availability is the limiting factor, then as long as the hydrograph reaches the same peak value and entrains the available sediment, then only the timing can be in error. Further if the correct amount of water is shifted by the model then again it is only timing that can be affected.

5. DISCUSSION

What did we learn from the TOPOG simulations? Firstly, the presence of roads, and their associated drains, increases both the flux of water moving over the surface, and the amount of water simulated as exported from hillslopes, but not necessarily by a large factor. Using any sediment transport assumption, the amount of sediment mobilised and transported by any increased flux must be increased. The presence of streams and channels, and gullies at drain outlets in particular, allows suspended sediment to bypass deposition on a vegetated surface and therefore to be exported from a hillslope.

What did we learn from observation, and a threshold based on some simple measurements? Road drain outlets are often the head of gullies. It is essential, therefore, that gully erosion at drain outlets be prevented through appropriate spacing of drains (see Mockler and Croke, this vol).

So what is the recommended sequence of steps for determining the impact of road drains on gullies? The first step should be a survey operation to rapidly assess the data required for a threshold technique, such as that of Mockler and Croke (1999), for the planned road network. This, along with some estimate of soil erodibility, would quickly establish the maximum desirable distance between drains along any section of road. Once the practicalities of installing the engineering works are observed, specific drains at sensitive (highly erodible soil or high slope) or critical parts (near existing drainage lines or associated with cut-fill roads) might be simulated using a model such as TOPOG to determine the likely magnitude of fluxes generated from design or historical storm data, and in concert with other empirical equations determine the likelihood of gully initiation and erosion.

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Tracing Sediment Flows into Buffers: Causes and Consequences

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Summary: This paper presents the results of an investigation into the frequency and nature of unmanaged sediment flows into stream buffer strips in recently logged coupes in the Tarago catchment in Victoria. Observed instances of unmanaged flows are outlined and the implications for forest management and buffer design discussed.

1. INTRODUCTION

Soil erosion is common during and following rainfall events on logged areas and a range of management actions are aimed at ensuring that sediment laden overland flows do not develop sufficiently to threaten either site productivity through soil loss or catchment water quality through pollution of streams. The general approach to water quality protection is to manage erosion processes on the logged area through dispersion and infiltration of developing overland flows, and to provide additional protection to the streams themselves by retaining undisturbed vegetation, or buffers, between them and the disturbed area. From a water quality point of view these buffers serve a number of functions, They protect the hydrologically active riparian zone from disturbance, contribute to streambank stability and act as infiltration zones for runoff generated in the logged area. This last function is the only one which usually requires that the buffer extend beyond the edge of the riparian zone and as such is the focus of a design criterion based on the premise that the buffer should be capable of infiltrating some or all of the runoff from the adjacent hillslope. In the case of forestry in particular, with its emphasis on in-coupe management and dispersion of runoff, the infiltration requirement of the buffer relates only to that which is generated close to the buffer. Under normal logging practices in Victoria, this usually means the boundary track of the logging coupe, a fire control track around the edge of the logged area, and the harvested area which drains onto it.

Since these areas are generally uncompacted the expectation is that concentrated overland flows will not occur and the buffer will only have to infiltrate dispersed or low energy flows directed into it by drainage measures such as crossbars on boundary tracks. Buffers, characterised by permeable soils and surface roughness, are efficient at containing such flows and generally do so over a relatively short distance (Hairsine, 1996).

As part of an evaluation of a range of design criteria for stream buffers in the Tarago catchment, the frequency and nature of sediment flows into buffers from sources other than boundary tracks and the adjacent uncompacted hillslope was investigated by carrying out a survey of recently established buffers. This paper details the circumstances in which observed sediment intrusions occurred and their fate. It discusses the observations in the context of buffer design and forest management.

2. STUDY AREA AND DATA COLLECTION

The West Tarago catchment, located in south-east Victoria, is a 4th order stream network which occupies an area of 65.4 km^2. The geology is predominantly granite and the topography is steep with deeply incised streams. Elevation of the watershed ranges from 200 m to 900 m with slopes up to 40°. The main forest type is mountain ash (*Eucalyptus regnans* F. Muell), with messmate (*E. obliqua* L'Herit.) on drier slopes and silvertop (*E. sieberii* L. Johnson) present on spurs and more exposed sites. The forests are important commercially and the area is an important watershed feeding a large reservoir.

A field survey of buffers adjacent to 12 coupes logged over the period Dec. 1992 to March 1998 was carried out between November 1997 and May 1998, approximately 10km in total. The buffer:coupe boundary was searched for evidence of excessive or unmanaged sediment movement into the buffer. Where located, unmanaged sediment flows into the buffer were traced to their source and mapped using DGPS. The sediment flows in the buffer were also traced as far as possible, using visual evidence of sediment deposition.

3. RESULTS AND DISCUSSION

A total of eight unmanaged sediment intrusions into the 10 km of stream buffer were found over the course of the study. All incidents were in logging coupes regenerated since 1996 (7.3 km of buffer surveyed), possibly due to difficulty recognising sediment flows on older coupes. Each incident was traceable to a primary source and in some cases secondary sources were also assessed as contributing to the sediment flow. Table 1 summarises the circumstances of each flow. The sediment sources can be roughly divided into failure of in-coupe sediment management and roading sources. The instances are presented as a series of individual case studies under these general headings.

In-coupe generated sediment flows

Case 1: The initial source of the sediment flow which entered the buffer was a large compacted area at the junction of the log landing, an internal road and a major snig track running downhill from the landing area. This source area had no evidence of vegetation development and soil surface exposure was close to 100%. The slope of the major snig track gradually increased from about 5° to 24°. The snig track was below the level of the adjacent *general harvested area*

(GHA) and while the first eight cross bars on the major snig track were still intact there was evidence of substantial sediment storage at the base of the bars and of sediment returning to the track from the GHA below the bars. There was deep rilling of the snig track surface between bars. Below the eighth crossbar the slope of the snig track increased sharply and the next seven crossbars failed due to breaching of the centre of the bar The next functioning crossbar was at a point where the snig track levelled out close to the coupe edge and directed the sediment flow onto and across the boundary track into the stream buffer, resulting in an extensive sediment fan at the entry point to the stream buffer and further deposition of coarse sediment for 14 m into the buffer. The slope of the buffer at the entry point and over the length of sediment deposition was relatively gentle (3° rising to 8°).

Case 2: This instance of in-coupe generated sediment flow entering the buffer was similar to that outlined above, though of lesser magnitude. The primary source was again the compacted area at the junction of a primary snig track and a log landing. The soil surface in this area was also close to 100% exposed. The first crossbar draining the primary snig track/landing junction was effectively neutralised by sediment deposition at its base allowing the development of sediment flow in around it. This flow then moved down an adjacent snig track running downhill at a slope of 8°, increasing to 22° further downslope. Vegetation development on this snig track was good, indicating that compaction and/or topsoil removal or displacement was not as severe as on the primary snig track. The six crossbars draining this snig track were all breached in a similar fashion to those in Case 1, although not as severely as some sediment remained in deposits at the base of the bars. Shallow rills were evident on the track between bars. The sediment crossed the boundary track into the buffer where a minor sediment fan developed with further deposition evident for a further 28 m over an average slope of 18°.

Case 3: The primary sediment source in this instance was again a substantial area of severe compaction at the junction of a log landing and a major snig track. This area and the snig track running to the buffer were gentle in slope (3-7°) and the failure of crossbars to effectively drain the compacted area appeared to be due to wide spacing of the bars, low bar height and sediment deposition at the base. The low height of the crossbars may have been due to frequent traffic following construction as there appeared to be frequent use of the coupe by firewood cutters. There was little vegetation or litter cover development for over 100 m downslope of the landing on the main snig track due to severe compaction and subsoil exposure. The sediment flow eventually drained due to outsloping onto a secondary snig track, breaching two crossbars on this track before crossing the boundary track and flowing into the buffer. Visual evidence of sediment deposition in the buffer continued for 12 m. The average buffer slope along the deposition zone was 7°.

Roading related sediment flows

Case 4 and 5: In both these cases culverts draining permanent roads had outlets close to the buffer, within 37 m and 27 m respectively. The area between the culverts and the buffer edge were GHA in both cases with average slopes of 15° and 14° respectively. The first flow was dispersed by logging slash just inside the buffer and sediment deposition was evident for a further 10 m into the buffer over an average slope of 15°. The second flow crossed the boundary track and created a sediment fan on entering the buffer. Sediment deposition was evident for a further 19 m at an average slope of 20°.

Case 6: The source of this sediment flow was a push-out drain draining a metalled logging road passing within 21 m of a buffer protecting the headwaters of a stream. The drain extended 5 m into the GHA , and despite substantial sediment detention in the GHA there was sediment discharge onto the boundary track over an average slope of 18°. The flow followed a shallow rill along the track to the next crossbar (10.7 m @ 15°), depositing sediment at the base of the bar and extending into the buffer for 22 m at an average slope of 28°.

Case 7: The most serious of the sediment flows originated on a section of unmetalled roading which was realigned to create a large S-bend. A table drain draining the road at the lower apex of the bend created deep rills in a large fill slope and discharged onto a snig track below the road. It breached one crossbar on this track, flowed down an adjoining track before being diverted onto the GHA and then to the boundary track through a series of deep rills. The sediment flow on the boundary track breached six successive crossbars, constrained in a deep rill along the inside edge of the track. The slope of the track ranged from 13° up to 34° over one section. A combination of an outsloping section of boundary track and a crossbar diverted the sediment flow into the buffer, where sediment deposition continued to the stream over a distance of 46 m at an average slope of 28°. A considerable sediment fan combined with streambank collapse partly diverted the stream at the point of entry.

The magnitude of the sediment flow resulted from a number of sources. The drained upslope section of road was 158 m with an average slope of 8°. Additional sediment was contributed from a junction with a log landing and sections of the original road alignment and a separate major snig track which drained onto the road across the cut batter. Deep rilling of the table drain was evident upslope of the discharge point. Discharge onto the fill slope mobilised further sediment, while deep rilling of the boundary track and crossdrain failure were additional sediment sources.

Case 8: The source of this sediment flow was a cross drain on a lower section of the road described above. This drain drained 118 m length of road at an average slope of 8°. The drain discharged onto GHA, forming deep rills over 15 m at a slope of 21° before discharging onto a severely compacted area at the junction of the landing and two primary snig tracks. The flow followed shallow rills for 20 m with no evidence of sediment deposition, crossed a 12 m strip of GHA, again forming deep rills with little deposition, discharged onto the boundary track before being diverted into the stream buffer at the next cross bank, 25 m down the boundary track at 18°. Considerable sediment deposition was evident at the entry point to the buffer and continued for a further 20 m into the buffer over an average slope of 23°.

4. CONCLUSION

The following points emerge from this set of case studies;

- Unmanaged sediment flows into stream buffers in logging coupes are relatively infrequent.
- Initiation of unmanaged sediment flows into buffers was often some distance from the buffer.
- Contiguous compacted areas at the junction of landings and primary snig tracks are an important source of sediment.
- Cross drains on snig tracks can fail due either to incorrect construction or flow exceeding design as upslope structures fail.
- More attention needs to be paid to the draining of roads passing through logging coupes.
- Sediment movement into the buffer frequently exceeded the minimum width (20 m), but only reached the stream in one case.
- Rill development was invariably stopped by buffers.
- Interception of unmanaged sediment flow is an important function of buffers but the nature of the flows reported here indicate that is would be difficult to predict where non-road sources are likely to occur.

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Table 1. Details of unmanaged sediment flows in recently logged coupes in the Tarago catchment.

Case No.	Year of regeneration	Primary sediment source	Secondary sediment source/s	Length of flow* (m)	Average slope	Buffer location	Depth of penetration (m)	Entry to stream
1	96	Compacted area (Log landing, snig track and internal road junction)	Snig track, failed crossbars	210	16.8°	Flank	14	no
2	96	Compacted area (Log landing/ snig track junction)	Snig track, failed crossbars	124	13.4°	Flank	28	no
3	96	Compacted area (Log landing/ snig track junction)	Snig track, failed crossbars	153	7°	Flank	12	no
4	96	Culvert, metalled road	None	53	10.7°	Headwater	10	no
5	96	Culvert, metalled road	None	28	14°	Flank	19	no
6	96	Pushout drain, metalled road	None	39	10°	Headwater	22	no
7	98	Table drain, unmetalled road	Road fill slope/ snig tracks, failed crossbars	123	16°	Flank	46	yes
8	98	Cross drain, unmetalled road	Compacted area (log landing /snig track junction)	76	17°	Headwater	20	no

* from source to buffer edge

The Soil Stability Factor in Erosion Hazard Assessment: the NSW Approach

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Summary: The assessment of erosion hazard for forestry operations in NSW state forests was recently revised, replacing a USLE based approach with a set of look up matrices developed by an expert panel. The erodibility factor in this new system is termed regolith stability, and combines both soil coherence and sediment delivery potential to give a relatively simple four class rating which is then combined with the other variables of slope and rainfall erosivity in the matrix tables. The classification is based on soil morphological information combined with experience of soil behaviour associated with logging. The regolith stability classification has been applied across all State Forests in the eastern half of NSW at a broad scale of mapping using existing soil and geological surveys, and is verified by more detailed survey during harvest plan preparation.

1. INTRODUCTION

Regolith stability is an expression of combined soil and substrate erodibility, and sediment delivery potential. It is one of the input variables to the soil erosion and water pollution hazard assessment for the Environment Protection Authority (EPA) 1998/99 Pollution Control Licence for State Forests' logging operations. The hazard assessment framework has been cooperatively developed by the EPA, the Department of Land and Water Conservation (DLWC) and NSW State Forests.

The concept of regolith stability was proposed by Ryan (1996) to focus the classification of soils and parent materials into classes that relate directly to hazard associated with logging operations. This classification system incorporates knowledge about the erosive response of soils to logging operations derived from both field experience and from research, with data on regolith properties and distribution.

2. BACKGROUND

The procedure for forest erosion hazard assessment in NSW was significantly revised over the period1996-97, resulting in a shift away from the former USLE based system to a set of look up tables developed by an expert panel. The look up tables comprise a set of matrices which combine the principal variables that influence the erosion and sediment delivery process. These are rainfall energy, slope gradient, regolith stability and the nature of disturbance associated with a particular operation.

The need for a new classification of soil stability reflects the lack of an appropriate quantitative predictor of forest soil erodibility. The USLE soil erodibility factor, K, has been shown by field experience in many situations to relate poorly to the behaviour of forest soils. The K factor relates specifically to the detachment of soil through sheet and rill erosion and not other processes of erosion, most notably gully erosion. The K factor also does not account for susceptibility of soil material to transport and delivery to receiving waters. The new concept of regolith stability overcomes these shortcomings by using expert assessment to relate the regolith stability classes to observed soil behaviour in the field.

A further advantage of the regolith stability approach is to change the scale of the initial determination of erosive resistance and sediment delivery potential from that of a soil sample to broader scales more appropriate for planning. In the previous USLE based hazard assessment system, considerable effort was frequently expended in field sampling and/or laboratory analysis on a theoretically representative set of samples in an attempt to determine a K value at the compartment scale. The new approach permits an initial, broader scale assessment which incorporates experience and knowledge of soil behaviour for the particular landscape unit from a range of similar sites. Subsequent site assessment at the harvest planning stage verifies the accuracy of the broader scale regolith stability classification for particular logging compartments and describe significant variability at a more localised scale.

3. CONCEPTS

Table 1 presents the conceptual framework for the classification and broadly outlines the various regolith types that fall into each class. The table shows that the classification is a two by two matrix, giving four possible classes of regolith stability. This level of detail corresponds with that of the other input variables for the hazard assessment matrices and the overall requirement to ensure the practicality of the matrix approach.

This includes both the soil as well as weathered parent materials and substrates down to hard bedrock. This is appropriate for forested land as large areas of state forest are situated on steep slopes where the mantle conventionally regarded as soil is shallow, while the material with which forest operations is interacting may be considerably deeper.

For the purpose of this classification regolith stability is assessed to a depth of one metre. This depth is considered to include most of the material likely to be exposed in a logging operation.

As table 1 shows, regolith stability comprises two components: i) coherence; and ii) sediment delivery potential.

Regolith coherence refers to the ability of the regolith to resist detachment due in particular to the erosive power of running water. Soil properties which have a strong influence on coherence are particle size composition, aggregate and matrix strength, penetration resistance and to some extent, matrix density. Soils which lack coherence are typically sandy soils in which there is low aggregate strength due to the lack of bonding agents or highly dispersive clay soils, such as solodics, in which both surface and subsoils are prone to physical collapse when wet.

Regolith sediment delivery refers to the potential for regolith to yield fine-grained (silt and clay) sediment that can be transported to receiving waters. Key physical properties influencing sediment delivery potential are particle size composition (including coarse fragment content), aggregate stability and the thickness of regolith with a high proportion of fine material.

4. APPLICATION

To date, regolith stability has been mapped at the broad scale of 1:100,000 for all state forests in eastern NSW, using a variety of existing data sources ranging from soil surveys, which cover approximately 75% of the area, and geological surveys to fill in the gaps (Murphy et al, 1998). This broad classification is the basis for more detailed field assessment which is made by EPA accredited assessors at the compartment scale, for the preparation of harvest plans. This data is the basis for the final designation of erosion and sediment delivery potential. During this checking, additional soil units may be mapped, or the designations changed, on the basis of field experience and evidence of their erosive behaviour. This field checking and any supporting documentation is compiled as part of the harvest plan which can be subject to audit by the EPA.

During a recent field verification of the regolith stability mapping of 12 areas of softwood plantation in northern NSW, the existing broadscale regolith stability designation was appropriate for 7 of the areas, while in the remaining five the designation was revised or other regolith stability units (usually small areas) were added. Other soil assessors are also finding the broadscale mapping to be adequate in around 60-70% of cases, and requiring some form of amendment in the remaining 30-40% of cases.

5. CONCLUSION

There is no perfect answer to classifying soil erodibility for forest erosion hazard assessment. The regolith stability concept has advantages, particularly relating to scale of application and relative simplicity in its application. It is somewhat subjective, but this has been addressed by requiring that its application is by accredited assessors. During the short period over which it has been used, the approach has been demonstrated to work well, and certainly gives a more meaningful result than did the USLE K factor. The regolith stability approach to soil stability therefore represents an advance, but not the last word.

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Table 1: conceptual framework for regolith stability classification

	Low sediment delivery	**High sediment delivery**
High Coherence	R1 High ferro-mangnesium regolith eg basalt, dolerite; Fine- grained argillaceous regolith with high gravel content eg siltstones, metasediments; Highly organic regolith eg peats.	R3 Fine-grained argillaceous (clay) regolith with low/no gravel contents; Fine-grained massive regolith.
Low coherence	R2 Unconsolidated sands; Medium to coarse-grained felspathic-quartzose regolith eg adamellite, quartz sandstone.	R4 Unconsolidated deposits of silt and clay; Unconsolidated fine-grained weathered regolith (saprolite).

Does Runoff from Snig Tracks Reach Streams? A Predictive Approach

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Summary: Snig tracks are a feature of most Australian forestry operations. When overland flow from a snig track reaches a stream, an off-site impact results. This paper uses findings from the experimental study concerning runoff redistribution of *Croke et al.* as input to a prediction of the probability of overland flow reaching streams from snig tracks of differing lengths and spatial locations within harvesting areas. Two examples of the predictions are provided for all snig tracks in a compartment in southeast Australia for plausible scenarios. Guidelines for simple snig track design which enable the use to chose the risk of overland flow reaching the stream are also provided.

1. INTRODUCTION

In most Australian forestry operations tracks within compartments are normally drained by constructing crossbanks, or water bars, across the tracks immediately following logging. Crossbanks serve four functions in controlling sediment movement within forestry compartments: they 1) define the specific catchment area of the snig track so that the overland flow does not develop sufficient energy to cause gullies, 2) reduce sheet and rill erosion through reduction in slope length, 3) cause deposition of some sediment as flow reduces at the crossbank and, 4) redirect overland flow into the adjacent general harvesting area (GHA) so that further sediment deposition may take place. This paper describes a simple model for the fourth of these functions in which the rate of runoff from the track is combined with spatial attributes of the track and stream network. This paper is an extract of Hairsine *et al.* (submitted).

2. RUNOFF FROM SNIG TRACKS

The generation of overland flow is uncommon on soil surfaces in forests with no or low levels of disturbance. Croke *et al.* (1999) found that general harvest areas on hillslopes away from permanent saturation zones generate small quantities of runoff with a very patchy spatial distribution even for very intense rainfall. Croke *et al.* also found that overland flow generated on snig tracks was reduced in volume by passage through a five-metre segment of the GHA. These findings are of major importance to the planning of forestry activities in order to minimise impact on in-stream water quality. Firstly, these findings suggest that the quantity of overland flow can be managed through manipulating the spatial layout of snig tracks and secondly, that this is a realistic mechanism for the reduction of fine sediment entering streams in forestry environments.

In the following development, a method is proposed to extend the above findings of the experimental study of Croke *et al.* to provide design rules for snig tracks and similar forms of intense disturbance.

Table 1: Summary of runoff volumes at crossbank and volume to breakthrough for all sites and all intensities.

Site	Soil Type	Age (years)	Rainfall Intensity ($mm\ hr^{-1}$)	Total Runoff from crossbank V_{ts} (litres)	Volume to break through V_{bts} (litres)
7	LG	0.6	54	1003	550
7	LG	0.6	68	2552	450
7	LG	0.6	123	3512	648
1	LG	1.5	49	1222	234
1	LG	1.5	67	1724	500
1	LG	1.5	113	2128	690
2	LG	4.7	56	no runoff	no connection
2	LG	4.7	69	227	no connection
2	LG	4.7	121	1679	345
6	MS	0.4	49	563	57
6	MS	0.4	78	1896	96
6	MS	0.4	144	4081	113
4	MS	1.5	43	35	no connection
4	MS	1.5	53	1055	200
4	MS	1.5	92	2861	459
5	MS	4.7	75	no runoff	no connection
5	MS	4.7	80	532	150
5	MS	4.7	148	1497	215
9	RG	0.6	43	414	no connection
9	RG	0.6	66	1559	390
9	RG	0.6	100	2664	513
8	RG	1.7	50	379	300
8	RG	1.7	64	1822	105
8	RG	1.7	117	2142	360
3	RG	4.6	53	no runoff	no connection
3	RG	4.6	65	144	no connection
3	RG	4.6	124	1110	350

Table 1 provides a summary of runoff volumes from the snig tracks on a range of soil types from south eastern Australia subjected to simulated rainfall of a range of intensities. All of the snig tracks were approximately fifteen metres long and five metres wide. All simulated storms were thirty minutes in duration. As described by Croke *et al.* (1999) the general trends in runoff were increasing runoff volume in proportion to rainfall intensity, decreasing runoff volume with increasing age (time since logging) of the sites, and no clear differentiation between soil types

3. WATER BALANCE ON THE GENERAL HARVEST AREA

A useful concept in describing the connectedness of hillslope sediment sources to downslope drainage features is the volume to breakthrough. Volume to breakthrough is the volume of runoff that may enter an area before a discharge is observed at the downslope boundary of that area. The volume is a combination of water lost to overland flow through infiltration, water stored above ground in depressional storage and water in transit between the upper and lower boundary of the area.

In the experiments of Croke *et al.* (1999, a, b) runoff from compacted snig tracks was measured at a flume at the exit of a crossbank. This runoff was then discharged onto the GHA, and collected in a trough at the lower boundary. Overland flow associated with the plume of water leaving the crossbank was observed to enter the trough as one or two concentrated flows of the order of 0.5 metre wide.

A further form of data collected in this is the elapsed time of rainfall before runoff arrived at the exit of the plot, 5 metres downslope of the crossbank outlet. By summing the runoff volume discharging from the crossbank up to this time the volume to breakthrough for a 5 metre length of the hillslope, Vbt5, is obtained. Table 1 provides measured values of Vbt5 for each of the runoff events simulated at the nine sites considered in the study of Croke *et al.*(1999). Regression analysis of these values showed that Vbt5 is weakly related to time since logging, runoff rate, and soil type. The value of Vbt5 is now taken as a single randomly distributed term for all sites with mean, μ_{vbt5}, and variance, σ_{vbt5}^2. Using all data where breakthrough did occur in Table 1, these values are μ_{vbt5} = 336.25 litres and σ_{vbt5}^2 = 35607 litres^2.

We now use the distribution of measured values of Vbt5 to predict the length and volume of overland flow of plumes resulting from crossbank discharges. In these predictions the following assumptions are made

a. The overland flow leaving the crossbank is non-eroding. This requires that the resistance of the GHA surface be such that incision does not occur.

b. The behaviour of the 5 metre segments of hillslope investigated by Croke *et al.* containing the plume is representative of the hillslopes within the compartments. This implies that the concentration of flow resulting from the crossbank and that occurring five metres downslope are identical in terms of their effect on the spatial distribution of Vbt5. It also implies that the distribution of soil hydraulic properties in the plume area, as influencing the calculated values of Vbt5, are representative of those of the compartment.

c. The values of Vbt5 for adjacent plume areas are spatially independent, though drawn from the same population.

d. Vbt5 describes all losses of overland flow. This assumption neglects any losses occurring after the time of breakthrough.

Making these assumptions, predicted plumes longer than the measured five metres are considered as a sequence of 5 metre long plumes in series, each of which has the capacity to store overland flow described by μ_{vbt5} and σ_{vbt5}^2. Mass balance of overland flow in the flow reducing conditions described above can be described by:

$$lpred = 5\frac{V_{out}}{vbt5} \qquad [1]$$

where $lpred$ is the predicted length of the plume and V_{out} is the volume of flow leaving the cross bank. The statistical properties of $lpred$ can be approximately obtained through describing the equation [1] as a Taylor series and using only the linear terms in deriving the new probability functions.

The mean predicted length of the plume, μ_{lpred}, is then approximately given by

$$\mu_{lpred} = 5\frac{V_{out}}{\mu_{vbt5}} \qquad [2],$$

and the variance of the predicted plume length, σ_{lpred}^2 is approximately given by

$$\sigma_{lpred}^2 = \frac{25\sigma_{vbt5}^2 V_{out}^2}{\mu_{vbt5}^4} \qquad [3].$$

In equations [2] and [3] a major input variable is V_{out}, the volume of overland flow leaving the crossbank during the runoff event. The volume of runoff from any snig track segment of length, L, is assumed to be related to the standard 15 metre length segment volume, V_{15} measured by Croke *et al.* (1999) by the relationship:

$$V_{out} = \frac{L}{15}V_{15} \qquad [4].$$

Combining the above equations enables predictions of the length of plumes from cross banks for combinations of runoff volumes (from Table 1), and snig track lengths. These predictions can then be compared with available lengths.

4. SAMPLE OUTPUT

Figure 1 gives a comparison of predictions of the mean length of plumes for three design storms applied to the range of existing snig tracks in compartment 2128 of Mumbulla State Forest. Mapped snig track lengths and distances downslope to the mapped drainage lines are used in these predictions. Snig track runoff volumes are taken from Croke *et al.* (1999) for the hillslope plot on this same soil and logged 0.4 years ago.

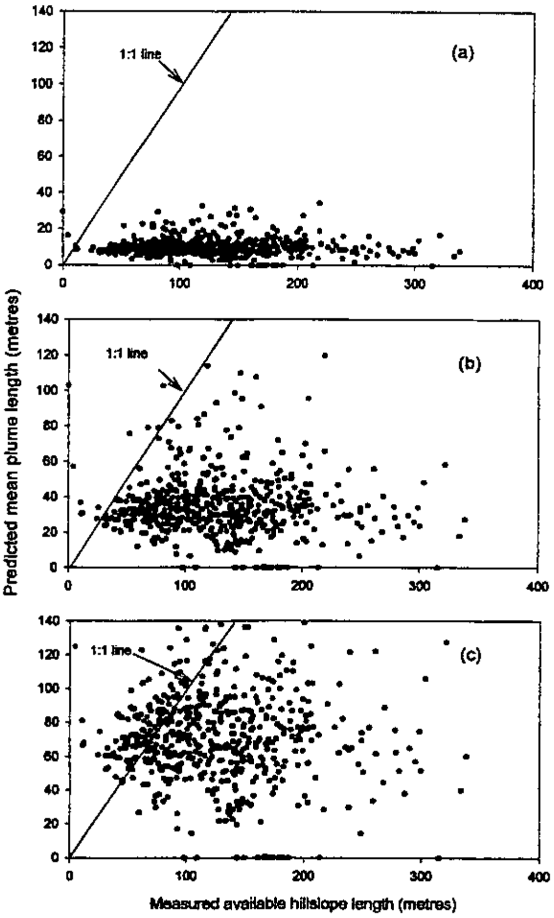

Figure 1. Predicted mean plume length versus actual available hillslope length for compartment 2128 of Mumbulla State Forest for a simulated event of 30 minutes duration and a forest age of 0.4 year, for three rainfall intensities used in the study of Croke *et al.* (1999): (a) 49 mm hr^{-1} (b) 78mm hr^{-1} (c) 144 mm hr^{-1}

In Figure 1 part (a) it is clear that for the low rainfall rate, all predicted mean plume lengths are less than 20 metres, so than only snig tracks with outlets within ~20 metres of the stream will contribute overland flow and associated suspended sediments directly to the stream. With increasing rainfall intensity an increasing number of plumes are predicted to connect into the stream network.

5. PRELIMINARY GUIDELINES

A simple probabilistic design approach can be constructed by setting a tolerable probability of overland flow reaching the stream. Assuming the predicted plume length is normally distributed, it may be given and compared with the available hillslope length, H, in the expression:

$$\mu l_{pred} + f\sqrt{\sigma l_{pred}^{2}} \le H \qquad [5],$$

where f is the number of standard derivations above the mean permitted (e.g. f =1.645 gives a 5% probability of excedence).

Substituting equations [2], [3] and [4] into equation [5] gives

$$\frac{LV_{15}}{3\mu_{vbt5}} + f\frac{LV_{15}\sqrt{\sigma_{vbt5}^{2}}}{3\mu_{vbt5}^{2}} \le H \qquad [6].$$

An example of the implementation of equation [6] is given in Figure 2 using three runoff rates from Croke *et al.* (1999). This figure provides a quantitative guide to the selection of inter-crossbank length for the range of hillslope positions of the crossbank outlet. Note that all required hillslope lengths are greater than the interbank length so there is a further assumption that overland flow plumes from sequences of crossbanks do not connect together.

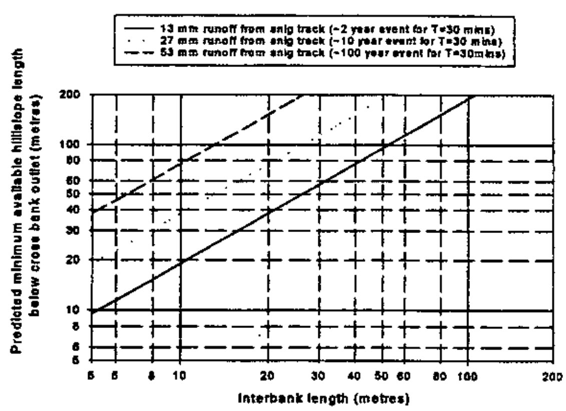

Figure 2. An example of the implementation of equation [7] using three runoff rates from Croke *et al.* (1999) typical of snig tracks in recently logged compartments with a rainge of rainfall intensities. In this example the probability of the plume not reaching the stream is set as 95% (f=1.645).

6. DISCUSSION AND CONCLUSIONS

This paper has outlined an approach for predicting plumes from snig tracks in environments where runoff is generated on snig tracks and net infiltration of overland flow occurs between the snig track and the stream. The approach draws on an expansion of the analysis of the runoff redistribution presented by Croke *et al.* using a volume to breakthrough concept.

The predictions of plume lengths provided here are preliminary and await rigorous examination of the assumptions used. However, the approach has the advantage only requiring data from the study of Croke *et al.* (1999) in combination with snig track lengths and distance to streams. A further advantage of this approach is that predictions can be associated with probability of plumes entering the stream. This may be appealing to decision-makers in a regulated environment where risk is a normal input to decisions.

7. ACKNOWLEDGEMENTS

I gratefully acknowledge the advice and assistance of Jacky Croke in this study. The Cooperative Research Centre for Catchment Hydrology and the NSW Department of Land and Water Conservation provided funding for this work. Thanks also to Heather Mathews, Peter Fogarty, Simon Mockler, Jeff Wood, Jim Brophy, Matt Nethery, Andrew Loughhead, Jason Thompson and Ross Allen for their assistance in this and related research.

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Successful Delivery of Forest Soil and Water Protection Outcomes.

The New South Wales Story

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Summary: The role of training, the use of codes of practice and their conditions, and the application of harvest planning and audit programs used by State Forests NSW to achieve successful delivery of forest soil and water protection outcomes are briefly described, discussed and evaluated.

1. INTRODUCTION

There are a number of perceptions that provide the impetus for successful forest soil and water protection in public forests. These perceptions include:

- An appreciation that national water quality standards include consideration of turbidity and the transport of fine clay suspension in drainage systems as pollutants;
- A community expectation that all public forests will contribute good quality runoff water into catchments;
- A realisation of the inseparable nature of actions that both mitigate soil erosion and control water pollution;
- An understanding that the best practice management approach to soil protection and water pollution control provides the most desirable but often difficult to implement solutions to water pollution control, especially where complex or unquantifiable site variables or benchmarks have to be adopted;
- Recognition that forest harvesting industry operators and supervisors need clear unambiguous instructions and relevant training to deliver on regulatory requirements.

State Forests' staff did not immediately recognise all these perceptions during the thirty-year development of the forest soil and water protection measures which are now in place.

Soil erosion mitigation controls alone were originally published in the *Standard Erosion Mitigation Conditions for Logging in NSW (1975)* as a consequence of the NSW *Catchment Areas Protection Act 1971*. The NSW Soil Conservation Service (now absorbed in to DLWC) and the Forestry Commission of NSW (now State Forests of NSW) jointly prepared the conditions in consultation. They were applied to both State forests and Crown-timber lands and to protected lands under the CAP Act. The SEMC were revised to "guidelines (SEMGL)" in 1990. State Forests were subsequently obliged to translate the guidelines back into conditions to meet various Environmental Impact Statement determinations in 1993-94.

The decision by State Forests to apply for a Pollution Control Licence in 1992 highlighted the inseparability of soil erosion from water pollution. The NSW Environment Protection Authority (EPA) therefore subsequently sought to develop its own set of licence conditions that whilst they built on the SEMGL, essentially rendered them redundant on public forests.

The need to introduce competency-based training of forest harvesting operators, and others, became an issue in 1993.

Apart from these self-initiated developments, State Forests has had to recognise that in NSW, the exercise of "Due Diligence" and adoption of the "Precautionary Principle" have become enshrined in all aspects of environmental management system development and are also formalised in NSW environmental legislation. Proven preventative and remedial action taken by an organisation or individual to overcome a pollution event is an essential means for demonstrating due diligence. It can be used as a defence against prosecution for water pollution. State Forests, over the past seven years in particular, has therefore also had to meet this challenge through a number of further initiatives.

These are:

- Training in forest harvesting at the operator, supervisor and planner level;
- Integration of training, the use of codes of practice, and planning;
- Designing and implementing adequate reporting and auditing procedures.

2. TRAINING IN FOREST HARVESTING

Background

The Soil Conservation Service of NSW conducted informal training in soil erosion mitigation and the use of the SEMCs over the period 1975-1990. This training especially concentrated on snigging and log dump management and track drainage. Because it was voluntary and infrequent, only a small proportion of the harvesting workforce attended.

In 1993 a steering committee comprising representation from State Forests, the EPA, DLWC the CFMEU, the NSW Forest Products Association and the NSW Loggers' Association initiated preparation of a training course in "Forest Soil and Water Protection". The title was deliberately chosen to emphasise the link between soil erosion and water pollution.

In December 1994, a State Forests' consultancy prepared the competency based training course application for accreditation by the NSW Vocational Education Training Accreditation Board (VETAB) and State Forests entered an agreement with NSW TAFE as the training provider. A pilot course for operators was held in June 1995. The full training program across coastal and plantation regions commenced in 1996, with a three year commitment to train all existing harvesting operators or supervisors from industry and State Forests' staff.

Current training

The course is a nested short course with three modules at different levels. Supervisors and planners have to attend an operator course before undertaking courses at their own levels. Similarly planners have to attend a supervisor course before being eligible to undertake a planners course. This approach guarantees commonality of knowledge and attitude among all groups. The course also allows for elective units to provide for native forest harvesting (emphasis on temporary road construction and snigging practices), softwood plantation harvesting (walkover techniques, log dump and landing management), and road construction and maintenance (for State Forests' employees).

Between 1996-1999, ninety-five courses have been conducted, leading to the assessment and accreditation of 1330 operators, supervisors and harvest planners at the operator and supervisor level. There were approximately ten students who failed the courses, even after extensive re-assessment. At the insistence of harvesting industry and union interests the training program especially targeted those operators who were employed by licensed contractors.

The cost of training for this group was subsidised by State Forests who provided instructors and assessors, and TAFE who provided the course as a mainstream one semester equivalent course. The cost to State Forests in developing the course and arranging accreditation and subsidising delivery to industry and its own employees is cumulatively estimated at $0.85 million to date.

Now that the core target group has been trained, the next phase of the program is aimed at tree fallers and log truck drivers, and forest operations employees who are engaged in forest activities other than harvesting.

International recognition of training

State Forests was gratified be awarded an Environmental Achievement Award from the International Erosion Control Association in 1998 for recognition of its commitment to forest soil and water protection training through development and delivery of the training program

Results of the present training program

The success of the training has been reflected in an almost complete elimination of non-compliance in harvesting operations at the in-compartment level. The training also revealed, along with other indicators, that water pollution in the forest environment comes mainly from roads, through failures in road drainage systems and lack of groundcover protection of earthworks. This has been addressed in recent revisions to the course.

The training experience has resolved a number of issues. At the supervisor level, the course is now focused entirely on interpretation and use of the Forest Practices Code and schedule conditions attached to the pollution control licence. Literacy, which was originally considered as an impediment for some of the more experienced operators, has been shown to be a smaller problem than was originally thought. However there is no doubt that operators (and many supervisors for that matter) do not read licences or codes of practice unless they are put on the spot to do so.

The training course does not provide a means for keeping operators and supervisors immediately updated on changes, eg to licence conditions, although the teaching content of courses is easily reviewed and modified as these changes occur.

Training issues

Only a few instructors have the appropriate qualifications to teach and assess students. All these people work for State Forests although three principal instructors are soil conservationists already seconded from the NSW Department of Land and Water Conservation. In the present deregulated technical education environment, this is not acceptable to some but alternative arrangements which ensure that instructors possess the essential depth of understanding required and a close knowledge of current licence requirements, are proving hard to achieve.

Also, State Forests has not yet conducted any planner courses. Up to now, the volatility of licence conditions, and the regulatory environment generally, have delayed our ability to put a comprehensive training package together for this level. State Forests hopes to achieve this by the end of 1999, following conclusions to arrangements under the Forest Agreements and the Integrated Forest Operations Approvals for the four forest regions in coastal areas.

3. CODES FOR FOREST HARVESTING OPERATIONS

Background

Since the introduction of the SEMC in 1975, there has been a steady increase in the use of Codes of Logging Practice and others cover forest harvesting and other forest operations. State Forests' present unified *Forest Practices Code* has four published sections and another five are proposed.

The present Code

We now face a number of difficulties in maintaining this *Forest Practices Code* because external regulators have adopted different or slightly variant rules and conditions. In order to promote and achieve compliance, a Code has to present stable, unified rules that everyone understands, and can be taught during training. Those who are directly accountable for particular activities need to be identified on a condition by condition basis even though some people claim that this is too threatening.

The first thing a reader does when reading a Code or other licence conditions is to ask," who is responsible or accountable for delivering on this condition?" If the condition indicates more than one type of person and implies more than one course of action then it may be too diffuse. My view is that conditions should, where possible, be grouped according to the person responsible for their compliance.

The other main challenges in achieving a single code are:

- Separating the standards and rules from the "how to" instructions;
- Identifying situations where special conditions are needed to address regional differences or unique operational methods (eg cable logging);
- Integration or rationalisation of conditions set out in different codes or licence conditions;
- Maintaining frequent updates in the face of prolonged consultation processes.

The last point needs some comment. In NSW we are moving towards an inclusive consultation process at the very time that code conditions need more frequent updating than the present two-three year timeframe allows. There is an imperative to synchronise code changes to common dates to allow for the full consultative process to proceed smoothly. Under current guidelines, the minimum period is usually about six months.

4. PLANNING FOR ACCEPTABLE FOREST SOIL AND WATER PROTECTION OUTCOMES

Background

Forest harvest planning in NSW has developed from a very basic timber production focus up to the mid 1980's to what must be arguably the most sophisticated and complex process ever devised for forest operational planning. Plan preparation has evolved from a relatively straightforward task that used standard conditions to manage environmental constraints, to an assessment process that is required to addresses all site-specific values at a fine scale.

Harvest planning issues

One of the unknowns that we face with current soil and water protection planning procedures is whether they influence on-site activities significantly apart from creating additional non-harvest areas within the compartment. This raises a number of immediate questions that are pertinent to harvest planning at the present time.

- Is there enough fine-tuning within the present planning process to refine our ability to remove all the useable resource available without compromising soil and water protection?
- Does the methodology used in the planning procedure for forest soil and water protection prejudice the practice of adequate silvicultural disturbance?
- Should we plan harvesting operations for optimum seed-fall or ground-cover recovery conditions?

However, during the introduction of site-assessment in the early 1990's, it was shown that there is always the danger that an intensive and expensive harvest planning process is blunted by the use of broad-scale conditions. This was highlighted in the application of the SEMGL 1993) and their derived conditions, where only one condition applied different requirements between low and moderate erosion hazard categories and high erosion hazard categories. In the SEMGL, the low and moderate hazard categories were combined for the purposes of applying the standard guidelines.

This situation begs the question of the relationship between site-specific investigation and the application of conditions even when the latter prescribe different parameters for different hazard categories – should we adopt highly refined survey and planning techniques when less sophisticated ones will satisfy the parameters set by the conditions?

My perception is that there is now an opportunity to scientifically review the planning process and the relationship between data collection and its analysis, and the determination of conditions. In addition, a pecking order has entered here to the extent that biological values now take spatial precedence over soil and water values. Therefore it can be contended that soil and micro-landscape survey should concentrate only on those in-compartment areas that still remain as candidate net harvest area, once the exclusions for other environmental purposes have been made. This may ease the burden on survey and other investigative costs.

5. INTEGRATION OF TRAINING, CONDITIONING AND PLANNING

State Forests is now almost at the point where the benefits of the past five-year training initiative, the refinements and near stability of the schedule conditions in the pollution control licence and a clearer procedural approach to harvest planning will hopefully pay off. Since the present pollution control licence was issued in April 1998, we have been able to train supervisors with some degree of confidence and certainty in the use of, and compliance with schedule conditions. The final negotiations with the Forest Agreements and Integrated Forest Operations approvals (the final outcome of the former NSW RFA process) will hopefully reinforce a consistent working environment in which harvest and other planning will proceed in a more orderly fashion.

6. INSPECTION, REPORTING AND AUDITING PROCEDURES.

Background

Much has been debated and much is now expected of inspection and reporting of forest soil and water protection outcomes for compliance and success, both during and after harvest operations. What is the point of going to so much trouble to implement soil and water protection practices if no-one verifies that they work. In 1992, State Forests introduced an internal four-tiered environmental audit and review system. This was refined through the introduction of a standardised Harvest Inspection Report system in 1996, which is currently under review.

One of the strengths of this reporting system has been through its role as a consistent source of data on compliance levels and its usefulness in identifying weaknesses in harvest outcomes where compliance has been demonstrated. The present reporting system gives prominence to soil and water protection (75% of all check items).

This need to check outcomes is now reflected more closely in the schedule conditions set out in the pollution control licence.

Issues in inspection, reporting and auditing

There is room for improvement. Checking harvest outcomes requires consistent application of standardised procedures that are difficult to apply in a broad natural landscape setting. Checking procedures undertaken at the first tier level use either transect, defined area or single condition compliance techniques (eg. specifically targeting road drainage structures and function). Inspection and reporting is a costly practice, especially of time, and often there has to be a compromise between the level of inspection and the extent of the area covered.

There is also a need for inspection techniques to adopt uniform standards, including those used by regulators, and wherever possible, State Forests endeavours to the same inspection standards. This raises the need for frequent consultation between supervisors and auditors to ensure that everyone has the same consistent approach. A conservative estimate of soil and water protection inspection activities associated with timber harvesting is that it is of the order of $4.5 million per year. (This translates as $1.35 per cubic metre of timber sold.)

Although not a research task, State Forests is obviously looking for more cost-effective means of delivering harvest inspection.

Before leaving the inspection and reporting system, it should be emphasised that the current system is not a substitute for effectiveness monitoring. Failures where adequate compliance with conditions has clearly been demonstrated always need to be followed up with further investigation. The data from harvest inspection can be used as a component of other monitoring system and considerable thought has gone into designing harvest inspection procedures that will allow:

- generation of statistics on compliance;
- feedback on emerging problems with compliance; and,
- cross-reference with the post harvest monitoring systems currently used by State Forests.

7. FUTURE RESEARCH FOR OPERATIONAL BEST PRACTICE

Research programs

Any future forest soil and protection research program will, I believe, need to address:

- The development of a range of effective road drainage guidelines that can be applied to more specific soil and landscape conditions;
- Effective cost-efficient armouring techniques to deliver clean run-off water from road drainage systems;
- Development of minimal disturbance extraction systems to complement snigging;
- Refinement of data on threshold parameters for steep slope harvesting;
- A review of categorisation systems for soil erodability and water pollution potential;
- Ground-cover disturbance in relation to seedbed establishment versus soil erosion risk.

National standards and measurement of soil erosion and water pollution factors

There also has to be a mid to long-term aim to come up with national standards for the definition of landscape and drainage features associated with forest soil and water protection and the terminology used to describe forest operations. A consistent national approach to measurement of drainage features and objects is also needed. For example, Strahler's Schematic Stream Order diagram (1964) makes a lot of sense if everyone uses it.

8. CONCLUSION

The future

Although forest soil and water protection presently focuses on timber harvesting in public forests in NSW, it is only a matter of time before the same approach to training, conditioning and practices will need to be applied to other forest operations. There is also good reason to expect that a similar approach to the overall management of natural surface roads in conservation reserves and other rural areas in general.

I believe that State Forests' contribution has been and will continue to be the development and use of good robust codes of practice and corporate guidelines, well crafted training programs and adequate and certifiable monitoring and auditing systems, which other organisations would do well to emulate.

Note that the opinions expressed in this paper are the author's and do not represent any position or policy adopted by State Forests NSW.

*The author is State Forest's Forest Practices Officer, Forest Policy and Programs Division, Sydney

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APPENDIX 1

State Forests of NSW

FOREST SOILS AND WATER PROTECTION COURSE – MODULES AND UNITS			
Operator **4 core + 1 elective minimum**		**Supervisor** **Core + 2 electives minimum**	**Planner** **2 core + 1 elective minimum**
Operator 1 (Core) Legal and other requirements for forest soil and water protection.	Operator 9 (Elective) Harvesting operations and procedures in river red gum forests	Supervisor 21 (Core) Supervision procedures for forest soil and water protection during forest operations.	Planner 31 (Core) Assessment of soil erosion and water pollution conditions.
Operator 2 (Core) Use of forest operation plans.	Operator 10 (Elective) Harvesting operations and procedures in western NSW native forests	Supervisor 22 (Elective) Procedures for harvesting coastal and tablelands native forests.	Planner 32 (Core) Application of hazard assessment to forest operation plans
Operator 3 (Core) Soil properties and factors influencing soil erosion and water pollution.	Operator 11 (Elective Road construction and maintenance operations.	Supervisor 23 (Elective) Procedures for harvesting plantations.	Planner 33 (Elective) Survey and verify and report forest soil factors.
Operator 4 (Core) Wet and other adverse weather conditions.	Operator 12 (Elective) Softwood plantation establishment operations.	Supervisor 24 (Elective) Procedures for road construction and maintenance.	Planner 34 (Elective) Manage and implement compliance with forest operation plans.
Operator 5 (Elective) Harvesting operations in coastal and tablelands native forests.	Operator 13 (Elective) Hardwood plantation establishment operations.	Supervisor 25 (Elective) Procedures for softwood and hardwood plantation establishment.	
Operator 6 (Elective) Manual tree felling and timber servicing during harvest operations.	Operator 14 (Elective) Prescribed burning operations and procedures for soil protection.		
Operator 7 (Elective) Using haulage vehicles on forest roads.	Operator 15 (Elective) Chemical pollution prevention and emergency action response procedures.		
Operator 8 (Elective) Harvesting operations in plantations.			

Water Quality and Sediment Transport of Radiata Pine Plantation and Eucalypt Forest Catchments in Victoria

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Summary: The Cropper Creek Hydrology project commenced in 1976 to study the hydrology and water quality of eucalypt forest catchments and to determine the impacts of conversion of one catchment to a Radiata pine plantation. The second phase of the project commenced in May 1997 to study long-term changes for the two forest types under different management regimes. Data collected to December 1998 showed that stream flow and water quality, turbidity and total suspended solids, for the catchments were consistent with relationships established for base flow conditions during the calibration phase prior to conversion. Water quality was uniform under base flow conditions but significant hysteresis was observed during a major storm event. Soil loss during this single event was substantial compared with the relatively low rate of erosion under base flow conditions.

1. INTRODUCTION

In the mid 1970's the Cropper Creek Hydrology project was undertaken to study the hydrology of three small forested catchments of mixed species eucalypts in north-eastern Victoria (Bren *et al.* 1979). In 1980, one of the catchments was cleared for the establishment of a Radiata pine plantation but a 30-m wide riparian zone of native vegetation was retained to protect water quality. In the short term this conversion in land use to plantation forestry increased annual water yield by 3.5 ML/ha but this was followed by a gradual decline in yield to pre-treatment levels (Bren and Papworth 1991). Changes in water quality were relatively minor however exports of suspended solids and nutrients in stream water increased because of higher discharge after clearing (Hopmans et al 1987).

In 1997, the Cropper Creek project was resumed to evaluate the long-term changes in hydrology and water quality of the 17 year-old pine plantation in comparison with historic data prior to the change in land use of Clem catchment (46 ha). As part of this second phase (Phase II), monitoring of the adjacent undisturbed catchment (Ella catchment, 113 ha) also resumed to enable evaluation of changes in the original relationships established for the paired catchments during the pre-treatment calibration phase from 1975 to 1980 (Phase I).

2. METHODS

Weirs and gauging equipment for monitoring rainfall and stream height were reinstated in April 1997. The traditional Leopold-Stevens chart recorders were fitted with shaft encoders for electronic logging of stream height. Details of the physiography of the catchments, climate, geology, native vegetation, instrumentation of the weirs and collection of field data are given by Bren *et al.*(1979). Regular collection of stream water for analysis commenced in May 1997. Samples were collected from a fixed location above the weirs at weekly intervals. Additional samples were collected at more frequent intervals (1 to 4 hours) during storm events in July and September 1998 using Sigma automatic sample collectors. Turbidity and total suspended solids (TSS) in water were measured generally within 48 hours after collection using standard procedures (APHA 1995).

3. RESULTS AND DISCUSSION

The first year of Phase II of the project was very dry with climatic conditions similar to those in 1982. Consequently water yield for this period were very low for Clem catchment (0.68 ML/ha) and especially Ella catchment (0.27 ML/ha). This compares with the range in annual yields of 0.1 to 7.7 ML/ha for Ella catchment from 1977 to 1987 (Bren and Papworth 1991). Stream flow at Ella ceased from December '97 to July '98 when flow resumed following a moderate storm event in July and a major storm of about 180 mm over 7 days in September '98 (Figure 1). Discharge from Clem and Ella catchments was highly correlated (R^2 0.99) over the wide range of flows during Phase II. This is consistent with the regression relationship for stream flow from these catchments reported by Bren and Papworth (1991).

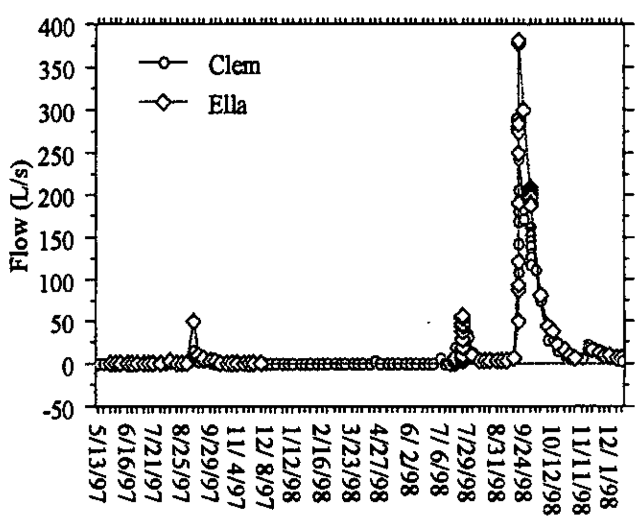

Figure 1. Instantaneous stream flow (L/s) at Clem and Ella catchments from May '97 until January '99.

Stream water from the Radiata pine and eucalypt forested catchments is of high quality as indicated by the low levels of turbidity and TSS at base flow conditions during Phase I and the present Phase II (Table 1). Clem catchment showed good agreement in these water quality parameters for the period prior to clearing of eucalypt forest and the present Radiata pine plantation. In contrast, present water quality values for the undisturbed Ella catchment are lower than observed during Phase I. This is attributed to the very low stream flow and the comparatively long dry period (7 months) at this catchment since the commencement of Phase II observations.

Table 1. Median values for turbidity and total suspended solids in stream water from Radiata pine and eucalypt forested catchments at Cropper Creek.

Catchment	Period	Turbidity (NTU)	TSS (mg/L)
Clem - Euc	1976-1980	2.0 (1.5)[#]	5.0 (3.1)
Clem – Pine	1997-1999	3.3 (1.3)	3.6 (1.6)
Ella – Euc	1976-1980	2.0 (1.0)	3.0 (2.0)
Ella – Euc	1997-1999	1.0 (0.4)	0.8 (0.4)

[#] median absolute deviation (MAD)

Monitoring of the storm events in July and September 1998 in addition to the regular weekly sampling enabled the relationship between turbidity and TSS in stream water to be evaluated across a wide range of values (turbidity 0.3 to 680 NTU and TSS 0.2 to 1190 mg/L). The two parameters were highly correlated (R^2 0.95) across most of this range and regression equations were the same for both catchments. Analysis of the data also showed that the variance of the regression relationship increased for turbidity values less than 10 NTU consistent with base flow conditions. However, above this value the relationship could be used to predict TSS with confidence from the more easily observed turbidity.

Under base flow conditions when discharge is generally less than 20 L/s (Figure 1), the relationship between turbidity and stream flow showed little, if any, concentration-discharge hysteresis consistent with earlier observations (Hopmans *et al.* 1987). In contrast, distinct clockwise, concave hysteresis was observed for turbidity (and TSS) during the major storm event in September 1998 (Figure 2). This suggests a change in runoff generation with a significant contribution from surface water during this episode (Evans and Davies 1998). It also showed that this effect was more pronounced for the steeper Clem catchment.

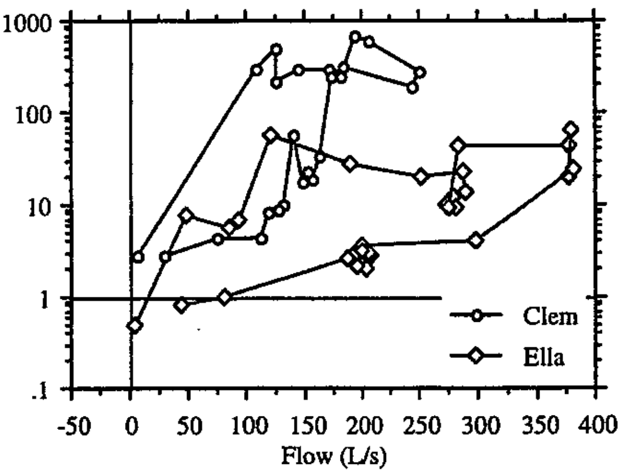

Figure 2. Turbidity (NTU) in stream water in relation to instantaneous flow at Clem and Ella catchments during a major storm event in September 1998.

Water yield and export of suspended solids showed significantly higher rates of transport from Clem catchment (Table 2). Results indicate that soil loss under base flow conditions was approximately 10 kg/ha at Clem compared with 2 kg/ha at Ella catchment. Export rates did not change significantly during a moderate storm event in July. In contrast, both discharge and sediment export changed substantially during the major storm event in September. Soil loss from suspended solids for this episode was estimated at 17 and 530 kg/ha for Ella and Clem catchments respectively. In addition accumulated coarse debris collected in the weir stilling pond was estimated at approximately 12 tonnes indicating a substantial shift in stream bedload during this event.

Table 2. Water yield and export of total suspended solids in stream water under base flow conditions (BF) and storm events in July and September 1998 at Cropper Creek.

Catchment	Time (days)	Volume (ML)	TSS Export (kg/ha)	TSS Export (kg/ML)
Clem (BF)	534	110	10.0	4.2
Clem (Jul)	7	12	1.2	4.6
Clem (Sept)	15	99	530	247
Ella (BF)	307	137	2.1	1.7
Ella (Jul)	7	13	0.1	1.1
Ella (Sept)	15	135	17.2	14.3

4. ACKNOWLEDGEMENTS

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4. ACKNOWLEDGEMENTS

The authors are grateful for the financial support provided by the Forest and Wood Products Research and Development Corporation and Hancock Victorian Plantations Pty Limited. We also wish to thank Mike McCormick and Gabi Szegedy for assistance with field work and water analysis.

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A Process for Identifying the Optimal Width of Buffer using Aquatic Macroinvertebrates

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Summary. Buffers (uncut forest on either side of streams) are widely used to minimise the environmental impact of timber harvesting on streams. If a buffer is too wide it will be an unnecessary use of the timber resource. While, if it is too narrow it will not be effective in preventing harvesting impacting on the stream and its ecosystem. The optimal width of a buffer would be the minimum width necessary to protect the stream and its biota. Although there is overwhelming evidence that a buffer of almost any width will offer some protection to a stream, there is very little objective information as to what is the optimal width of buffers. The project described here involved the sampling of aquatic macroinvertebrates at paired sites in commercial native forests of Victoria in order to estimate the optimal width of buffers to protect instream biota. Each pair of sites consisted of a control site located upstream of logging (and associated roading) and a treatment site located downstream of logging (and roading). We also used pairs of control sites, which consist of two sites in the same stream (approximately the same distance apart as the logged paired sites) with neither site having logging within its catchment. These control pairs are to account for any difference between two sites unrelated to timber harvesting. We aim to compare the similarity of the macroinvertebrate communities between each pair of sites, to the width of the buffer protecting the treatment sites. We also aim to ascertain whether the protective properties of buffers of a particular width are affected by such factors as slope of the coupe, soil type and forest type.

1. INTRODUCTION

We will be presenting an alternative approach of ascertaining the minimum buffer width or the optimal buffer width necessary to protect streams. This approach involves the use macroinvertebrates as a biotic indicator, sampled at wide range of sites. We are using this approach in a project that is currently underway in Victoria. The project is not yet complete and minimal data is presented here. However, we believe the approach is useful and should be adopted more widely.

What buffers do

If appropriate measures are not put in place, the harvesting of forests has the potential to have a large effect on the aquatic environment. It is common to leave an area of un-disturbed forest on the side of stream called a buffer or streamside reserve. The aim of such a buffer is to reduce or even prevent timber harvesting (and associated roading) from impacting on the health of the stream. Numerous studies have shown that buffers reduce the adverse impact of timber harvesting on stream health (Dignan *et al.* unpublished). One of the mechanisms that allow for buffers to prevent impacts is their ability to trap and store sediments that run-off from coupes. However, in the context of preventing impacts to the biological health of streams, there are several additional mechanisms by which buffers prevent impacts occurring, including:

4. Altered temperature and light regimes
5. Changes in the amount, size and type of terrestrial organic material entering the stream
6. Stream channel alteration
7. Changes in nutrient concentrations within streams
8. Edge effects on the riparian ecosystem

It is worth remembering that these additional mechanisms may be more important than reducing sediment entering a stream.

The dilemma

It is generally accepted that wider buffers will be more effective at preventing impacts than narrower buffers. Therefore in terms of stream protection wider buffers are better. However, if buffers are wider than is necessary to protect stream health they will use more timber resource than is necessary. Leon Bren (1995) investigated the effect of buffer width and the percentage of the catchment occupied with buffers in the Tarago River Catchment (Figure 1). In terms of the volume of timber harvested, narrower buffers are better. Thus there is a conflict in management aims.

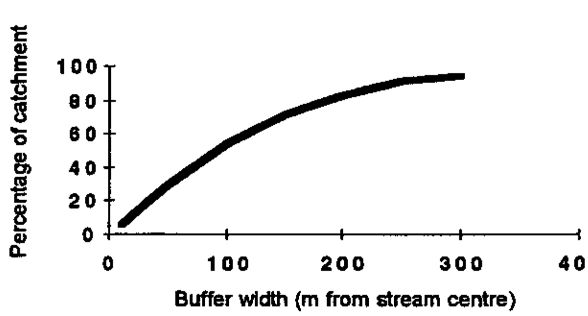

Figure 1. The effect of increasing buffer width on the percentage of the Tarago catchment within buffers, after Bren (1995).

If one of the aims of timber harvesting is to prevent impacts occurring in the aquatic environment then the optimal width of a buffer would be the minimum width necessary to protect the stream and its biota. Although there is overwhelming evidence that a buffer of almost any width will offer some protection to a stream, there is very little objective information as to what is the optimal width of buffers.

2. OUR APPROACH

Why biological indicators?

If one confines oneself only to look at the mechanism behind which buffers offer protective ability, establishing the optimal width of buffers is quite a complex task. For example, the optimal buffer width necessary to prevent sediment inputs to streams is likely to be quite different from the buffer width necessary to prevent changes to the inputs of large woody debris (LWD). The USA Department of Agriculture (USDA), Forest Ecosystem Management Assessment Team (FEMAT) estimated the buffer width necessary to perform some of the various functions of buffers (Figure 2). Although, there is some room for debate on the accuracy of FEMAT's estimates, it is highly likely that the different functions of buffers will require different widths.

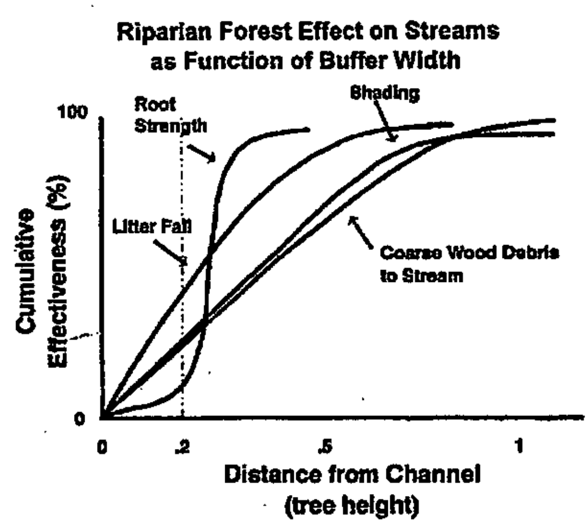

Figure 2. The effectiveness of various functions of buffers, after FEMAT (1993).

Furthermore, the amount of protection that each function (of a buffer) that is needed to prevent impacts on stream biota is unknown. For example, to be effective in stream protection, buffers may need to prevent 90% of sediments from entering the stream but only to maintain 60% of LWD. In order to establish the amount of protection needed for each of a buffer's functions would require a large amount of basic scientific research.

Using biotic indicators to establish the optimal size of buffers has the advantage of eliminating the need to completely understand the complex relationships between buffer widths and their ability to perform their various functions and the degree of protection of each function required. If the biological indicator changes, it can be assumed important changes to water and/or habitat quality occurred.

Why multi-sites?

Most past studies on the effectiveness of buffers have conducted intensive observations at a small number of coupes. These studies suffer the disadvantage of being highly site specific and thus providing little information on the width of buffers necessary for a wide range coupes/streams.

Additionally, studies that show deleterious impacts with some buffer strip widths (e.g. 20 m) in one catchment, merely indicate that an appropriate width suitable for similar conditions must be greater than 20 m, but cannot indicate how much greater than 20 m is required. Similarly, if no observed impact is recorded with a particular buffer width, there is no indication of how much the buffer width could be reduced before an impact would occur.

Davies and Nelson, in Tasmania (1994), conducted one of the few studies that collected data from a large number of coupes. Davies and Nelson found that buffer widths less than 30 m were not effective at preventing impacts to macroinvertebrates, while those buffer wider than 30 m were effective. Our study adopted comparable methods to Davies and Nelson in order to establish if similar patterns occur in Victoria. However, in order to overcome some criticisms levelled at Davies and Nelson, our study adopted some additions to their study.

3. OUR STUDY

Design

Our study involved the sampling of aquatic macroinvertebrates at paired sites in commercial native forests of Victoria in order to estimate the optimal width of buffers to protect instream biota. Each pair of sites consisted of a control site located upstream of logging (and associated roading) and a treatment site located downstream of logging (and roading). If a buffer is effective at a preventing impact from a specific coupe then there should be no difference in the macroinvertebrate fauna between the paired sites. By ascertaining which buffers were/were not effective at preventing impacts and relating the specific properties of these buffers and their coupes we hope establish the minimum buffer width required to protect a stream.

We also have paired sites along a stream with no recent harvesting within both of their catchment. These paired sites serve as double controls because they will allow for any consistent difference between pairs to be accounted. These control pairs are located approximately the same distance apart as the harvested pairs.

Analysis

We plan to relate the similarity of the harvested paired sites to buffer width (Figure 3). As a secondary aim we will also relate this relationship to factors such as slope, soil erodibility, whether one or two sides of the stream was harvested and the number of years since harvesting.

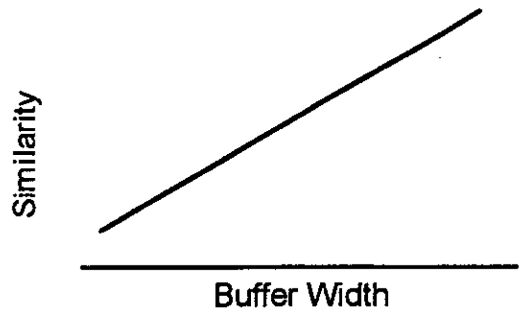

Figure 3. Simplified diagram of analysis to relate the effectiveness of buffers to their width.

If the above approach does not indicate any relationships, we will look for common patterns or characteristics between streams where harvesting affected macroinvertebrates and streams where macroinvertebrates were unaffected. This second approach may indicate certain types of forests, streams and/or activities within coupes are more like to result in impacts to the aquatic environment.

4. PROGRESS TO DATE

At present we have no data on the effectiveness of buffers. However, we can present some data that illustrates some of the approaches we plan to take. Specifically, data is given on the distribution of buffer widths within a coupe; the relationship between buffer width and slope; and the proportion of a buffer that occurs after the break of slope.

Variation in buffer width

Local conditions usually result in actual widths of a buffer being quite variable. There is a reported case in the literature of 100 m buffer not being totally effective at preventing an impact (Growns and Davis 1991). However, in this particular case there the buffer was crossed by at least one road (Borg *et al.* 1987). In another study, Growns and Davis (1994) showed a 100 m buffer was not effective, although one tributary had a buffer of less than 30 m width on one bank and gaps in the buffer were created by machinery crossing the stream. These and other examples may be due to narrower width at critical places in the catchment. Consequently, we plan to look not only at mean buffer width but also minimum buffer width. The variation in buffer widths that we have encountered to date is shown in Figure 4.

Relationship between buffer width and slope

The study by Davies and Nelson found no relationship between the effectiveness of a buffer and slope. However, one criticism of their study was that in planning the coupe forest officers would give wider buffers to coupes on steeper slopes. If this were the case results would be confounded and one would not expect a relationship between buffer effectiveness and slope. In our study we plan to test for this potential confounding factor. Figure 4 shows that there is little relationship between buffer width and the slope within the buffer.

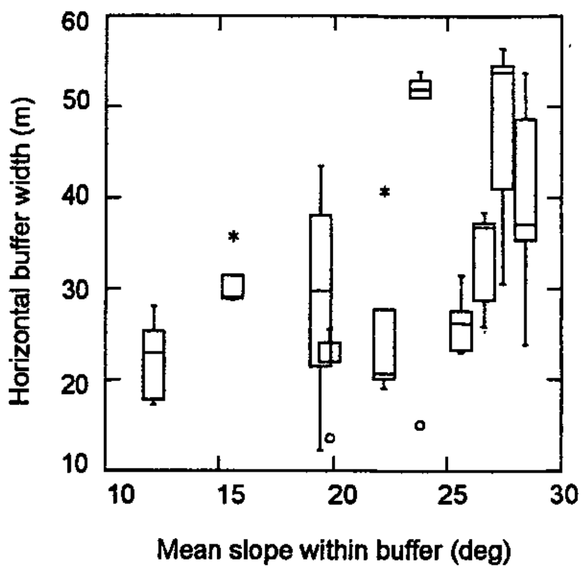

Figure 4. The (a) relationship between buffer width and slope within the buffer, and, (b) the spread of buffer widths within a coupe.

The proportion of a buffer that occurs after the break of slope

It has been suggested that buffers may be better defined not from the edge of the saturated zone# but from the edge of break in slope (Dignan *et al.* unpublished). The rationale for this is that after heavy rains the area after the break of slope may temporally become saturated. If this occurs the area after the break of slope is likely to have minimal ability to reduce suspended sediments and other pollutants from entering the stream. However, there is minimal information on the consequence, of defining buffers by this definition, with respect to the total width of buffers and thus the area of forests that buffers would occupy. Preliminary data on the percentage of the buffer occurring after the break of slope is presented in Table 1. We plan to assess the effectiveness of buffers using both definitions.

The saturated zone is the wet area surrounding the stream channel or if no evidence such an area from the edge of the stream channel (NRE 1996).

Table 1. Percentage of the buffer that occurring after the break of slope (by mean for each coupe).

Minimum:	11%
Maximum:	37%
Median:	26%
Mean:	26%
Standard Deviation:	7.7%

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Runoff and Soil Loss from Steep Snig Tracks under Natural and Simulated Rainfall

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Summary: Soil erosion processes under natural and high intensity simulated rainfall were measured on 22° snig tracks in northern NSW. Sediment yields of up to 23 t ha^{-1} resulted from single simulated rainfall runs and up to 52 t ha^{-1} from natural rainfall. Sediment depletion was apparent and erosion under most circumstances was supply limited. We conclude that erosion potential declines rapidly with successive erosion events and discuss some management implications.

1. INTRODUCTION

Ground-based logging in northern NSW is commonly carried out on slopes of up to 30 degrees. Rainfall erosivity is high and seasonally weighted towards summer months. The region is generally regarded as having a high erosion and water pollution hazard for logging[1] and an aware community is concerned to ensure adequate protection of environmental values.

In order to assess the actual water pollution threat posed by logging disturbance, it is first necessary to quantify runoff and sediment generation. We need to answer questions such as how much rainfall becomes runoff? How much sediment is mobilised by this runoff? Do the amounts change with time? A major study has recently examined some of these questions for several soil types in south eastern NSW[2]. The answers, however, may not necessarily apply to the steeper slopes and higher rainfall intensities of the north coast. In this paper we present data from field plot studies in this high-erosion environment. Data from both natural and simulated rainfall are presented.

2. STUDY SITE

The study site was located on an east facing side slope of a narrow valley located 15 km inland of Coffs Harbour in the foothills of the Great Dividing Range. The region has a warm temperate climate with an annual average rainfall of 1,731 mm (Coffs Harbour airport, 80 years of record). Hillslope gradient was a generally uniform 21°. Soil parent material was a lithic sandstone of the Lower Permian Coramba Beds[3] and colluvial influences were present. The soil profile was uniform to gradational with a dark brown silty loam surface layer and yellowish brown silty clay loam sub-soil (27% coarse sand, 23% fine sand, 33% silt, 18% clay). Structure grade ranged from moderate at the surface to weak or massive in the subsoil. The soil was a yellow earth[4] or Kandosol[5] with about 7% organic matter in the topsoil and 2 % in the subsoil. Coarse rock content was variable across the site, reflecting the colluvial origin. Vegetation at the site was in an ecotone between dry and moist hardwood forest and contained only a few tall trees in the overstory (~45 m). Most of the tree layer was of moderate height (20-30 m) with main species being *Eucalyptus acmenioides*, *E. siderophloia*, *Corymbia maculata*, and *Lophostemon confertus*. The midstorey was well developed and species-rich (54 recorded) consisting mainly of mesic tree species including several species of palm.

3. METHODS

Snig tracks in this environment are required by law to be drained by cross banks at a maximum spacing of 20 m. The experiment was therefore conducted by installing 20 m long by 5 m wide runoff plots aligned perpendicular to contours. Under natural rainfall, runoff was directed into a large modified Gerlach trough and passed through an electronically metered tipping bucket (≈4 L tip^{-1}). Rainfall intensity was measured simultaneously by a tipping bucket rain gauge. Sediment accumulated in the trough was weighed and corrected to dry weight basis. A small fraction of runoff was collected from every second tip of the tipping bucket for calculation of total sediment yield. Sediment yield was measured as soon as practicable after significant events or to reset accumulation to zero during longer periods with low rainfall. Data were therefore collected on a service period, rather than individual event, basis.

Three replicate plots were monitored between December 1995 and August 1998. Plot slope varied between 20-25° with an average slope of 22°. The plot surfaces were re-disturbed by a bulldozer in October 1996, October 1997 and immediately prior to the rainfall simulator experiments conducted in August 1998.

In August 1998 three snig track plots were selected for simulated rainfall measurements. These were not the same plots used for natural rainfall measurements, though located within the same experimental area. Two plots were freshly disturbed after previously being monitored under natural rainfall (replicates 1 and 2). The third plot (replicate 3) was freshly created after previously supporting natural vegetation. These plots were also used for studies of filter strip effectiveness[6]. The rainfall simulator consisted of standpipes fitted with pressure regulated nozzles. The system was calibrated to provide a range of intensities and drop size distribution representative of natural rainfall and has been described in detail previously[2]. At the lower end of each plot, runoff was directed through an RBC flume which was fitted with a capacitance probe to record runoff height. All simulation runs were of 30-minute duration with 10 runoff samples taken for sediment analysis at approximately 3-minute intervals. Replicates 1 and 3 received events of ≈75 mm h^{-1} and ≈135 mm h^{-1} with a separation of 1-1.5 h between runs. Replicate 2 received the same followed by a third event repeating the higher intensity. Rainfall intensity varied slightly between events (Table 1). The smaller and larger intensities represented 30-minute duration events with a recurrence interval of 3 and 50 years respectively.

4. RESULTS

Runoff coefficients under natural rainfall were generally around 30 to 50% for the largest events and much lower for small events. Under simulated rainfall runoff coefficients were higher (50 to 80%) but only on replicates 2 and 3 (Table 1). There was generally an increase in runoff coefficient with successive simulation runs due to both rainfall intensity and soil moisture being greater. Runoff coefficients of replicate 3 were unusually low compared with the other two plots.

Table 1. Summary of simulated rainfall results.

Run	Rain intensity (mm h^{-1})	Total runoff (mm)	Runoff Coeff. (%)	Mean sed. Conc. (g L^{-1})	Total sed. yield (t ha^{-1})
Plot 1 (old plot, freshly disturbed)					
1	78	23	60	19.5	4.6
2	134	56	83	18.9	10.6
Plot 2 (old plot, freshly disturbed)					
1	74	19	52	91.4	17.8
2	124	39	63	58.9	23.4
3	134	47	71	31	14.8
Plot 3 (new plot)					
1	74	12	31	200.6	23.5
2	132	20	30	102.4	20.1

Plot sediment yields for individual service periods under natural rainfall ranged from 0.01 to 52 t ha^{-1} (Figure 1). Only a few service periods produced sediment yields over 10 t ha^{-1} and all of these contained 30-minute rainfall intensities above 30 mm h^{-1}. The largest yields occurred in period three which contained the highest recorded 30-minute rainfall intensity of 67 mm h^{-1}. Low runoff and sediment yields in plot replicate 1 resulted from a high coarse rock content (~50% by volume) in the soil.

Under simulated rainfall the highest sediment yields of around 23 t ha^{-1} in 130 mm h^{-1} runs were only marginally higher than the lower intensity first run in plots 1 and 2 (Table 1). In plot 3 the sediment yield actually fell slightly in the high intensity run. These trends occurred despite higher rainfall intensity (erosivity) and higher runoff coefficients suggesting exponentially stronger erosive energy. That the erosion process was proportionately weaker in plots 2 and 3 is reflected in the substantial decline in sediment concentration with successive runs on the same plot. These trends reveal a pronounced sediment depletion process with successive events. The sediment concentrations of plots 2 and 3 were, however, extreme at 100 and 200 g L^{-1} reflecting the abundance of loose material available for transport.

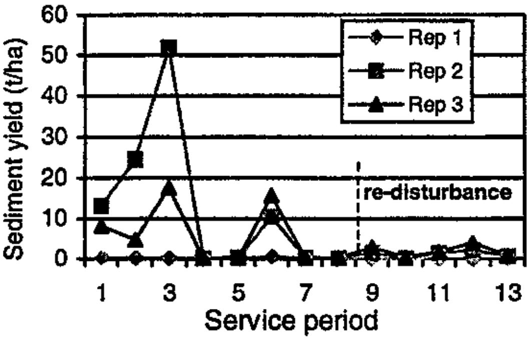

Figure 1. Plot sediment yields for each service period under natural rainfall.

Sediment depletion was also recorded within simulation runs. Figure 2 shows sediment flux over time for each run. Sediment flux is a measure of the mass of sediment per unit width of flow per unit time and is the product of sediment concentration and discharge. Under steady state rainfall and discharge it is an indicator of sediment availability. In plot 3, where there was initially an abundance of sediment available for transport, flux during the high-discharge second run was less than the first run resulting in a lower total yield.

Similar patterns were observed under natural rainfall. After the commencement of natural rainfall observations, one storm event containing a peak 30-minute rainfall intensity of 67 mm h^{-1} resulted in a sediment yield of 23 t ha^{-1} (mean of three snig track plots). A subsequent event with 655 mm of rainfall over three days, produced a mean of only around 10 t ha^{-1}. The highest 30-minute intensity throughout that event was, however, only 37 mm h^{-1}.

The difference in initial sediment yields between the three simulated rainfall plots was attributable to differences in soil surface conditions. Plot 3 had a loose, powdery surface whilst plot 1 was cohesive and plot 2 was intermediate. Having supported vegetation when disturbed, the soil of plot 3 was drier than the soil of plot 1 and, to a lesser extent plot 2. The effect of bulldozer traffic on the same soil in these different moisture states resulted in surfaces with different soil erodibility. Plot 3 was the only plot in which distinct rills formed.

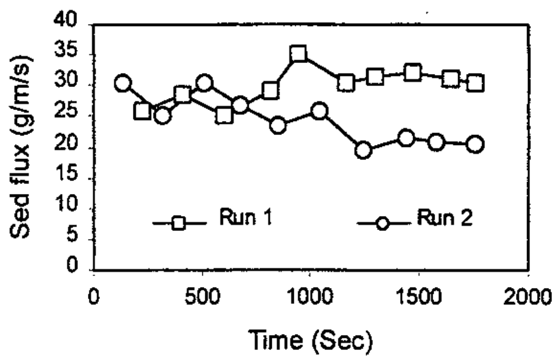

Figure 2. Sediment flux of plot 3 measured at three-minute intervals during simulated rainfall run 1 (74 mm h^{-1}) and run 2 (132 mm h^{-1}).

Natural rainfall observations supported simulated rainfall observations concerning the effect of soil moisture at disturbance on sediment availability. After re-disturbance in October 1996, high soil moisture resulted in a more cohesive surface. Subsequent sediment yields were consistently low. The highest single yield was only 4 t ha^{-1} from a period containing an event with a maximum 30-minute intensity of 63 mm h^{-1}. This compares with the yields of 20-50 t ha^{-1} from similar and smaller events in the first post-disturbance period.

5. DISCUSSION

Our data provide some insight to the questions posed earlier.

How much rainfall becomes runoff?

The lower runoff coefficients from natural rainfall probably stemmed from a higher proportion of simulated rainfall occurring under high intensity. These data represent the range of runoff coefficients for large events on snig tracks and accord with previous reports[2].

We found that whilst runoff coefficient on these steep slopes is variable, the situation may have been confounded by the fact that very high coefficients were associated with plots that had received several disturbance episodes. On the other hand, the freshly disturbed snig track did not experience as much compaction as an operationally created snig track where compressive and shear forces on the soil would cause greater reductions in infiltration rate. On this basis runoff coefficients of 30-40 % might be expected from low trafficked snig tracks whilst 60-80% may be more likely from highly trafficked snig tracks. These values are in general accordance with those observed on 15° snig tracks across three different soil types in south eastern NSW[2].

How much sediment is mobilised?

We have shown that under high intensity rainfall, considerable sediment loads can be mobilised from snig tracks. Sediment yields from events of similar size were two to three times those measured in south eastern NSW[2]. Similarly, mean sediment concentrations were generally higher in the current study, with the highest concentrations from plot 3 being an order of magnitude greater than the highest concentrations in the earlier study. There were two important differences in south eastern NSW study. Firstly, slope was less, around 15°. Secondly, simulations did not occur until 6 months after disturbance and evidence from the current study suggests that considerable sediment depletion may have occurred in that time.

Are responses influenced by rainfall intensity or temporal trends?

Whilst total sediment yields increased with rainfall intensity, the general trend was a decline in sediment concentration. This suggests that either particle detachment by overland flow was more important than by rainfall impact, or that sediment availability had declined. The dramatic fall in sediment concentration and overall erosion on plot 3 suggests a potentially very rapid temporal reduction in erosion susceptibility of snig track surfaces. A similar sediment depletion process observed in the southern NSW[2] suggests that a constant decline in soil erodibility state will confound attempts to define an erosion versus rainfall intensity relationship for logging disturbed sites.

Management implications

Sediment depletion suggests that the highest sediment yields are possible when the track is freshly disturbed. This highlights the importance of progressive installation of soil conservation measures such as drainage banks rather than waiting until operations have been completed.

From an erosion hazard perspective, there are two important implications from this work. Firstly, that sediment availability can vary considerably on the same site depending on the soil condition when it was disturbed. This illustrates some of the problems with soil erodibility concepts if state factors can be as important as inherent soil properties. The second implication relates to the role of sediment depletion in site recovery from raised erosion hazard. Whilst the recovery of vegetation is often viewed as a critical factor in surface stabilisation, it is suggested that initial decline in erosion may be strongly influenced by sediment depletion well before revegetation commences. This mechanism would appear to apply to the current soil type and there is some evidence for it on other sites[2]. Data from slopes of 25 to 30° are lacking and soils of low aggregate stability may display less pronounced sediment depletion.

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Sediment Generation and Management on Forest Roads – A Queensland Perspective

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Summary: For much of a forest plantation rotation, forest roads represent the major area of bare soil, and therefore have potential to be significant long term sources of sediment. Control of sediment and runoff from roads is thus a major priority for sustainable management of forest plantations. This paper outlines results from field studies investigating the generation and management of runoff and sediment from forest roads in South East Queensland. Experiments included; a) rainfall simulation to determine erosion from road surfaces, b) overland flow experiments to investigate erosion in table drains, c) sediment laden overland flow to assess the effectiveness of vegetated filter strips, and, d) a range of modeling approaches to predict the infiltration of overland flows discharged onto hillslopes. A simple technique to predict the effectiveness of vegetated filter strips is also presented.

1. INTRODUCTION

In a review of impacts of timber harvesting on streams, Campbell and Doeg (1989) noted that roads in forested country, particularly those constructed during harvesting, were one of the major sources of suspended sediment. Reasons for this include their lack of vegetation cover and surface of loose, fine, material overlaying an impermeable compact layer. Because road networks in managed plantations remain in place throughout the cropping cycle, they can act as sources of sediment throughout a forest rotation.

The preferred method for controlling sediment movement from forest roads is to spread runoff water and avoid channelisation. However, where concentration of runoff cannot be avoided, standard design techniques include table drains that run along roads sides and turn-out drains at intervals to limit flows in drains to non erosive levels. In forest plantations, turnout drains typically discharge runoff into areas of vegetation or surface litter where entrained sediment will be deposited rather than being transported off-site. When roads occur in close proximity to watercourses, vegetated filter strips (VFS) can provide an additional barrier to the movement of sediment into waterways.

2. OBJECTIVES

The objective of this research is to improve the sustainability of forestry operations in South East Queensland. Specifically, field research was conducted to:

- determine the levels and characteristics of sediment generated from unsealed forest roads and table drains; and
- evaluate and predict the effectiveness of sediment trapping and water quality improvement measures, including hillslope discharge and vegetative filter strips.

Results from this research are reported in detail by Loch *et al.* (in press) and Costantini *et al.* (in press.)

3. SEDIMENT GENERATION ON ROADS

Simulated rainfall was applied at approximately 100 mm/h for 20 minutes to sections of unsealed road 4 m long and 1.5 m wide. Road surfaces included gravelled, ungraveled and partially grassed. Runoff samples were taken over the duration of the rainfall event and analysed for total sediment load (gravimetrically) and for sediment size distributions. Surface samples from the rainfall plot were also collected and analysed for size distribution. The effect of vehicles traversing the road when wet was also investigated.

The results showed a high level of enrichment of fines under rainfall and relatively little effect of road slope on erosion rates. This is consistent with predominantly interill transport at low slopes. Gravel surfaces were found to reduce the generation of fine sediment (< 0.050 mm) by more than 500%. Roads with 50% grass reduced sediment generation by more than 400%. Wheel tracking the plot when wet increased sediment concentration in runoff by approximately 25%.

4. SEDIMENT GENERATION IN TABLE DRAINS

Ten metre lengths of two table drains were selected at each site to investigate sediment generation from table drains. Flow rates of 2, 4, 6, and 8 l/s were applied to the drains, each for four minutes. Runoff samples were collected and analysed for total sediment. The table drains tested were found to be largely resistant to scour; the only drain in this study to actively erode was both steep and in a highly erodible condition following re-forming maintenance activities. It was concluded that in the area studied, erosion of properly designed table drains is unlikely.

5. FILTER STRIP TRIALS

Sediment laden flows (2 and 4 l/s) produced using soils from two sites (Imbil and Toolara) were introduced to plots 0.3m wide at range of lengths, vegetation types (6) and slopes (from 6 to 29%). Samples of inlet and outlet flow were sampled and sediment concentration and characteristics determined. Infiltration was not considered to be significant at these high flow rates and was not considered in calculations.

Results showed that most deposition occurred within the filter strip, with the sedimentation fan extending downslope as the run progressed. The effectiveness of VFS was found to be variable in terms of total sediment. However, the effectiveness was relatively consistent when the sediment was considered in terms of its constituent size fractions. The results showed:

- 95% of particles < 0.050 mm were not trapped in filter strips;
- most particles > 0.125 mm were trapped by filter strip;
- there was a high variation in the proportion of sediment trapped in the size class between 0.05 mm and 0.125 mm.

These results indicate that removing sediment < 0.050 mm will be difficult to achieve using VFS. To enable prediction of the effectiveness of VFS, a simple conceptual approach was used equating the filter strip to a sediment pond with variables of settling velocity, flow depth and residence time. Model predictions compared well with field observations, indicating that most of the sediment > 0.125 mm would be trapped by the filter strips, and most of particles < 0.05 mm would not be deposited under these conditions. For the 0.125 to 0.050 mm size class, predicted and observed sediment yields were compared using regression analysis, resulting in a significant r^2 of 0.62 and a slope close to 1.

These results suggest that this simple approach may provide an easy to use and effective method for practitioners to predict sediment movement through a filter strip.

6. HILLSLOPE DISCHARGE

Data from road rainfall simulation experiments and table drain flow studies were used to parameterize hydrology and erosion models to allow simulations of infiltration downslope of table drain discharge points. The scenario modeled was a 60 m long 2 m wide road with 4% crossfall draining into a table drain with a longitudinal gradient of 7%. The table drain discharged to an area 6 m wide. Runoff from a 20 minute rainfall event with an recurrence interval of 10 years was simulated.

The KINCON model (Connolly and Barton 1990) was used to create a hydrology pass file to allow the CREAMS (Knisel 1980) model to predict sediment detachment and transport within the road/drain system. The ANSWERS model (Beasley *et al.* 1980), using a Green and Ampt representation of infiltration, was used to model flow and infiltration of runoff following discharge to the hillslope.

Predictions of hillslope runoff patterns for scenarios with and without flow spreading demonstrated the importance of this aspect of hillslope discharge. Results from simulations showed that appropriate flow spreading could reduce the distance surface flow moved downslope from 70 m to 20 m in some cases.

7. CONCLUSION

This study provides information to assist with the development of practices to improve the sustainability of the forestry sector in South East Queensland. Specific conclusions drawn from this study include:

- Hillslope infiltration provides the best option for the management of sediment laden road runoff.
- Only short lengths of road surface approaching water courses should be directed to water course filter strips.
- Roads with table drains discharging in the vicinity of watercourses should be graveled.
- Table drains should discharge to areas with high infiltration rates and flow spreading should be maximised.
- Residue retention practices should be adopted during inter-rotational periods to maintain high roughness and infiltration rates.
- Graveling, grass cover and consolidation of road surfaces reduced sediment loads.
- Avoid pushing loose material generated from road maintenance into table drains as this material will be easily transported.
- Fine sediment will be very difficult to remove using VFS – need to reduce sediment generation, prevent flow concentration and discharge runoff as high as possible in the landscape instead.

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Analysis of the Fish River Road Landslide, Mersey Valley, Tasmania

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Summary: The Fish River Road landslide and associated fill batter failure are developed in dolerite talus (colluvium) overlying glacial deposits. This combination of geological materials occurs on high-altitude steep land in Tasmania and has not previously been identified as posing a landslide hazard. Landslides may occur if roads are cut through impermeable glacial till overlain by water-saturated dolerite colluvium - on the Fish River Road removal of the toe of a pre-existing landslide has reactivated landslide movement. Fill batter failure may occur when road builders unwittingly construct roads from fill containing a high proportion of unconsolidated fluvio-glacial sands - on the Fish River Road incorrect culvert placement and consequent water-saturation of the fill batter sands has resulted in road collapse. At-risk areas in dolerite colluvium are zones of subsurface drainage that can be identified by boulder lines contained within slightly concave slopes.

1. INTRODUCTION

Landslides can have catastrophic and expensive effects on river systems. For example, a landslide on 3 March 1999 in the Great Western Tiers, Tasmania deposited about 20 000 m^3 of sediment into the Meander River and the Tamar, and about $200 000 worth of dredging will be required to restore a channel in the Lower Tamar. A landslide and an associated area of fill batter failure on the Fish River Road in the Mersey catchment have a much smaller volume. However, catastrophic movement of this landslide and collapse of the road would have important implications for the safety of forest workers, ecological values, production, sediment supply into Lake Rowallan, access to the forest, and public perception of the forestry industry.

Landslide risk is recognised in steepland soils formed in dolerite talus overlying Paleozoic sedimentary rocks (Grant et al. 1995). However, this geological combination does not occur at the Fish River Road landslide. When tension cracks were noted in the Fish River Road engineers reported on the problem (Weldon and Sloane 1997) and identified two separate zones of instability: (1) a zone of Fill Batter Failure; (2) a zone of inferred Active Landslide Movement. A blocked culvert was considered partly responsible for the landslide movement and fill batter failure. Because of plans to log in the area, and because a new landslide had occurred between 1997 and 1999, we made a detailed field study of the soils and the geology of the landslide location to determine what factors were causing instability. The study was requested by Forestry Tasmania.

2. LOCATION AND SITE CHARACTERISTICS

The new landslide and the associated area of fill batter failure are on the east side of the glaciated upper Mersey Valley, Tasmania at an altitude of 840 m. Slopes are uniform and about 28° with a westerly aspect. The valley side is slightly concave at the landslide location and the centre of the concavity contains very large boulders, but no flowing water. Vegetation is wet sclerophyll forest dominated by *Eucalyptus delegatensis*. Mean annual rainfall at the nearest weather station (Moina) is 1817 mm and is likely to be similar at the landslide site. Despite the high rainfall there are few streams. Most drainage appears to be subsurface. Streams that are evident may emerge from the ground and disappear after a few metres of flow.

3. OBSERVATIONS AND DISCUSSION

Landslide

The volume of material in the new landslide is estimated to be in the range 1000-1400 m^3.

Examination of a soil profile in the road cutting showed that the unstable landslide material is weathered brown bouldery dolerite colluvium incorporating silty sediments. Approximately 2 m below the present surface of this material is a waterlogged zone which is identified as a paleosol consisting of a litter layer, and an A horizon overlying a B horizon). The paleosol represents the previous ground surface overlain by landslide material. The litter layer and A horizon of this paleosol are water-saturated and have gley colours as a result of reducing conditions induced by organic matter decomposition. The brown B horizon is water-saturated. Below the paleosol is a grey cemented gravelly till that is impermeable. The weakly weathered nature of the till suggestes that it is a deposit of the Last Glaciation. Elsewhere in road cuttings along Fish River Road streams can be seen emerging at the contact of the brown bouldery colluvium and the grey cemented gravelly till.

The immediate reason for the new landslide was removal of the toe of the pre-existing slump. Removal of the landslide toe has reactivated the old landslide.

Fill Batter

In the area of fill batter failure the road bench has been cut into dolerite colluvium overlying uncemented matrix-supported bouldery sands of low strength. We interpret these bouldery sands to be fluvio-glacial sands of the Last Glaciation. Road batters have been constructed from a mixture of the dolerite colluvium and the sands. After heavy rain both the in-situ bouldery sands and the fill sands have become water-saturated and unstable. As a result the batter has collapsed, taking with it some of the road. A blocked culvert, the wide spacing between culverts, and the absence of a culvert at the lowest point of the road (which is probably over a line of underground drainage) are likely to be contributory causes for the fill batter failure.

4. CONCLUSIONS AND RECOMMENDATIONS

- In high-altitude soils developed in dolerite talus (colluvium) the geological and hydrological conditions that predispose land to instability are (1) impermeable cemented till overlain by water-saturated dolerite colluvium; (2) unconsolidated fluvio-glacial sands.
- Landslides may be induced when roading operations remove the toe of old landslides and in this way reactivate landslides. Fill batter failure may occur when roads on steep slopes are partly constructed using unconsolidated sands, and road drainage does not take account of the risk of this material becoming water-saturated.
- Areas that are "at risk" of landslide movement on high-altitude slopes mantled by dolerite colluvium are hard to identify, but concave areas on valley sides, with linear boulder fields, are probably zones of underground drainage having potential landslide risk.
- To avoid inducing landslides particular care must be taken during roading on glaciated valley sides. Minimum disturbance is recommended for boulder lines and associated concave areas, which may indicate subsurface drainage. Such boulder lines should be treated as Class 3 streams and a 40 m (total width) streamside reserve is appropriate.
- To avoid fill batter failure, low points on roads must be drained by culverts. Where unconsolidated sands are encountered during road building, culverts must be closely spaced to ensure that sands are never saturated.

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Road to Stream Connectivity: The Relative Roles of Slope, Discharge and Soil Erodibility

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Summary: Unsealed roads are recognised as a dominant source of sediment in forested catchments. However, it is the degree to which these sources are directly connected to the stream system that will determine the likely impact on water quality. Recent studies have identified a consistent and significant relationship between the contributing area and slope at the discharge outlet in explaining gully erosion at road outlets. This paper examines the application of this area:slope threshold model in two contrasting areas in southeastern NSW with very different soil types. Results highlight the need for caution in using this relationship in areas where the critical shear stresses required to initiate a channel are predominantly a function of soil cohesion rather than contributing area and slope.

1. INTRODUCTION

The importance of roads as a source of excess sediment and runoff is well recognised in the literature (Reid and Dunne, 1984; Grayson *et al.*, 1993, Croke *et al.*, this volume). Roads are highly compacted surfaces with low infiltration capacities that generate high volumes of runoff compared to other, less compacted parts of the landscape. The significance of gullies and other channelised pathways as persistent and efficient delivery routes for road-derived sediment has also been recognised (Montgomery, 1994; Wemple *et al.*, 1996; Mockler and Croke, 1999). Conceptually, road to stream linkage can be perceived as the expansion of the existing natural channel network through the connection of road segments to streams by gullies. Table drains or roadside ditches which route road generated runoff from the road surface to the drain outlets, and the gullies that link the drain outlets to streams essentially function as first order channels. Mockler and Croke (1999) report for example that a road network occupying < 1% of the catchment area resulted in direct connectivity along ~14km of the road length and contributed to a total increase in drainage density of ~6%. These persistent delivery pathways can have a negative impact on stream health and ecosystem diversity.

A number of studies have now quantified the dominant factors responsible for gully initiation at road drainage outlets (Montgommery, 1994; Wemple *et al.*, 1996, Mockler and Croke, 1999). Collectively these studies, have identified a consistent and significant relationship between the contributing area or length of road surface and the gradient of the hillside at the discharge point. The area-slope threshold relationship, which was used previously as a conceptual framework to quantify channel initiation processes on natural hillslopes (Dietrich *et al.*, 1992; Montgommery and Dietrich, 1994) appears to offer considerable potential as a management tool in road design and planning. In particular, the model offers the potential to be used to predict both the spatial extent of gully erosion at road drainage outlets, and thereby assess the degree of connectivity for a catchment, and to plan road drainage to prevent erosion occurring at these points in the landscape. The fact that the model has been applied to catchments in the United States and southeastern Australia lends support to its widespread application in a range of forest environments, irrespective of climate or soil type. The purpose of this paper, therefore, is to explore the robustness of this model across two different catchment topographies and soil types in southeastern NSW. We summarise the findings of a detailed road-to-stream connectivity analysis undertaken in the Cuttagee Creek catchment near Bermagui on the coastal lowlands of NSW, and compare these with more recent research results from catchments in the granite terrain around Bombala, south eastern NSW. The results of the Cuttagee Creek analysis have been reported in detail elsewhere (Mockler and Croke, 1999). Our application of the threshold model in two contrasting catchments provides important insights into the relative importance of factors such as slope, road contributing area and soil erosion in explaining gully erosion and road-to-stream linkage in catchments. Importantly, the study allows us to distinguish between environments where this model may be quite appropriate and those where other factors may be more dominant in explaining road-to-stream linkage.

2. STUDY SITES AND METHODS

Data relating to road-to-stream connectivity were collected in two broad regions of the Eden Management Area in southeastern NSW. A detailed analysis of road connectivity was undertaken firstly within the 60km^2 Cuttagee Creek catchment which is located approximately 7km south of Bermagui (see Mockler and Croke, 1999). Sediment erosion rates on these road surfaces are reported in Croke *et al.*, (this vol). Additional work was undertaken in forested catchments near the town of Bombala on the southern Tablelands of NSW. Cuttagee Creek is a predominantly forested catchment with a characteristic steeply incised drainage network composed predominantly of first and second order streams which drain into Cuttagee Creek, the main 5th order stream draining to Lake Cuttagee, a small estuarine lake at the coast. Soils have developed from Ordovician Metasediment parent materials and are predominantly gravelly-duplex loams, which are not considered to be highly erodible. In contrast, the areas around Bombala reflect a gently rolling topography with less steeply incised valley and drainage networks. Soils have developed primarily from the intrusive granitoids of the Bega batholith. Croke *et al.*, (1997) compared sediment erosion rates on forest snig tracks in both geological regions and confirmed the highly erodible nature of the granites, with sediment yields almost one order of magnitude higher than the metasediment sites for similar rainfall intensities and slopes. However, field data indicated that the metasediment derived soils had high sediment delivery potential, that is, the fine-grained nature of these soils contributes to higher sediment concentrations in surface runoff which are transported longer distances than the coarser granite particles. In terms of potential impacts to in-stream water quality, the particle size distribution of soils in these coastal lowland catchments may pose a greater threat to stream turbidity than the more erodible granite soils which travel relatively short distances before being redeposited on the hillslope. Rainfall patterns are broadly similar but the Tableland region around Bombala is characterised by slightly lower rainfall intensities with no marked seasonality compared with the short duration, high-intensity storms that occur during summer months along the coastal lowlands (NSW EIS, 1994).

The location of all road drainage structures within selected sections of both regions was recorded using a Differential Global Positioning System (DGPS). Drain locations were stored as a digital coverage on an ArcView Geographic Information System (GIS). A more detailed outline of the methodology is presented in Mockler and Croke (1999). A survey of the road surface, drain structure and linkage features was then conducted on selected road segments. Contributing length and width of road segments, batter heights, drain and gully lengths were measured to 0.1m with surveyors' tape and road, drain, gully, and hillside gradients were recorded with a clinometer to the nearest degree. Each road drainage outlet was assigned to a specific category of road to stream linkage as outlined in Table 1. These linkage classes were assigned on the basis of the observed geomorphological impact of the erosional features and the perceived sediment delivery potential of the pathways and are consistent with the definitions used in the other overseas studies.

The three main types of drain used on the road network in this region are: mitre drains, push-outs, and culverts. Mitre drains are extensions of the road table drain constructed by outwardly curving the table drain away from the road alignment and onto adjacent hillsides. Push-out drains are very similar to mitre drains in construction but are generally located along low gradient ridges and saddles serving to relieve accumulated runoff from the table drain. Culverts are usually found in association with cut and fill type road alignments and drain the cut batter side of the road, via cement piping, which discharges water onto the fill batter side of the road and then down hill. In total, 171 mitre drains, 23 culverts, 20 push out drains and 4 near stream cross-bank drainage structures were surveyed along approximately 13.6km of road network in the Cuttagee Creek study area. Ninety-eight culverts and 143 mitre drains were surveyed along almost the same length of road (~13.4km) in the Bombala survey.

Table 1: Classification of road to stream linkage used to classify geomorphological features found at drain outlets (Mockler and Croke, 1999).

Class	Linkage	Visible Geomorphological impact
1	None	Channelised flow pathways or rills extend <10m from drain outlet.
2	Partial	Discontinuous rills and gullies extending >10 m from drain outlet and terminating >10 m from stream
3	Gullied	Continuous rills and gullies extend from drain outlet into stream channel.
4	Direct	Drains discharge directly into stream or road runoff flows directly into streams.

3. ROLE OF AREA AND SLOPE

Following the approach of Montgomery (1994), a threshold curve for these two variables, contributing area and hillside gradient, was found to successfully discriminate between gullied and ungullied drain outlets in the Cuttagee Creek study catchment (Figure 2a).

The threshold curve in Figure 2(a) has the function:

$$L(m) = \frac{25(m)}{\sin\theta}$$

where: L(m) is the maximum contributing length of the road segment, 25(m) is the calculated coefficient for the threshold curve, and, θ is the discharge hillslope gradient in degrees. Although using contributing road area yields a more distinct separation of linked from unlinked points contributing length is used here because it simplifies the practical application of the threshold into a design table for implementation by road construction managers. In the Cuttagee Creek catchment, most culvert outlets (91 %) were found to have gullies because they tend to drain water from longer segments of road, capture water from upslope contributing areas and discharge the runoff through pipes, which tend to focus the discharge into a single flow path. Only 7 % of mitre drains and 22 % of push-out drains were found to have gullied outlets. Discharge from a push-out or mitre drain tends to be spread out onto the hillside, as the outlets of these two drains are wider and lower in gradient than culvert pipes.

The threshold relationship evident in the Cuttagee Creek catchment provides further support to the dominant role of slope and area in explaining gully erosion across a range of environments. The Cuttagee Creek relationship was used to construct a practical drain design table, which outlines maximum contributing length required to prevent gully erosion occurring at road drainage outlets (Table 2). The coefficient of determination (eg 25m for Cuttagee Creek) will vary on a catchment by catchment basis together with the shape of the fitted curve which varied together with the shape of the fitted curve which varied between the studies conducted in the United States (eg Wemple et al., 1996) and Cuttagee Creek.

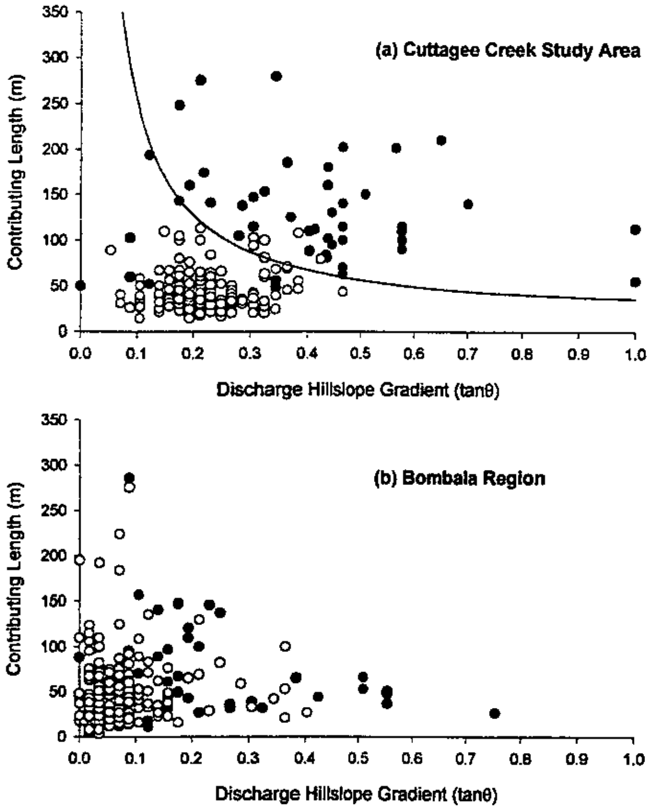

Figure 2: Plot of contributing length (m) versus discharge hillslope gradient ($\tan\theta$) for surveyed sections roads in (a) the Cuttagee Creek catchment, and, (b) in NSW State Forests in the region south of Bombala, NSW. Filled circles are gullied drain outlets and open circles are ungullied drain outlets.

Table 2: Proposed maximum drain spacing design table for Cuttagee Creek study area. The drain spacings are based on satisfying combined criteria of avoiding road surface erosion and gully erosion at drain outlets.

Roadway Gradient (degrees)	Drain Discharge Hillslope Gradient (degrees) 2.5	5.0	7.5	10.0	15.0	20.0	25.0	30.0
0	-	285	190	145	95	70	55	50
1	200	200	190	145	95	70	55	50
2	175	175	175	145	95	70	55	50
3	150	150	150	145	95	70	55	50
4	125	125	125	125	95	70	55	50
5	100	100	100	100	95	70	55	50
6	90	90	90	90	90	70	55	50
7	80	80	80	80	80	70	55	50
8	70	70	70	70	70	70	55	50
9	65	65	65	65	65	65	55	50
10	60	60	60	60	60	60	55	50
11	55	55	55	55	55	55	55	50
12	50	50	50	50	50	50	50	50
13	45	45	45	45	45	45	45	45
14	40	40	40	40	40	40	40	40
15	40	40	40	40	40	40	40	40

4. VALIDATION OF THE THRESHOLD MODEL

In contrast to the well-defined threshold relationship which appeared from road linkage data in the Cuttagee Creek catchment, there was no obvious relationship between these two factors for data in the Bombala region (Fig. 2b). There are two possible reasons why a threshold curve could not be established for the Bombala data. The first recognises the more erodible nature of soils in this region so that the critical shear stresses required to incise hillslopes and initiate channels are considerably less than the more gravelly and resistant metasediment soils. Additional data from a study on gully initiation in the pine plantations in the Kapunda area around Bombala also report the lack of a significant area:slope threshold relationship and suggest soil erosion as the most likely explanation (Prosser and Soufi, 1998). The authors also suggest that because of the low shear stresses characteristic of the granite soils, relatively small scale rainfall events with a recurrence interval of <2-y can be sufficient to initiate gully erosion on these hillslopes, particularly in the period immediately after plantation harvesting where soil cover is low and disturbance is at a maximum. Thus for any given road segment in the area around Bombala, it is not possible to quantify or predict the discharge or equivalent area required to initiate a channel, as this is also dependent upon soil cohesion, which the model does not explicitly consider.

The second compounding factor in the Bombala region is the presence of mixed forestry operations in the form of pinus radiata plantations and native eucalypt forest harvesting. Cuttagee Creek is managed only for native hardwood timbers and roadside hillslopes have not been intensively disturbed as part of the logging operation. In contrast, we observed that the majority of connected pathways or gully road segments in the Bombala region occurred on roads within the pine plantations. Plantation forestry is recognised to be significantly more intensive than selective logging of native eucalypts and involves heavy disturbance through planting, successive thinning operations and then clear cut harvesting itself. Prosser and Soufi (1998) also report differences in the strength of surface soils in the eucalypt and pine forests around this region lending support to some compounding influence of forest harvesting regime on the lack of a clear area:slope threshold relationship in Bombala.

The lack of a clear relationship in these granite regions highlights the additional contribution of poor soil strength and cohesion in defining conditions for gully erosion at road drainage outlets. This alerts us to the steps required before we can universally apply this model to a range of catchments in diverse topography and soil types. The significance of soil erodibility as a factor in explaining gully erosion suggests that in areas of known soil weakness, a threshold relationship would have to be constructed using some index of soil strength. Modelling tools which can assist in quantifying the critical shear stresses required for channel initiation in these environments may have considerable potential in assisting with the spacing guidelines in these environments (see Prosser and Soufi, 1998, Dawes and Croke, this vol).

5. MANAGEMENT OPTIONS

This study improved our knowledge of the factors that can lead to road to stream linkage. As with all cases of gully erosion, prevention is the best option, as once formed gullies are difficult to remediate and their effects may be long lasting (Prosser and Soufi, 1998). Some practical solutions that may be considered in environments where road to stream linkage is an existing, or potential problem include (1) effective **planning** of road alignments, and, (2) **prevention** of gully formation at road drain outlets through appropriate drain spacing and road location.

The most preferable options with regard to road planning, includes avoidance of high risk areas for road to stream linkage, particularly at stream crossings, on steep hillsides or on roads adjacent to streams and buffer zones. This could be achieved by planning roads to utilise ridge-top roading alignments where the distance between drain outlet and stream or buffer zone is maximised. Avoiding unnecessary stream crossings and road alignments where steep drain discharge gradients cannot be avoided are also desirable.

Current drain spacing guidelines used by NSW State Forests outline maximum separation of drains based on road travelway gradient which are aimed at preventing road surface erosion. However, to prevent erosion at the drain outlet, the gradient of the hillslope where road drain runoff is discharged must also be considered. Road drain outlet gully and erosion of the road surface may be prevented by utilising a suitably derived drain spacing table as part of the road construction, design and maintenance procedures (eg Table 4).

6. CONCLUSIONS

Gullies at road drain outlets can lead to high delivery of runoff from the surface of unsealed roads directly to streams. A number of studies had identified a significant relationship between contributing road area and slope and gully erosion at drain outlets. This suggested application of the model to a wide range of catchment topographies and geologies. Data from two contrasting soil types in the region around southeastern NSW suggest, however, that the compounding effects of soil strength in some areas may preclude the widespread use of this model for gully erosion prediction. In areas where the soils are relatively resistant, data from Cuttagee Creek and other studies overseas, suggest that the model should be robust which allows it to be used in the development of very valuable road design tables and drainage spacing guidelines.

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Victorian Soil Erosion Hazard Assessment Scheme

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Summary: The Victorian soil erosion assessment scheme is a newly developed method. It has two components. The first is a broadscale categorisation of inherent soil erosion hazard. Currently, this may be inferred from Land Systems mapping and associate Land Study documents and is intended for use by Planners. The second component of the scheme is the assessment of soil erosion hazard at the local (coupe) scale and is intended for use by Forest Officers who are not trained soil scientists. This component is a categorical points score system in which soil and site factors are assessed and scored using predetermined weightings. A series of tables are worked through and the results form a matrix that yields an erosion hazard rating of Low, Medium or High.

1. INTRODUCTION

The Victorian soil erosion assessment scheme is a newly developed method for NRE's Forests Service, and as such has not been fully field tested, and is being trialed over the next 9 months. The expectation is that it will be implemented to some extent during this autumn and winter as next years logging coupes and roading works are more fully reconnoitred. Full implementation will follow successful trialing. The scheme has been developed to provide a single system for state-wide use, which replaces a number of local systems that vary considerably in rigor and in application.

The consideration of soil erosion hazard is required by the Victorian Code of Forest Practices in several sections of the code: coupe planning, water quality and habitat protection, construction and siting of snig tracks and forwarding tracks, road drainage, and batter design. There is no requirement at present within the code for a particular methodology to be used. Consequently, the use of the scheme outlined below is not compulsory. Within the Code, there are a number of prescriptions and guidelines that act to minimise sediment movement and soil disturbance, both during and at the completion of logging operations. These include the construction and spacing of breaches and bars, design of road and snig networks and drainage, and protection of water courses by retention of vegetation in buffer and filter strips. The erosion hazard assessment is designed to be used in this context, consequently it is not a complete stand-alone scheme for erosion control.

2. DESIGN

The identification of soil erosion hazard has two components. The first is a broadscale categorisation of inherent soil erosion hazard. Currently, this may be inferred from Land Systems mapping (Rowan, 1990) and associate Land Study documents, for example "Land Inventory of East Gippsland - A Reconnaissance Survey" (Rees, 1996). These surveys are designed to provide a broad initial coverage of the State of Victoria. Their purpose is to map, describe and evaluate the bio-physical nature of the land as basic information needed to develop sound management systems. One element generally covered in this work is land deterioration (i.e. sheet, gully, streambank and wind erosion, soil compaction, mass movement and salting), including susceptibility to land deterioration. Much of the interpretation work is based on perceived differences in landform pattern and geology, with some consideration of climate, soils, native vegetation, land use and land deterioration. A variable level of ground-truthing is carried out, but some characterization of soils and land deterioration is always incorporated (see Rowan 1990). The quality, accuracy and reliability of the Land Systems mapping is very variable even when used at the Forest Management Area scale. The maps are intended for interpretation at a regional or landscape level only, and not for specific sites. Consequently, they are not suitable for use at the coupe planning level, except to provide general guidance. The Forest Management Plan for East Gippsland is an example of where Land System information has been used in conjunction with local knowledge to delineate areas with high erosion hazard. In this example, all logging coupes, new roads and upgrading of major roads in the high erosion hazard area will only be permitted in accordance with specialist advice, and excluded from areas with extreme erosion hazard. Additionally, there will be a general limitation on harvesting operations in areas with granitic-based soils and slopes steeper than 25 degrees. The need to seek specialist advice recognises the inherent variability within landscape mapping, particularly in relation to inherent soil erosion hazard.

The use of Land System information in conjunction with local knowledge to delineate areas with higher erosion hazard at the FMP level is a method that potentially has much merit. The major limitation with the method is the quality of the Land System information relating to inherent soil erosion hazard on forested public land, and the level of variability within landscape classes. Consequently, there will need to be additional input from specialists and practitioners who are familiar with the local forest conditions.

The second component of the scheme is the assessment of soil erosion hazard at the local (coupe) scale. The final hazard assessment is the product of two processes; **(1)** the assessment of inherent soil properties: (i) soil susceptibility to breakdown and (ii) infiltration and drainage characteristics, combined to give a soil erodibility classification, and **(2)** the soil erosion site factor rating, where the factors that impact on mobilisation and transport of sediment are considered. The overall system is a categorical points score system in which soil and site factors are assessed and scored using predetermined weightings. A series of tables are worked through and the results form a matrix that yields an erosion hazard rating of Low, Medium or High.

The factors used in this scheme have been chosen to give a meaningful assessment of erosion hazard, and are readily measured or estimated by people who are not trained soil scientists. Forest Officers employed by the Department of Natural Resources and Environment and responsible for logging coupe operations will implement the assessment. Training will be given to enable the various factors to be addressed confidently. This scheme has been designed so that it is workable within the present forestry structure and level of resources.

3. SOIL ERODIBILITY CLASSIFICATION

(i) Assessment of soil susceptibility to breakdown:
This is achieved following the approach taken by the Tasmanian system (Laffan et. al., 1996). The factors addressed in this table are: soil texture, soil aggregate stability, soil structure, soil colour, soil organic matter and the stoniness of the soil layer. The assessment is designed to be completed in the field rather than the laboratory. Soil texture is determined by the bolus method; soil aggregate stability by tests of dispersibility and slaking; soil structure by visual assessment of pedality, colour by chart guide, and organic matter and stoniness by visual inspection.

(ii) Soil permeability: This assessment is to yield surrogates for infiltration and drainage characteristics, as usual techniques are not suitable for the practitioners that will be doing the assessments and the time frames involved. The rationale behind this part is simply that where overland flow occurs the potential for sediment detachment and transport is high. Soils with low permeability and slow rates of internal drainage generate greater surface runoff than well drained soils if topographic effects are ignored. The following factors are assessed: soil texture, organic matter, depth to flow impeding layer, soil mottling, stoniness of soil layer, and mature tree stand height. It is considered that these easily assessed factors give a reasonable indication of water movement through the soil. Stand height is included as it gives an indication of the drainage status in that tall trees rarely grow well in waterlogged soil.

Soil erodibility classification is obtained from a matrix of the results of (i) and (ii). The assessment of soil susceptibility to breakdown needs to be made to a realistic depth, generally to the depth that soil disturbance is likely to occur. In most logging operations soil disturbance is unlikely to extend beyond a depth of 80 cm, with some operations disturbing to a lesser depth. In some soil types there are specific soil horizons which are at high risk of erosion if exposed/disturbed. If the logging operations do not specifically disturb these horizons then they need not be included in an assessment of soil susceptibility to breakdown. In relation to road construction, assessment should be to the depth that soil disturbance is likely to occur, which may be significantly more than 80 cm in the case of side-cutting. Within this context, each main soil horizon needs to be assessed, with the overall rating of soil susceptibility to breakdown by rainfall and runoff equivalent to the layer with the lowest rating.. Thin layers (generally < 10 cm thick) are not generally considered to be significant unless they occur at the soil surface and are therefore particularly important.

4. SOIL EROSION SITE FACTORS

The factors are: rainfall erosivity index, slope gradient, slope length and revegetation capacity. Erosivity is mapped for the state and the other factors are easily measured or estimated on-site. These factors consider the variables that influence the mobilisation and transport of soil particles. Slope uniformity and length influences the potential rate and length of sediment movement, and revegetation capacity refers to the capacity of the site to revegetate rapidly (within 2 growing seasons) with good ground cover following disturbance. Revegetation capacity is assessed by the potential for regeneration at the site as indicated by local experience, and by the vitality and structural composition of the pre logging vegetation.

The frequency of assessment needed to characterise each coupe will vary. In landscapes with relatively homogenous soils and gentle topography, a single test of susceptibility for soil breakdown, and permeability may suffice. Where there is a change in soil characteristics indicated by vegetation or evidence of natural erosion, or where operations indicate there are areas of concern, additional assessments should be made. Local knowledge is an important aspect of these assessments, and will assist in classifying areas that require greater or lesser intensity of assessment.

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A Tool for Assessing Stream Health in Forested Areas using Aquatic Macroinvertebrates

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Summary: The ecological integrity or 'health' of stream ecosystems has been traditionally measured through the collection of abiotic data such as turbidity, suspended solids and nutrients, with minimal assessment using instream biota. Biotic indices, which incorporate macroinvertebrates, fish, diatoms, algae and macrophytes have been gaining popularity as a tool for the assessment of stream health, they have a number of advantages over physico-chemical indices. In our study we are developing a tool, using aquatic macroinvertebrates, to provide an assessment of the ecological health of specific forested sites in a stream, stream systems and forested catchments. Macroinvertebrates were chosen over other biota, as they were the most practical biotic group to use in the smaller forest streams, where the study is focussed. In addition they are considered the most commonly used biotic indicator group in Australia. The tool is a predictive model that is built from macroinvertebrate data collected from streams in undisturbed-forested regions across Victoria. At a given location the model will be able to predict the suite of taxa that should be present at that site and by comparing this to the macroinvertebrate taxa that is actually present an assessment of the health at that site can be obtained. Where there is a much-reduced fauna than predicted by the model the site is classified as impacted. The model will be able to: (a) provide maps of stream health which will enable the pinpointing of potential disturbance(s); (b) target areas with more intensive sampling regimes to establish if specific activities are responsible for degrading the aquatic environment; and (c), be used in active adaptive management to monitor the effectiveness of management actions.

1. INTRODUCTION

It is important that the ecological integrity or 'health' of stream ecosystems be maintained and in situations where there has been a disturbance, improved. Traditional measures of stream health have been obtained through the collection of abiotic, physico-chemical data such as turbidity, suspended solids and nutrients, with minimal assessment using instream biota. Macroinvertebrates have a number of advantages over other biotic groups. Biotic indices and predictive models using macroinvertebrates provide a method for the assessment of the health of streams and for the ongoing monitoring of such systems.

The aim of our project is to develop a predictive model based on macroinvertebrates to assist in the assessment of stream health in forested areas of Victoria. The model will be based on both the Australian River Assessment System (AusRivAS) and the Invertebrate Prediction and Classification System (RIVPACS) used in the United Kingdom (see Wright *et al.* 1984).

2. BACKGROUND

Biotic indices

Biotic indices, which include macroinvertebrates, fish, diatoms, algae and macrophytes have been gaining popularity as a tool for the assessment of stream health. They have a number of advantages over physico-chemical indices which include:

- the ability to detect impacts such as altered flow regimes from dam construction or water abstraction;
- the detection of impacts long after they have occurred, which may be missed through chemical water sampling alone; and,
- ultimately the condition of the stream biota, including macroinvertebrates, is a reflection on the health of the stream.

Some examples/references of biotic indicators that are currently in use around the world include:

- diatoms, eg., NSW and far eastern Victoria (Chessman *et al.* [in press])
- phytoplankton, eg., Croome and Hötzel (1997)
- benthic algae, eg., Sonneman and Breen (1997)
- fish, eg., Karr (1981)

Macroinvertebrates as a biotic indicator

Macroinvertebrates are small animals without backbones and include aquatic worms, insects such as beetles and flies and crustaceans such as freshwater crayfish. They are an important component of the food chain, by breaking down organic matter and providing a food source for larger predators such as fish and birds.

Some specific advantages of macroinvertebrates over other biological indicators, include:

- Sensitivity to impacts - some macroinvertebrates are tolerant of impacts and their presence in a stream may be useful as an indicator, whereas some macroinvertebrates are sensitive and their absence may be indicative of an impact. What's more, different taxa are tolerant/sensitive to different types of pollutants and disturbances, making the identification and assessment of a wide range of impacts possible.

- Large range of life histories - macroinvertebrates have life histories ranging from weeks to years. This allows us to examine the fauna sometime after a disturbance has occurred to determine if it has had an impact on the fauna. It also allows consideration of disturbances that take place at many different time scales. This is in contrast to chemical water sampling and groups that have a smaller range of life histories such as algae.
- Cost effective - the collection, identification and analysis of macroinvertebrates is very cost effective compared to other biotic indicators and chemical water analysis.
- They are the most common biotic indicator used in Australia and the world in general. This has resulted in a large body of knowledge on the taxonomy, ecology, biology and ecotoxicology of macroinvertebrates.
- Most macroinvertebrates are non-migratory, numerous and diverse (unlike fish) allowing for replicate samples to be easily taken.

3. PREDICTIVE MODELS AS A TOOL FOR STREAM HEALTH ASSESSMENT

In order to monitor effectively for environmental impacts using biotic indicators, we need to know what taxa should be present at a site if the site is healthy. This has traditionally been done by using a series of control sites (eg. upstream of an impact) and/or by sampling at different times (eg. before and after the start of an impact). This procedure can work well, but in many common situations it is difficult or even impossible to find appropriate control sites close enough to an area where assessment is required. As a result, predictive modelling is becoming a more widely used tool which overcomes this problem.

A predictive model using macroinvertebrates

We are developing a predictive model using macroinvertebrates to assess the health of streams in forested catchments. The methods were first used in the United Kingdom but have been successfully adopted as a tool for the assessment of stream health in a number of countries including the USA, Canada, Spain and Australia.

The model is based upon the presence or absence of macroinvertebrate taxa recorded from sites that are of the "best possible condition", these are termed reference sites. At these sites, habitat, geographical (e.g. location), physical and chemical parameters are also measured. From this information a statistical model can be constructed that predicts the macroinvertebrate fauna that should occur at a site given it is in similar condition to the reference sites.

In addition to reference sites there are also monitoring sites, whose conditions are not known. From the model it is possible to predict the macroinvertebrate fauna at monitoring sites and compare this prediction is the macroinvertebrate fauna actually present. The more similar the predicted and observed macroinvertebrate fauna are, the closer a monitoring site(s) is to reference condition.

The steps to our model development

- Macroinvertebrate (biotic) and geographical, physical and chemical (collectively referred to as abiotic) data is collected from reference sites in forested streams, which are minimally disturbed, using a standard sampling protocol.
- The macroinvertebrate data is formed into biological groups using multivariate analysis (due to the nature of the type of data). These groups are clustered according to which taxa are present between sites.
- Variables are selected that explain the groupings partly from statistical analysis and partly from a knowledge of how the variables will effect the presence of macroinvertebrates at a site
- Discriminant functions are made that relate the selected abiotic variables with the site groupings. From these functions, probabilities are calculated which give the probability of a site belonging to a particular group.
- The output of this method is a list of taxa, with their probabilities, that are predicted to occur at a new site. This list of taxa expected at a new site is reduced to only include those that have a probability of occurrence greater than 50%.
- The expected taxa are compared to the taxa collected. One common method of making this comparison is to provide an observed to expected taxa ratio (O/E).
- This ratio is used for the assessment of the "health" of that site, the higher the ratio, the healthier the site.

Uses of the model

The model will act as a useful tool in these three main areas:

- <u>Mapping</u> of stream health - to provide large or small scale maps of stream health in a visual manner – such maps can be useful as a reporting technique for the dissemination of data in an easy-to-interpret format and for the identification of potential problems
- <u>Monitoring</u> for specific disturbances, e.g. sedimentation, nutrients, etc. -the model can be used to monitor for such disturbances after they have been pinpointed from the maps or by other means.
- <u>Management</u> of streams – as a component of active adaptive management, i.e., a decision tool for a change in management practices (Figure 1).

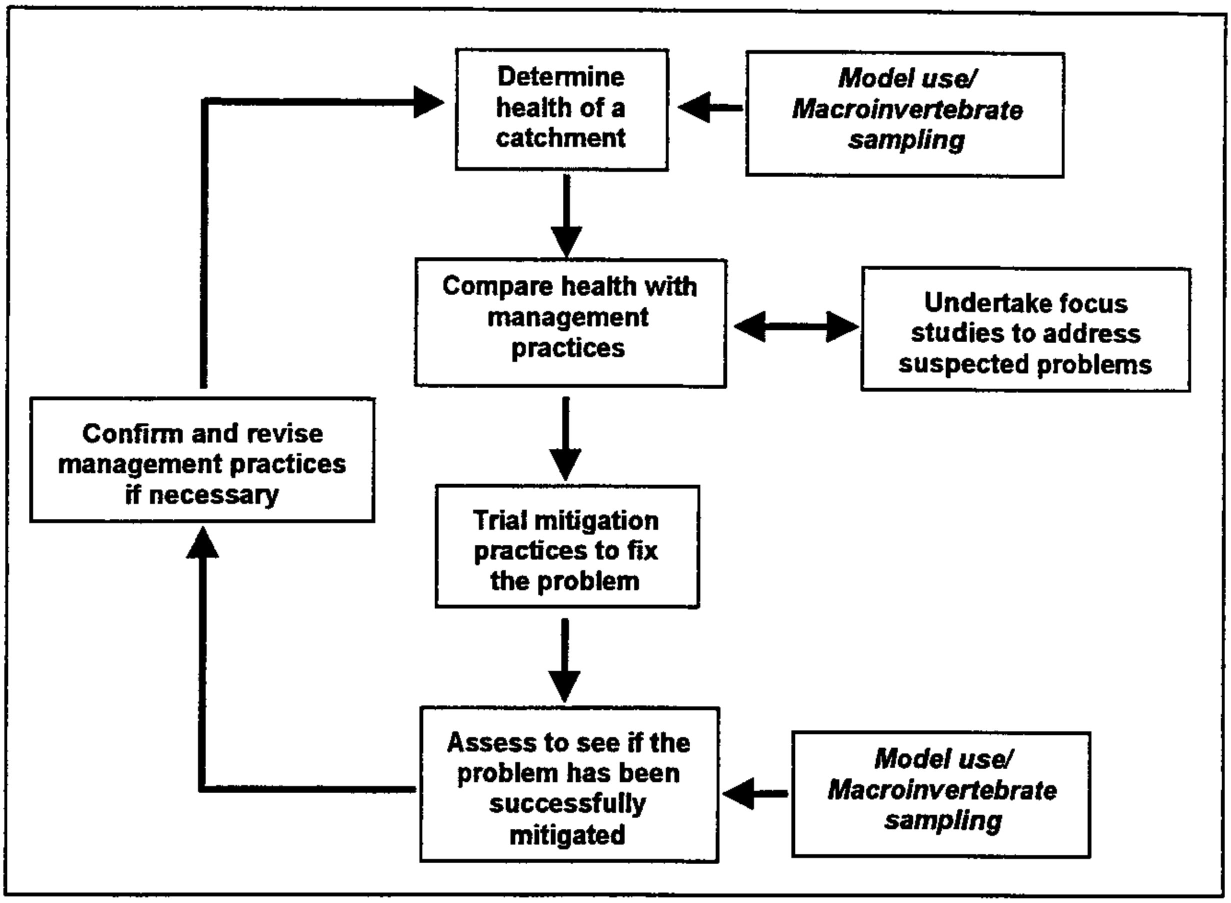

Figure 1. An active adaptive management framework

4. PROJECT STATUS

We have completed the collection of macroinvertebrate and abiotic data from over 200 sites in forested areas of Victoria. We plan to augment this with data collected by the Victorian Environmental Protection Authority. Macroinvertebrates have been identified to the genus level and data is currently undergoing analysis.

We expect the predictive model(s) to be produced by August 1999 and hope to be able to use the model(s) for: providing information to the government, the scientific community and the general public, and, by using the model as a monitoring tool, assist in the overall improvement of stream health in Victoria.

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Anticipating the Negative Hydrologic Effects of Plantation Expansion: Results From a GIS-Based Analysis on the Murrumbidgee Basin

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Summary: There are plans to raise the plantation area in Australia to over 3 million hectares, with much of the new area displacing what is currently grassland. It is well known that evapotranspiration rates in plantations are higher than in grasslands, yet there is scant acknowledgment of the fact that streamflows will decline significantly from afforested catchments. Here we evaluate an empirical relationship between mean annual runoff, mean annual rainfall and vegetation cover. It arose from work published by Holmes and Sinclair (1986) for a group of Victorian catchments, and is here evaluated for 28 sub-catchments of the Murrumbidgee basin in NSW where plantation expansion is occurring. We implemented the relationship in the ARC/INFO GIS, to predict mean annual runoff across the basin for the period 1986-1995 and obtained good results. We show how the model can be used to predict the average annual water yield changes that would ensue from afforestation in different parts of the basin.

1. INTRODUCTION

The Commonwealth and State governments of Australia have committed themselves to a vision of trebling the area of timber plantations by the year 2020 (DPIE 1997). This would raise the plantation area in Australia to over 3 million hectares, displacing what is currently grassland. The architects of the '2020 vision' argue that plantation expansion will help ameliorate environmental problems such as waterlogging and dryland salinity. Whilst plantation expansion does offer some promise in this regard, little consideration has been given to the possible dis-benefits of grassland conversion to plantations. For instance, it is well known that evapotranspiration rates in plantations are higher than in grasslands (Cornish 1989; Ruprecht and Schofield 1989; Schofield 1996). Hence, a significant dis-benefit of afforestation will be a reduction in runoff from catchments.

Runoff from forests is generally lower than from grasslands because forests have higher evapotranspiration (ET) rates. 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
- deeper rooting than grasses, enabling better access to soil water, leading to greater transpiration
- a greater ability to extract moisture from the soil under dry conditions, leading to greater transpiration
- lower albedo than grasses, meaning more absorption of radiant energy, leading to greater transpiration

In southern Australia, mean annual ET is usually less than 650 mm in grasslands, but can exceed 1300 mm in forests (Vertessy et al. 1998, Cornish and Vertessy 1999). Streamflow from forests is negligible in areas with annual rainfall less than 600 mm, unless rainfall is highly concentrated in time. 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.

In this paper, we describe and evaluate a simple hydrologic model and use it to predict the streamflow impacts of afforestation in the Middle Murrumbidgee Basin, defined here as the entire catchment area upstream of Wagga Wagga in NSW.

2. THE HOLMES-SINCLAIR RELATIONSHIP

The Holmes and Sinclair (1986) relationship (HSR) relates mean annual ET (and hence runoff) to mean annual rainfall. It is based on long term annual rainfall/runoff relationships for 19 large catchments situated across Victoria, with mean annual rainfalls ranging between 500 and 2500 mm, and varying mixtures of grass and native eucalypt forest cover. Holmes and Sinclair (1986) demonstrated that there were clear differences between ET rates for grassland and eucalypt forest catchments, and illustrated this with a pair of curves that denoted the differences along a rainfall gradient (Figure 1). According to these curves, a fully forested eucalypt catchment would evaporate 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) of different hydrographic data yielded very similar ET estimates to those reported by Holmes and Sinclair (1986).

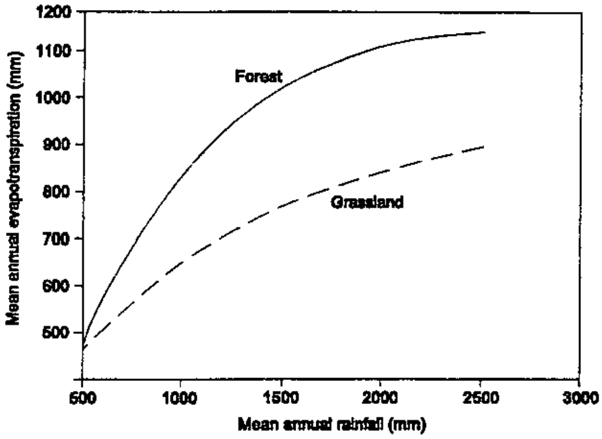

Figure 1. Mean annual ET from grasslands and forest as a function of mean annual rainfall, adapted from Holmes and Sinclair (1986).

3. THE MIDDLE MURRUMBIDGEE BASIN

The Middle Murrumbidgee Basin covers an area of 26,863 km^2 upstream and to the east of the town of Wagga Wagga in south-eastern New South Wales, Australia (Figure 2). Elevations across the basin range between 165 and 2061 m above sea level. During the period 1986-1995, mean annual rainfall across the basin varied between 514 and 2472 mm. Mean annual runoff data were obtained for 28 sub-catchments within the basin, from the PINNEENA data base, distributed by the Department of Land and Water Conservation, New South Wales. These sub-catchments had areas ranging between 10 and 5033 km^2. During the period 1986-95, mean annual runoff from the 28 sub-catchments varied between 49 and 832 mm, with a median of 135 mm.

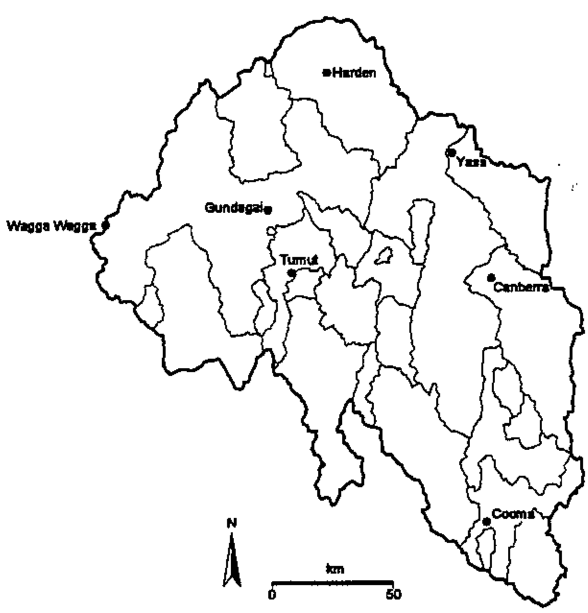

Figure 2. The Middle Murrumbidgee Basin, indicating boundaries for the 28 sub-catchments investigated in this study.

4. IMPLEMENTATION OF THE HSR IN ARC/INFO

Our objective was to estimate mean annual runoff from each of the 28 sub-catchments and to compare the estimates against the mean annual runoff observations we obtained from PINNEENA. To achieve this, we coded HSR into the ARC/INFO GIS. The analysis required three 'data layers', these being, (i) a digital elevation model (DEM), (ii) a mean annual rainfall surface, and (iii) a land cover surface.

The DEM was derived from the AUSLIG 9' elevation data set, and built using ANUDEM (Hutchinson 1997). We adopted a grid size of 250 x 250 m, resulting in a total of 429,816 grid cells for the basin. The rainfall surface was derived from the 'long-term' (1900-1975) mean annual rainfall surface generated by ANUCLIM (Hutchinson et al. 1998), but was modulated by another surface fitted to 1986-1995 rainfall records for 37 sites within the basin. The land cover surface was a 'snapshot' of the land cover status in 1991, compiled by the Bureau of Rural Sciences and the Murray-Darling Basin Commission (Michelle Barson, personal communication).

Mean annual runoff from grassland (Q_{grass}) and eucalypt forest (Q_{forest}) were estimated using the following equations:

$$Q_{grass} = 0.0001 * P^2 + 0.5074(P) - 260.01 \quad \text{... (1)}$$

and

$$Q_{forest} = 0.0002 * P^2 + 0.0584(P) - 129.91 \quad \text{... (2)}$$

where P is the mean annual rainfall, and all terms are expressed in mm. Equations (1) and (2) were developed by converting the ET curves in Figure 1 to runoff curves, assuming runoff equals P-ET.

Any treed area that was not dense forest was considered to be 'woodland'. Mean annual runoff from woodland ($Q_{woodland}$) was estimated as:

$$Q_{woodland} = (Q_{grass} + Q_{forest})/2 \quad \text{... (3)}$$

Holmes and Sinclair (1986) did not distinguish between eucalypt forest and pine plantations, though Smith et al. (1974), Feller (1981), Dunin and Mackay (1982) and Cornish (1989) have all demonstrated that pine plantations have higher ET rates, primarily because of higher leaf area leading to enhanced rainfall interception. Hence, in our analysis we assumed that runoff from pine plantations (Q_{pine}) could be estimated by:

$$Q_{pine} = Q_{forest} - (0.083P + 12.5) \quad \text{... (4)}$$

Equation (4) is derived from a graph of comparative runoff volumes for eucalypt and pine forest, given by Cornish (1989) (his Figure 4). As can be noted, differences in ET between the two forest types increase linearly with mean annual rainfall.

Some areas within various sub-catchments were defined as urban (up to 2%) and lake (up to 4%). We assumed that 80% of all rain falling onto urban areas was converted to runoff and that all rain that fell onto lake areas did not contribute to runoff.

5. RESULTS

The predicted pattern of mean annual runoff across the basin is shown in Figure 3. This pattern reflects the variation in both mean annual rainfall and the type of surface cover.

To test the validity of the HSR for our study area, we plotted the observed mean annual runoff (Q_{obs}) for each of the 28 sub-catchments in the basin against mean annual rainfall (Figure 4). The Q_{obs} values are divided into cases where forest cover was greater than or less than 50%. Also shown are the HSR curves defined by equations (1), (2), (3) and (4). Most of the Q_{obs} values fell within the envelope defined by the curves, though vegetation cover appears to be only a weak discriminator of runoff amount. This was confirmed in a multiple regression analysis (results not reported here). Some catchments with significant forest cover plotted closer to the curve for grassland, and vice versa. The outlier in the upper right hand corner of the graph is for an alpine catchment (median elevation > 1400 m) subject to significant snowfall and affected by very shallow soils.

Mean annual runoff values were estimated (Q_{est}) for all cells in the basin using the methodology described above. The Q_{est} values were aggregated at the sub-catchment level to enable comparison with the Q_{obs} values. Q_{est} for the 28 sub-catchments varied between 46 and 548 mm, with a mean of 207 mm. Q_{obs} varied between 49 and 832 mm, with a mean of 207 mm. The estimated and observed mean annual runoff values for the entire basin were 186 and 185 mm, respectively.

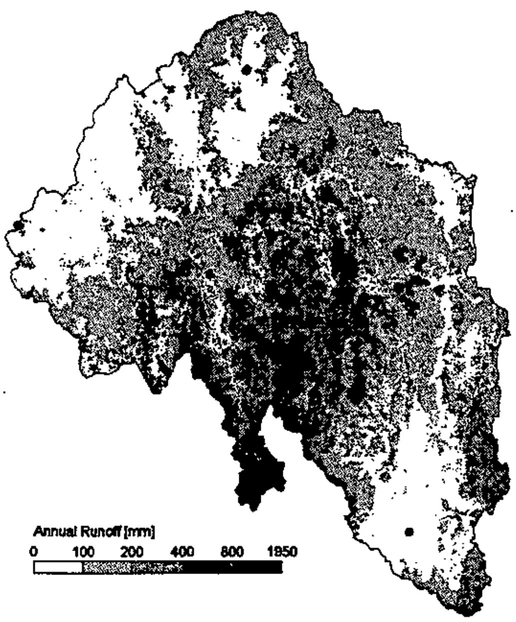

Figure 3. Predicted pattern of mean annual runoff across the Middle Murrumbidgee Basin, assuming surface cover status in 1991.

The aggregated Q_{est} values were regressed against the Q_{obs} values in Figure 5, indicating that the HSR model accounted for 55% of the variation in mean annual runoff from the 28 sub-catchments we examined. Q_{est} errors ranged between +149 mm (an overprediction) and -284 mm (an underprediction). When expressed as a percentage of mean annual rainfall, the Q_{est} error range was +16 to -20%, with a mean error of +1%.

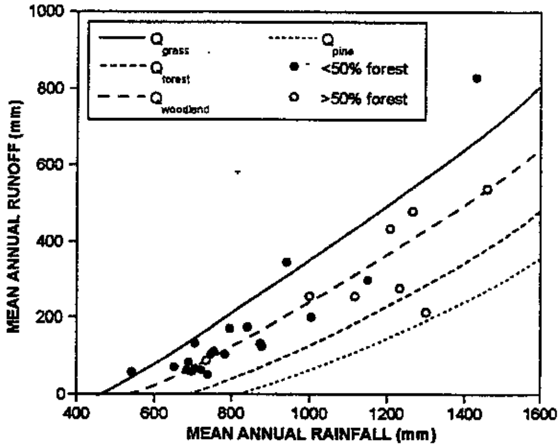

Figure 4. Observed mean annual runoff versus mean annual rainfall, in relation to the HSR equations for different surface cover.

We noted that the HSR model tended to overpredict runoff in low elevation areas and underpredict runoff in areas with high elevation. Figure 6 shows that the error in the runoff estimate (expressed as a percentage of mean annual rainfall) was significantly and negatively correlated with the median elevation of each sub-catchment. This could be due to the fact that the Middle Murrumbidgee Basin has a slightly wider variety of climate types than encountered in the catchments considered by Holmes and Sinclair (1986). For instance, it includes some drier, low elevation sites as well as some very high rainfall and snow affected alpine sites. Some areas are also tend to have relatively higher summer rainfall than experienced in Victoria. Hence, the ET patterns in the Middle Murrumbidgee Basin should differ from those underpinning the Holmes and Sinclair (1986) data set.

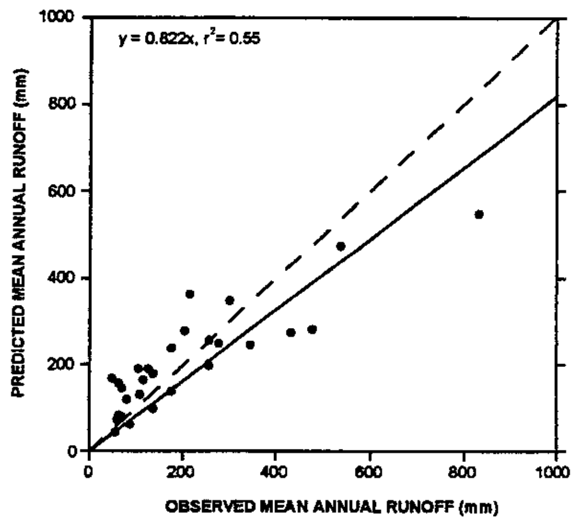

Figure 5. Linear regression of observed mean annual runoff (Q_{obs}) versus HSR estimates (Q_{est}) for the 28 sub-catchments. Dashed line denotes a 1:1 relationship.

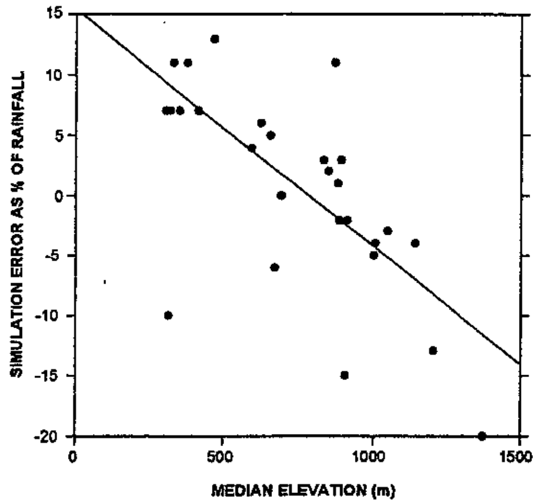

Figure 6. Error in estimate of mean annual runoff as a percentage of rainfall, in relation to the median elevation of each of the 28 sub-catchments.

The next step in our analysis was to modify our HSR estimates on the basis of the line of best fit shown in Figure 6, thus yielding a $Q_{m\text{-}est}$ surface for the basin. This was calculated by:

$$Q_{m\text{-}est} = Q_{est} + (0.02*h - 15.47)*P/100 \quad \ldots (5)$$

where h is the elevation (m) of each cell, and P is the mean annual rainfall for each cell. The resulting $Q_{m\text{-}est}$ values were aggregated at the sub-catchment level and are plotted against the Q_{obs} values in Figure 7, showing that the modified relationship accounts for 82% of the variation in mean annual runoff from the 28 sub-catchments. Errors in the estimate of mean annual runoff for the 28 sub-catchments were significantly reduced by modifying the HSR model with an elevation correction. The error range for $Q_{m\text{-}est}$ was +177 to −174 mm, compared to +149 mm to -284 mm for Q_{est}. When expressed as a percentage of mean annual rainfall, the $Q_{m\text{-}est}$ error range was +14 to -19%, with a mean error of +1%, compared to the Q_{est} error range of +16 to -20%, and a mean error of +1%.

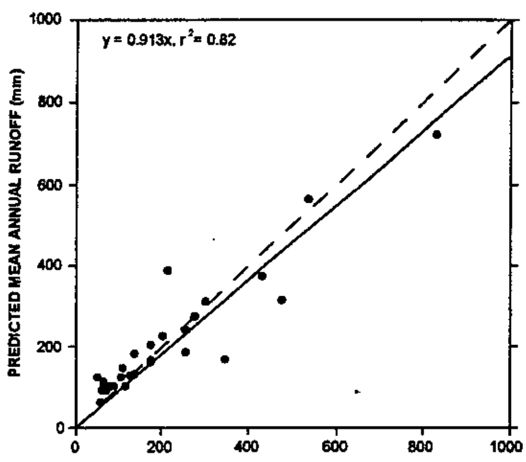

Figure 7. Linear regression of observed mean annual runoff (Q_{obs}) versus modified-HSR estimates ($Q_{m\text{-}est}$) for the 28 sub-catchments. Dashed line denotes a 1:1 relationship.

As is evident from Figure 1, the greatest hydrologic impacts of afforestation will be experienced in the wettest parts of the catchment. Figure 8 provides a map of the Middle Murrumbidgee Basin, indicating the predicted differences in runoff between grasslands and pine plantations across the basin. This shows that mean annual runoff declines of up to 500 mm are possible if grasslands are converted to pine plantations. Figure 9 shows that mean annual runoff would decline by 100 mm or more over 75% of the basin area if grassland were replaced by pines. Under such a land cover change, mean annual runoff would decline by more than 290 mm over 25% of the catchment. However, it should be noted that plantation expansion will affect a relatively small percentage of the total catchment area. Furthermore, some of the new plantation area may be converted from pre-existing woodland, which would lead to lesser declines in mean annual runoff, compared to the grassland case illustrated here.

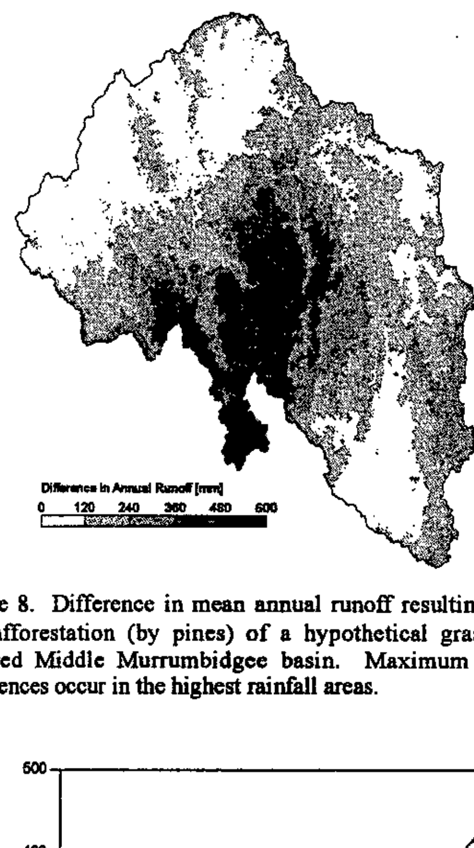

Figure 8. Difference in mean annual runoff resulting from full afforestation (by pines) of a hypothetical grassland-covered Middle Murrumbidgee basin. Maximum runoff differences occur in the highest rainfall areas.

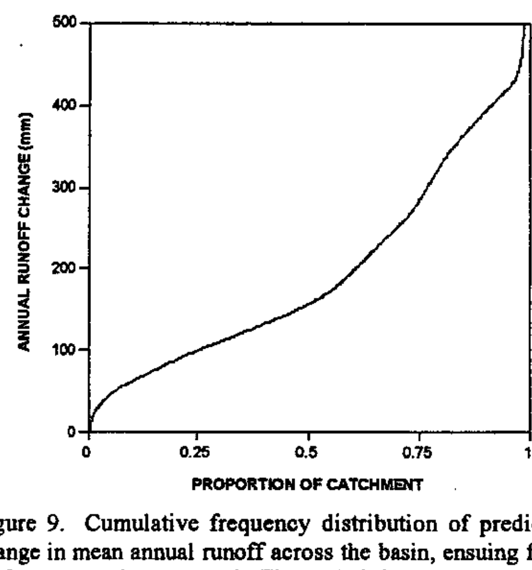

Figure 9. Cumulative frequency distribution of predicted change in mean annual runoff across the basin, ensuing from surface cover change. As in Figure 8, it is assumed that the whole basin is initially grassland, then planted with pines.

6. DISCUSSION

Although based on rather crude assumptions, the HSR explains most of the observed variation in mean annual runoff amongst the 28 sub-catchments within the Middle Murrumbidgee Basin. This is despite the fact that the topography, soils, vegetation and land use practices differ from those operating in the Victorian catchments that Holmes and Sinclair (1986) examined. To obtain a better fit, we modified the HSR by applying an elevation correction. This produced more runoff in high elevation areas and less in low elevation areas, than would be computed by the original HSR equations. We recommend further applications of the HSR to a wider variety of sites, to establish its robustness.

Our analysis has focussed solely on mean annual runoff, though we anticipate that the runoff frequency distribution will change markedly after afforestation. There are sure to be less extreme runoff values, but also far less low flows (Scott and Smith 1997). Some streams in the basin that are currently perennial will become intermittent as a consequence of afforestation. This may have even greater impacts than reductions in mean annual runoff.

Accepting the limits of our data and the modelling methodology adopted, it is still evident that there are major runoff changes to be expected from broadscale afforestation in the Middle Murrumbidgee basin and other catchments of southern Australia. In planning afforestation programs, catchment managers need to give careful consideration to the needs of downstream users dependant on streamflows. Sound hydrologic forecasting is necessary, but so too is a policy and legislative framework to manage the potentially competing needs of timber production and water security.

7. ACKNOWLEDGEMENTS

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The Role of Large Woody Debris and Trees in Controlling Channel Stability on Sand-Bed, Forest Streams

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Summary: Sand-bed streams are usually sensitive to any form of disturbance because of the highly erodible nature of the channel boundary sediments. In forests, such streams are stabilised by a high resistance to flow caused by dense loadings of large woody debris, woody debris dams and living vegetation within the channel. Forestry activities should not target trees within the potential recruitment zone of forest streams and should preserve woody debris loadings within the channel.

1. INTRODUCTION

Best management practices, codes of practice and various regulatory conditions for forestry operations in south eastern Australia include the protection of riparian zones for distances up to 80 m away from the banks of streams. One of the aims of such exclusion zones is to maintain channel stability. The role of large woody debris (LWD) and trees in controlling channel stability on sand-bed, forest streams in south eastern Australia has not been investigated in detail although the extensive removal of riparian vegetation has certainly contributed to large scale channel changes since European settlement (Erskine, 1999). The purpose of our work is to understand why forest streams with highly erodible sand boundaries are generally stable and to frame management practices, which maintain such stability.

Two study sites with riparian vegetation communities dominated by *Tristaniopsis laurina* have been investigated to date. Bruces Creek is a tributary of the Wallagaraugh River in Nadgee State Forest, NSW. Tonghi Creek is a tributary of the Cann River, near the township of Cann River, Victoria. A 310 m long reach of Bruces Creek where the catchment area is 61 km^2 and a 716 m long reach of Tonghi Creek where the catchment area is 182 km^2 have been surveyed.

2. RIPARIAN VEGETATION

There is a distinct vertical zonation of species from the stream bed to the floodplain at both sites that is slightly different to the nearby riparian communities described by Melick (1988). Shallow sections of the bed (riffles, runs and backwaters) are dominated by *Triglochin* spp. with occasional *Tristaniopsis laurina,* whereas pools are essentially free of vegetation, except for *Tristaniopsis laurina* attached to the lower stream bank. Living vegetation with extensive root systems, including fibrous mats, protects the banks. This vegetation is dominated by *Tristaniopsis laurina* trees but also includes an understorey of *Tristaniopsis laurina* seedlings, *Lomandra longifolia, Lomatia myricoides* and *Blechnum* sp. There are only rare, isolated small patches of bank erosion in the study reaches because of the continuous bank vegetation. Floodplain vegetation varies laterally away from the stream. *Tristaniopsis laurina, Notalaea longifolia* and *Pittosporum undulatum* dominate next to the channel where there has been no fire disturbance (Melick, 1988). Where there has been fire disturbance, *Pomaderris aspera, Acacia mearnsii* and *Kunzea ericoides* have colonised the riparian corridor. Emergent trees occur on higher parts of the floodplain away from the immediate stream bank and include *Eucalyptus cypellocarpa, E. viminalis, E. elata, Acacia melanoxylon* and *Angophora* sp. Floodplain understorey species include *Lomandra longifolia, Smilax australis, Gahnia* sp., *Hymenophyllum* sp., *Eustrephis latifolius, Bursaria* sp., *Adiantum* sp., *Blechnum* sp. and *Culcita* sp.

3. LARGE WOODY DEBRIS

LWD refers to snags, logs, pieces of wood, large branches and coarse roots with a minimum diameter of 0.1 m. The results of an inventory of the total amount of large woody debris and trees in the bed of both study reaches are shown in Figures 1 and 2. *Tristaniopsis laurina* (Water gum) is the only live tree in the bed with a diameter greater than 0.1 m. The total LWD and tree loadings are 711 and 550 m^3/ha for Bruces and Tonghi Creeks, respectively. Alive *T. laurina* comprises 16.9 and 17.6 % of the total LWD volume and there are 1.41 and 0.66 pieces of LWD per metre length of channel for Bruces and Tonghi Creeks, respectively. While there are more small pieces of LWD (diameters between 0.1 and 0.3 m) at both sites, the volume is dominated by medium pieces (diameters between 0.3 and 0.7 m). On Bruces Creek, a single large dead *Eucalyptus cypellocarpa* trunk has formed a woody debris dam, which contributes 20.6 % of the total LWD volume. Tree throw of large *E. cypellocarpa, E. viminalis* and *E. elata* on the floodplain at least 10 m from the top of the channel banks during the storm of June 1998 contributed 14 % of the total LWD volume in Tonghi Creek. Episodic wind throw and bank erosion are the most important processes of LWD supply to these channels. *T. laurina* coppices readily when damaged by floods and hence can survive for relatively long periods of time in the channel (Melick, 1988).

4. CHANNEL STABILITY

Log steps are individual pieces of LWD that span the active channel bed, forming a natural wooden dropstructure. Alive *T. laurina* can also form tree steps. Such wooden bed steps are important energy dissipating mechanisms in forest streams. On Bruces Creek, 56 % of the total head loss occurred over log steps under base flow conditions when the discharge was 0.014 m^3/s.

LWD and alive *T. laurina* are important roughness elements, which often form woody debris dams. Such dams are important for organic matter retention and processing, as well as sediment storage. There is, on average, one debris dam every 40 m of channel.

Figure 1: Large woody debris in Bruces Ck

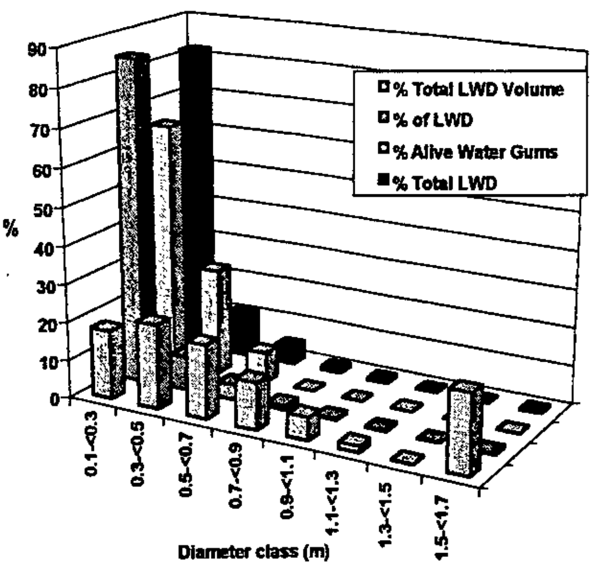
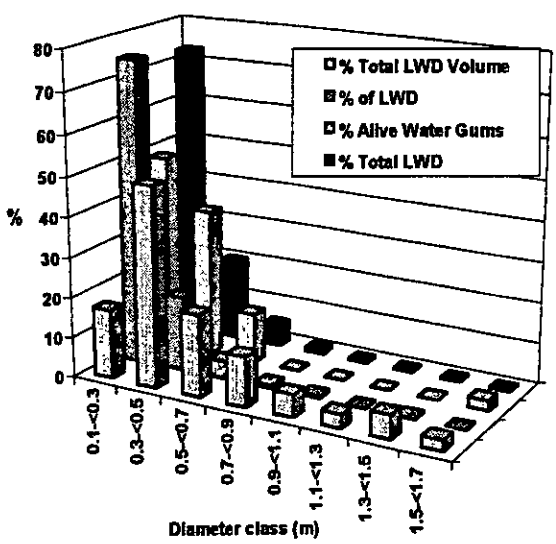
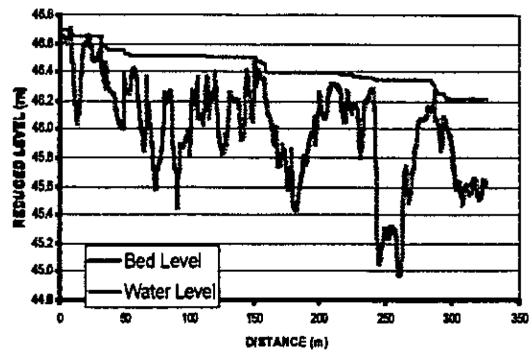


Figure 2: Large woody debris in Tonghi Ck



Scour associated with LWD, woody debris dams and alive *T. laurina* is important for creating step pools (eroded on the downstream side of a log step) and scour pools (eroded under an immobile piece of LWD). Figure 3 shows the detailed longitudinal profile of Bruces Creek. Bedform diversity is dependent on LWD, woody debris dams and alive *T. laurina* as well as on planform. Only the five deepest pools are associated with rhythmic bed disturbances forming the well-known pool-riffle sequence.

FIGURE 3. BRUCES CREEK LONGITUDINAL PROFILE



5. IMPLICATIONS FOR FOREST MANAGEMENT

Channel stability, ecological functioning and habitat diversity on forest streams are dependent on the presence and recruitment of LWD and particular indigenous species of riparian vegetation. Both LWD and trees are also required for the formation of woody debris dams. Tree felling should be excluded from the "recruitment zone" to ensure that the distinctive character of forest streams is maintained. This recruitment zone is likely to be highly variable depending on the species present and the recruitment processes. Furthermore, LWD and alive trees should not be removed because of the consequent mobilisation of stored sediment and organic matter.

6. CONCLUSIONS

Existing regulation of forestry activities next to sand-bed forest streams protects the riparian zone from disturbance. The width of the riparian corridor from which the recruitment of LWD occurs also needs to be considered in framing stream protection protocols.

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Improving Forest Road Management

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Summary: Forest road management, including the maintenance of drainage systems and minimisation of erosion, could be improved through the adoption of Road Management Systems. These systems are now commonly used by shires and highway managers, but most are sealed pavement orientated. A pilot scale Forest Road Management Systems (FRMS) system was developed that can be used to collect, store and analyse forest road information and produce maps and reports describing their current condition of the forest road network. The FRMS also supports systematic road inspection by producing reports that summarise road conditions and help field staff relocate road features. Documenting drainage and erosion due to culverts are an integral part of the data collection system. These data support the sequencing of repair and maintenance to ensure effective performance of the road drainage system. FRMS will greatly enhance a forest manager's capability to maintain and demonstrate environmental compliance. Collecting road data will be a once-off problem whenever forest managers first move to more systematic road management. A cost study was performed that measured data collection productivity and estimated the total data collection task for a large plantation area. The results of this study indicate that FRMS implementation cost will be approximately $15/km of road or 95 cents per ha of plantation.

1. INTRODUCTION

Construction and maintenance of roads account for a large part of the cost of delivering logs to the mill door. Forest roads are usually unsealed and require regular maintenance to prevent deterioration. These roads also impact heavily on log transport costs because they are generally of a lower standard and cause trucks to travel more slowly, use more fuel and incur greater mechanical wear. However, there is a trade-off between road expenditure and transport efficiency. Recent moves by forest managers towards mill door delivery, in both native forest and softwood plantation areas, will provide an opportunity for more integrated management of roads and trucks to capitalise on the potential to minimise haulage and roading costs.

Forest roads have also been identified as being the major source of sediment arising from forest operations (Haydon et.al., 1991). Sediment produced from forest roads can degrade the quality of water flowing from forested catchments. Poor road management practices (e.g. incorrect drain spacing or failure to implement routine maintenance) can exacerbate the problem.

2. ROAD MANAGEMENT SYSTEMS

Successful management of assets and infrastructure requires a sound knowledge of what exists and what condition it is in. Road management systems are inventories of road network components with descriptions of their condition. They can then make information available to the road manager for use in determining management priorities for activities such as maintenance or upgrading. Computerised road management systems have been developed over the last 20 years and are now in common use by shires and highway managers, largely for pavement management. The practicality and utility of road management systems has recently increasing rapidly due to advances in GIS and GPS technology. GIS provides the database for storing road information and allows data to be analysed, and viewed at various scales, from the individual road section up to the regional network. GPS is being used to dramatically increase the efficiency of data collection where location is important. Some types of GPS are integrated with electronic data loggers and allow the user to capture information about a point as it location is recorded. Information can then be electronically transferred and stored within a GIS for display and analysis.

In the longer term, development of a system tailored to forest roads (eg. FRMS) would also have a range of potential benefits in managing roads systems and transport (Douglas and McCormack, 1995) including planning road upgrades and determining maintenance schedules. Detailed road condition information would be available for harvest managers to use in optimising haulage routes for logging operations. Impediments to haulage operations (e.g. unsound bridge decking) can be identified and repaired well in advance. Records of the locations of loading bays, truck turnarounds and landings would be stored within the FRMS to facilitate their reuse in future harvesting operations.

3. DEVELOPMENT OF AN FRMS

The goal of this project was to develop and test a pilot-scale FRMS (Winter, 1999). This involved:

- Definition of what data needs to be collected
- Design of data collection procedures
- Development of FRMS within ArcView GIS
- Demonstrating FRMS use
- Designing methods to support road condition auditing and updating FRMS
- Evaluating FRMS implementation cost

3.1 *Road Data Definition*

The FRMS has been developed to accommodate 4 types of data:

- Road pavement and formation data.
- Road asset data – location, description and condition of features such as culverts, bridges, rollovers, mitres, guardrails.
- Road defect data – location and description of problems in the road network – eg. corrugations, potholes, erosion, aggregate stripping on sealed roads.
- Log truck travel time information

The FRMS can also store information on expected or actual road use over defined periods however, a convenient way of obtaining this data has yet to be developed. An important step in FRMS development was to define procedures for systematically naming and numbering road segments, assets and defects.

3.2 *Road Data Collection Procedures*

The FRMS stores and manages information about pavements, road defects and the location and states of individual road features. The most efficient method of collecting information about defects and road features is by using an integrated GPS/electronic data logger. Information recorded with this device can be loaded directly into the GIS.

3.3 *Development of FRMS within ArcView GIS*

A significant part of the work has been developing a pilot implementation of FRMS has within the ArcView GIS framework. Many of its operations use the functions provided by this GIS program. Specific procedures were devised to:

- Add road segment information
- Add data collected by GPS
- Process and organise data within the system
- Name features using the chosen convention
- Add distance along road to road feature points
- Display data

Once the initial data has been collected for a forest area and then organised within the GIS, the FRMS is ready to be used to support road management operations. The addition of information such as economic data (e.g. construction and maintenance costs) and projected road use will further enhance the FRMS.

A high degree of customisation is possible and over time it is expected that a considerable number of procedures and tools will be developed. Several management related enquiry functions have been developed as examples. A procedure in FRMS was developed to check whether the existing drainage spacing met specification by calculating inter-drain spacing and road slopes, and comparing to a stored spacing tables. A demonstration 'querying module' was also added to the FRMS, showing how the system can be tailored to specific needs. This module automates the retrieval of a status report listing the features, pavement width and type and a list of any defects occurring on roads selected by the user.

3.4 *Road Condition Auditing*

Forest roads should be inspected on a regular basis and the ability of the FRMS to produces condition reports to assist in this assessment is a major advantage. For each road feature, the pre-inspection report lists the distance from the roads origin, the information currently stored within the FRMS for that feature. Fields in the data-capture device allow recording of amended or new data. Re-doing the condition assessment routinely serves to update the information stored in the FRMS.

4. EVALUATING THE IMPLEMENTATION COST

Many road management systems are abandoned at the implementation phase because it proves too costly and time consuming to collect the initial dataset. A study was undertaken to determine the likely cost of initial data collection for the FRMS in a large softwood plantation area. Field staff were used to assess the feasibility of using a GPS + data logger collection method and to measure its productivity. After an introduction to the technique, field staff had little trouble in using it to record the location and attributes of road features. The 4 day productivity trial revealed that there was no advantage gained by having 2 people working as a team rather than just 1 person because the time taken for the GPS to record a features location often exceeded the time it took to assess the feature.

Collection of data for culverts and bridges took significantly longer than for other road features due to the complexity of assessment and their being commonly obscured by vegetation. Productivity of data recording was significantly poorer in un-thinned pine than in other forest types because dense canopy interfered with the satellite signals required by the GPS.

A second study was undertaken to estimate the rate of occurrence of features along roads that would need to be mapped. The road system in a large plantation district was selected as an example. The frequency of road features on 120 randomly selected segments of forest road (all >500m in length) were measured. The segments were stratified into three terrain slope classes ($<5^0$; $5^0 - 10^0$; and $>10^0$) and into three road standard classes. Approximately 80 kilometres of road were assessed, covering:

- Sealed, 2 –lane road (Road Class 1)
- Major unsealed road with formation width ≥ 5.5m, usually gravelled (Road Class 2)
- Minor unsealed road with formation width < 5.5m, usually natural surfaced (Road Class 3)

The frequencies of occurrence for each class were then applied across all forest roads in the plantation according to their classification to estimate the total number of road features requiring assessment. These were multiplied by time based cost to estimate the expense for the initial data collection of for the FRMS in one forest district. The overall average cost was calculated as approximately $15/km of road or 95 cents per ha of plantation for about 11 road features per kilometre.

5. CONCLUSION

The pilot-scale version of the FRMS has shown potential to be a useful management tool. In particular:

- Data collection procedures were demonstrated to be suited for use by field staff and able to accommodate the variety of conditions and road features found in production forests
- Systematic inspection of roads and road structures is made possible by the condition reports produced by the FRMS
- The ArcView GIS provides a suitable base for the FRMS and it has the advantage that many forestry staff are already familiar with its use
- While there would be a significant cost to collect the original location data, once running, the FRMS would greatly enhance a forest managers capability to maintain and demonstrate environmental compliance.
- The FRMS would provide a good basis for more intensive management of roading and transport operations.

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Preliminary Assessment of the Impacts of Landuse Changes on Sediment Production in a Mixed Pastureland/Forestry Catchment in S.E. NSW.

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The point source erosion rates from high yielding areas in both pastoral and forestry regions are well documented. However, their influence to downstream water quality is largely unknown firstly, in terms of their accumulation rates and secondly, in terms of their impact relative to one another. In this forestry F01 project we investigate these questions and report here our results on the first issue regarding historical pre-european and contemporary accumulation rates of sediment material in a mixed landuse pastureland/forestry catchment on the s.e. coastal floodplains of NSW. Primary landuses in this catchment were roughly equal proportions of land cleared for pastural/ dairy activities, and native forests managed by State Forests for timber production. The pastureland areas are based on granitic soils, the forested areas on metasediments. Both soil types are classified as highly erodible. A two metre box section monolith was taken from sediment deposits in the downstream floodplain of this catchment. The exposure was described in the field, and the monolith returned to the laboratory for analysis. Three independent methods were used to define a chronolgy for the material within the monolith, thus enabling a history of sediment accumulation to be constructed. These methods were:

- OSL dating
- Presence and absence of fallout Cs-137
- Presence and abundance of pollen species

The OSL method effectively measures the number of years before today that quartz grains were last exposed to sunlight. Thus presumably recording the year in which the grains were deposited onto the surface of the floodplain. In this way, a sequence of samples throughout the monolith can effectively date various horizons to points in time and so is particularly useful in terms of defining the pre-European/post European boundary. Alternatively Cs-137 was globally distributed as fallout during the 1960's. Thus all soil material that was exposd during this time carries this particular label. Finally pollen grains are also distributed and carried within sediment bodies, and so an analysis of the pollen record can reveal much about the history of plant and tree abundances in catchments. In particular the presence or absence of *Pinus* grains can be used to date sediment horizons from the known introduction date of this species in Australia. In addition, the ratio of pollen abundance from grass to tree species can be used to map rates of catchment clearing.

These three methods are combined to reveal that most of the sediment accumulated within the floodplain of this catchment were deposited within the last 120 years. This was much higher than anticipated and reflects significantly greater impact on our catchments through european arrival and landuse change than previously expected. It also appears that a single event (the 1971 flood) was responsible for a substantial component of this deposit (~40 cm). Records reveal that this event probably coincided with a few years of preceding drought, possible overstocking in upland areas, and initiation of channels in prior valley fill deposits in the granitic regions upstream. Contemporary accumulation rates appear to have declined.

The evidence points to most of this deposit being derived from erosion processes active in the upper granitic portion of the catchment. Much of this erosion can probably be attributed to land clearing, destabilization of prior valley fill deposits in valley floors, the removal of riparian vegetation in upland areas, the consequent change in hydraulic behavior of flood discharges and the coincidence of all this with large floods. Historical diaries reveal this river was probably originally narrow, characterized by deep pools, there was much woody debris. In contrast today, the river is shallow and wide, this is mostly due to a large and extensive quartz deposit draped along much of its length filling prior holes up to 6-7 metres depth, woody debris is largely absent. The river form is much changed. The historical influence on this system from forestry activities is probably minimal although selective removal of sawlog timber was occuring throughout this century. A more complete roading network was commenced in forestry regions in the late 1960's and integrated harvesting practices were introduced in subsequent years after that. The relative influence of contemporary sediment production from the forestry and pastoral areas is now under investigation.

Impact of Timber Harvesting Activities on Water Quality; What Can We Confidently Say from 30 Years of Research?

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Summary: This paper synthesises over 30 years of scientific research in order to identify the issues that remain poorly addressed in our understanding of the impact of forest harvesting on water quality. The approach consists of categorising findings into those that are well established by the research, those that are supported by limited evidence and those that remain speculative. This approach provides a more meaningful insight into the state of our knowledge and perhaps more importantly into the willingness of the industry and researchers to implement guidelines and management practices based on research in general and not just site specific data. The paper is used as a constructive framework to talk about future issues in forest erosion research and the effective management of our forests for environmental protection.

1. INTRODUCTION

Many of us have undertaken literature reviews in the hope that we can effectively synthesise research findings to construct definitive statements on the known impacts of forest harvesting activities on water quality. Forest research has been very active for 30 years or more and there is certainly a wealth of published material. However, it remains very difficult to conclude from these findings;

- exactly what we do know about the impact of harvesting on water quality,
- what are the underlying assumptions of many of our research approaches and the implications for result interpretation and,
- whether advanced technologies and a greater public and industry awareness of the issue has contributed to our ability to effectively manage these landscapes.

Much of what we can say consists of *broad generalisations* and the extent to which these are supported by the literature is highly variable. Some are well established reflecting a consensus from a wide number of studies irrespective of site factors, while others are the result of limited evidence and finally some are largely speculative. This paper summaries a recent review of the literature using this three-tiered assessment approach in order to look more closely at issues that remain unresolved or poorly addressed by existing studies. It provides a very useful framework within which to talk about future research issues and the way forward in both technology transfer and forest management. This issue of more effective technology transfer underpins our ability to have a large influence on the way these environments are managed. For the sake of brevity, this paper does not refer to the original papers but a detailed bibliography of the articles consulted is attached here for future reference.

1.2. Water Quality and Timber Harvesting: 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 and scaling approaches to assess changes in catchment water quality due to forest harvest disturbances. 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 almost 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. 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. 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 fully understand the issue of forest management impacts on water quality.

2. WHAT WE NEED TO KNOW

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 to 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 transient if best management practices are employed and proven to be successful.

Data on these three key variables allow us to construct a more meaningful and accurate assessment of potential impacts. The problem lies in acquiring data for any specific 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 and modelling using data from different scales and regions. The following section summarises our understanding of these key areas and concludes by identifying knowledge gaps and issues of immediate concern for present and future management of forested catchments.

3. WHAT WE KNOW

3.1 Sediment and Nutrient Sources

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 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 landuses, primarily agriculture.
- The dominant cause of increased nutrients in streams if observed, is due to the effects of prescribed burning and wildfire.
- Sediment yields from forested (managed and un-managed) watersheds are considerably lower than those from other landuses, particularly agriculture.

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.
- 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.
- Observed impacts of sediment and nutrient delivery are short-lived and transient with no long-term effect.

Speculative

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

3.2 Delivery Pathways

Broad Generalisations

Well established

- Channelised pathways forming at road drainage outlets form the most efficient conduit for sediment and nutrient delivery to streams.
- Sediment delivery ratios 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 in determining 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

3.3 Effects Of Best Management Practices

Broad Generalisations

Well established

- BMP's 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 water and sediment/nutrients delivered to a stream.

Limited Evidence

- The best location and design of buffer strips in forested catchments of varying topography and landuse.
- The specific role and effectiveness of BMP's on the hillslope versus those in the near-stream area.

4. KNOWLEDGE GAPS

A number of important issues remain relatively unexplored in the literature. These focus primarily on the lack of studies examining the issues at a range of temporal and spatial scales. The following issues in particular require further attention.

Temporal trends in impact assessment.

The magnitude of disturbance and sediment generation is considerably greater on recently constructed roads than stable well-established surfaces, although data in the literature comparing this temporal trend are scarce. There are few new roads currently being constructed for forestry operations due to the economic expense and the presence of a relatively large existing infrastructure in many catchments and regional forest areas. The major impact of road infrastructure in Australian forested catchments, therefore, is likely to have occurred some 30 years or more ago when large areas of forest roads were built to meet increased harvesting demands. Possible impacts may have included increased channel destabilisation, in-stream storage of both coarse and fine material, and siltation of estuaries. These impacts may be emerging as major issues of concern in some catchments today due to the rate of temporal response and readjustment of the channel system. The large increases in sandy bed load material observed in many of south eastern NSW streams may be a function of 'historic' landuse impacts or more-recent landuse changes such as forest roading, and the clearing of forests for agriculture etc. The community remains divided with regard to their perceptions, many have anecdotal evidence that relates to a particular time period, and scientists have failed to adequately address the issue through well-defined, large-scale catchment monitoring studies.

Spatial-scale of analysis.

The review also pointed to the scarcity of catchment or regional scale analysis of potential impacts. Whilst acknowledging the problems with in-stream monitoring studies, there is a clear need to redefine the objectives of water quality monitoring programs to address specific issues. This forces both scientists and interested parties to identify the areas that require protection and the means to ensure their protection. This is not a simple task but is one that researchers and community groups must make if we are to progress in our development of acceptable water values. The literature overwhelmingly suggests that even if small-scale increases in stream turbidity are observed these are short-lived (within the time frame of 2-3 years) and that the persistent effects of increased sediment yields are rarely reported. There must also be increased public awareness that stream turbidity is a natural phenomenon that even in pristine forests, reflects the dynamic processes of sediment transport and channel adjustment. There is an urgent need for clearer, well-defined thresholds of concern that can be used to manage and regulate these systems. These thresholds must be established for all aspects of stream health and not just sediment loads or turbidity but also biological indicators. In Australia, there are very few 'baseline' data sets of biological and chemical indicators of stream health. Consequently, studies examining the potential impacts of land use change on stream systems have limited opportunity to assess the magnitude of the impact.

Predictive tools- the way forward?

The third aspect that remains unresolved from the literature is the type of predictive tools that should be applied to the management of harvesting/roading practices in forested catchments. The debate about simple empirical relationships or sophisticated models is likely to continue. The first step in evaluating this question for Australian forested catchments lies in data assimilation and archiving. The common use of GIS by many of the state agencies should mean that it is now possible to collate available soils, topographic, infrastructure data and develop catchment-scale characteristics. It is clear, however, that predictive tools must be clearly linked to identified areas or thresholds of concern; what it is that requires protection and the acceptable limits of change.

5. SUMMARY

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 to 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. Croke *et al.*, (this vol) highlight the large variability in sediment concentrations with road usage and traffic intensity but also suggest that the specific impact of roads within a catchment is more likely to be influenced by the degree of road-to-stream connectivity and not just a measure of road density.
- The most persistent and threatening impact of sediment delivery due to roading relates specifically to gully erosion 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 gully erosion 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. The development and refinement of BMP's within the forest industry over the past 10 years has significantly improved methods of timber harvesting, in particular, selective logging on an alternate coupe basis and the introduction of the recent inherent hazard system. Continuing efforts must be applied to developing environmentally sound practices of timber removal.

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The Impacts of Forestry on Streamflows: A Review

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1. INTRODUCTION

There is now broad agreement amongst researchers that forestry activities 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 throughout the world to ascertain the nature and extent of streamflow change likely to result 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 on streamflows. The focus in this review is mainly on *water yield* or the amount of streamflow, though attention is also given to streamflow seasonality and flow frequency, and the magnitude of peak flows.

Our knowledge of the effects of forest clearance on streamflows in Australia stems mainly from ten major sets of catchment treatment experiments, located in diverse geographic settings. 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, unlike in the USA, Australia is not endowed with catchment treatment experiments which have been monitored over long time periods. The 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 summarises the state of knowledge regarding the effects of forest cover on catchment water balances and streamflows. A broad geographic focus is adopted, though emphasis is placed on forest types and climatic conditions prevailing in south-eastern Australia. This review concludes with a statement of current knowledge gaps and recommendations for future research in the field of forestry impacts on streamflow.

2. COMPARISON OF STREAMFLOWS FROM FORESTS AND GRASSLANDS

Streamflows from forests are generally lower than from grasslands because forests have higher evapotranspiration (ET) rates than grasslands. Table 1 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. Table 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

Table 1: Mean annual rainfall and evapotranspiration (ET) for catchments with various vegetation covers in south-eastern Australia.

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

3. WHAT EFFECT DOES FOREST CLEARANCE HAVE ON STREAMFLOWS?

Almost all of the catchment water balance studies reported to date have shown that streamflow increases as forest cover decreases, and vice versa. They also show that the magnitude of 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, so only the first few years of data following treatment are used in building such 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.

Northern hemisphere studies

Hibbert (1967) summarized 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
- 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. Figure 1 shows their conclusion that water yield increases proportionately with the percent area of forest cleared, but indicating a fair degree of scatter in the relationship. Bosch and Hewlett (1982) related much of this scatter to species differences. Figure 2 shows the trend lines they fitted to the data in Figure 1 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 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.

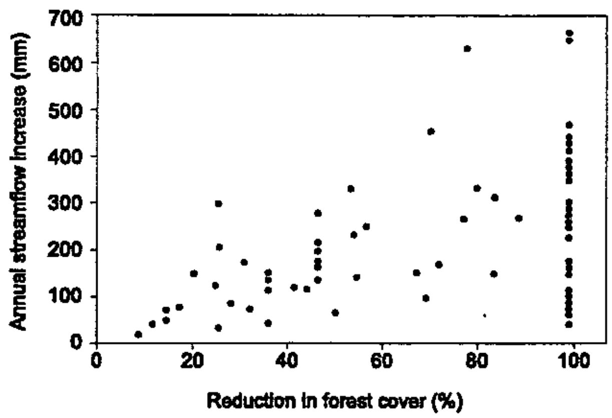

Figure 1: Relationship between reduction in forest cover and increase in water yield (adapted from Bosch and Hewlett, 1982).

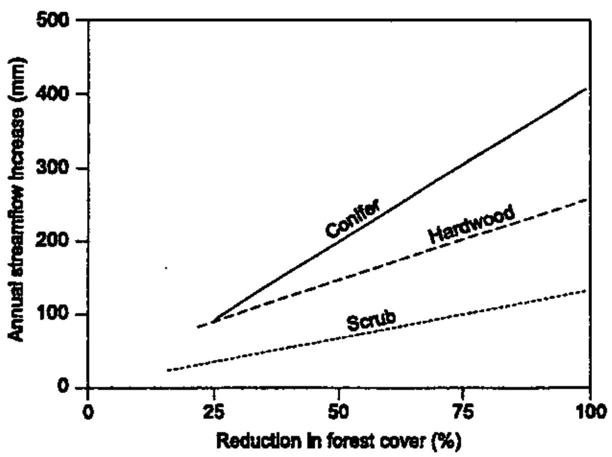

Figure 2: General trend lines showing the relationship between cleared area and yield increase for three different types of woody vegetation (adapted from Bosch and Hewlett, 1982 and Cornish, 1989).

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) 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 r^2 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%.

Table 2: Estimated annual yield increase per 10% of catchment area cleared of forest (AYI/10%) for various geographic regions in the USA (after Stednick, 1996). # 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.

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

Stednick (1996) fitted a similar relationship to the Bosch and Hewlett (1982) data set shown in Figure 1, 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 r^2 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 the data shown in Figure 1 to represent yield increases that would occur if the entire forest area was cleared in the catchments examined. Figure 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 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 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).

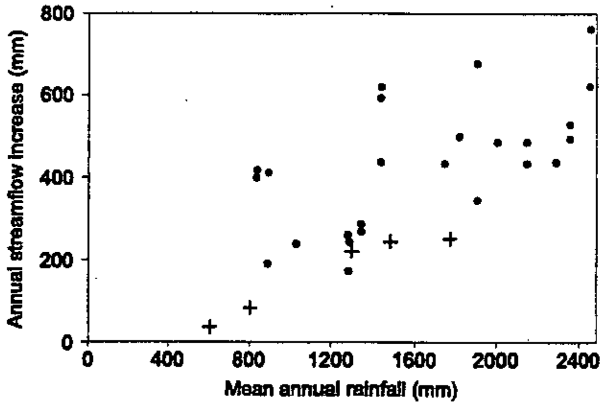

Figure 3: Effect of mean annual rainfall on water yield increases caused by total clearance of conifer and scrub vegetation from catchments (adapted from Bosch and Hewlett, 1982). Crosses denote the expected water yield gains from conversion of eucalypt forest to grassland, based on the Holmes and Sinclair (1986) relationship.

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 affronted with pines. This equates to a streamflow decline of 30 mm per 10% of catchment affronted. 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 3). For the seven catchments they examined, the average streamflow increase was 44 mm per 10% of forest cleared

Table 3: 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

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.

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 speed and magnitude of response varied depending on the type of vegetation change and the mean annual rainfall of the catchment (Figure 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. As will be shown later in this review, this conclusion contradicts Australian findings on comparative rates of ET and streamflow for pine plantations and eucalypt forests.

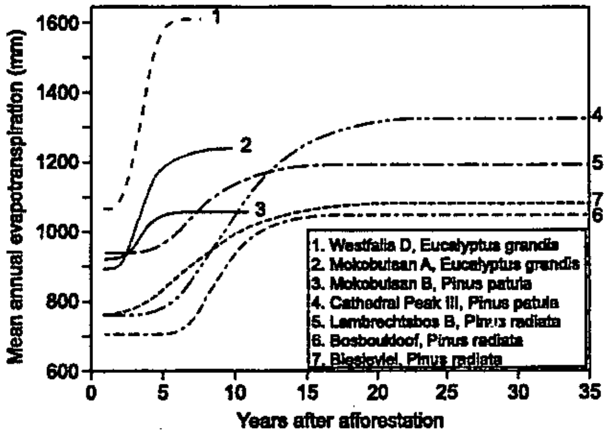

Figure 4: Evapotranspiration trends in afforested South African catchments (after Dye 1996).

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 5). They determined that streamflow increased by 33 mm for each 10% of forest area cleared. The linear regression underpinning this relationship had an r^2 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 suggests that conversion of forest to grassland at this rainfall isohyet should yield an additional 240 mm of streamflow, whereas Figure 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 3).

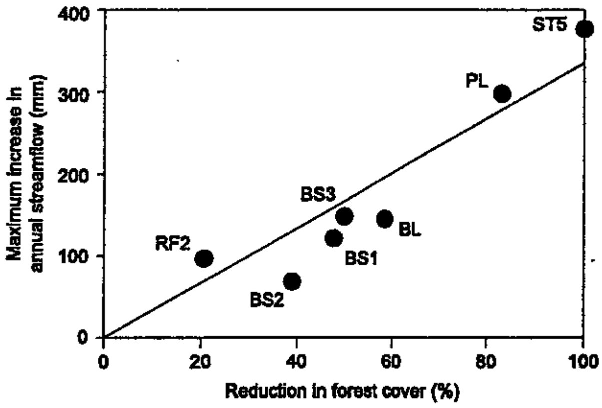

Figure 5: Relationship between maximum annual streamflow increase and percent area of forest cleared for seven catchments in Victoria (after Nandakumar and Mein 1993).

Cornish and Vertessy (submitted) 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 4). 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 logging-induced streamflow increases reported for Karuah are slightly higher than reported for other Australian studies.

Table 4: Maximum annual increases in streamflow after clearance of eucalypt forest at Karuah, NSW (based on data from Cornish and Vertessy, submitted).

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
Barrata	1657	0

Cornish (1991) reported a peak annual streamflow increase of 35 mm per 10% of logged area for the Eden 2 catchment, part of the Yambulla group of catchments. However, he also showed that the Eden 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 Eden 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 Eden 5 and 6 catchments. She determined maximum annual streamflow increases of 38 and 16 mm per 10% of forest area treated in Eden 5 and 6, respectively.

Transience and persistence of streamflow increases after clearance

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 clearance. 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 cleared *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 5). 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 5: Magnitudes and time scales of streamflow increases following forest clearance in six Victorian catchments (after

Nandakumar and Mein 1993). The catchments had variable areas of forest cleared. The time to recovery is a predicted value, based on a logarithmic equation.

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

For the Karuah catchments, Cornish and Vertessy (submitted) 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 shortlived 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.

In her analysis of streamflow changes at Eden 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 Eden 6 and by year 13 in Eden 5. This is surprising given that forest regeneration was allegedly far more vigorous in Eden 6.

Comparative effects of uniform thinning and patch cutting

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.

4. DO STREAMFLOWS FROM NATIVE EUCALYPT FORESTS AND PINE PLANTATIONS DIFFER?

In the preceding section, some reference was made to studies which compared streamflows from eucalypt and pine forested catchments. Almost all of the South African data suggests that streamflows from pine afforested catchments are *greater* than those from eucalypt afforested catchments (Bosch 1979, Van Wyk 1987, Smith and Scott 1992, Dye 1996, Scott and Smith 1997), due to the relatively *higher* ET rates of eucalypts in South Africa. However, the opposite trend has been observed in Australia where higher ET rates are normally ascribed to pines, primarily because of their higher leaf area index and rainfall interception rates (Smith et al. 1974, Pilgrim et al. 1982, Dunin and Mackay 1982, Cornish 1989). Unlike in South Africa, pines generally seem to grow better than most eucalypt species in Australia.

Smith et al. (1974) compared rainfall interception and streamflow rates of pine (P. radiata) plantations and mixed species eucalypt forest at Lidsdale, NSW. They reported that the pine plantation generated lower streamflow than the eucalypt forest, primarily because of much higher rainfall interception rates in the pines. Similar findings for the same catchments were provided by Pilgrim et al. (1982) who used a more extensive data set than Smith et al. (1974). Pilgrim et al. (1982) argued that the streamflow differences between the two catchments could be entirely explained by differences in rainfall interception (Table 6).

Table 6: Differences in the mean annual water balance between a pine plantation and a eucalypt forest at Lidsdale, NSW, over the period 1974-1976 (after Pilgrim et al. 1982).

	Pine plantation	Eucalypt forest
Rainfall (mm)	842	870
Rainfall interception (mm)	183	99
Transpiration and soil evaporation (mm)	472	501
Soil moisture change (mm)	-3	1
Streamflow (mm)	190	269

Dunin and Mackay (1982) compared the rainfall interception and transpiration rates of pine plantations (P. Radiata) and eucalypt (E. Maculata) forests in south east region of NSW under conditions of abundant soil moisture and similar radiation input. They found that pines transpired at a greater rate than eucalypts during winter, but that the annual transpiration totals for the two forest types were similar. However, the rainfall interception rates of pines were determined to be three to four times greater than those measured for the eucalypts. Dunin and Mackay (1982) found that the pines intercepted about 10% more of rainfall than the eucalypts.

Similar findings were reported by Feller (1981), who compared throughfall, stemflow and interception rates of two eucalypt species (E. Regnans and E. Obliqua) with those for pine (P. Radiata) plantations in the Maroondah catchment. He found that the pines intercepted 25.5% of gross rainfall over a two-year period, compared to values of 18.5% and 15% for E. Regnans and E. Obliqua forest, respectively (Table 7).

Table 7: Comparative rates of throughfall, stemflow and interception in eucalypt forests and pine plantations in the Maroondah basin, expressed as a percentage of gross rainfall over a two-year period (data from Feller 1981).

Parameter	E. Regnans	E. Obliqua	P. Radiata
Throughfall (%)	74.5	84.5	73.5
Stemflow (%)	7	0.5	1.0
Interception (%)	18.5	15	25.5

Summarising several studies from Australia and New Zealand, Cornish (1989) ranked a range of factors affecting evapotranspiration rates in pine plantations, eucalypt forests and pastures (Table 8). His findings support those reported above, namely that pines and eucalypts seem to be differentiated mainly by rainfall interception rate. He ascribed enhanced interception rates in pines not only to higher leaf area index, but also to the lower albedo of pines, meaning that more net radiant energy is available to drive the evaporation process. A similar argument was made by Dunin and Mackay (1982).

Table 8: Ranking of factors affecting evapotranspiration rates in pine plantations, eucalypt forest and pasture (after Cornish 1989).

Characteristic	Ranking
Transpiration	Eucalypt = Pine > Pasture
Interception	Pine > Eucalypt >> Pasture
Available energy for evaporation (albedo)	Pasture > Eucalypt > Pine
Energy input (combined albedo and temp. profile)	Pine = Eucalypt > Pasture
Evapotranspiration (combined effects)	Pine > Eucalypts >> Pasture

Cornish (1989) estimated the streamflow reductions likely to arise from afforestation of pasture by pine plantations and eucalypt forest as a function of mean annual rainfall (Figure 6). Figure 6 shows that because of the higher rainfall interception rate of pines, streamflow differences between pine and eucalypt afforested catchments can be expected to increase mean annual rainfall increases.

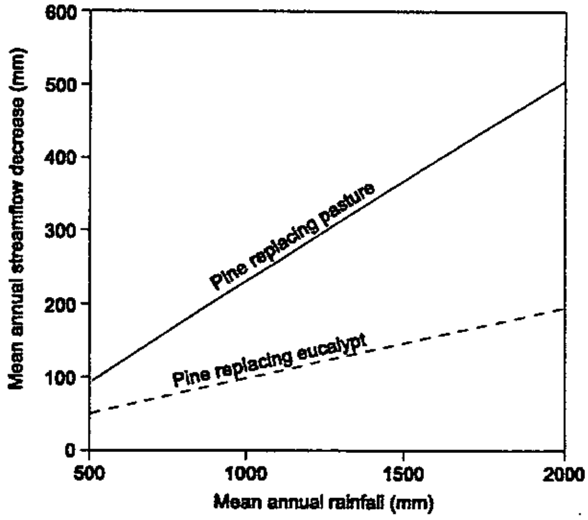

Figure 6: Estimated streamflow reductions likely to arise from afforestation of pasture catchments with pines and eucalypts, as a function of mean annual rainfall (after Cornish 1989).

5. HOW DOES FOREST AGE AFFECT STREAMFLOWS?

Australian forest hydrology is distinguished by a concern for the effect of forest age on streamflow. It is widely assumed that regrowth forests yield less streamflow than old growth forests, because of differences in ET, yet this trend has only been verified for two forest areas in Australia (Maroondah and Karuah) and in some pine plantations in South Africa. 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. Below, we review the data supporting a link between forest age and streamflow.

Mountain ash forest

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, mountain 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 only regenerate 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 7). The curve combines the known hydrologic responses of eight large (14–900 km^2) basins to 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, taking as long as 150 years to recover fully.

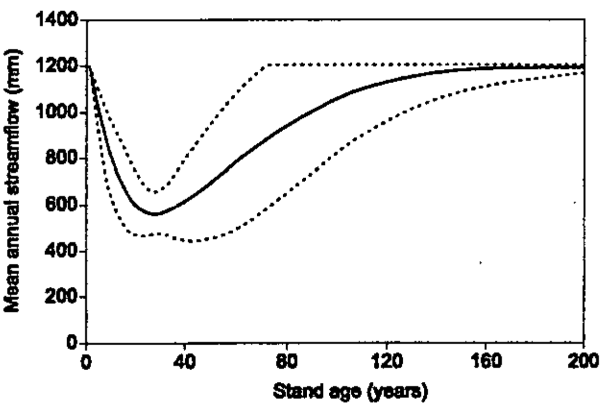

Figure 7: Relationship between forest age and mean annual runoff from mountain ash forest catchments (after Kuczera 1985). Dotted line denotes the 95% confidence limits on the relationship.

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 7), 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.

To emphasise this last point, Figure 8 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).

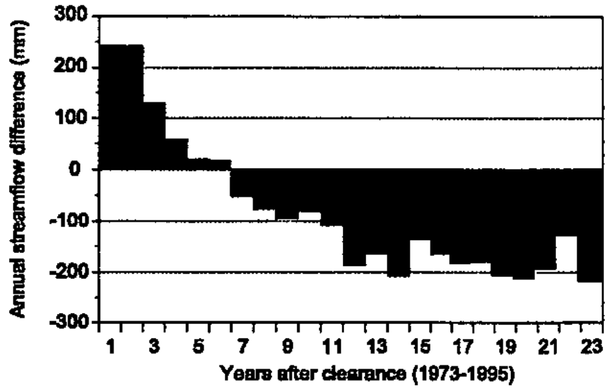

Figure 8: Annual streamflow change from the Picaninny catchment, Victoria, from 1973 to 1995 (after Vertessy et al. 1998). Picaninny had 78% of its forest area clearfelled in 1972. The changes shown are relative to streamflows from the undisturbed Slip Creek catchment, sited adjacent to Picaninny.

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 9.

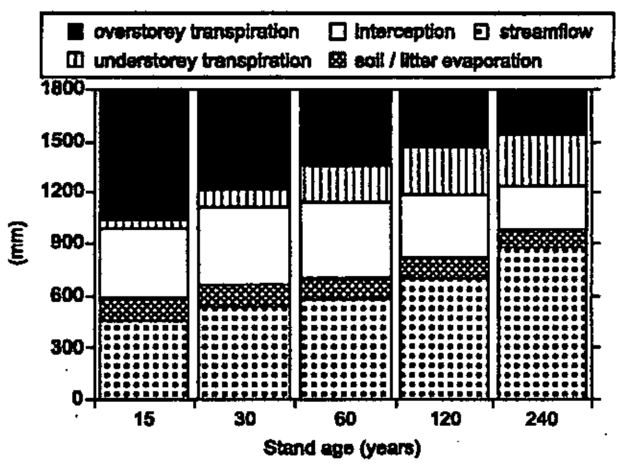

Figure 9: Water balance for mountain ash forest stands of various ages, assuming annual rainfall of 1800 mm (after Vertessy et al. 1998).

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
- 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.

Using the small catchment experimental data yielded from the Maroondah basin study, Watson et al. (1999) developed an alternative forest age-streamflow relationship (Figure 10). 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. (1999) 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 10.

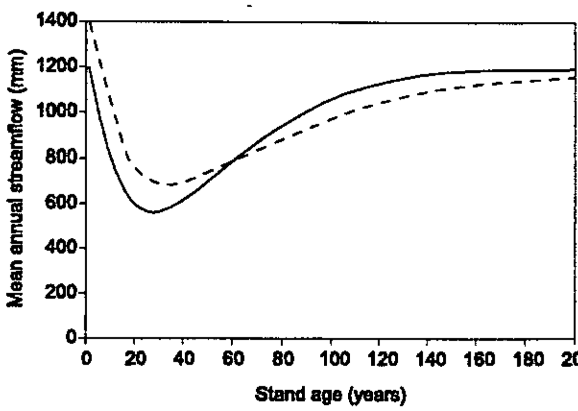

Figure 10: An alternative 'model' of the streamflow-forest age relationship for mountain ash forests (dashed line) (after Watson et al. 1999). The Kuczera curve (solid line) is shown for reference.

The Karuah catchments

The only other Australian studies which have demonstrated a link between streamflow and forest age are those of Cornish (1993) and Cornish and Vertessy (submitted). 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, deep and permeable soils, and strong baseflow. Cornish and Vertessy (submitted) analysed streamflow records for six catchments, logged to varying extents (25-79%). Their analysis is founded on 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 (submitted) 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 (submitted) showed that these three catchments were affected by insect attack, leading to decreased leaf area and ET rates and enhanced streamflows.

Figure 11 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.

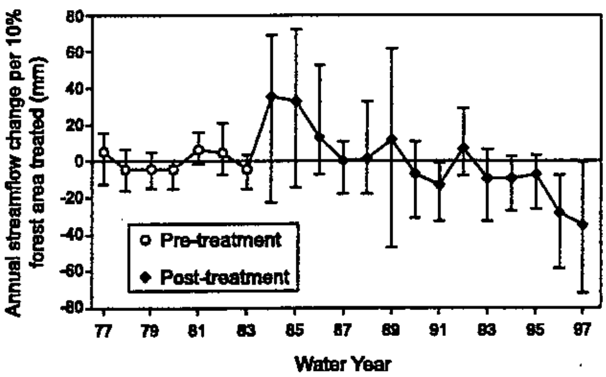

Figure 11: Annual streamflow changes amongst the six treated Karuah catchments in the NSW lower north east region (after Cornish and Vertessy submitted). Ends of bars denote maximum and minimum changes, symbols denote mean change.

Cornish and Vertessy (submitted) 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:

$$ASR = 1368 - 480.8*SD - 37.418*BAI - 16.76*CC \quad (1)$$

where

ASR = reduction in annual streamflow for the final year of record (mm)

SD = soil depth (m)

BAI = mean annual basal area increment

CC = canopy cover (%)

The model fit obtained by Equation 1 is shown in Figure 12. Equation 1 was shown to account for almost 82% 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.

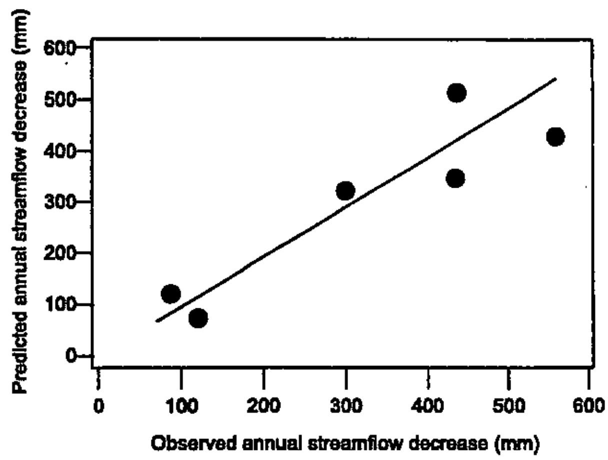

Figure 12: Observed and predicted reductions in annual streamflow for the six treated Karuah catchments (after Cornish and Vertessy submitted). The predictions are based on Equation 1 and assume that the entire catchment area was treated.

The Yambulla catchments

Roberts et al. (1998) 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 2.6, 1.5 and 0.7 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 *increase* as the E. Sieberi forest ages.

Roberts (pers. comm.) has analysed streamflow data for the Eden 1, 5 and 6 catchments. Eden 1 is an undisturbed control catchment, whereas Eden 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 of reduced streamflows*, despite fairly vigorous regeneration of forest in Eden 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.

Pine plantations

There is no Australian data linking pine plantation age to the amount of streamflow from catchments, though some evidence is available from South Africa. Van Wyk (1987) showed that streamflow reductions in the pine afforested Bosboukloof catchment (relative to those from a grassland control catchment) changed through the life cycle of the plantation. Figure 13 shows that peak reductions in streamflow from this catchment occurred around age 24-27 and declined significantly by age 36-39. This finding is consistent with hydrometric evidence from the mountain ash forests which indicates that streamflow minima are normally attained when the forest is aged about 27 years. Ryan et al. (1997) summarised the world literature on the chronosequence of growth rates and stand leaf area for different forest species. They showed that peak forest growth rate can be attained anywhere between 3 and 68 years of age, depending on forest type and site conditions. They also showed that leaf area index (a major determinant of rainfall interception and transpiration rate) for most forest types tails off sharply after the time of peak growth rate has been passed. It thus follows that streamflows should increase as a forest or plantation ages.

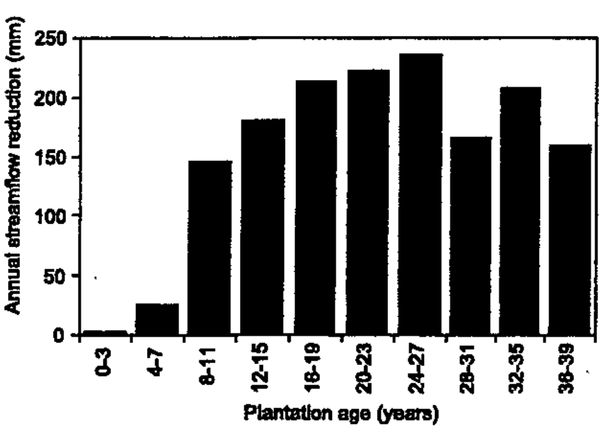

Figure 13: Temporal changes in streamflow reductions in the pine afforested Bosboukloof catchment in South Africa, relative to a grassland control catchment (data from Van Wyk 1987).

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.

6. WHAT AFFECT DO FORESTRY ACTIVITIES HAVE ON 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 clearance (Hewlett and Helvey 1970, Burt and Swank 1992, Schofield 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.

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 proportionally 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 14). They demonstrated that absolute reductions in streamflow were greatest during the wet months, but that the reductions were *proportionally* greatest during the low flow periods. For example, Figure 14 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 proportionally more even 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 14 is that monthly streamflow differences are 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.

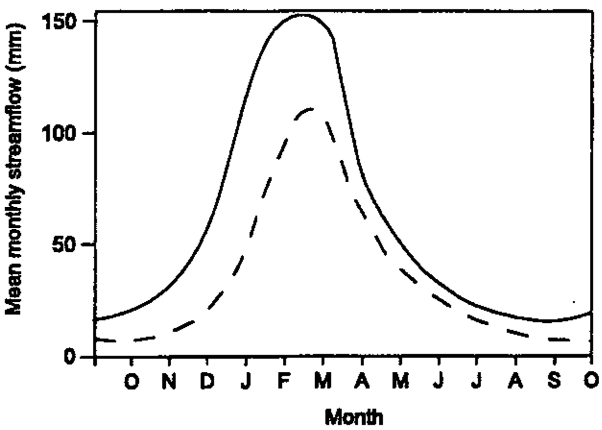

Figure 14: Mean monthly streamflows in the Cathedral Peak catchment, South Africa, before and after afforestation of the grassland vegetation with pines (after Bosch and von Gadow 1990). Solid (upper) line denotes grassland condition, dashed (lower) line denotes pine plantation condition.

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 afforesation 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 15).

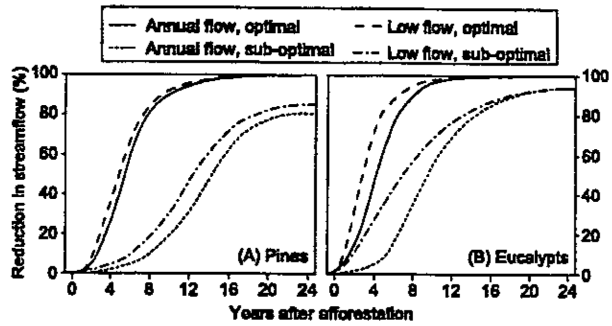

Figure 15: Generalised curves for predicting annual and low reductions as a function of age in pine and eucalypt plantations in South Africa (after Scott and Smith 1997). Separate curves are shown for optimal and sub-optimal growing regions for both forest types.

Figure 15 shows that the effects of afforestation on annual and low flows were more 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 15 shows that low flows were reduced proportionally 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.

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 16). Figure 16 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.

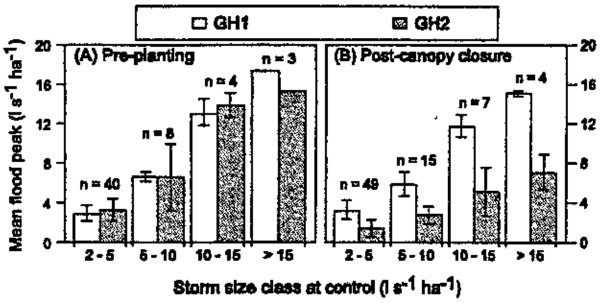

Figure 16: Comparison of mean flood peaks for four size classes of storms in the G1 (control) and G2 (treated) catchment, (a) prior to the treatment of G2, and (b) after plantation canopy closure in G2 (after Fahey and Jackson 1997). Bars denote one standard deviation on each mean value, n denotes the number of storms in each class.

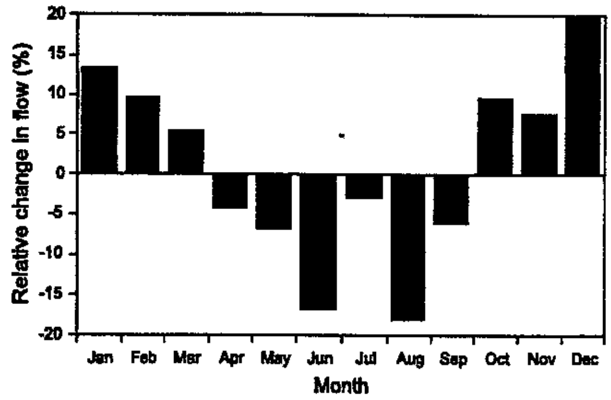

Figure 17: Effect of a 50% basal area thinning on monthly streamflows in the Crotty Creek catchment, Victoria (data from Table 6.1 in Haydon 1993). To de-emphasise the effects of streamflow variability, monthly streamflow is expressed as a percentage of annual streamflow. Positive values denote an enhanced 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.

Clearance 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 1998). 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 17). Figure 17 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.

Watson et al. (1999) 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 18. When assessing the flow regime changes at Picaninny, as illustrated in Figure 18a, it is necessary to compare them with rainfall-induced changes evident in the flow duration curves for Slip Creek (Figure 18b). An important conclusion of the Watson et al. (1999) 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 (submitted) 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 (submitted) attributed most annual streamflow changes in the Karuah catchments to changes in baseflows.

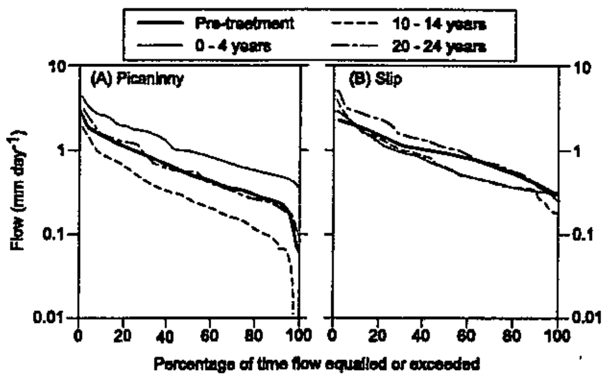

Figure 18: Flow duration curves for the Picaninny and Slip Creek catchments, Maroondah basin, Victoria (after Watson et al. 1999). Separate curves are shown for the pre-treatment period and discrete five-year blocks in the post-treatment period; only a sub-set of the original data are shown here for 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 (1983) 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.

7. KNOWLEDGE GAPS AND RECOMMENDATIONS FOR FURTHER WORK

There are several areas demanding further research before the impacts of forestry on streamflows can be properly understood and forecast in Australia.

A greater number of studies are needed comparing the water balance of old growth and regrowth eucalypt forests. The dynamics of mountain ash are now well understood but a similar knowledge base is required for other eucalypt communities. Process studies such as those conducted by Vertessy et al. (1998) are recommended to elucidate the streamflow-age relationships observed in the Karuah group of catchments (Cornish and Vertessy submitted). Recent field measurements by Sandra Roberts (University of Melbourne) and Rob Vertessy (CSIRO) indicate that transpiration per unit leaf area declines with forest age in E. Sieberi and E. Regnans forests, respectively. These trends need to be verified in the forests where they were determined and checked for in other eucalypt communities.

Almost all of the data reported in this review are of an empirical nature. Such data underpins our knowledge of how catchments respond to forest disturbance but is compromised by significant inter-site variability. There are real difficulties in extrapolating empirical relationships between sites with different physical characteristics (climate, topography, soils and vegetation). In recent years there have been significant advances made in process-based hydrologic modelling of forest catchments (Vertessy et al. 1993, Vertessy et al. 1996, Watson et al. 1997, Watson et al. 1998). This new generation of models enable site factors to be taken into account in a way that is not possible with empirical models. However, before they can be used, a quantum leap is needed in the way that we collect environmental data, needed to drive these models. Accurate maps of forest age, forest type, soil type and soil depth are still surprisingly difficult to obtain, as are good temporal sequences of rainfall, humidity, temperature and radiation. Such data are critical if process-based models are to be employed to forecast the streamflow impacts of forest disturbance and climate change.

This review has highlighted the lack of Australian reports on seasonal changes in streamflow resulting from forestry activities. Raw data is widely available to evaluate forestry-induced changes in flow regime, and it is recommended that appropriate analyses are conducted to test for such changes across a range of catchments. One particularly useful application of process-based models that should be explored is the simulation of changes to flow frequency spectra that result from forestry activities. Process-based models, if properly calibrated on a site, can be used to predict how flows of varying magnitudes may be differentially altered by changed forest cover. If these models also have a forest growth component embedded in them (as in the model described by Vertessy et al. 1996) then the persistence and transience of these changes in streamflow spectra can also be ascertained. Even more importantly, the effects of variable climate (droughts and floods) can also be determined. These may well have an over-riding influence on streamflow dynamics when compared to the effects of forest cover and forest age.

Finally, whilst Australia has several good paired catchment experiments, few of these are set up to properly compare water balance differences between varying landuses. For instance, there are very few well instrumented cropping, pasture and pine plantation catchments in Australia that are proximal to one another. Given the likelihood that Australian grazing and cropping lands will undergo extensive afforestation in the next three decades (DPIE 1997), it is essential that we establish a range sites throughout the country to measure the water balance differences between cropped, grazed and afforested catchments. There is a particular need to establish such comparative sites in areas of intermediate rainfall (700-1400 mm per year).

8. SUMMARY

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
- 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
- the degree of streamflow increases following forest logging is 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

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 proportionally 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

Speculative

- forest age affects ET rates in dry eucalypt forest, though there is no hydrometric evidence to show that streamflows are affected as well
- transpiration per unit leaf area of forest declines as eucalypt forests age

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