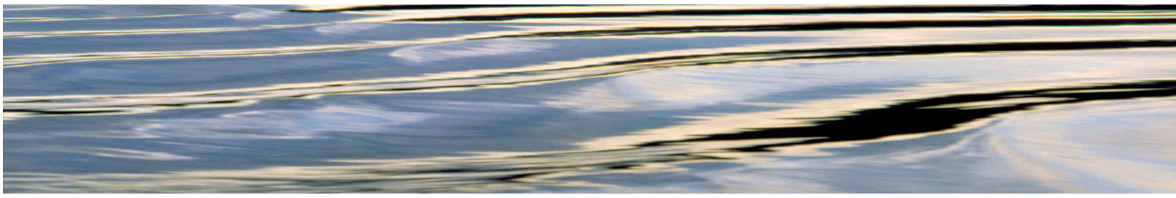
COOPERATIVE RESEARCH CENTRE FOR **CATCHMENT HYDROLOGY**

EROSION IN FORESTS: PROCEEDINGS OF THE FOREST WORKSHOP - MARCH 2004

TECHNICAL REPORT
Report 04/10

December 2004

Jacky Croke / Ingrid Takken / Simon Mockler



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Preface

Welcome to the 3rd Forest Management Workshop in Canberra, March 23rd-25th, 2004. Following on from the last workshop 5 years ago in Warburton, Victoria, (doesn't time fly!) we are again availing of this opportunity to expose aspects of forest 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 exchange ideas and discuss mutual needs. The healthy registration list of over seventy people reflects a willingness to participate on the part of many and on the genuine concern and interest that people have in managing our forest systems.

The collection of papers in this volume illustrates that indeed new research is progressing. This can only contribute in a very positive way to an improved understanding of the processes and complexities of measuring and modelling these managed landscapes. This workshop attempts to incorporate the diverse range of forest management themes including water quantity, quality, fire management and sustainability. As such, we believe the program represents an interesting collection of research in the major areas of forest management. We are very grateful to the presenters who have contributed to this workshop by sharing their research and management outcomes with us[1].

As with previous workshops, the field trip on Wednesday will provide an excellent forum for discussing many of these themes in greater detail. This will present us with the opportunity to examine the real life challenge of managing forestry environments, most particularly in this instance, those that have suffered the enormous impacts of wildfire.

This workshop also marks the end of a three year project funded by the Australian Research Council and Industry Partners (State Forests of NSW and The Forest Stewardship Council of Victoria). We would like to thank all the people who have contributed both to the ARC SPIRT Project and to this workshop. In

[1] The views and opinions expressed in the following abstracts and papers are those of the author(s) and have not been subject to an independent external review.

particular, we acknowledge our colleagues Dr Pat Lane, Dr Peter Hairsine, Dr Peter Wallbrink, Dr Wayne Erskine and Mr Jim Brophy. As with previous workshops, we appreciate the assistance and skills of the Cooperative Research Centre (CRC) for Catchment Hydrology in the marketing of this event. We hope that the more formal venue will not detract from past traditions of a relaxed atmosphere where scientists, industry and agency people could mix and discuss aspects in a constructive manner. We look forward to hearing your comments and please feel free to contribute to the full range of discussions throughout the presentations and field day.

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Tree Water Use of a Young Plantation Targeted at Salinity Control

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Summary: In Australia, the rapid expansion of agriculture, with extensive clearing of deep-rooted native vegetation, has caused changes in landscape water balances, leading to rapidly expanding areas of salinization and increased salinity in rivers. The reintroduction of trees into dryland salinity-prone landscapes is seen as a means of halting the spread of salinity. By transpiring more water than the agricultural land use, carefully targeted planting of trees can help reduce salt mobilisation and stream salinity levels. In 2001 State Forests of New South Wales established a 60 ha plantation in a small salinity prone sub-catchment of the Little River, NSW. Over the next few years the impact of the plantation on the hydrology of the sub-catchment and the quality of runoff and groundwater will be monitored, along with estimates of tree water use. This paper presents a brief overview of the project and describes the measurement of water use by individual trees and the scaling up from tree to stand being applied on the site.

1. INTRODUCTION

In parts of Australia the replacement of deep-rooted native vegetation with crops and pasture has led to a change in the water balance of large tracts of land. A higher proportion of the rainfall is now draining below the rooting zone leading to rising watertables and the mobilisation of salt stored deep in the soil profile. The salt may rise to within the rooting zone resulting in salt scalds or move laterally through the soil profile into watercourses thereby increasing the salt content of streams and rivers. Dryland salinity is widespread across NSW in the < 800 mm-rainfall zone. Some 120,000 ha of agricultural land has become salinized in NSW (Coram, 1998) and if action is not taken to address the problem the area with a potential to develop dryland salinity is predicted to grow to some 1.3 million hectares by 2050 (National Land and Water Resources Audit 2000).

One solution to the problem of dryland salinity is to reintroduce trees to the landscape (Morris and Collopy, 1999). Trees are able to access water deep in the soil and regolith and in these low to medium rainfall areas can utilise most of the available rainfall thus reducing the deep drainage that leads to rising watertables and salinization. State Forests of NSW is investigating the potential for commercial forestry in the dryland region however, in this rainfall zone trees tend to be slow growing and it is unlikely that commercial forestry based on wood products alone would be viable. To encourage investment in forestry projects, the New South Wales Government is actively engaged in developing markets for non-traditional forest products and environmental services, such as salinity control, carbon sequestration and biodiversity enhancement.

As a pilot study into a possible market driven mechanism for salinity control, State Forests and Macquarie River Food and Fibre (MRFF) have entered into an agreement whereby MRFF will pay State Forests for water transpired by trees established on two small catchments known to be sources of salt in the Upper Macquarie catchment. The basis of the trade is that the water transpired by the trees will not be available to mobilise salt into the river so will lead to improved water quality downstream (Walsh *et al.,* 2003).

This paper focuses on one of the sites where instrumentation has been installed to monitor the water use of the trees and the impact of the forest on groundwater levels and stream water quality.

2. THE SITE

The site is a 260 ha sub-catchment in the Little River Catchment near Baldry, 70 km North West of Orange, NSW. The geology of the site consists of fine-grained Devonian-age granites and granodiorites. These crop out in several parts of the catchment and are not limited to the hilltops. The granites appear to be well fractured and deeply weathered - at least in parts. Yellow solodic soils dominate the subcatchment. These soils are characterised by a strong texture contrast between the A and B horizons, low to very low fertility and high sodium levels. Topsoil textures range from loamy sands and sandy loams. There is usually a strongly bleached A2 horizon present with loamy sand to sandy clay loam texture. The B-horizon has yellowish-brown colour with sandy clay texture with moderate to strongly developed coarse columnar peds, prismatic or sub-angular blocky structure with a tough consistency (Murphy and Lawrie, 1998). The subsoils are dispersible and susceptible to severe gully erosion.

An eroded ephemeral creek drains northwards into the Little River. Since clearing in the 1930s the site has been alternately cropped and grazed for periods of three to four years. A large salt scald developed on the lower slopes and salt has been expressing all along the eroded creek line. In September 2001 State Forests established 60 ha of *Eucalyptus camaldulensis* and *Corymbia maculata* on the western side of the creek line from the salt scald up to the ridge top, while the eastern side of the creek has continued to be cropped.

The well defined nature of the subcatchment, combined with the planting design and the fact that this is one of the largest areas of block tree planting in this salinity prone region of NSW makes this site valuable for studying the impact of re-establishing trees on water and salt movement. State Forests of NSW are collaborating with the Department of Infrastructure, Planning and Natural Resources and the University of NSW to collect data that will be used to validate and improve models that can then be used to help target tree planting for optimum salinity control.

To this end a range of instrumentation was installed across the site during 2003 with a view to monitoring ground water levels and quality, stream flow and quality, tree water use and growth, soil moisture and evapotranspiration from the cropped area over the next few years as the trees grow. Ten pairs of nested piezometers are distributed across the site, monitoring deep and shallow ground water, while eight shallow piezometers monitor ground water levels along the creek line. A network of neutron probe access tubes permit regular measurement of soil moisture across the site. These are complemented with ten sets of continuously logged soil moisture probes at various depths. A V-notch weir monitors the intermittent stream flow and stream EC levels. A baseline EM31 survey was conducted in 2002 and periodic resistivity surveys will be carried out on transects across the site to monitor any changes in salt concentrations in the soil profile.

3. MEASUREMENT OF TREE WATER USE

As part of the pilot salinity trade it is necessary to assess tree water use, since payment is based on the amount of water transpired by the trees. Heat pulse sapflow sensors are being used to measure the water use of a number of trees of various sizes. The relationship between water use and tree size is then combined with tree size distribution and stocking to estimate tree water use across the whole site. Three locations were selected across the site to cover a range of tree sizes and landscape position. The sites chosen were in close proximity to where soil moisture measurements were being made.

The Compensation Heat Pulse method employs two temperature sensors, implanted radially into xylem tissue and spaced asymmetrically from a similarly inserted line heater element. The midpoint of the two temperature sensors is located a specific distance downstream (i.e. above) from the heater. A pulse of heat (typically one-second duration) is applied to the sapwood and is carried with the sap stream toward the midpoint of the temperature sensors. When both sensors reach the same temperature the heat pulse has travelled the known distance from the heater to the midpoint, and the heat pulse velocity can be calculated as a function of distance and time (Marshall 1958, Swanson and Whitfield 1981). The sap flux density (cm^3 sap / cm^2 sapwood/hr) can be calculated from the heat pulse velocity and knowledge of the wood and water volume fractions of the sapwood. Multiplying sap-flux density by cross sectional sapwood area of the stem gives sap flux (cm^3/hr).

The base of the live crown on the two-year-old trees was very low (20-40 cm), even on trees taller than three metres, hence probe implantation had to be below this to capture all of the flow. Sap flux velocity is not constant throughout the sapwood of a tree and so more than one probe set is often required to estimate water use of large trees. The diameter of trees at probe implantation varied from 2 to 10 cm and no heartwood was present. Two probes were used in stems larger than 5 cm diameter and one probe in stems smaller. A total of 16 stems were instrumented and sap-flux has been recorded every half-hour from 28th July 2003. Each month the diameter of the stems at ground level, 30 cm and 1.3 m have been measured along with tree height and crown extent.

4. EARLY RESULTS

Individual tree water use

The daily water use of four trees and the rainfall events during the first six months are shown in Figure 1. As would be expected large trees used more water than small trees. Daily water use increased from winter through spring to summer primarily in response to increased evaporative demand but in part due to increased tree size. Interestingly the smaller trees showed an increase in water use after rain but the larger trees didn't. This indicates the small trees were more water limited and may explain part of the variation in growth rates across the site, even between neighbouring trees. The day to day variation in tree water use between rain events is in response to daily weather and gradual soil moisture depletion.

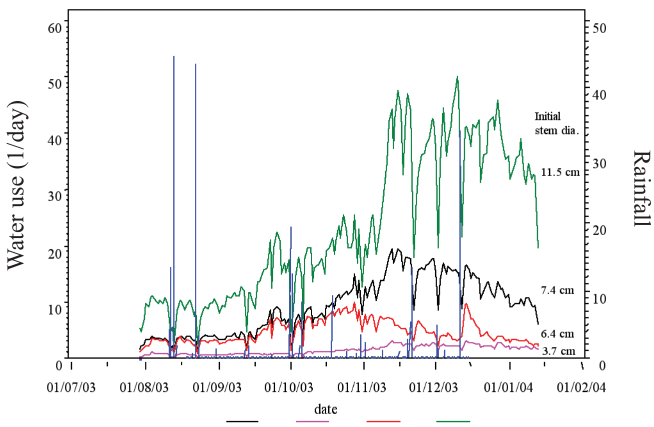

Figure 1. Daily water use of four of the 16 trees between 28 July 2003 and 12 January 2004. Vertical lines show rainfall events.

In order to estimate the water use of non-instrumented trees across the site an empirical three-parameter power function relating D30 (diameter at 30 cm) to tree water use was fitted to each day's data. This was then used to estimate daily water use of trees of various sizes along transects on each day during the period.

Scaling

There was variation in tree size and stocking across the site resulting from, among other things, varying soil depth, moisture availability and frost susceptibility. In order to scale water use up from the instrumented trees to the site it was necessary to stratify this variability hence nine permanent transects were installed in August 2003. Each transect was four metres wide, seven transects were 100 m long and two 50 m long. In August 2003 the position, height, crown extent and diameters at ground, 30 cm and 1.3 m were measured on all trees within each transect (see Table 1).

Table 1. Nine transects across the site measured in August 2003.

No.	Stocking stems/ ha	Projected Crown area m^2/ ha	Canopy cover %	Basal area at 30cm m^2/ha
1	825	973	10	1.0
2	525	265	3	0.5
3	450	138	1	0.2
4	725	870	9	0.6
5	500	178	2	0.2
6	800	2833	28	1.8
7	500	3260	33	1.6
8	600	875	9	0.6
9	650	305	3	0.4

The daily water use of each tree in the transects was estimated using the relationship derived above and summed to give transect water use over the 169 day period. The rainfall during the period was 265 mm and the potential evapotranspiration was 670 mm. When transect water use is expressed as a percentage of rainfall the value is similar to percent canopy cover (Table 2). The average water use across transects of 32 mm was 12.4% of the rainfall while the average percent canopy cover was 11%.

Table 2. Estimated water use of trees along nine transects across the site between 28th July 2003 and 12th January 2004.

No.	WU mm	WU mm/day	Canopy cover %	% of rainfall
1	42.5	0.25	10	16.3
2	19.7	0.12	3	7.6
3	9.7	0.06	1	3.7
4	30.6	0.18	9	11.8
5	11.8	0.07	2	4.5
6	72.8	0.43	28	28.0
7	65.0	0.38	33	25.0
8	22.7	0.13	9	8.7
9	14.6	0.09	3	5.6
mean	32.2	0.19	11	12.4

The water use of the trees across the whole site will depend on the proportion of the site represented by each transect, which has yet to be assessed. But based on the average across transects it appears the trees used slightly more water than fell as rain onto the tree canopy covered area. Between 25 and 30 months old the trees transpired 12 % of rainfall falling on the site. There was good cover of weeds between trees during most of the period, which will have transpired a significant proportion of the rainfall, and some stream flow was observed but the weir was not yet operational.

5. FUTURE WORK

This paper presents the first analysis of part of the data set being collected at the Baldry site. Tree water use measurements will be continued and expanded to include physiological measurements such as leaf water potential, stomatal response to vapour pressure deficit and stem hydraulic conductance. These measurements will contribute to process based model development and testing.

An areal photograph of the site will be used to determine tree size distribution and stocking thus enabling improved stratification of the site to enable the estimates of transect water use to be distributed.

Measurements of evapotranspiration of the understorey and inter-row vegetation will be carried out. Comparison of soil water budgets from soil moisture sensors with evapotranspiration estimates will be undertaken.

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Water Use By Plantations Over Shallow, Fresh Watertables

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Summary: Forestry plantations can reduce groundwater recharge compared with pasture and many rain fed agricultural crops and in rare circumstances can be net users of groundwater. This paper describes methods used to quantify groundwater recharge and uptake in plantations located over shallow watertables in southeast South Australia and the Riverina and presents summary data illustrating maximum groundwater uptake rates by plantations and effects on plantation productivity.

In regions of relatively low relief and low to medium rainfall (400 to 800 mm annual mean), where soils are permeable and where potential evapotranspiration substantially exceeds rainfall for the majority of the year, plantations generally use all the available rainfall once the canopy has closed. Under certain combinations of permeable soil over shallow, fresh groundwater in a transmissive unconfined or semi-confined aquifer, tree plantations can be net users of groundwater. Nationally, this situation is uncommon, but in southeast South Australia it might, in the near future, occur over a large enough area of plantations to have a significant impact on the groundwater balance over several hundred km^2. In the past 4 years, annual net rates of groundwater uptake of between 190 and 750 mm year^{-1} have been measured in *Eucalyptus globulus, E. grandis, Corymbia maculata* and *Pinus radiata* plantations in the Green Triangle and the Riverina, at locations where depth to the water table is less than 6 m and groundwater salinity is low. These plantations also have high productivity, with current annual stem volume increment ranging from 35 to 65 m^3 ha^{-1} year^{-1} for annual rainfall of between 370 and 770 mm.

1. INTRODUCTION

Tree planting provides a means for minimising recharge of groundwater systems to slow or reverse rising saline water tables in regions affected by dryland salinity. Conversely, trees planted in locations where the groundwater is fresh may compete with agricultural, domestic and industrial users of groundwater by reducing aquifer recharge, or using groundwater directly.

It has long been known from studies of catchment water yields, that replacement of grassland with forest reduces stream flows (Hibbert 1967, Bosch and Hewlett 1982, Cornish 1989). It is generally accepted that forests use more water than shallow rooted vegetation due to forests having lower albedo, greater interception losses (particularly conifers), deeper root systems, the evergreen habit of non deciduous trees and the greater canopy height and roughness of forests (Holmes and Sinclair 1986, Zhang *et al.,* 1999, 2001).

Trees located over shallow watertables, particularly where groundwater aquifers are highly transmissive and low in salinity, may access additional water to that received from rainfall with potential to substantially increase plantation productivity.

There are two key questions concerning the effects of such tree plantations on groundwater:

1. Do trees use all the rainfall? (is there any groundwater recharge), and
2. Is there net aquifer discharge through direct extraction by tree roots?

This paper presents summary results of up to 4 years of water balance studies aimed at quantifying groundwater net recharge and discharge under tree plantations. Some of these studies are ongoing, and the data presented here will be revised, refined and added to as additional measurements proceed.

2. METHODS

Ground water recharge and uptake can be inferred using a combination of measurements of water balance components (net rainfall and evapotranspiration) and changes in soil water storage. At locations in Southeast South Australia and the Riverina with light textured surface soils and flat to gently undulating topography it is often reasonable to assume there is little net loss or gain of water from study plots via surface or subsurface lateral flows. In these circumstances, periodic measurements of total net rainfall, transpiration, soil evaporation and change in volumetric water content of the root zone measured or estimated in square or rectangular sample plots containing around 40 to 50 trees have provided estimates of plantation annual evapotranspiration rates and groundwater net recharge or uptake.

In this case, for each period of several weeks between soil water measurements, net deep drainage to or uptake from below the deepest soil water measurement under the plot can be calculated as:

$$Q_{wt} = P - I - T - E - (S_c - S_p)$$

where: Q_{wt} represents either drainage (a positive value) or water uptake (a negative value) below the maximum depth of soil water measurement;

P represents gross total precipitation for the period, measured in a rain gauge in the open nearby;

I represents rainfall interception losses, measured using throughfall gauges;

T represents transpiration determined by direct measurement using sap flow sensors;

E represents evaporative losses from the soil and weeds measured using mini lysimeters;

Sc represents the current volumetric water content of the root zone measured using a neutron moisture meter;

Sp represents the previous volumetric water content of the root zone.

Rainfall

A rain gauge in an open area near each site (located within 2 km for most sites) was read every time the site was visited (nominally every 2 to 4 weeks).

Interception and soil evaporation

At nine sites, throughfall was measured using eight collection troughs randomly located within each plot. The collection troughs were 90 degree, V shaped pieces of aluminium, each 1.2 m long by 0.14 m wide, draining into a collection drum. The volume of water collected in each drum was measured every 2 to 4 weeks. Stem flow was not measured, but estimated assuming 10% of rainfall for pines and 3% of rainfall for eucalypts. Interception loss was calculated as rainfall minus throughfall (including the 10% or 3% allowance for stem flow).

The soil evaporation data from mini lysimeters at nine sites was used to derive a regression equation for each plot relating soil evaporation to plant available water in the top 0.3 m of soil. This relationship was applied using the soil water data to estimate soil evaporation.

Transpiration

Transpiration was measured continuously in 4 to 8 trees per plot using sapflow sensors (models SF100 and SF300, Greenspan Technology, Warwick, QLD). To select the sample trees from each plot, the stem diameters of all trees in the plot were measured and stem basal area over bark calculated. Plot total basal area was also calculated and the plot was divided into three classes of tree size. Each class contained an equal total basal area. Two trees for sap flow measurement were randomly selected from each tree size class.

Estimates of wound size around the drill holes were based on previously measured values and were assumed to be 3.0 mm for eucalypts and 2.4 mm for pines. Wood and water volume fractions were determined on several occasions using 5 mm diameter cores. Within plots, these did not change significantly with time. Heat pulse velocities were converted to sap velocities after Swanson and Whitfield (1981) and Edwards and Warwick (1984). To account for radial variation, the sapwood area was divided into two (SF100 sensors) or four (SF300 sensors) concentric rings of equal area, and one sensor was located at a random depth and azimuth within each ring, after Benyon (1999). Zero flows were identified using the method described by Benyon (1999). Tree mean sap velocity was calculated as the average of the two or four sample points within the tree.

In eucalypts, sapwood thickness was determined every 3 to 4 months using the sap flow sensors after Hatton *et al.,* (1995) and Benyon (1999). Younger radiata pines (< 15 years old) contained little or no heartwood. At older pine sites sapwood thickness was determined by examining the colour change in 12 mm diameter wood cores. Tree sapwood area at the sapflow measurement height was determined based on measured stem diameter, bark thickness and sapwood thickness. For trees in which sap velocity and sapwood thickness were not measured, sapwood area was estimated based on a regression with tree basal area over bark. Plot sapwood area was calculated by summing the estimated sapwood areas of all trees in the plot.

Plot daily transpiration was estimated as the product of the 24 hour mean sap velocity (6am to 6am) of the sample trees and plot sapwood area.

Soil water

Soil water to maximum depths of 3 m at some sites and 6 m at others was measured using a neutron probe. Measurements were collected every 2 to 4 weeks in five randomly located access holes in each plot at depths of 0.075, 0.15, 0.3 m and then every 0.3 m to the maximum depth of measurement. For each site the neutron probe measurements were calibrated based on the volumetric water content of soil samples collected during installation of the access tubes.

3. RESULTS

Summary characteristics of 20 sites are detailed in Table 1. All the New South Wales (NSW) sites were located within 50 km of Deniliquin in soils ranging from heavy sodic clays to sandy loams. In Table 1, the three NSW sites on heavy clay soils have been grouped together as these behaved differently compared to C. maculata (C. mac) and E. grandis (E. gran) located in a light textured, sandy loam soil. All the South Australian (SA) sites were located in the lower southeast on duplex soils with sandy A-horizons 0.4 to 2.5 m depth overlying sandy clay B-horizons. The SA sites have been grouped by species and two classes of depth-to-watertable. Components of the water balance averaged over 1 to 4 years of measurement in each group of sites are summarised in Table 1. A negative value for net drainage indicates net uptake from below the deepest soil water measurement point, while a positive value indicates runoff or deep drainage below this depth. All plantations had closed canopies, with LAI of 3.5 to 7 in the eucalypts and 4 to 6 in the pines.

The four species were able to take up substantial quantities of fresh groundwater at sites with light textured soils, where a fresh watertable was within 3 to 6 m of the ground surface. Transmissivities of the unconfined aquifer supplying ground water in SA and of the semi-confined aquifer near Deniliquin are generally very high. There was little or no groundwater uptake at six sites in SA with deeper watertables, with the exception of two sites where there appeared to be a small amount of groundwater extraction from a watertable at about 9 m depth. There was also one pine site in SA that did not extract water from a watertable at 5 m depth.

Increased water use resulting from groundwater uptake had a strong influence on plantation productivity. In both locations, current annual stem volume increment (CAI) was substantially higher at the sites using fresh groundwater (Table 1).

4. DISCUSSION

Factors influencing deep drainage and discharge

The technique has been used successfully to identify plantations which are net users of groundwater, enabling examination of site factors associated with groundwater use. The observed annual rates of ET at the sites using groundwater are within the ranges of potential ET (ETp) estimated for these regions by Wang *et al.,* (2001) or by the Penman-Moneith evapotranspiration model (Theiveyanathan *et al.,* 1998). Thus, it appears that where the roots of tree plantations have easy access to a highly transmissive, unconfined or semi-confined freshwater aquifer within about 3 to 6 m of the ground surface, their water use rates approximate potential ET as defined by Wang *et al.,* (2001) or the Penman-Monteith model. At these sites, the trees used groundwater to make up the difference between ETp and water available from rainfall.

There was little difference in water use between pines and blue gums in SA with the exception of percentage interception loss. This is in keeping with other observations in the world's literature. Interception loss

Table 1. Annual water balance components and annual stem volume increment

	P. rad	*P. rad*	*E. glob*	*E. glob*	*E. gran*	*E. gran*	*C. mac*
Locations	**SA**	**SA**	**SA**	**SA**	**NSW**	**NSW**	**NSW**
Watertable depth class (m)	> 6 m	< 6 m	> 6 m	<6 m	< 3 m	< 3 m	< 3 m
Number of sites	4	3	2	6	3	1	1
Soil texture	light	light	light	light	heavy	light	light
Rain + net Irrigation[1] (mm)	673	586	614	675	530	628	583
I (mm)	161	205	101	125	44	60	60
E (mm)	99	111	169	176	105	253	253
T (mm)	458	669	342	758	308	675	1018
ET[2] (mm)	718	985	612	1059	457	988	1331
Dw[3] (mm)	19	-28	-6	-16	0	0	0
Annual Net drainage[4] (mm)	**-63**	**-371**	**8**	**-367**	**73**	**-360**	**-748**
CAI (m3 ha-1 year-1)	27	32	19	43	10	50	41

at six blue gum sites averaged 18% compared with 29% at three radiata pine sites. For regional average annual rainfall of about 700 mm this equates to a difference of about 80 mm year^{-1}.

The sites near Deniliquin, NSW had similarly shallow water tables, but widely differing soil properties. Sites with heavy-textured, sodic clay soils did not use groundwater, while those with light-textured, sandy loam soil used large amounts of groundwater. Root coring and excavation studies indicated roots were unable to penetrate the heavy clay soils. Rooting depth, and hence the degree to which tree roots were extracting groundwater, was thus controlled by soil properties. Any impenetrable soil layer above the watertable, due to adverse physical or chemical properties, prevented the trees from using groundwater.

In southeast SA, radiata pine roots in a cave penetrated to at least 14 m below-ground and individual roots from one large tree extracted groundwater from this depth (Doody and Benyon 2003). However, this is the only site so far examined in SA, where groundwater extraction has been detected from any watertable at more than 9 m depth. Where the watertable is between 6 m and 9 m, groundwater extraction appears less likely and in smaller amounts. This may mean tree roots are unlikely to penetrate deeper than about 6 to 9 m without meeting an impenetrable layer. There will also be increasing hydraulic resistance to water uptake with increasing water table depth.

Measurement errors

In this study, deep drainage or uptake from below about 3 or 6 m was estimated as the difference between measurements of ET and changes in soil water. Any errors inherent in the measurements have been included in the estimates of deep drainage or uptake. Because the measurements of transpiration, throughfall, soil evaporation and soil water were based on samples, it is possible to place statistical confidence limits on these estimates. These indicate uncertainty in estimates of annual ET of about 50 to 100 mm year^{-1} using this method.

No studies of water use in these plantations have been undertaken prior to canopy closure when LAI is below that carried by the stand after canopy closure. Plantation forestry in Australia usually involves

1. Sites near Deniliquin NSW were irrigated once or twice times each summer, those in SA were not.
2. ET = I + E + T
3. Dw = net increase (Dw positive) or decrease (Dw negative) in stored soil water.
4. Negative net drainage represents net water uptake from below the deepest soil water measurement depth.

rotations of between 10 and 40 years. In plantations grown commercially for wood production, planting densities are such that canopy closure usually occurs in 5 to 8 years in radiata pine and 2 to 5 years in eucalypts. There is a period of several years between rotations and in the early part of each rotation when, due to the small size of the trees and rigorous control of competing weeds, LAI and hence ET is probably less than that of rainfed pasture. In tree crops grown for sawlogs, the plantation is thinned (some trees removed to increase growth of others) two to four times. Thinning also reduces LAI and hence can reduce plantation water use if applied appropriately. During a full crop cycle, maximum LAI will only be present for 60% to 80% of the rotation. Under all plantations there is likely to be some net recharge early in the rotation and between rotations and possibly for a period after thinning. These factors need to be accounted for in calculating regional water balances.

5. CONCLUSIONS

Water balance studies can be used reliably to quantify plantation water use and net groundwater recharge and uptake in plantations established over shallow water tables. At some sites trees can use water from shallow, transmissive, fresh aquifers. At these sites the difference between ETp and rainfall is supplied from groundwater, resulting in a substantial increase in productivity.

6. ACKNOWLEDGEMENTS

This work could not have been undertaken without the financial and in-kind support of The SA Dept of Water, Land and Biodiversity Conservation, the South East Catchment Water Management Board, Forestry SA, Auspine, Green Triangle Forest Products, The National Heritage Trust, Rural Industries Research and Development Corporation and the Forest and Wood Products Research and Development Corporation. We also thank the many scientific and technical staff who have been involved with these projects.

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Some Observations on the Water Use of Mature Eucalypt and Immature Radiata Pine Forests in Victoria, Australia

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Summary: This project involved multiple catchment measurement of tree water use for radiata pine of up to 23 years of age, including the effects of a pine thinning at age 18 years and water use of the mature eucalypt catchments in south-eastern Australia. One of the catchments was converted to radiata pine in 1980. The control catchment showed that annual water yield was basically a function of annual rainfall, and that annual rainfalls below 900 mm generated little runoff. The analysis showed an increased water yield of up to 300 mm per annum immediately after clearing but that this was progressively declining. For the seventeen year old trees the radiata pine water use appeared to be close to that of the original eucalypt forest. However a non-commercial thinning increased the yield by about 100 mm, and this yield has then declined. Comparison with other Australian results suggested that the radiata pine water use (relative to the native vegetation) could be approximated as a function of age and rainfall. Older trees and low-rainfall years are associated with a higher water use by the radiata pine than the native forest, but conversely younger trees and higher rainfalls give lower water use. This relationship would be useful for estimation of the net change associated with radiata pine in larger catchments.

1. INTRODUCTION

The work reported in this paper resulted from community concern about the lack of knowledge of water use and nutrient impacts of *Pinus radiata* D. Don plantations in southern Australia. These concerns are not new, and in the mid 1970s the Croppers Creek multiple catchment project in north-east Victoria was initiated to provide answers to allegations concerning the impact of newly cleared plantations on storm flow hydrology. This paper updates results to May, 2003. This project was designed to determine the hydrologic impact of intensively managed radiata pine on water flows, catchment dynamics, and water quality properties compared to native (eucalypt) forest. One catchment (Clem Creek) was converted from native forest to plantation in 1980 (Figure 1). Measurement on the initial Croppers Creek project ceased in 1987 because of cut-backs in Government funding. At the resumption of this project in 1997 this contained seventeen-year old radiata pine. Adjoining “control” catchments (Ella and Betsy Creek) consists of native, mixed species eucalypt forest. The present project involved a rehabilitation of the defunct catchment project to obtain data on the catchment water use and nutrient export. It is hoped to continue these measurements over the remaining rotation of the radiata pine. The results reported in this paper are a continuation of the results reported by Leitch and Flinn (1986), Bren and Papworth (1991), and Bren (1997). Specifically this paper examines (1) the water yield of the catchments carrying native eucalypt forest and (2) the changes in water yield associated with the growth of radiata pine. Some simple comparisons are made using data available from two other catchment studies. Work concerned with nutrient balances is being separately reported.

2. METHOD

Analysis of collected data followed conventional multiple-catchment approach, as detailed in Bren and Papworth (1991). Flow in Ella Creek was used as a real-time model to provide an estimate of flow in Clem Creek if the catchment had not been converted to radiata pine. In general Ella Creek provides an excellent control catchment except that it can dry up at low flows. A fuller account of the work is available from the author on request.

Figure 1. Clem Creek catchment with 17-year old pines

3. RESULTS

Figure 2 shows the measured annual stream flow from the native forest catchments as a function of rainfall. For reference these are compared to the results of Holmes and Sinclair (1986). It can be seen that, in this environment the native vegetation is a heavy water user. Thus a minimum annual rainfall of about 800mm is needed before there is measurable stream flow.

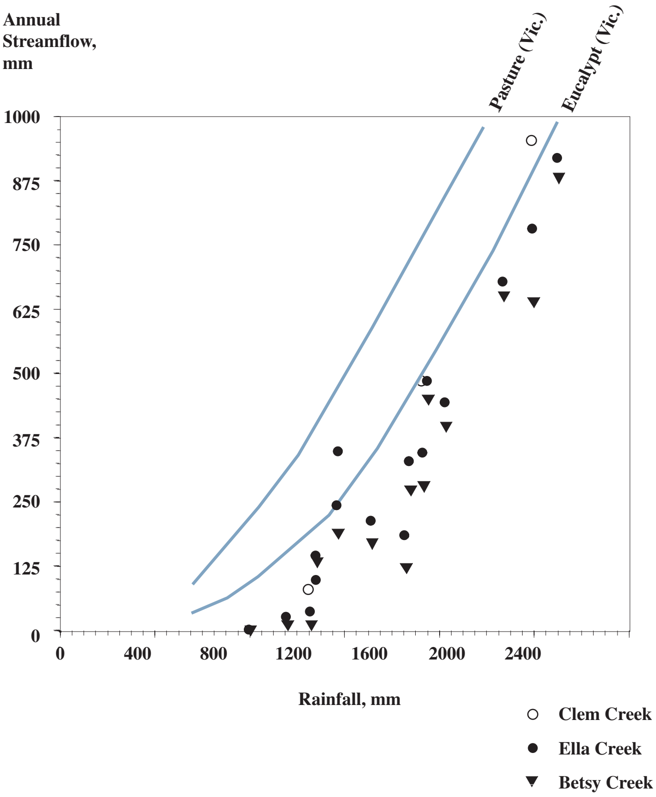

Figure 2. Water use as a function of rainfall for the three catchments within the Cropper Creek Project.

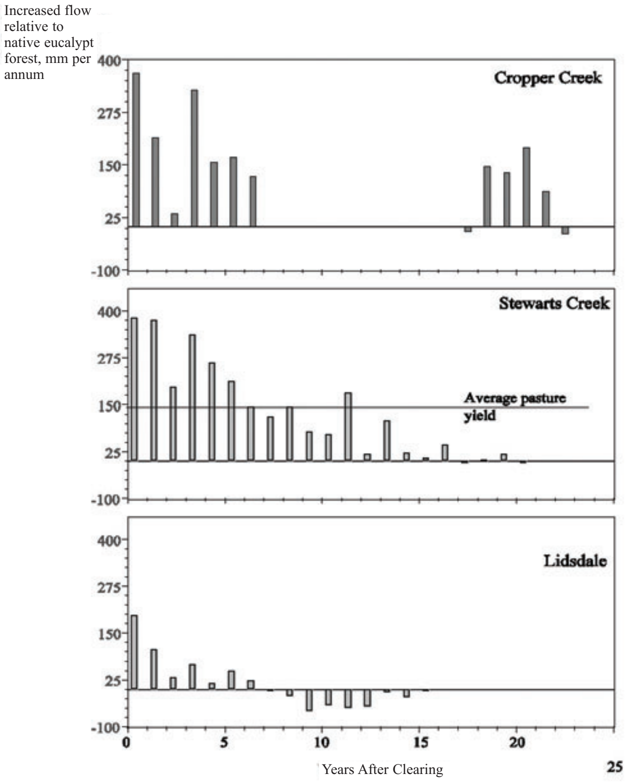

Figure 3. Increase in flow (relative to radiata pine) resulting from conversion of a catchment from eucalypt to radiata pine as a function of age. Data is for Croppers Creek (N.E. Victoria), Stewarts Creek (Central Victoria) and Lidsdale (Central NSW).

Recently there has been community concern about the "excessive" water use of regrowth forest. It is likely that the water-using habits of these mature–age mixed species eucalypt forests may prove a threat to their long-term status as water becomes more valuable.

Figure 3 shows the stream flow relative to the flow from a catchment carrying native forest after the conversion to radiata pine. The utility of the results are compromised by the break in measurement between 1987 and 1997. The illustration also includes results from two other multiple catchment projects concerned with radiata pine. The first of these is the Stewarts Creek experiment in Central Victoria (Nandakumar and Mein 1993). The second is the Lidsdale Project near Bathurst, NSW (Putuhena and Cordery, 2000). It can be seen that there is a diminished yield associated with the growth of the trees over time, and that in times of low annual rainfall the Clem Creek yield seems slightly less than that of the catchment under native forest. However there are many similarities in response. The increase in yield at Cropper Creek after

1988 (year 18) is associated with a non-commercial thinning of the pines.

If we pool the data we can derive the increase in flow as a function of the annual rainfall and the age of the trees:

$$\Delta y = -238.6 + 13.58\, r^{0.5} - 10.36\, a \qquad (1)$$

(R^2 = 0.674), in which Dy is the yield increase relative to the native eucalypt forest (mm), r is the annual rainfall in mm, and a is the age of the tree in years. Notwithstanding the dissimilarities between sites, the relationship (as shown by the R2 value) is surprisingly strong. Figure 4 shows a contour plot of this relationship. Thus the relation suggests that, with either young trees or heavy rainfall radiata pine tends to give more runoff than the native eucalypt forest, but with older trees and lower rainfall the stream flow is less than that of the native eucalypt forest. Thus, for instance, 15 year old trees receiving 1,200 mm annually of rainfall would yield 73 mm of runoff more than the native eucalypt forest. If the same trees received only 800 mm of runoff then the water yield would be 13 mm less than that of native forest. The relationship also suggests that attempts to find a "water-usage" curve are doomed to failure because the rainfall is a key factor.

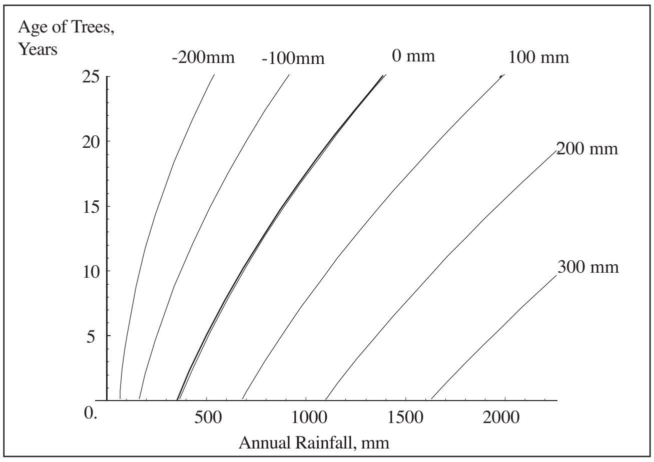

Figure 4. Contour plot of increase in flow as a function of annual rainfall and age of radiata pine.

The relationship offers the potential for a direct approach to the question of "how much water will the radiata pine plantations in a large catchment use". Thus, for each year one would need the distribution of age classes. One could then feed this information in, together with the annual rainfall and by weighting the contributions by area, determine the net amount relative to native forest. If required an estimate of the absolute amount of runoff generated by native forest could be derived from Figure 2. If pasture land was involved then an adjustment factor could be used. Ideally other factors such as the thinning status of the forest could be included but this data is rarely available.

4. CONCLUSIONS

The Croppers Creek project has shown that the native eucalypt forest in N.E. Victoria is a heavy water user, and that effectively 800 mm of rainfall per annum is needed to generate any stream flow. The conversion of a catchment to radiata pine has generally increased flows but this effect appears to be diminishing with increased age. Pooling of the results with two other hydrologic experiments indicates that the radiata pine water use (relative to that of the native forest) is a function of age of the trees and annual rainfall. In high rainfall years and with young ages the radiata pine uses less than the native forest. With low rainfalls or older age classes the radiata pine uses slightly more water than the native forests.

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Turbidity Probes – Can They Reliably Estimate Suspended Sediment Concentrations in Forest Streams?

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State Forests of NSW is required to monitor water quality and suspended sediment loads via an EPA monitoring program in certain forested catchments. This program was initiated in 1998 utilizing automatic water sampling over storms for sediment concentration, and the continuous measurement of stream discharge. However inadequate sediment sampling during the more extreme events has resulted in unacceptable errors in load calculation.

As site-specific relationships have frequently been found to exist between turbidity and suspended sediment concentration, the program was expanded in 2000 to determine these relationships via in-situ turbidity measuring probes, with the ultimate aim of indirectly estimating sediment concentrations and hence sediment loads. This paper examines relationships obtained for certain monitoring sites, and quantifies errors in estimating loads.

Sediment concentration and turbidity data were log-transformed because of skewed distributions, and multi-site or single-site regressions were developed between log-transformed sediment concentration and log-transformed turbidity data obtained at certain monitoring sites. Prediction errors, corrected for bias introduced through the use of log-transformed data, varied but were generally within the range ±200 mg/L, while the interquartile error range was generally less than 50 mg/L.

Analyses of prediction errors have not been presented in comparable studies, making it difficult to compare outcomes and to estimate whether these relationships could improve following longer investigation periods with more data. It is probable, however, that even with relationships containing much additional data, prediction errors may not decline because of the inherent nature of the processes which produce turbidity and sediment concentration values in streamwater.

Forest Harvesting Activities and Water Quality: A New Approach to an Old Problem

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Summary: This paper examines how recent research has contributed to forest management and in-stream water quality. It follows on from a previous review paper presented at the Second Forest Management Workshop in 1999 where three broad areas of research were identified as significant. These included Forest Roads; Catchment Connectivity and Managing water quality for catastrophic events. These three research themes are looked at within the context of changing research trends and industry demands. With a decrease in the harvesting of native hardwoods, much of the recent research must now address the specific needs and demands of plantation harvesting. As such the issues of roading density and connectivity are critical. As recent wildfires in the ACT, NSW and Victoria have demonstrated, there will be scenarios when even the best management practices will fail, simply due to the enormity of the impact and the catchments response.

1. INTRODUCTION

In May 1999, we convened a similar gathering of the forest industry, scientists and relevant stakeholders in Warburton, Victoria to discuss aspects of water quantity and quality management in forests. We subsequently applied for Australian Research Council (ARC) funding to support the continuation of forest-related research in the area of water quality protection. Some key stakeholders in the forest industry (NSW State Forests, CFS Victoria) supported the project, which had the major objective of delivering "an innovative and cost-effective framework for the protection of water resources in forestry environments". We provided the above title for our grant application and as we have found out over the last three-years, it proved an ambitious project.

Some of the key findings of this project are presented in more detail in accompanying papers (Takken and Croke, 2004a, Takken and Croke, 2004b), Thompson *et al.*, 2004, Lane *et al.*, 2004). The purpose of this paper is to review this research within the broader context of forest harvesting activities and water quality protection. In essence, the paper examines the fundamental questions of whether research has actually changed the way the forest industry approach an old problem?

1.1 Broader Literature Review

A review article published in the proceedings of the Warburton meeting entitled "Impact of timber harvesting activities on water quality: what can we confidently say after 30 years of research?" (Croke, 1999) provides a useful context and structure for this paper. The purpose of the 1999 review paper was to synthesise the literature for some definitive conclusions on the nature and magnitude of water quality impacts associated directly with harvesting activities. There has certainly been an abundance of research on, or closely related to, the topic over the 30-yr-time period. Perhaps not surprisingly, the review did not achieve this objective; in part due to the enormous diversity in the way 'impact' is reported and indeed measured. Overall there was no specific trend in water quality data or in-stream response that could be related unequivocally to harvesting activities.

The review was more useful in identifying a change in research focus, which depicted a departure from traditional soil erosion studies at the small plot scale and a move to sediment delivery type research at the catchment scale. While there has always been implicit recognition that timber-harvesting activities result in soil erosion, the extent to which these activities contribute to water pollution has always been more ambiguous. The review exposed some fundamental gaps in our understanding, and quantification of, processes that linked soil loss (ie particle detachment) to in-stream water pollution. The review concluded that there were three main research areas which should be developed to enhance our knowledge on harvesting activities and water quality protection. These were:

- Road Use and Sediment Production;
- Connectivity in Forest Catchments;
- Managing for Worst-Case-Scenarios.

Since the 1999 paper significant changes have occurred within the forest industry that affect the application and relevance of some of our research. The forest industry in both NSW and Victoria has progressively declined operations in native hardwoods (NSWSF, *pers. comm.*). The industry now has greater financial and operational investments in plantation harvesting, a trend that is likely to expand in future years. This growth area in plantations brings with it a different environmental protection philosophy and potentially, research needs.

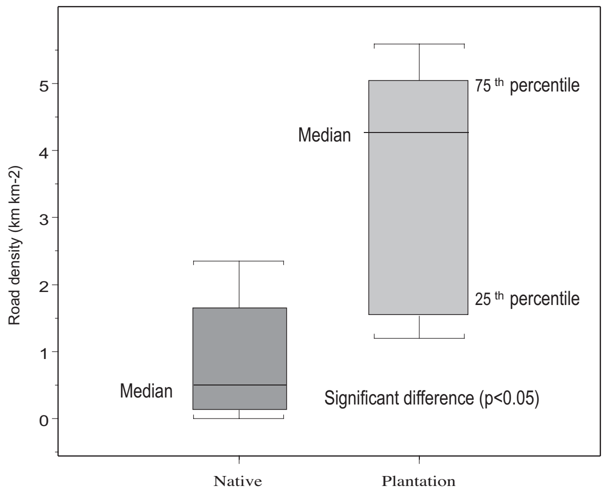
This paper will now examine advances in these three areas of (1) roads, (2) connectivity and (3) catastrophic events and specifically their contribution to future issues of forest management and water quality protection.

2. ROADS: THE DOMINANT SOURCE OF SEDIMENT

The potential contribution of forest roads as a source of sediment was recognised by research in the early 60s (Haupt, 1959). Later experimental work in the United States (Reid and Dunne, 1984) and Australia (Croke *et al*., 1999, Lane and Sheridan, 2002, Sheridan *et al*., 2004) added further to our understanding of generation processes. We now have a reasonable understanding of the factors, such as road usage and road surface materials, which contribute to enhanced sediment production from these features. Croke *et al.,* (1999) reported that in general, unsealed forest roads generate one order of magnitude more sediment than a forest snig track and two orders of magnitude more than a disturbed general harvesting area (GHA) for the same simulated rainfall event. Sediment production rates were also closely correlated with road usage or traffic intensity.

As the papers presented to this theme at this workshop testify, forest roads have continued to be emphasised as a source of sediment in our research (see Lane *et al.,* 2004, Sheridan *et al*., 2004). Roads are now commonly regarded as the most significant sediment source and, equally as importantly, the dominant pathway for runoff and sediment delivery to streams (see Section 2).

There has been a concerted effort on the part of the industry to address some of the issues associated with roading and road drainage. However, with an increase in plantation operations, the issue of road density, location and sediment production deserves particular attention. Recent surveys in both native hardwood and pine plantations in NSW and Victorian forests reveals a statistically significant difference in road densities between these two types of harvesting operations (Fig. 1).



Catchment Forest Type

Figure 1. Box-plots of roading densities in plantation and native hardwood sites in NSW and ACT. From Thompson *et al.,* 2004 (unpublished data).

In addition to the obvious implications for sediment production, the criss-crossed pattern of high road densities within plantations also has significant implications for the delivery of this material to the stream network (Fig 2). Many plantations are now dealing with the potential affects of an inherited road network, where poor initial planning has created a highly connected system of roads and streams. In recognition of plantations high roading densities, and as a result of devastating bushfires of January 2003 in

Canberra, there has been a commitment by ACT Forests to reduce roading density by 60% in the next 3-5 years. The landscape scale of such road rehabilitation is considerable, and to my knowledge never before attempted in Australia.

3. CONNECTIVITY AND WATER QUALITY

The second major research focus is the recognition of the linkage between sediment sources and the stream network. This is now commonly referred to as 'connectivity' and is a major conceptual advance in the way both scientists and forest managers have addressed water quality protection in forested catchments. Several papers at this workshop highlight the importance of this concept to water quality protection at the catchment scale (Takken and Croke, 2004a, Lane *et al*., 2004).

3.1 Definition of Connectivity

The concept of connectivity has been widely used in both ecology and geomorphology to describe some spatial structure between landscape attributes. Interestingly, biologists define landscape connectivity as the degree to which a landscape facilitates or impedes the movement of individuals (Taylor *et al*., 1993). In the hydrological and geomorphological sciences, it has been used to convey some sense of spatial integration in the landscape. For example, runoff generated on a hillslope is connected to the stream system through the physical linkage of these two landscape components. One definition of connectivity, which explicitly recognises the hydrological component, is that of Bracken and Croke (2004), which defines it as the volume of water necessary to result in connected flow within a catchment. Connectivity is therefore a function of the potential runoff produced by an area, transmission losses, and the travel distance of runoff. The latter will be affected by the existence of features (natural or constructed) that either prevent runoff reaching channels such as dams and terraces or those that encourage runoff connecting with channels such as roads and tracks (Bracken and Croke, 2004). In reality, there is considerable variability in the degree to which landscape elements are actually connected to the stream. As such, connectivity at the scale of entire catchments has been rarely measured and remains difficult to incorporate into any physically-based process models. Another approach is to investigate connectivity from the perspective of the various types

Figure 2. Example of the roading pattern in a plantation site in Victoria.

of pathways that may link runoff or sediment sources (Fig 3). This approach has now been widely adopted to examine the specific impact of forest roads on runoff and sediment delivery (Wemple *et al.*, 1996, Croke and Mockler, 2001, La Marche and Lettenmaier, 2001).

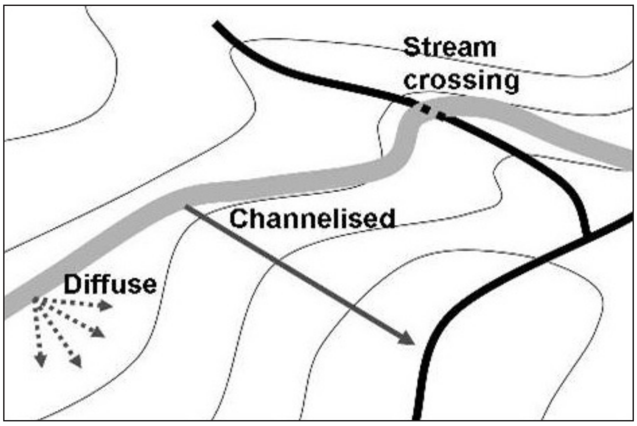


Figure 3. Range of connectivity pathways in a catchment with road drains (adapted from Wemple *et al.,* 1996 and Takken and Croke, 2004a).

3.2 Types of Connectivity

By far the greatest threat to water quality is the 'direct' forms of connectivity such as stream crossings and continuous gully development from drain outlets to streams. These pathways spatially integrate the road and stream network and provide efficient pathways for the transfer of runoff, sediment and associated pollutants. Each catchment may display a different spatial pattern of connectivity due to road position, type, density and drain frequency. Takken and Croke (2004a) report a range of connectivity scenarios from catchments in both NSW and Victoria and demonstrate the impact associated with stream crossings, gullies and diffuse overland flow pathways. Road-related mass movement, as commonly occur on the fill batters of cut-and-fill roads, are another form of connectivity that has been recognised in forested catchments (see Erskine, 2004).

Overall, the concept of connectivity has made a significant difference to the scale at which both scientists and forest managers view potential road-related water quality impacts. It encourages catchment-scale analysis. This, in itself, forces forest managers to address not just the isolated impact of a poorly constructed road or road drainage, but also view its overall catchment impact. For example, previous experimental work in some coastal forests in NSW demonstrated that all of the road related gullies were located on a single road in the northern part of the catchment. However, 28% of the downstream stream network was affected by runoff from this one road (see Takken and Croke, 2004a).

While considerable effort has gone into the maintenance and upgrade of the road surface in recognition of their dominance as runoff/sediment sources, continuing effort must be made on the re-direction of road runoff. It remains common practice, and logistically easier, to discharge road runoff directly into convergent parts of the landscape, such as stream hollows, that drain to first order streams. We envisage that the development of a road-connectivity GIS model (see Takken and Croke, 2004b) will allow forest managers to better plan the location of forest roads and their drainage, with due consideration of their delivery and connectivity potential.

4. CATASTROPHIC EVENTS AND WATER QUALITY

Water quality protection in many areas is now managed through the imposition of water quality standards or set targets. These are typically defined in terms of turbidity, typically measured solely in terms of NTU's. This reflects a growing trend in water management for agencies to achieve a reduction in overall sediment loads in rivers. As the review paper of 1999 discussed, one significant problem with water quality management is determining a measure of impact using a mean value of turbidity. The range of turbidity values reported for forested and non-forested catchments in the literature is understandably very diverse. The values reflect a range of climatic and disturbance environments. Sediment concentrations or loads for logging operations in the Tropics are commonly several orders of magnitude higher than those even for highly intensive operations in temperate climates (Croke, 2004). Not withstanding problems of in-stream turbidity measurement and their interpretation (see Noske *et al.*, 2004, Webb, 2004), we can not confidently say that once a stream exceeds a particular level of turbidity, some notable in-stream response will be observed. 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. Thompson *et al.*, (2004) proposes the use of a process classification of stream morphology to develop a template, which can be used to investigate the impacts of increased sediment load on in-stream ecology.

The 1999 review clearly outlined case studies where sediment concentrations or turbidity were significantly different in catchments where standard prescriptive measures were applied. Suspended concentrations were found to range from 56,000 mg/L to 15 mg/L depending on the application of no prescriptions or standard management practices respectively. It seems unlikely that we can effectively manage water quality protection in forestry environments using the application of 'set targets'. This seems particularly relevant in the aftermath of the catastrophic events as occurred recently throughout much of NSW, ACT and Victoria in the form of wildfires. As the field day associated with this workshop will demonstrate, there are no management practices that can effectively prevent the large-scale landscape readjustments that follow events such as wildfire.

Although there are no data sets available as yet for the ACT regions, reports of extensive in-stream sedimentation following rainfall events which occurred immediately after the fires, are several orders of magnitude larger than that observed or measured for timber harvesting activities alone. The extent of erosion and sedimentation we have witnessed as a result of these fires raises interesting challenges for forest managers in terms of deciding what sort of rehabilitation practices will limit future adverse impacts. The literature is clear that where best management practices are applied, then the observed impacts on water quality are significantly reduced. The occurrence of such catastrophic events can not be accommodated for in the design of these practices. Rehabilitation of these disturbed landscapes will, however, have to consider alternative management options, which may assist in limiting overall catchment connectivity. A key contribution will undoubtedly be the re-design of the existing road network and the careful planning of the location of future tracks.

5. SUMMARY

In returning to the three themes of (1) roads, (2) connectivity and (3) catastrophic events, it appears that research has continued to make a contribution to our understanding of these aspects as they relate to forest management and water quality protection. Roads are now regraded as the most significant sources of sediment but just as importantly, as the dominant pathway for runoff and sediment delivery in forests. The concept of connectivity has allowed us to better view the catchment-scale impacts of poor road location and drainage. This will inevitably lead to a more holistic basis for water quality protection and provide the industry with practical guidance on priority areas for road rehabilitation and water quality protection. While it is widely acknowledged that the most important issues in relation to water quality relate to the standard to forest management practices, events of the last year in many states also highlight the disturbance due to natural catastrophic events. The challenge now is to work with the industry in developing and modifying best management practices for the recovery of many forested catchments.

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Assessment of Mass Movement Hazard in Canobolas State Forest, NSW

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Summary: Clearfall harvesting of 1962 age class *Pinus radiata* plantation in Canobolas State Forest triggered an assessment of mass movements according to the Environment Protection Licence. Only a small number of rain storm-induced mass movements, particularly debris slides and debris flows, have occurred episodically in Canobolas State Forest both before and after European settlement. Natural shallow landsliding was restricted to steep convergent areas, such as hollows, when rain storms caused soil saturation. Management-induced mass movements have only been found to date on cut road batters where either the road was re-routed through diamictons formed by large rotational slides of probable Pleistocene age or where the road removed toe support. The risk of mass movements is low and largely associated with roading works.

1. INTRODUCTION

State Forests of New South Wales (SFNSW) carries out forestry activities under a number of approvals including an Environment Protection Licence issued by the Environment Protection Authority. Mass movement assessment must be conducted according to a standard procedure contained in the licence when forestry activities involve the maintenance and upgrading of existing roads or the construction of new roads. Recent clearfall harvesting of the 1962 age class of Monterey Pine (*Pinus radiata* D. Don) in Canobolas State Forest involved the construction of 0.15 km of new road and the maintenance and/or upgrading of 10.1 km of existing roads. Therefore, the mass movement assessment provisions of the licence were triggered. This paper presents the results of a detailed mass movement hazard assessment of harvesting activities in Canobolas State Forest (Erskine 2004).

2. CANOBOLAS STATE FOREST

Canobolas State Forest No. 901 forms part of a post-World War II government reforestation project with softwood plantations in the Orange area. The first plantings occurred in 1947. The 1962 age class is mapped as three informal stratigraphic units of the Canobolas Volcanic Complex, namely, unnamed basaltic intrusions and flows (Tb), unnamed alkali rhyolite (Tp) and unnamed trachyte intrusions, flows and volcaniclastics (Tt). A series of K-Ar dates on these volcanic rocks gave ages ranging from 12.7 to 10.9 Ma (Middle Miocene). The Canobolas soil landscape has been mapped by Kovac *et al.,* (1989) in the 1962 age class and comprises a toposequence of lithosols and skeletal soils on steep slopes and rocky pinnacles; krasnozems on gentler slopes; and yellow solodics and occasional yellow podzolics in depressions and on footslopes. The dominant soil regolith stability class for the 1962 age class was assessed by Murphy *et al.,* (1998) as class 3 with a sub-dominant class of 1. Recent soil assessments by David Bell and the author are consistent and indicate that the soil regolith stability class for the Canobolas soil landscape should be revised to a dominant class of 1 with a sub-dominant class of 3. Furthermore, the author has identified *subplastic* gravelly clay loams and gravelly light clays for the first time in the Canobolas soil landscape and such materials exhibit high mechanical and chemical stability.

3. HISTORICAL MASS MOVEMENTS

Although Fell's (1992) extensive review of landslides in Australia did not cite a single reference to mass movements on Tertiary basalt in the Orange area, the following four mass movement events have been recorded recently in Canobolas State Forest:

1. Lee (1999) reported a major landslide (debris slide) that involved some 1.3 ha on a 28.8^{o} slope of 1969 age class pine in compartment 259 of Canobolas State Forest in about October 1998. The debris slide was located in a slope planform concavity (hillslope hollow) which concentrated overland

flow and soil water, and which would, therefore, cause soil saturation during prolonged rain. From analysis of rainfall records, it was concluded that the 1998 landslide actually occurred at some time between June and August rather than when first observed in October. The debris slide resulted in a second mass movement in the drainage line downstream of the toe of the debris slide. Temporary damming of some of the material eroded by the debris slide in the drainage line was effected by a series of interlocked pine trees (debris dam) at the side of the debris slide. Initiation of renewed movement of some of the temporarily stored debris as a flow was probably caused by a combination of liquefaction and partial failure of the debris dam soon after formation. The debris flow extended about 500 m along the drainage line and downstream channels.

2. Complex debris flows occurred during the night of 16/17 November 2000 in the catchment of water quality monitoring site CNBL06 in the 1962 age class. The flows consisted of two discrete parts, an upper slope flow and a lower channel flow. The ***upper slope flow*** failed first and eroded a very steep hillslope hollow on the right side of the valley where the contours concentrated surface and subsurface water. The slope angle where the upper flow occurred was 33 to 35^{o} but there was no well-defined failure plane in the soil regolith. The upper debris flow occurred because the soil was saturated due to prolonged rainfall over five consecutive days; overland flow and streamflow were concentrated in a hillslope hollow downstream of an upper slope amphitheatre; and slopes were very steep due to the location at the edge of vertically stacked trachyte flows. The ***lower channel flow*** was restricted to the main channel on which water quality monitoring site CNBL06 was located. The material eroded by the upper flow was deposited in the small bedrock-confined main channel at the foot of the slope. This sediment and large woody debris caused a temporary dam which subsequently failed by overtopping, initiating a second confined debris flow downstream of the original slope debris flow. The resultant channel debris flow severely eroded the stream margins and transported many pine trees. The main channel slope is about 10^{o} and the lower flow moved about 140 m downstream until it reached a local valley expansion on a bend where the transported pine trees wedged to form a 3.3m high debris dam which trapped most of the mobilised sediment. The structural integrity of the dam is dependent on the longevity of the pine trees.

3. Repeated translational slides have occurred on part of a steep (33^{o}) cut road batter on Cadiangullong Road. Upslope infiltration and seepage were important hydrologic processes that led to a high water table and positive pore water pressure of the earthen embankment, causing slope failure. When Cadiangullong Road was reconstructed, it was re-routed to higher ground through a short section of ancient landslide deposits. These landslide deposits are diamictons, which are nonsorted sediments consisting of sand and/or larger particles dispersed through a muddy matrix. The diamicton was probably formed by at least two late Pleistocene phases of deep-seated rotational sliding of the steep slope above the road cutting.

4. A recent (ie within last 20 years) small mass movement in compartment 156 which crossed a road and necessitated local road works.

4. PREHISTORICAL MASS MOVEMENTS

A field assessment of mass movements in the 1962 age class found the following evidence of landslides:

- An old diamicton consisting of an irregular hummocky area of cemented angular volcanic gravels below a log landing in compartment 158.
- A series of at least four vertically stacked landslide deposits up to 2.5 m thick in a cut road batter in compartment 155.
- A very old but poorly exposed landslide deposit in a road cutting in compartment 156.

5. AIR PHOTOGRAPH INTERPRETATION

Air photograph interpretation was also conducted of a rectangular area of approximately 80 km^2 which included the 1962 age class and the same geological mapping units. Three sets of vertical air photographs (1991, 1997 and 2000) were used. At least eleven mass movements were identified in areas not hidden under a continuous pine canopy:

- Five possible debris flows in the steep headwaters of streams draining the central elevated core of the Canobolas Volcanic Complex. Only one is located in Glenwood State Forest while the others are located on freehold land or in Mt Canobolas Park.
- Six possible translational and rotational slides on steep slopes in the same area. Only two are located in Glenwood State Forest while the others are located on freehold land or in Mt Canobolas Park.

6. RAINFALL TRIGGER OF MASS MOVEMENTS

Detailed analysis of the Canobolas State Forest (Station No. 063018) rainfall record revealed a poor correlation between monthly rainfall and the occurrence of historical mass movements because:

- the location of the rain gauge is possibly unrepresentative of rainfall at higher elevations;
- monthly rainfall may not be an accurate measure of mass movement-inducing rainfall events. However, Reid (1998) concluded that the potentially important effects of antecedent moisture may not be accounted for by shorter durations;
- the rainfall-slope stability relationship is indirect via rainfall effects on pore water pressure; and
- the mass movement record for Canobolas State Forest is possibly incomplete.

A large statistically significant increase in annual rainfall was detected at the Canobolas State Forest station since 1949 and was also associated with a number of very high annual totals. The years 1950, 1952, 1956 and 1960 exhibited greatly above average rainfall but the period 1973-1978 was extraordinarily wet. However, no mass movements were recorded during these years. Figure 1 shows changes in annual rainfall at Orange Post Office which has the longest record for the area and is also highly correlated with rainfall at Canobolas State Forest. While mass movements do occur infrequently in Canobolas State Forest, the risk is minor, even during extended wet periods.

The results of a simple water balance model based on monthly rainfall at Canobolas State Forest gauge suggest that soil moisture storage capacity is only exceeded during August under median monthly rainfall but is exceeded between July and September under mean monthly rainfall. Clearfall harvesting, road works and site preparation should not be conducted when total soil moisture storage has been exceeded.

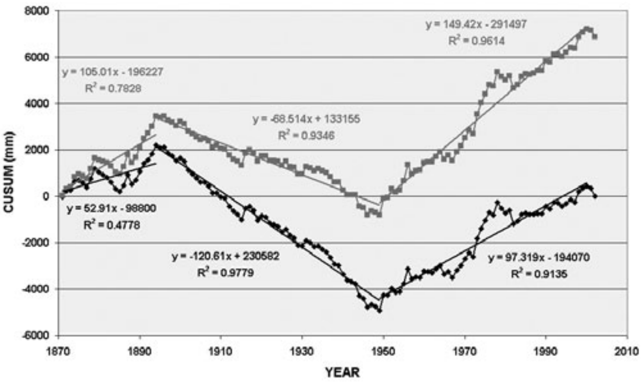

Figure 1. Cusum based on mean (bottom) and median annual rainfall (top) at Orange Post Office.

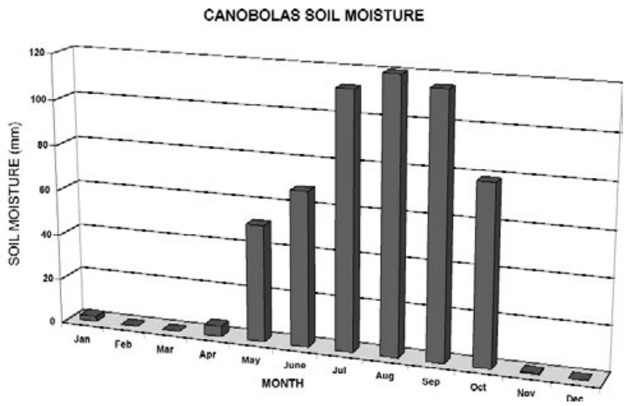

Figure 2. Soil moisture storage in the soil profile at Mt Canobolas for median monthly rainfall, assuming a maximum soil moisture storage value of 120 mm to produce soil saturation.

7. CONCLUSIONS AND RECOMMENDATIONS FOR FORESTRY ACTIVITIES

The essential conditions for the initiation of historical mass movements in Canobolas State Forest are:

- Relatively deep soil (about 1m minimum depth) that develops a failure plane either between soil layers/horizons or between soil and bedrock.
- Saturated soil and build up of pore water pressure due to prolonged rainfall, with or without an intense rain storm.

- Flow concentration in hillslope hollows or planform concavities, which further promotes soil saturation and increased pore water pressures during rain storms.
- Very steep slopes (>25^{o}) on the edge or snout of individual trachyte flows or a series of vertically stacked flows or on the side of steep volcanic cones and domes.
- Formation and subsequent rapid failure of landslide dams which cause channel debris flows and, in some case, the formation of slot channels on slopes of 10^{o}.

All forestry activities undertaken in steep hillslope hollows with relatively deep soils should not cause soil saturation and/or artificial loading of the slope. Runoff dispersal from roads should not be directed to steep hillslope hollows and the use of machinery on such areas should be prohibited when soils are saturated. Management-induced mass movements have only been found to date on a cut road batter on Cadiangullong Road where the road was re-routed through diamictons formed by a large rotational slide of probable late Pleistocene age and on another cut road batter in compartment 158. It is recommended that new roads or the upgrading of existing roads should avoid excavating cut batters in thick diamictons (>2 m deep). Where there are existing cut batters in such diamictons, appropriate remedial works should be installed, where required.

Plantations are a recommended land use for mass movement-prone areas because rapid mass movements occur more frequently per unit land area under grass than under forest. Soil regolith strength is increased by tree roots which extend into either joints and fractures in bedrock or into weathered soil.

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Plantation Impacts on Stream Flow - Putting Things in Proportion

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Summary: The discussion on water use by forest plantations should be seen in a wider "whole of landscape" context. Perceptions on the impacts of plantation on water flow may be inconsistent with the small proportion of catchments planted. National Forest Inventory figures on Australia's plantations show that most catchments have a very low proportion of plantations - usually below 4% of the total catchment area. Water use impacts from land use changes on the other 96% of land also need to be considered, especially the 64% of land (on average) used for agriculture. Indications are that the agricultural land use changes are larger than forestry in area and in most cases are towards higher water use practices. Future plantation expansion scenarios are uncertain. Some unrealistically high estimates have raised concerns over the potential water impacts. Industry estimates are for more moderate expansion and into lower rainfall zones that are likely to have less impact on water yield than past plantations. The challenge is to improve our understanding of these impacts and take a multidisciplinary approach to plan future plantations to deliver social, economic and environmental benefits (e.g. salinity reduction) with least impact on water flow. The paper will report on recent work done by the Bureau of Rural Sciences on the potential impacts of plantations on stream flow and discusses the implications for the forest industry and for water policy makers.

1. INTRODUCTION

There are many benefits that can be delivered by forest plantations and replanting trees has perhaps been seen unquestioningly by some as "a good thing". However, it is well established that forested catchments generally produce reduced runoff compared to those under pasture or other crops (Zhang, Dawes *et al.,* 1999). Greater water use by tree crops means that plantation development in a predominantly cleared catchment could reduce river flows and recharge to groundwater. Given that many of Australia's important catchments and waterways are stressed and overallocated (National Land and Water Resources Audit 2001; Vertessy *et al.,* 2003) it is important that forestry impact be considered as part of land changes that may affect water availability.

2. SMALL PROPORTION OF CATCHMENTS COVERED BY PLANTATIONS

There is a perception by some that plantations already, or will in the future, cover significant areas of catchments that supply irrigation areas. In reality, the proportion of large catchments covered by plantations is usually very small. National Plantation Inventory figures show that plantations are generally below 4% of the total catchment (Table 1). The highest proportion under plantations is 6% – in northern Tasmania (Keenan, *et al.,* 2004).

Annual crops, pastures and native grassland are the major land cover in the drier of these regions (Green Triangle 73% of total area and Western Australia, 57%). Native forest and woodland are the major cover in Tasmania (47%) and Queensland (48%). Annual crops, pastures and native grassland are correspondingly lower in the wetter regions but even the most heavily forested region (south-east Queensland) had 39% crops, pastures or grasses (Table 1).

A recent meeting of forest scientists and catchment hydrology experts agreed that:

"There is strong scientific evidence that the magnitude of catchment response is proportional to the percentage of the catchment planted. This relationship is less certain where only small proportions of catchments are planted. In catchments under 1,000ha, where less than 20 % is planted to forest plantations and there is no rainfall gradient within that area, it is

Table 1. Vegetation Cover for 5 Catchment Based Regions Covering Major Plantation Areas in Australia

Vegetation Type	**Region**				
	South-east Queensland	**Murray Valley**	**Green Triangle**	**South-west WA**	**Northern Tasmania**
Annual crop, pastures, native grasslands	39%	66%	73%	57%	32%
Native forest or woodland	48%	26%	15%	29%	47%
Other	10%	7%	8%	12%	15%
Plantations (%)	2%	1%	4%	3%	6%
(ha)	166,000	202,000	240,000	269,000	157,000
Total area of catchments (ha)	8,158,000	16,647,000	6,047,000	10,117,000	2,807,000

Source: Keenan *et al.,* (2004). Note these regions include over 1 million ha of plantations which is about 2/3 of Australia's total.

difficult to measure a statistically significant effect on catchment yield. In larger catchments, this proportional relationship breaks down for a number of reasons, particularly the variation in annual rainfall across the catchment." (Anon, 2003).

3. WHOLE OF LANDSCAPE APPROACH TO WATER USE ISSUES

We need to consider a whole of landscape approach that includes all land use changes that may affect water yield. Plantations should be seen as part of bigger land use changes and all considered for their water impacts. A detailed analysis of land use change is beyond the scope of this article but a brief national overview is provided and one example from existing work indicates that change in agricultural land use areas and practices needs to be considered along with the changes in forest plantations.

Agriculture dominates land use in Australia. Approximately 485 million hectares (63%) of Australia is used for agriculture (National Land and Water Resources Audit 2001) compared with about 1.6 million hectares (0.2%) for plantation forests (NFI 2003). The vast majority (91%) of agricultural land is minimally modified native pasture and a further 8% is improved pastures and cropland. About 2 million hectares is irrigated (NLWRA 2001). There are major changes occurring in land uses in agricultural land use and the magnitude of these changes is far larger in area than those for forestry. There are substantial increases in areas of irrigated land (up 26%) from 1.6 m ha to 2.1 m ha, with increases in cotton, sugar cane and rice. In many regions perennial pastures have replaced annual grasses and in some areas grapes and dairying have increased significantly.

A study of the Green Triangle region (Petheram *et al.,* 2000) found that the use of freehold rural land in that region has changed markedly in the last decade. Cropping, dairying and timber production have all expanded at the expense of sheep and beef cattle enterprises (Figure 1). Sheep numbers declined from 10 to 8 million from 1990 to 2000, while dairy cow numbers increased from 307,000 to 424,000, cropped area from 62,000 to 205,000 ha and blue gum plantations from 0 to 40,000 ha.

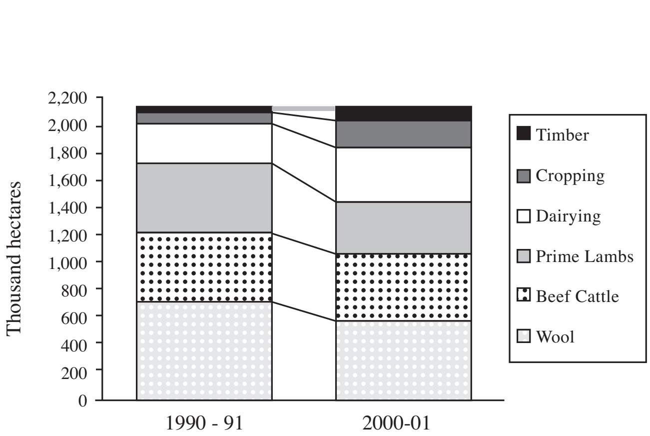

Figure 1. Change in land use across S.W. Victoria

Source: Petherham *et al.,* (2000)

These land use changes have potential to significantly affect the water runoff and recharge to groundwater. The impacts of increases in cropping and dairying at the expense of dryland sheep farming may be far bigger than the relatively modest increases in plantation forestry. More work needs to be done to confirm this but it would appear likely if the crops or dairying are irrigated where the water use may be 7 M L/ha (NLWRA, 2001) which is over double the amount used by a pine plantation (estimated to be between 2 and 3 M L/ha between 800 – 1,000 mm rainfall (Vertessy *et al.,* 2003).

The trend appears to be towards more intensification of agricultural practices (NLWRA 2001) and many of these have higher water use than the previous land use. Recognising that many of these are high value crops per unit area we need to develop methods that allow the trade-offs and comparisons to be made in socio-economic terms to determine policy options aiming at arriving at a socially optimum balance. Preliminary data for the Green Triangle region indicate that agriculture uses ten times as much land in the 600-1,000 mm rainfall zone as the forestry and forest products industries but produces only 1.3 times as much economic output (Keenan *et al.,* 2004). This is likely to be due to the regional manufacturing industries that typically accompany mature plantations. More research is required to assess the extent to which this is the case in other regions.

4. PLANTATION EXPANSION MOVING TOWARDS DRIER AREAS

Trees generally grow better in moderate to higher rainfall areas and, other things being equal, plantations in those situations are thus more profitable. Plantations are generally in upper catchment areas because these areas are less suited to intensive agriculture, which historically has been the priority land use in most landscapes. The result is that plantations often occupy higher rainfall land between the cleared agricultural lands and native forest on upper slopes.

While industry and governments have set a national goal of 3 million hectares of plantations by 2020 Commonwealth of Australia (1997), the distribution of these new plantings is uncertain. Most plantations established in the last 10 years have been outside the areas used to supply major irrigation areas (Wood *et al.,* 2001). Plantation expansion projections suggest that a trebling of the current plantation area in higher rainfall areas of the Murray Darling Basin is unlikely (Ferguson *et al.,* 2002). New plantations in major irrigation catchments are more likely to be in lower rainfall zones and therefore will have less impact on water yield than existing plantations.

An analysis in the Murray Darling Basin (MDB) shows that 71% of the areas planted to the end of 2000 are located in the 800-1,000 mm/yr rainfall zone and 93% are in areas receiving more than 800 mm/yr (Gerrand 2004). Most remaining higher rainfall areas are now conservation reserves or are covered with native forests and are not available for conversion to plantations. Consequently, plantation expansion in the MDB is most likely to occur in lower rainfall zones than areas planted in the past. Different species or management regimes are likely to be used in these regions and payments for environmental services, such as salinity mitigation, may compensate for lower commercial returns from wood production.

5. SELECTING WHERE TO PLANT TO BALANCE THE BENEFITS AND IMPACTS

Vertessy *et al.,* (2003) present results from theoretical models that indicate planting 50% of a catchment from 'the bottom up' (i.e. starting near the streams) will have a much greater impact on streamflow than planting from the 'top-down' (i.e. planting upper slopes and ridges). This relationship is thought to work well for small catchments that do not have much lateral flow, but does not hold for wetter catchments or where lateral water flow is high (Anon 2003).

Stirzaker *et al.,* (2002) show that plantations established in contour-banded configurations may also use more water than the same area of plantations established in blocks or perpendicular to the contour. Their theoretical modelling indicates that the same area of trees planted in different arrangements can have quite different growth and water use impacts. Their modelling indicates that if the same areas of trees had been planted as four narrow belts across the contour the trees would have had access to, and used, more water and suffered less drought death.

While more research is required to improve our ability to predict runoff and streamflow changes due to afforestation, there does appear to be good potential for reducing the impacts on water yield by the targeted location and design of plantations. If we choose those catchments that are producing salt then we may want to maximize water use. This would mean planting close to drainage lines and on lower slopes. Planting away from drainage lines and upslope is an option for those catchments contributing freshwater where the objective is to minimize impact on streamflows. It is likely to be easier to target parts of the landscape for afforestation under an integrated farm forestry approach, rather than broad acre or whole-farm plantings. This is because farm forestry provides some flexibility to choose specific sites within a farm and keep other areas under grassland or cropping.

6. CONCLUSIONS

Caution is required in drawing general conclusions about the impact of plantations on stream flow, as impacts from a similar activity may vary from one region to the next and even locally within a catchment. Ideally, water allocation policies should take into account regional variability, be based on the best available knowledge and consider the full range of environmental, economic and social factors from plantations and other dryland agricultural land uses.

Changing land use and the resulting impacts on water yield are complex. These changes in land use vary across regions and catchments and agricultural land use change is often more significant in area than forestry. Market forces and social factors are driving these changes. We need to improve our ability to model the impacts of these changes. Simple solutions that target only one land use may be inequitable across different sectors. We also need to improve our understanding of the trade-offs involved and develop ways to assess the impacts and benefits in a "whole of landscape approach" to evaluate the best policy mix and outcomes for society as a whole. This will require a multidisciplinary approach combining skills from at least the forestry, hydrology, social sciences and economics professions.

We have moved on from an era where "any tree, anywhere was a good thing" (Alexandra and Campbell 2003) and we now need to be able to plan to put the right trees in the right place for the right reason.

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Fire Management - An Integral Part of Forest Management

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Summary: Fire management must form an integral part of forest management. It encompasses all those activities that involve the use of fire and the suppression of fires. These activities include the provision of access and firebreaks, communications, risk assessment and threat analysis, burning for fuel management, species regeneration and habitat management, wildfires suppression and rehabilitation after wildfires. Fire management is the legal responsibility of State agencies who manage government land and individuals or corporations who manage private land. For specific activities such as fire suppression and fuel management volunteer bushfire brigades have been formed throughout the rural and peri-urban areas and are coordinated by State rural and metropolitan fire authorities.

1. INTRODUCTION

Australian forests are dominated by eucalypts and cover 124.5 million hectares (National Forest Inventory 1997) in the north, the east coast, Tasmania and the southwest corner of Western Australia. These forest areas of Australia, composed of fire-dependent and fire-adapted species and ecosystems, have evolved in the presence of a fire regime driven originally by natural sources of fire ignition, by cultural practices of aboriginal people and most recently by the European influence over the last two hundred years. The forests are a source of raw material for forest industry, and a source of many tangible and intangible products and services, including recreational and cultural opportunities for all Australians. In recognition of these values, forest protection efforts commenced in the early 1900s, and have steadily developed to the point where Australian State public land management agencies are recognized among the world's leaders in fire management.

Forest fire management in Australia is the responsibility of the State and Territorial governments. Fire management on public lands (e.g. State forests, National parks, State parks, Crown lands, etc.) is the responsibility of the State agency charged with managing those areas. Fire suppression may be carried out by individual agencies or placed with one agency, e.g. in Victoria suppression an all State lands are carried out by the Fire Management Section of the Department of Sustainability and Environment. Fire management on private lands is carried out by volunteer bushfire brigades or industry brigades that are co-coordinated and supported by the State rural fire agencies. In recent years there has been an increase in the corporatisation of State-owned plantations and the fire management responsibility for these forests, along with new plantation forests established on private land, rests increasingly with the State rural fire authorities. This shift in fire responsibility has mainly occurred in South Australia and Victoria over the last three years.

Most of the States provide fire management directly as a government service, generally by the departments that manage lands, forests and other natural resources. Their fire management programs provide for varying levels of planning, fuel management (i.e. prescribed burning), detection, pre-suppression and suppression operations. The level and type of activity in each category vary with each agency's natural resource polices, protection priorities, financial resources and in particular the ecological and biogeographical conditions of the forest itself. Consistent with the statutory obligations and policies of public management agencies, their fire management objectives include:

- Protection of people from bushfire.
- Protection of buildings and facilities from bushfire.
- Prevention of bushfire burning onto neighbouring property.

- Conservation of natural and cultural values including:
  - Native plant and animal species, habitats and communities;
  - Soil and water resources;
  - Scenic and landscape values; and,
  - Aboriginal and European heritage values.

All agencies deliver an organised detection program. Fire towers are the most common detection system offering regular surveillance of high-value areas and community assets. The use of fixed wing aircraft for detection has increased in the past 15 years. There are recent attempts to use satellite based remote sensing as a tool for fire detection (ref Sentinel and Savannah CRC system).

Suppression strategies use a mix of resources from the land management agencies with support from rural bushfire authorities. Ground crews using fire appliances (fire tankers), heavy equipment (dozers), and hand tools are the backbone of the suppression system. Aircraft for aerial suppression has been used in Victoria for more than thirty years, and over the past decade other forest agencies have increasingly used air attack on forest fires.

Different suppression strategies are used by the agencies, which are based on the nature of the forest and fire regimes that they deal with and, to some extent, on the organisational philosophy. Some agencies, such as those in Victoria and Western Australia, have relatively large full-time fire management organisations compared to those in other States.

All the State fire management agencies face a similar array of challenges in meeting their fire management objectives and the task is becoming increasingly difficult. As a government service, fire management has traditionally been combined with other forest management skills, notably sustainable timber production. Financial pressures and changes in policy relating to timber production from native forests are resulting in staff reductions and erosion of traditional levels of the fire management skills base and resources. Resources are declining at a time when demands for protection by the general community are increasing. Concurrently, the demands for ecologically appropriate forest management practices and concerns about the long-term impacts of prescribed burning practices have led to the suggestion that, in some areas, fire is adversely affecting biodiversity and long-term sustainability of forest ecosystems. These issues are overlain by debate about how fire can affect climate change, greenhouse gas balance at the landscape and national level, as to whether those changes are being exacerbated by managed and/or wildland fires.

This paper discusses the various aspects of fire management practices in Australia's public forest.

The fire management practices that may have an impact on forest management are:

- Prescribed burning
  - for fuel reduction in native forests and conifer plantations
  - for habitat management (ecological burning)
  - for native forest regeneration
- Fire suppression

Each of these operations is carried out under different conditions that result in different levels of fuel consumption and fire intensity so in addition to knowing the area burnt it is also necessary to know:

- The amount of fuel potentially available;
- The proportion of potentially available fuel burnt; and,
- The fuel dynamics before and after the fire events.

2. FIRE MANAGEMENT PROGRAM

A framework for fire management can be grouped into five components list below:

i. Hazards and Risk Assessment (analysis of the fire problem),

ii. Prevention (of fires),

iii. Preparedness (for Response),

iv. Suppression (controlling bushfires) and,

v. Recovery (of fire damage).

i. Hazard and Risk Assessment

Hazard can be defined as "a state that may result in an undesired event, the cause of risk" (Sampson *et al.* 2000). The level of hazard will increase if changes to the arrangement of vegetation and fuels contribute to bushfires that are larger, more intense, and more severe in their impacts than the majority of historical fire events. In fire management, hazard is a fuel complex defined by the type, arrangement, volume, condition and location that form a special threat of ignition or suppression difficulty.

Risk is the chance of a fire starting as determined by the presence and activity of causative agents. A risk is something that adds to the hazard. It is generally expressed as a probability of ignition and of weather conditions conducive to fire spread. Fires need an ignition source, and the weather conditions at the moment of ignition and some time thereafter have a critical influence on fire behaviour. Conditions suitable for ignition and fire spread can be characterised by the probability of their mutual occurrence, based on historical records.

Risk must be assessed in terms of the potential consequences to environmental, economic and social values. Bushfire risk must incorporate the complex interactions between the dynamic fire environment, the complex ecological environment and the complex and politically sensitive social and economic environments.

ii. Prevention

Fire prevention seeks to eliminate unplanned ignition, i.e. the accidental fires. By their very nature, such events cannot be removed totally. There are two main aspects of fire prevention. The first is the knowledge of fire causes and occurrence. The second is prevention action based on this knowledge. Since humans are major contributors to the number of unplanned ignitions, the latter includes selection of the best approach or approaches, to influence people and use of effective media in reaching them. In its application, fire prevention is fundamentally a problem of regulation, education and persuading people act responsibly with fire.

iii. Preparedness

Preparedness includes the tasks, monitoring and preparation needed to make suppression most efficient and rapid. It includes mainly infrastructure and engineering such as building and maintaining of roads, tracks, firebreaks, fire towers, making assets and homes ready in case of fire. Also included is the ongoing monitoring of fuels, weather conditions and ignition sources.

iv. Suppression

The main objectives of fire suppression are first to stop the fire from spreading and causing more damage, and secondly to make it absolutely safe. Contrary to common belief, fires are rarely ‘put out’ at first, but most often ‘secured’, the usual procedure is:

1. Knocking down blazing fronts and control of dangerous trees and vegetation likely to cause spot fires.
2. Cutting off the extreme outer edges of the fire from access to new fuel by building a fire line or extinguishing them
3. Extinguishing major fires in the interior of the fire perimeter
4. Mopping-up persistent fires along the outer edges of the fire perimeter

v. Recovery

Rebuilding of assets and rehabilitation of affected ecosystems are included in the recovery phase. Recovery can be particularly important to prevent future fires, as burnt areas may be more prone to fire in the years following a fire from increased fuel and debris in the forest and invasive species providing altered fuel or fuel that more readily ignites and burns.

An adaptive fire management plan cycle informed by sound scientific knowledge and well disseminated information provides the more appropriate framework for decision making and implementing a fire management program. The plan should focus on fire management and mitigation as part of a continuous cycle of activities rather than a disjunct of events, with particular attention on responding to and recovering and learning from serious fire events. This approach is illustrated in Figure 1. The key requirements are:

- Effective stakeholder involvement;
- Measurable targets;
- Commitment to monitoring and evaluation, and;
- Transparency, public reporting.

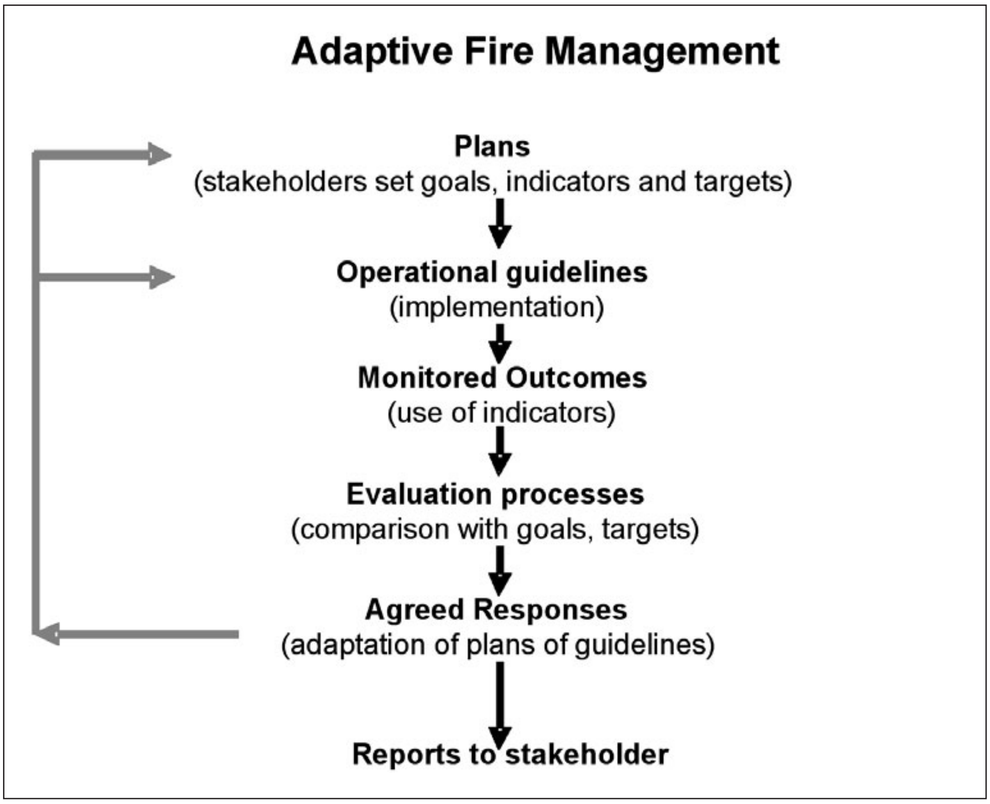

Figure 1. Cycling theme for an adaptive fire management program.

3. FUEL MANAGEMENT STRATEGY

The damage caused by wildfires and the ability of suppression forces to control them is strongly linked to fire intensity, which is governed by fuel, weather and topography. Of these factors, only the fuel level can be manipulated, and fuel management is the basis of wildfire prevention throughout much of Australia. In the natural landscape, this requires the periodic removal of part of the surface litter and understorey vegetation. This can be achieved by manual, mechanical, or chemical methods or by the use of fire.

Prescribed burning is defined as burning under specified environmental conditions and within a predetermined area to achieve some predetermined objective. The objective may include habitat management for native fauna, species regeneration, and maintenance of specific eco-types or hazard reduction, etc.

Studies conducted by McArthur (1962), Peet (1965), and others since the 1960s (Cheney *et al.,* 1992) have provided the technology for fire to be used effectively to manage fuels. These studies enable the behaviour of fires that are lit under given conditions to be predicted. A range of operational procedures provides a high level of security against fire escape. Due to the improvements in techniques and the application of fire behaviour knowledge, prescribed burning has become a reliable fuel management tool. To date the only effective way of reducing fuels over large areas is by low-intensity prescribed fires and, in Australia, this is generally synonymous with broad-area fuel reduction. In most of the eucalypt forests the aim of fuel-reduction programs is to keep the load of fine fuel (fuels less than 6 mm in diameter) on the forest floor to < 10 t ha^{-1}. This will prevent the development of crown fires in medium to tall forests and will limit the rate of spread and damage done by wildfires. The frequency of burning is determined by litter accumulation rates so that burning rotations are normally between 5 and 10 years.

Prescribed fire is also used in native forests to remove slash accumulations and to prepare a seed bed for the regeneration of native forest species and more recently to regenerate understorey species and manipulate vegetation to provide suitable habitat for native fauna. Although these operations also remove fuels, they are generally of higher intensity than low-intensity prescribed burning specifically for fuel reduction and the intensity prescribed is determined by the requirements for good regeneration.

i. Hazard reduction burning

Hazard reduction burning will reduce the total load of fine fuel and is also effective in reducing the height and flammability of elevated fine fuels such as shrubs and suspended dead material. Burning is the only practical way of reducing the fibrous bark on trees, which is the prime source of firebrands that causes spotting. Hazard reduction reduces fire behaviour by:

- reducing the speed of growth of the fire from its ignition point;
- reducing the height of flames and rate of spread;
- reducing the spotting potential by reducing the number of firebrands and the distance they are carried downwind; and,
- reducing the total heat output or intensity of the fire.

Prescribed burning is not intended to stop forest fires but it does reduce their intensity and this makes fire suppression safer and more efficient. Prescribed burning does not provide a panacea, nor does it work in isolation. It must be used in conjunction with an efficient fire fighting force.

Hand crews can suppress a fire up to a maximum intensity of 1000 kilowatts per m. If the fuel load is > 15 t/ha (which is typical of dry eucalypt forests between 8 - 15 years since the last fire) this intensity will be exceeded under low to moderate fire danger conditions. If the fuels are reduced to 10 t/ha fires will not develop an intensity of 1000 kW/m until fire danger gets into the moderate to high range. This means that the range of weather conditions that fire fighting with hand tools is effective, is increased and more time is available to bring the fire under control. If the fuels are reduced further to a less than 7.5 t/ha then suppression with hand tools is effective under weather conditions of very high fire danger. Under extreme conditions, provided there is sufficient fuel to carry fire, fire suppression by any means is virtually impossible because the strong dry winds associated with conditions will cause burning embers to breach any fire line. Nevertheless the result of the light fuel load will reduce the rate of spread of the fire and the area burnt so that the fire suppression task will be easier when the weather conditions ameliorate.

ii. Silvicultural burning

Silvicultural burning is usually a moderate-intensity prescribed burn carried out after a partial-cut logging operation designed to remove logging slash, prepare the seed bed and stimulate regeneration and/or the growth of rootstock regeneration. Silvicultural burning is conducted in the jarrah forest of Western Australia and the silvertop ash forests of NSW.

iii. Ecological burning

The main aim of using fire for ecological management is to provide an appropriate fire regime (of specific fire frequency, intensity, seasonality and patchiness) to meet specific goals for the management of a specific species, populations or communities (e.g. as part of a recovery plan for a threatened species). Since fire has a fundamental role in the development of forest ecosystems, it follows that fire has a place in maintaining them. Good (1981) indicated that because fire is the major and only environmental factor over which some control can be exercised, and many native species depend on fire for their continued existence, and the use of fire will always have a place in ecological management. Fire has a place in both flora and fauna management but its effective application has been infrequent.

4. APPLICATION OF PRESCRIBED BURNING

There is a perception among people unfamiliar with forest fire management that prescribed burning is simply lighting fires to burn-off the undergrowth and that this can be carried out with only a basic understanding of fire behaviour. Indeed, where burning-off has been carried out in this way the results have been less than optimal and has resulted in escapes, injury and/or death (e.g. Kur-Ring Gai National Park 2000). Like any land management operation, prescribed burning requires setting clear priorities and objectives, planning and the application of technical guidelines to meet those objectives. In general terms of the process of conducting a prescribed burn is as follows:

- Set the objectives and desired outcome for the fire.
- Determine the fire intensity and the associated heat pulse that is required to meet that objective (in forestry and for fuel management this may be determined by an acceptable height of scorch on the overstorey canopy or an acceptable level of heat damage to the cambium of regenerating trees).
- Determine the level of fire behaviour that will produce this heat pulse for the particular fuel type.
- Determine the weather conditions and the ignition pattern that will produce this fire behaviour.
- Light the fire in a planned way and confine it to a predetermined area.

The key to conducting the operation is a good fire behaviour guide that predicts fire behaviour in the selected fuel type. In Western Australia, the Department of Conservation and Land Management has been conducting prescribed burning to meet fire protection, forestry and ecological objectives in a scientific way since mid-60s. The planning process starts seven years in advance of each prescribed burn. Individual burning guides have been developed through empirical research for all their major fuel types including dry jarrah forest, tall wet to Karri forest, conifer plantations and Mallee shrublands.

In the eastern states prescribed burning is largely carried out using rules of thumb based on a MacArthur's original burning guide for dry eucalypt forests produced in the 1960s (MacArthur 1962). Only one specific new burning guide has been developed and that was for burning under young regeneration of silver top ash in New South Wales State Forests (Cheney *et al.,* 1992). Clearly, if prescribed burning is to be conducted in a more professional way, there is an urgent need for new and better burning guides that can be applied to a whole range of different fuel types.

5. ADVANCES IN FUEL MANAGEMENT

The development of more sophisticated burning guides requires a better understanding of fire behaviour in fuels of different structure and composition. Recent work by CSIRO (Project Vesta (Cheney *et al.,* 1998, Gould *et al.,* 2001, McCaw *et al.,* 2003) -- work in progress) has identified the importance of fuel structure in determining fire behaviour and has developed a system for quantifying fuel structure with a numerical index that can be used as a fuel predictor variable to replace fuel load.

Although fuel structure is difficult, if not impossible, to measure reliably and consistently, all natural fuels can be divided to easily recognisable layers. It is the characteristics of these layers that determine the particular fuel type and its characteristic fire behaviour and the difficulty of suppression.

For example, the simplest fuel type is annual grassland like wheat. This is a single layer of relatively uniform compaction. The main factor that determines rate of spread is the continuity of the grass. Although the height of the sward affects the flame height and thereby the suppression difficulty, it has only a minor effect on the rate of spread.

A dry eucalypt forest with a tall shrub understorey has fuel that can be identified into several layers of different compaction. These are in order of decreasing compaction:

- Compacted surface litter bed of leaves twigs and bark that makes up about 60 percent of the total fuel load,
- Near surface layer above it of the low shrubs containing suspended litter and bark,
- Elevated layer of tall shrubs,
- Intermediate layer of small trees,
- Fibrous bark of the over story trees, and
- Canopy of the over story trees.

All of these layers make an important contribution to the fire behaviour and each layer becomes progressively involved in fire as the intensity increases. The hazard rating system is being developed (Gould *et al.,* 2001) takes into account the height continuity and a fraction of dead flammable material in each layer. The most forests the delay of that appears to be most important in determining fire spread is the near surface fuel layer and the best fuel variable for predicting the rate of spread is an index based on the hazard score and height of the near surface fuel layer (Project Vesta - work in progress).

Table 1. Period that fuel reduction burning will assist suppression activities and the main factors that contributing to difficulty of suppression.

Fuel type	Persistence of reduced fire behaviour (years)	Factors contributing to difficulty of suppression
Annual grass	1 (year of burning)	
Tussock grassland	5	Development of persistent tussock fuel
Tall shrubland	10 - 15	Height of shrubs accumulation of dead material (ROS, flame height)
Forest, short shrubs, gum bark	10 - 15	Surface fuel, near-surface fuels structure (ROS flame height)
Forest, tall shrubs, stringybark	15 - 25	Near- surface fuel, shrub height and senescence, bark accumulation (ROS, flame height, spotting potential)

6. EFFECTIVENESS OF FUEL REDUCTION OVER TIME

The period that fuel reduction remains effective to assist suppression depends upon the number of fuel layers involved, the rate of accumulation of fuels and the time that it takes for the key layer to build up to its full potential hazard for the site. This may be a relatively short time for fuels with a simple structure or take many years in more complex fuel types - see Table 1.

Although prescribed burning may persist for a considerable time most fire management agencies consider that sufficient fuels have accumulated after 5-8 years to warrant re-burning.

7. SUPPRESSION STRATEGY

The fundamental requirement for extinguishing a forest fire is that the fire must be totally surrounded by a trail that completely separates the fire from unburnt fuel and is constructed down to the mineral earth. The entire fine fuel within the control line must be burnt out and smouldering logs and trees within 50 m of the line extinguished. This basic requirement has not changed since man successfully started suppressing forest fires. It is labour-intensive and any failure to fulfil this requirement will mean that the fire will escape from control if very high to extreme weather subsequently occurs.

Water is very effective in suppressing the flames but a waterline cannot be relied upon, because smouldering fuels will reignite as the water dries out. Fires that are suppressed by water, be it from ground or air tankers, must be followed up with a bare-earth fire line around the fire and mopped up within the fire perimeter to ensure that embers will not be blown out and breach the fire line.

Because fire suppression can only be successful at relatively low fire intensity and under relatively mild weather, it is essential that fire fighters attempt to suppress the fire when it is small before it has reached its potential intensity for the prevailing conditions and take advantage of periods of mild weather when the fire line intensity is low. The fire intensity is normally low during night-time conditions and fire management agencies must train and equip their fire fighters to carry out fire fighting at night.

The most effective fire fighting will be enhanced if there is good access and preparedness to ensure that fire fighters get to the fire without delay. Fire trails should be maintained and cleared prior to the fire season so that valuable time is not lost opening up overgrown or abandoned roads and repairing unsafe bridges.

The primary objective of an initial attack is to control fire at minimum size in minimum time. If this is not possible because of the remote location of the fire or the nature of the terrain then the primary objective must be to control fire in minimum time before the next period of dangerous fire weather occurs. This decision will determine whether the fire fighting strategy is direct or indirect.

i. Direct attack

This is primarily deployed during an initial attack when the objective is to contain the fire in minimum size. The flames are either suppressed directly or a fire line is constructed parallel and close to the fire edge. Different techniques of direct fire fighting are:

Fire line construction with hand tools: a bare earth fire line between 0.5 and 1.0 m wide is constructed with a combination of hand tools (generally an axe or slasher, several McLeod tools and chainsaw) around fires of low intensity less than 1000 kW/m.

Ground tankers and hose-lay: water applied directly to extinguish the flames is probably the fastest and safest way of containing a forest fire. Water can be pumped to the fire edge over more than 1000 m using multiple sections of hose and relay pumping with portable pumps. Direct suppression with water can control fires up to 2000 kW / m and is most effective in flashy shrubland fuels when fire-fighters work systematically from the base of the fire and extinguish the flames from inside the burnt area. Often fires in this type of fuel are too dangerous to attack by any other tactic.

Bulldozers: once the fire is more than a few hectares in area there is really no alternative but to set about systematic fire line construction using bulldozers. Rates of fire line construction can exceed 2000 m per hour although the use of bulldozers is limited when slopes exceed 25 degrees. In moderate or undulating terrain bulldozers can be supported by ground tankers for an immediate mop-up and vehicular patrol.

Aircraft: Small to medium helicopters are no more effective in suppressing fires than crews with hand tools. Helicopters have the advantage that they can be deployed rapidly to fires while they are small, but they have the disadvantage that the attack is not systematic or continuous. Even when multiple helicopters were used, the intermittent nature of their attack meant that sections of the fire that had been partially extinguished by the drops re-lit or burnt around the area of dampened fuel.

Helicopters are most effective when supporting ground crews who can work in a continuous manner to systematically surround fire. Helicopters can be used to reduce the intensity of the fire, but more importantly they can detect and immediately suppress spot fires as soon as they occur beyond the fire line.

A wide range of fixed-wing and aircraft helicopters are available for aerial suppression delivering both chemical suppressant and/or water. The effectiveness of the retardant application depends on:

- Amount of retardant actually need on the critical fuel,
- Interception of retardant by the forest canopy above the critical fuel,
- Pattern of the retardant drop, and
- Chemical characteristics of the retardant drop reaching the fuel.

ii. Indirect attack

Indirect fire fighting involves constructing a fire line, or using an established road as the control line, located at some distance from fire. The same principles that apply to direct fire fighting apply to indirect fire fighting: the fire must be surrounded by a bare-earth trail or road and the fuels between the trail and fire burnt out and mopped up before the next period of dangerous fire weather re-occurs.

Close indirect fire fighting: this is when the fire line is constructed relatively close to the fire primarily taking the easiest route to construct the fire line quickly. This is the most dangerous form of fire fighting particularly for inexperienced fire-fighters. Because the line is relatively close to the fire edge and there is no immediate access to the safety of the ground fire-fighters are vulnerable to sudden changes of wind direction that can escalate the speed and intensity of the fire.

Remote indirect fire fighting: this tactic is employed when the fire is burning in inaccessible terrain and to bring it under control in minimum time fire-fighters need to fall back to use established roads and fire trails as control lines and immediately commit a much larger area to fire. In recent years this technique appears to have been adopted more frequently in National Parks to avoid using bulldozers within the park.

The tactic is appropriate if it can be accomplished speedily and the entire area burnt out before the onset of severe weather. However there are disadvantages. In some cases the area committed can mean a twenty-fold increase in the planned final fire size, say from 500 hectares to 10,000 hectares. This means that a large amount of manpower and resources are required to secure the perimeter roads and skill to burn-out inside the control lines, usually with aerial incendiaries, without increasing the fire intensity to a point where the fire throws spot fires beyond the control line.

The tactic may have attraction because of the availability of a large number of tanker-based volunteers, but if they do not have the skill in forest fire behaviour they may not be able to secure the control line in sufficient time to allow burning out to proceed before the weather deteriorates.

8. CONCLUSION

Suppression of wildfire is a forestry management practice. The impact of fuel reduction burning on wildfire suppression, including the training, safety and well-being of firefighters is difficult to quantify but has contributed to the decline in the area burnt by wildfires of land managed by forestry agencies. The author believes that, on a national scale, proactive fuel management can still be successful in decreasing area burned by wildfire in Australia.

The impact of fire on forest management still depends on having better estimates of post and pre-fire data on forest resources and impact of fire through a combination of field measurements and modelling. Accurate interpretation of the effect of fire management practices on forest management requires not only accurate measurement of area burnt, but also the classification of all fires by vegetation type and burning conditions, the measurement of the fuel dynamics and equilibrium fuel loads for each type and the measurement of consumption rates under a wider range of burning conditions than is currently available.

Fire is a dominant vegetation-renewing agent in most of the eucalyptus forests and ecologically these forests need fire. A changing climate with increased carbon dioxide concentrations in the atmosphere is likely to affect both fire weather and vegetation (fuel) types. However, feedbacks have not been estimated, so that it is difficult to predict whether interaction between weather, fuels and land management practices including fire exclusion will change fire regimes drastically. Also, fuel management using prescribed fire has an important role in protection of forests, community assets, other valued resources and biodiversity. Australian forest landscapes are becoming increasingly more fragmented because of human activities; this also has an impact on the fire management practices that could contribute more to the amount of area burnt by wildfires.

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Early Steps in Assessing the Water and Stream Salinity Outcomes of Commercial and Environmental Forestry

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Summary: Commercial and Environmental Forestry is commercial forestry that delivers net positive environmental outcomes. These environmental outcomes include carbon sequestration, reduced stream flow, changed stream salinity, reduced soil erosion and associated improvement in water quality. New plantation development on mainland Australia will largely occur on land currently used for grazing. Net positive environmental benefits of these plantations are more likely to be achieved in medium rainfall zones (mean annual rainfall 600 to 800 millimetres) rather than high rainfall zones. The results of preliminary analysis suggest that not all catchments will be suitable for this style of forestry. Also, the analysis suggests that it is important to provide a high level of spatial resolution to the specification of environmental service payment, or incentives, if stream salinity benefits are to be realised.

1. INTRODUCTION

Most of Australia's plantation forestry resources are in high rainfall zones (Gerrand, 2003). Recently there have been proposals to expand plantations into medium rainfall zones (mean annual rainfall 600 to 800 mm) (Burns *et al.*, 1999). CSIRO's Commercial and Environmental Forestry program aims to "Establish a framework that enables private investors and farmers to optimise commercial outcomes in low-to-medium rainfall forestry, while delivering long term environmental benefits" (Anon., 2003).

2. MOVING BEYOND PLANTATIONS VERSUS WATER

It is the universal conclusion of international studies of catchment water balances, that for the same climate and land, forested land produces less stream flow than grazed or cropped land (Zhang *et al.*, 2001). Australian studies for plantations in moderate rainfall zones, including McJannet *et al.,* (2000) and Stolte *et al.* , (1997), support this conclusion. Recently Vertessy *et al.,* (2003) reviewed the available literature and concluded that reduced water yield can manifest itself as reduced low flows, reduced water security for downstream water use as well as reduced mean annual flow.

This finding has resulted in extended discussion of the environmental benefits in establishing new plantations on lands currently vegetated with either pastures or crops. Foresters have argued that this land was vegetated with forests, woodland or native grasses prior to European clearing, so they are simply returning the type of vegetation towards what is natural. Water resource managers have argued that the current allocations of water to users, both consumptive and non-consumptive, require runoff rates as per the current vegetation cover.

The Commercial and Environmental Forestry Program seeks to provide a technical basis to a broader consideration of the outcomes of new plantations in medium rainfall zones. It is likely that net environmental benefits may be achieved by establishing new plantations in parts of this zone. Only through applying our understanding of the multiple outcomes can such an analysis be completed.

3. ENVIRONMENTAL OUTCOMES SOUGHT

As well as enhancing rural economies, establishing new plantations on land that is currently grazed will have a range of environmental outcomes. These include carbon sequestration, reduced soil erosion, reduced groundwater recharge – that may be associated with reduced dryland salinity, reduced stream flow, and changes to stream salinity.

Changes in stream salinity as a result of afforestation can be either positive or negative (Vertessy *et al.,* 2003). Replacing pastures or crops with plantations results in two changes that are important to the change in stream salinity in creeks and rivers draining the catchment. First, the change in vegetation reduces the volume of the streamflow. This effect occurs relatively quickly and has its full effect when the canopy of the plantation is established (3 to 6 years after planting). Second, the establishment of the plantation reduces the recharge to the groundwater system. In most instances groundwater movement is the mechanism by which salt is transported to the stream. Thus a reduction in recharge also will reduce the rate at which salt is mobilized. As groundwater flow systems have relatively long lag time associated with the slow movement of water through the aquifers (ten years to thousands of years), the effect of reducing salt mobilisation on salt loads in the stream will be delayed by a significant period.

In some medium rainfall areas the effect of reducing salt mobilisation is greater than the effect of reducing stream flow, so that stream salinity (measured as salt concentration or EC units) is reduced. In other areas this is not the case and stream salinity increases. The outcome for stream salinity may vary greatly over time as a result of the differing time lags in the surface flow and groundwater flow. Vertessy *et al.* (2003) present a fuller description of this effect. Dowling *et al.* (in press) describe the factors that lead to these two outcomes for possible afforestation in the Murray Darling Basin.

4. SCREENING CATCHMENTS FOR FOCUSSING FURTHER INVESTIGATION

In the early phases of the CEF program, van Dijk *et al.* (2004a) examined the potential of six catchments to meet two screening criteria, one indicative of forest productivity goals and the other a key environmental goal. These criteria were: to have a predicted mean annual growth increment of greater than 15 cubic meters per year for a model describing the spatial distribution of growth of pinus radiata (using the forest productivity model PROMOD - Battaglia and Sands, 1997) and second, to result in a reduced stream salinity 15 years after planting (using the catchment salinity model BC2C – Dawes and Gilfedder, 2004).

The six catchment examined were Bet Bet Creek (Vic), Billabong (NSW), SW Goulburn (Vic), Kyeamba/ Tarcutta (NSW), Upper Little River (NSW) and Booroowa/Jugiong (NSW). The analysis found that both of the screening criteria were only present for significant areas within the South West Goulburn catchment. This finding does not exclude the use of plantations for either environmental or productivity purposes in the remaining catchments. It simply suggests that a single plantation within these catchments will not meet both of these goals.

5. MODELLING PLANTATION UPTAKE

The project has also conducted an analysis of the hydrologic consequences of different spatial patterns of establishing new plantations within the South West Goulburn. This work is reported in detail in van Dijk *et al.,* (2004b).

The spatial models of catchment salinity (BC2C) and forest productivity (PROMOD) were used to examine the outcomes of partial afforestation on total forest productivity, mean annual stream flow and changes to mean annual stream salinity at the catchment outlet. It was recognised that 100% afforestation of these areas that met both screening criteria (a subset of the SW Goulburn catchment) would not be achieved. Therefore, we examined the outcomes of different spatial patterns of uptake that may result from spatially-specific incentives. We found that if stream salinity benefits were used to rank areas to be planted then the improvement against random planting was large (25% afforestation gave approximately 60% of total benefits) and that the forest productivity and water yield outcomes did not change substantially from the random uptake case.

While this analysis is preliminary, it does suggest that maps of potential outcomes at the 1 km spatial resolution are useful in guiding where plantations should be established for these outcomes to be realised. Thus if a tender-based market for environmental services was established, the use of such maps of outcomes, would make the market a more effective means to distribute public funds.

6. CONCLUDING REMARKS

The CEF program is still in its early phases but the analyses performed so far suggest that assessing the net benefit of new plantations will be complex. The results to date suggest that not all catchments will be suitable for this style of forestry. Also the analysis suggests that it is important to enable a high level of spatial resolution to the specification of environmental service payment or incentives, if stream salinity benefits are to be realised.

Our analysis so far has considered three of several outcomes from establishing new plantations on lands currently vegetated with pastures: forest productivity, streamflow and changes to stream salinity. For an expanded analysis to be made more outcomes will need to be included and some method of valuing the outcomes introduced.

7. ACKNOWLEDGEMENTS

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Solutes in Stream Water as Montreal Process Indicators of Sustainable Management of Forested Catchments: Sampling Strategies to Evaluate a Change in Water Quality

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Summary: Australia has adopted the Montreal Process of criteria and indicators as a national framework for assessing the sustainable management of forests. Maintenance of soil and water resources is one of the criteria with several indicators including yield and quality of water from catchments as indicators of the environmental impacts of forest management. Impacts are evaluated against the historic range of variation in yield and water quality. Water quality data from forested catchments of the Cropper Creek Hydrology project in north-eastern Victoria were used to evaluate relationships between solute concentrations and stream flow and to compare fixed interval and peak flow event sampling conducted over a 6-year period. The range of variation and threshold values of solutes (N, P, S, Ca) in stream water were defined for an undisturbed native forest catchment. Threshold values for phosphorus were used in a case study to evaluate the long-term impact of treatment of a radiata pine catchment with phosphate fertiliser on P levels in stream water.

1. INTRODUCTION

In recent years southern Australia has seen a significant expansion of industrial plantations for wood production mainly on cleared agricultural land. Associated with this change in land use to intensively managed, fast-growing plantations are changes in hydrology and environmental conditions potentially affecting both water yield and water quality. Australia has adopted the Montreal Process of socio-economic and environmental criteria and indicators for the sustainable management of native forests and industrial plantations (DPIE 1998). The maintenance of water quality within the range of historic variation is one of the environmental indicators of sustainable management included in the Montreal Process. This process is consistent with the aims and objectives of Australia's national water quality management strategy for the protection of water resources (ANZECC, 2000).

The Cropper Creek Hydrology project established in 1976 and comprising three native forest catchments (Bren *et al.,* 1979) was resumed in 1997 to evaluate long-term changes in hydrology and water quality. A radiata pine plantation was established following clearing one of the native forest catchments in 1980. A 30 m wide strip of undisturbed native forest was retained as a buffer along the stream. In the short term this change in land use (Phase I, 1976-1982) resulted in only minor changes in water quality but export of suspended solids and nutrients in stream water increased because of higher water yield in response to clearing (Hopmans *et al.,* 1987). As part of the second phase (Phase II) of the project, monitoring of the radiata pine catchment (46 ha) and the adjacent undisturbed native forest catchment (113 ha) resumed to evaluate the effects of thinning and fertiliser treatment of radiata pine.

2. METHODS

Average annual rainfall at Cropper Creek is 1300 mm and details of the physiography of the catchments, climate, geology, soils, native vegetation, instrumentation of the weirs, and monitoring of stream flow and water quality are given by Bren *et al.,* (1979).

Stream gauging for Phase II of the project and collection of stream water for analysis commenced in May 1997. Samples were taken from a fixed location above the weirs at weekly intervals. Additional samples were collected at more frequent intervals (1 to 4 hours) during peak flow events using Sigma automatic water sample collectors. Sampling protocols were consistent with the Australian Guidelines for Water Quality Monitoring and Reporting (ANZECC, 2000)

In 1998 the plantation was thinned to a stocking of approximately 600 stems per hectare. Foliage diagnostic testing indicated that radiata pine was deficient in phosphorus and fertiliser (P:S:Ca, 18:9:14) was applied to the entire pine plantation a rate 570 kg/ha in April 1998. This was followed in September 2000 by an application of urea (46% N) at a rate of 670 kg/ha to 20 ha of the pine plantation. The 30-m wide buffer zone of native forest along the stream of the radiata pine catchment remained untreated.

Solute concentrations in stream water were measured after filtration including total N, P, S and Ca using standard procedures (APHA, 1995). Total N was determined colorimetrically (Cd reduction method) after oxidation of all nitrogenous compounds to nitrate using persulphate. Concentrations of total P, S and Ca in filtrate was determined by Inductively Coupled Plasma Atomic Emission Spectroscopy (APHA, 1995).

3. RESULTS AND DISCUSSION

Solute concentrations in stream water for the period May 1997 to October 2002 (300 weeks) were generally not normally distributed and consistently showed skewness towards lower concentrations (Figure 1). Therefore average concentrations of solutes are best expressed as the median values and sample variability as the median absolute deviation or MAD (ANZECC, 2000).

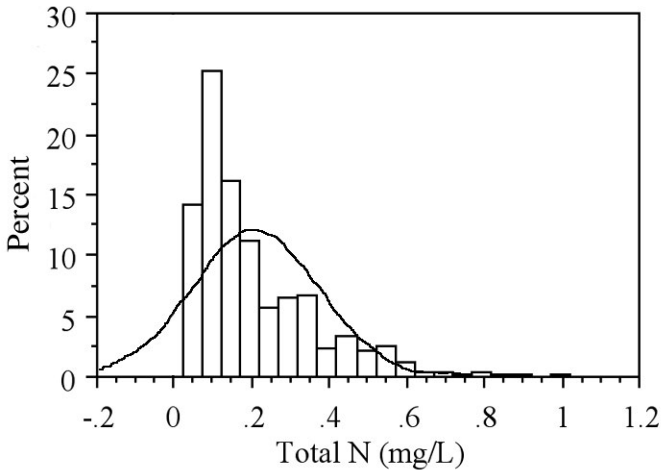

Figure 1. Frequency distribution of total N in stream water of the eucalypt forest catchment (solid line indicates normal distribution).

Concentrations of total N varied according to flow conditions and showed distinct clockwise concave hysteresis across the hydrograph (Evans and Davies 1998) of a major peak flow event in the radiata pine catchment in September 1998 (Figure 2). During this event levels of N increased rapidly to around 0.7 mg/L followed by a decline to 0.1 mg/L. In contrast, concentrations of total P, S and Ca in stream water were relatively uniform across a wide range of flow conditions (Table 1).

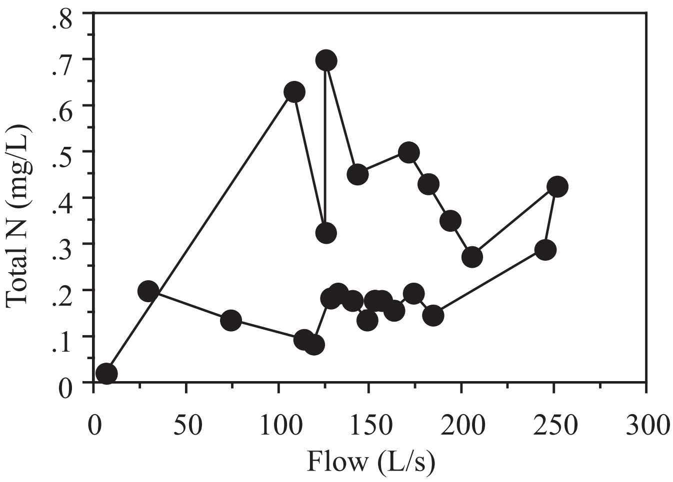

Figure 2. Hysteresis of N concentrations in stream water of the radiata pine catchment for a peak flow hydrograph in September 1998.

This has significant implications for sampling strategies aimed at determining changes in solute concentrations in stream water in response to forest management practices. Monitoring of solutes showing significant hysteresis needs to be linked to flow conditions, while sampling at fixed time intervals would be appropriate for solutes not affected by the rate of discharge.

Table 1. Levels of N, P, S, and Ca in stream water of the eucalypt forest catchment (untreated) based on (A) fixed interval (weekly) and (B) event sampling (five major peak flow events) from May 1997 to October 2002.

Solute	Sample	Median (mg/L)	MAD
Nitrogen	A / B	0.09 / 0.25	0.04 / 0.13
Phosphorus	A / B	0.005 / 0.005	0.003 / 0.004
Sulphur	A / B	0.10 / 0.11	0.01 / 0.01
Calcium	A / B	0.44 / 0.55	0.06 / 0.05

Discharge in these steep forested catchments varied enormously with maximum peak flows of around 250 L/s and 380 L/s recorded at the radiata pine and eucalypt catchments respectively. Discharge has been continuous in the pine catchment, in contrast flow often ceased over summer and resumed during autumn in the eucalypt catchment. The two sampling regimes (weekly interval and peak flow events) used at Cropper Creek essentially covered two ranges of flow conditions (Table 2). During 300 weeks of monitoring (May 1997 to October 2002), peak flow conditions prevailed for a total of 16 weeks. No discharge was observed for about 70 weeks in the eucalypt catchment.

Table 2. Average flows (L/s) and range for weekly (A) and peak flow event (B) sampling regimes from May 1997 to April 2003 (n, number of samples).

Catchment	Mean	Median	Range	n
Pine – A	6.0	1.6	0.1 - 96	284
Pine – B	78	65	31 - 251	106
Euc – A	11.3	2.8	0.1 - 194	203
Euc – B	103	74	14 – 382	118

The Montreal Process indicator for water quality relies on being able to measure significant deviations in water quality parameters from the historic range of variation (DPIE, 1998). Threshold values of 20th and 80th percentiles (lower and upper values) of the median, or one standard deviation from the mean for normally distributed data, have been recommended as low risk trigger levels for the protection of undisturbed aquatic ecosystems in Australia (ANZECC, 2000). It is also recommended that reference conditions or the historic range of variation be defined using data collected over 3 to 5 years at intervals of 4 weeks or less.

The range of variation in solute concentrations for the undisturbed eucalypt catchment adequately meets the requirements for reference data in order to evaluate impacts of forest management practices on water quality.

These data were also used to estimate the minimum number of samples required to measure a statistically significant change in solute concentrations at a confidence level of 95% and for a range of one median absolute deviation. This indicated that a minimum of 18 samples are required to detect a change in concentrations of the four solutes (N, P, S, Ca) in stream water with confidence. This equates to minimum monitoring periods of 18 weeks (weekly sampling) to detect any significant changes in water quality in response to treatment.

At Cropper Creek sequential monitoring periods of 26 weeks were used to determine changes in median P levels in stream water of the radiata pine catchment following treatment with phosphate fertiliser in April 1998 (Figure 3). Using the 80th percentile value of 0.008 mgP/L for the undisturbed the eucalypt catchment as the upper threshold level; median values of P exceeded this level only during the first 6-month period after treatment and again 2 years later. Median P values for the entire 4-year period after treatment were 0.005 mg/L with a median absolute deviation of 0.004 mg/L. These results show that the application of fertiliser to radiata pine has had little impact on P levels in stream water to date.

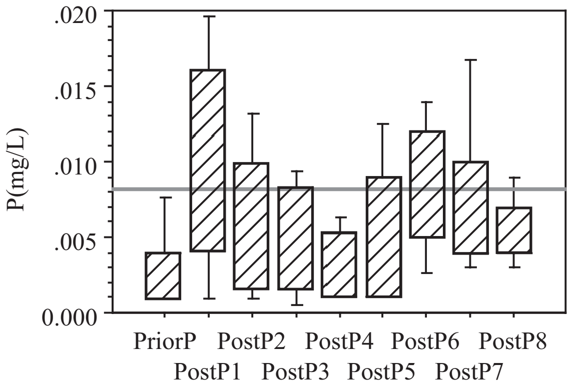

Figure 3. Distributions of total P in stream water (box plots indicating 10th, 25th, 50th(median), 75th, and 90th percentiles) at the radiata pine catchment prior to treatment with P fertiliser in April 1998 and for eight 6-month periods thereafter. The solid line indicates the 80th percentile threshold value.

The levels of P in stream water at the radiata pine catchment remained well below the low risk threshold value of 0.04 mg/L for slightly disturbed aquatic ecosystems adopted in the National Water Quality Management Strategy (ANZECC 2000). In New Zealand application of phosphate fertiliser to radiata pine at low rates (36 kg P/ha) and excluding stream channels increased P levels to a maximum of 0.08 mg/L but peak values of 1.7 mg/L were observed when the entire catchment was treated (Neary and Leonard, 1978).

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Critique of Forest Management and Implications for Water Yield. Case study: Thomson Catchment, Melbourne's Water Supply

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Summary: A critique, from a Non Government Organisation perspective, of the forest management issues arising from water yield reductions following timber harvesting in Victorian wet forests. Rotation length and timber royalty rates have a key role to play in balancing wood and water yields in the best long-term interest of the community. There are particularly important management issues in sensitive, small domestic supplies and systems with a shortage of supply. A comparison of management scenarios is undertaken in Melbourne's water supply catchments.

1. INTRODUCTION

An internationally respected research program spanning three decades, undertaken in the Victorian Central Highlands has shown that logging activities in wet forests can reduce water yield by 50%, 30 years after logging and are projected to take about 150 years to return to pre-logged levels (Kuczera 1985).

These findings have important implications for the management of timber harvesting in water supply catchments, yet a number of key issues identified by the science remain to be adequately addressed.

2. THE NEED TO ACT ON THE SCIENCE

Two decades ago, the science progressed beyond simply proving that water yield reductions occur following harvesting to identifying further key implications for forest management.

Key issues of importance to catchment management include: clearfelling wet forests will cause large reductions in long term streamflow (MMBW 1980); logging impacts are cumulative and proportional to the area of catchment logged (Kuczera 1985) therefore small catchments and catchments with a shortage of supply are particularly sensitive to logging (Moran 1988); intensive logging rotations in the 40-80 year range produce the largest yield reductions (O'Shaughnessy & Jayasuriya 1987); logging in higher rainfall areas has larger impact on yield (Moran 1988). More recent work has supported and built upon these findings.

There has been a lack of decisive action from politicians and land managers in responding to issues raised by the science. For example, small, sensitive catchments continue to be intensively harvested - such as the Steavenson River catchment supplying Marysville (SFRI 2000). Further issues are illustrated in detail in the following case study.

3. CASE STUDY: MELBOURNE'S WATER SUPPLY

Mountain forests (of the type researched in the Central Highlands program) to the north east of the city form Melbourne's water supply.

The Department of Sustainability and Environment, in consultation with Melbourne Water, manage logging operations in five of the city's catchments, including the Thomson, the city's largest catchment - and the Yarra Tributary catchments (Armstrong Creek, Cement Creek, McMahon's Creek and Starvation Creek). A sixth catchment, the Tarago - is available for harvesting but is currently disconnected from the water supply system (WRSCMA 2002a).

3.1 The Thomson Catchment

In 1983, after the completion of the reservoir – the management of the Thomson catchment became more important. Unfortunately, harvesting rates were increased in the early 1990s to an average of 150 hectares per year (NRE 1998, NRE 1998a). Clearfelling over the 1970s and 80s had occurred at a rate of less than 40 ha per year (RSA 1992). This harvesting increase occurred despite both clear science showing important water yield reductions (Kuczera 1985) and economics indicating the catchment was more valuable for water production than logs (RSA 1992). Increased harvesting was justified as necessary in meeting existing log licence commitments that would not be re-negotiated until the year 2002 (NRE 1998). No decision on harvest rates was made in 2002, and it is unclear when such a decision will be made. Log licences have been extended two years to 2004 as an interim measure (URS 2002).

The Victorian Government recently appointed the Water Resource Strategy Committee for the Melbourne Area (WRSCMA), to consider measures to increase water supply security. The committee recommended that the Government investigate in detail, and release a report by October 2004 on the potential to establish plantations to phase out of logging in the Thomson Reservoir catchments (WRSCMA 2002a). The State Government responded to every recommendation made by the committee, except this one (SoV 2003), however a response is expected later this year. This approach is disappointing in that it favours timber interests over water - logging is permitted to continue as in interim measure at the elevated rate of the 1990s - while the investigation of plantation alternatives outside the catchment has been stalled.

Given the very long tern nature of harvesting impacts on water, the State and Federal governments have recognised the need for a long term water production strategy, agreeing one be developed when timber resource data became available in 1999 (RFA 1998). This was a milestone in the Regional Forest Agreement, yet a strategy has not been developed, and it is unclear if or when this may occur.

3.2 Thomson Water Yield Losses

The WRSCMA predicts that phasing logging out of the Thomson would produce an extra 20,000 ML of water by 2050 (WRSCMA 2002). In order to model three scenarios – no logging, a 200 year rotation and continued harvesting at current rates, the author conservatively applied the yield curves developed by Peel *et al.,* (2002) for the Integrated Forest Planning System (IFPS) for use in the Thomson. If harvesting continues at current rates, a 20,00 ML water loss projected per annum by 2050 rising to 40,000 ML within 200 years relative to what would be saved if logging ceased this year (Illustration 1). This result underestimates yield loss in comparison to an earlier study in the catchment which projected a 50,000 ML loss for this period (RSA 1992) based on the Kuczera curve rather than the Macaque model.

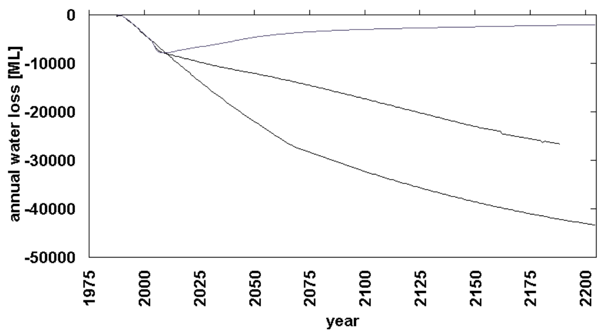

Illustration 1. Thomson water loss under three management scenarios, relative to 1939 regrowth. Top curve is no logging scenario, middle curve status quo, bottom curve 200 year rotation.

These projections assume that the forests are not disturbed by fire which has the potential to reduce water yields in ash forest. Should fire occur, the projections would need to be revised dependant on the size and intensity of the fire.

The aspects of harvesting that can be managed include harvesting extent, rotation length and site selection relative to rainfall. An analysis of post harvesting yields using the IFPS curves strongly suggests that site selection relative to rainfall under current harvesting intensities is negligible within the currently fully allocated ash resource and that rotation length and harvesting extent are much more important variables.

3.3 Thomson Water Loss Economics

The water lost following harvesting has an economic value, and in the Melbourne system water is precious. Melbourne is facing a shortage of supply with demand expected to exceed supply by 2012 on current consumption trends (WRSCMA 2001). Water availability is also a limiting factor in the economic growth of Australian cities (CSIRO 2001). The Thomson provides water to irrigators downstream and environmental flows need to be increased by 47,000 ML per year (ET 2003). Wood production therefore competes with water production from the Thomson.

Log prices do not include a fee for water loss following harvesting operations (NRE 2001), nor are there plans to do so.

In 1992, consultants Read Sturgess and Associates compared the value of water and wood in the catchment, concluding that the Victorian community would be $147 million better off if logging ceased (RSA 1992).

While a full economic analysis is beyond the scope of this paper, the price paid by alternative users of water from the Thomson provides insight into the management choices being made. Melbourne households would currently pay 27 million dollars for the 40,000 ML water loss projected to occur per annum, 200 years in the future, if logging continues in the Thomson.

In the authors' opinion, catchment managers are favouring timber interests over water by allowing harvesting to impact on water values at no cost to the industry. The timber industry should ideally pay water loss relative to the benchmark set by natural vegetation.

3.4 Yarra Tributaries and Tarago

The author estimates that continued harvesting at current rates in the Yarra Tributaries and Tarago would result in water losses of at least 10,000 ML by 2050 and 20,000 ML in the long term relative to mature forest (Hughes unpub 2002, Hughes unpub 2003). There is less confidence in these projections than those for the Thomson due to lack of previous research and data.

4. ALTERNATIVES AND OPTIONS

Consultants URS forestry have concluded that plantations can be grown to substitute for the 85,000 m^3 pulp log sourced from the catchments each year and that further investigation is needed on substituting the 55,000 m^3 sawlogs with plantation Mountain Ash or Shining Gum (URS 2002).

Shining Gum and other species can be grown on cleared land in lower Gippsland catchment areas (BRS 1999). Growing wood at these lower rainfall sites, about 800-1000 mm, is water efficient because less water is used for every cubic metre of wood produced.

5. DISCUSSION AND CONCLUSIONS

If harvesting continues at current rates in the Thomson catchment, IFPS modelling indicates yield loss is expected to reach 20,000 ML per annum by 2050 and 40,000 ML in 200 years, relative to a no logging scenario. This loss is important in the Thomson system and represents 3 to 7% of Melbourne's current water supply.

The modelling indicates that ceasing logging in the catchment would benefit water resources most, and reducing harvesting rates would provide less benefit. Choice of harvesting site relative to rainfall would have little influence without reductions to harvesting.

Caution needs to be taken in interpreting the results of this study. The IFPS curves were derived from Macaque which is a relatively new model with recognised limitations in validating results (Peel *et al.,* 2000). Differences exist between the IFPS curves and Macaque which cannot currently be explained. This desktop study modelled harvesting scenarios simply but reasonably without precise information on future harvesting sites. Previous modelling based on the Kuczera curve has indicated larger water losses in the catchment under similar scenarios, and note should be taken that water yield losses may exceed those projected here.

There remains a gulf between the science and current catchment management, with timber interests being favoured over water in selection of rotation length, with no requirement for the industry to pay for water losses following harvesting. Decisive action from politicians and land managers is appropriate in order to balance the interests of water and timber more fairly.

In the authors' opinion, timber production should be avoided in high rainfall forested upper catchments, which could be prioritised for water production. The potential to produce wood in a more water efficient manner in lower catchment plantations should be further explored, with potential additional benefits to water quality and biodiversity.

It is strongly recommended that harvesting be minimised in Melbourne's catchments while the potential of plantation grown alternatives is further investigated, as recommended by the Water Resource Strategy Committee for the Melbourne Area. A cautious approach is warranted given the long term implications - today's catchment management decisions will have the greatest impact on future generations.

6. ACKNOWLEDGEMENTS

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Forest Health Decline in Coastal New South Wales

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Summary: Forest health decline with unnatural fire regimes is an increasing problem that is gaining increasing worldwide recognition. This paper summarises some information from various surveys of forest health in coastal New South Wales. It is estimated that hundreds of thousands of hectares of forest are declining, and millions of hectares potentially may be affected. Decline affects a high proportion of some grassy forest types, including many that have been substantially reduced in extent because they were targeted for agricultural development. Current regulations, frameworks and practices are not enhancing conservation of these forests. Some changes are suggested to encourage sustainable forest management in accordance with Regional Forest Agreements.

1. INTRODUCTION

Mount (1969) suggested that unnatural fire exclusion could cause tree decline whereas periodic fire could reduce the effects of drought, fungi and insects. Ellis and Pennington (1989) found that increasing nitrification in soils with exclusion of fire was associated with decline of alpine ash, Eucalyptus delegatensis, in Tasmania. Lunt (1998) documented eucalypt decline in Victorian coastal woodlands with long term fire exclusion. Many other reports of forest decline with fire exclusion were cited by Jurskis *et al.,* (2003), including tuart (Eucalyptus gomphocephala) decline in Western Australia, and decline associated with heavy koala (Phascolarctos cinereus) browsing in Victoria. Jurskis and Turner (2002) proposed that exclusion of low intensity fire was a common cause of eucalypt decline in southeastern Australia. Exclusion of low intensity fire has been found to increase topsoil moisture, organic matter and nitrogen levels, forest floor litter, rates of nitrogen cycling and shrub density. (Raison *et al.,* 1993, York 1999). Fire exclusion has also reduced ground level solar radiation and available phosphorus in soil. (York 1999, Guinto *et al.,* 2001). These changes are likely to cause decline in forests that are adapted to arid and infertile conditions (Jurskis and Turner 2002). Forest decline and increasing fire control problems with post European changes in fire regimes were documented for the Inland North West of North America (Hessberg and Agee 2003).

2. EXTENT OF FOREST DECLINE

Some estimates of the extent of forest decline are available from recent surveys of forest health in coastal NSW. In 2002 a brief aerial reconnaissance of part of the Eden region identified about 5,000 ha of declining forest. More than 4% of the survey area was severely affected. Similar surveys on the north coast also identified many thousands of hectares of declining forest. Based on the conservative assumptions that 25% of the more susceptible forest types and 4% of the less susceptible forest types were affected, it was estimated that more than 120,000 ha of State forest in coastal New South Wales were declining in health. A larger area of private forest and National Park was likely to be affected.

Jaggers (2004) estimated that around 100,000 ha or 20% of forests in the Eden region were declining in health. About 40,000 ha of declining forest were on private land, about 40,000 ha were in National Parks and Nature Reserves, and about 20,000 ha were in State forest. Another 50,000 ha, or 10% of the region's forests were types that were susceptible to decline but were not yet affected because they were young regrowth forests. A limited (0.5%) systematic sample in the Batemans Bay region (State Forests unpublished data) indicated that about 28% of the State forests were in decline, and that the majority of the area of susceptible types, such as gum – box – woollybutt and peppermint types, was affected.

In 2003 an area of about 800 ha in Donaldson State Forest in northern NSW was being assessed for silvicultural operations. About 60% of the area was affected by decline (F. Bugno pers. comm.). Targeted surveys in 1992 had identified about 2,000 ha of declining forest within State forests north of Sydney (Stone *et al.,* 1996), but the area of about 500 ha of declining forest at Donaldson was not identified by those surveys. The average patch size of declining forest identified by the targeted surveys in 1992 was about 16 ha, whereas the size of patches identified by aerial reconnaissance during 2002 averaged about 40 ha. In 2002, unhealthy trees extended widely across the landscape with a gradation from severe decline in depressions and flats to early symptoms on upper slopes and ridgetops. Decline was not apparent in younger regrowth stands. Areas of declining forests extended as drought intensified during 2002.

In coastal NSW, decline is apparent in private and public forests across the landscape. It is particularly associated with concave topography, depositional soils and naturally grassy forest types. These open grassy forests evolved with frequent low intensity fire from lightning and aboriginal burning. An estimate of the extent of these forest types using NPWS compilation map of native vegetation in NSW (Keith 2002) suggests that about 2.4 million ha of susceptible forest occur in coastal NSW.

3. CONSERVATION OF BIODIVERSITY

Following European settlement, grassy forests were selected for grazing whereas shrubby forests were not suitable and remained largely undeveloped (e.g. Keith and Bedward 1999, Jurskis 2000). The majority of forests in conservation reserves are shrubby forests, and the majority of grassy forests are privately owned. The grassy forests are remnants of the more extensive forests that were mostly cleared for agriculture after European settlement. These 'under-reserved' forest types are the types that are particularly prone to decline under unnatural fire regimes.

Unfortunately, many ecologists hold an unrealistic view that fire was naturally rare before European settlement of Australia, and that fires lit by people are unnatural and ecologically damaging (Jurskis 2003). Although the frequency of intense fires increased following European settlement of Australia, it is not generally appreciated that this was associated with a reduction in low intensity fires (Jurskis *et al*. 2003).

Grassy eucalypt ecosystems are flammable under mild conditions conducive to low intensity fire. However their flammability in mild conditions is reduced as dense shrub layers develop after low intensity fire has been excluded for a while. The inevitable result is that these ecosystems burn more intensely and uniformly in more severe conditions and these intense fires carry into naturally less fire prone ecosystems containing more sensitive species. Natural edaphic controls on fire have been weakened by fire exclusion and consequential high intensity fire regimes (Jurskis *et al.,* 2003).

It has been suggested (e.g. Keith 2002, Keith and Henderson 2002) that the understorey of many grassy forests has been simplified by grazing and burning. However management that favours woody shrubs, litter and associated fauna may not enhance conservation of biodiversity. For example the shrubs that Keith and Henderson (2002) suggested were contributing to biodiversity in ungrazed and unburnt forests were common and widespead species that appear to be increasing in numbers and distribution across the landscape (Jurskis 2003). The declining health of the dominant components in these 'undisturbed' forests suggests that natural processes have been disrupted by fire exclusion (Jurskis 2003).

Coastal rainshadow grassy woodlands are amongst the vegetation types that have been most depleted by clearing in NSW. Keith (2002) estimated that between 60% and 90% of these types have been lost since European settlement. He included four vegetation types from the Eden region (Keith and Bedward 1999) in this broad vegetation category. All four types are declining (Jaggers 2004). Three of them are listed as endangered ecological communities under the NSW Threatened Species Conservation Act. Two of these grassy woodland types, Bega wet shrub forest and Brogo wet vine forest (Keith and Bedward 1999), have been invaded by shrubs and vines as a result of decades of fire exclusion (Jurskis 2003). Ironically NSW Scientific Committee and Keith and Bedward (1999) consider that these types are potentially threatened by frequent burning and grazing.

4. POLICIES AND REGULATIONS

Australia's National Forest Policy aims to conserve examples of natural ecosystems as they existed immediately prior to European settlement. Under this policy, Regional Forest Agreements specify that a key principle of sustainable forest management is to maintain healthy ecosystems by reducing threats from diseased weeds and unnatural fire regimes. However conservation management currently seems to be directed mainly towards designating reserves rather than physically conserving ecosystems and maintaining their health. Forest health decline and increasing fire control problems are two facets of a problem arising from passive management philosophies (Jurskis *et al.,* 2003, Jurskis 2000). State government policies and regulations encourage passive management and discourage active management for conservation. They require assessments of the environmental consequences of active management but not of passive management even though the consequences of passive management can be very severe (Jurskis *et al.,* 2003).

New South Wales Biodiversity Strategy and draft Native Vegetation Conservation Strategy fail to deal effectively with forest health and fire control issues. Objective 3.2 under the Biodiversity Strategy is to "minimise the modification of natural ecosystems"; Action 43 is to manage fire in accordance with principles of ecologically sustainable development. The absence of further guidance in the strategy implies that prescribed burning is unnatural. This was reinforced by the listing of frequent fire as a key threatening process under the Threatened Species Conservation Act.

Regulations in NSW exclude low intensity burning from much of the landscape including wilderness, old growth, rare ecosystems, habitats of rare plants or animals, and drainage lines. They focus on individual organisms, target species and fire frequency. They don't require assessments of the consequences of not burning. This regulatory environment encourages forest decline and more widespread high intensity fire regimes.

5. MONITORING

Despite the extent of forest decline that is occurring in coastal NSW and more widely in temperate Australia, state of the environment reports have not identified this as an issue of concern. Two Montreal Process indicators are particularly relevant to this issue. Indicator 3.1a is the area and per cent of forest affected by processes or agents that may change ecosystem health and vitality, and Indicator 3.1.c is the area and per cent of forest land with diminished or improved biological, physical or chemical components. Australia's 2003 State of the Forests Report did not include any estimates of the extent of native forest decline under either of these indicators (Parsons 2004). Parsons (2004) stated that no State or Territory has the capacity to monitor the health and vitality of forest ecosystems on a state-wide scale and suitable indicators are still being developed for future reporting. However I believe that existing evidence of broadscale deterioration in native forest health should be reported. Apparent reluctance to report this evidence may reflect a lack of resources or knowledge to address the problem.

6. SUSTAINABLE FOREST MANAGEMENT

Policies and regulations should aim to maintain healthy ecosystem processes, focussing on the ecosystems and species that have been most depleted since European settlement.

Sustainable forest management should be encouraged by:

- recognising and reporting forest health as a primary indicator of sustainable forest management;
- developing guidelines and prescriptions to manage landscapes, not individual plants and animals;
- developing prescriptions to control the extent and spatial variability of fires by controlling fire behaviour, rather than prescribing artificial exclusion zones and fire intervals;
- recognising that low intensity burning maintains edaphic controls and protects fire sensitive species;
- recognising increasingly extensive high intensity fire regimes and eucalypt decline as consequences of fire exclusion that must be considered in planning.

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Sustainable Forest Management in Tasmania

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Summary: Sustainability indicators at the State level have been reported for the first five yearly review of the RFA in 2002. On State forests, Forestry Tasmania has its own set of indicators which support the RFA indicators and also address a number of identified community concerns such as harvesting of old growth, chemical usage, pest control, strategic management of threatened species and compliance with the Tasmanian Forest Practices Code (2000). This paper presents some results and early trends from the implementation of a range of Forestry Tasmania's indicators.

The multi-disciplinary long-term ecological research site established at Warra in 1995 in the southern forests is supporting sustainable forest management. The RFA requires the results of research at Warra to be taken into account in the ongoing development of sustainability indicators. Two of the major themes being studied, alternatives to clearfell, burn and sow silviculture and hydrological and nutrient cycling studies in pristine and managed forests, are briefly described.

1. INTRODUCTION

The Tasmanian Regional Forest Agreement (RFA) (1997) underpins sustainable forest management (SFM) in Tasmania. The RFA required the implementation of the most comprehensive forest reserve system in Australia, sustainable forest management and the growth of a viable industry. The implementation of certified environmental management systems and a set of sustainability indicators ensure that forest managers are accountable in meeting sustainability requirements.

2. FOREST COVER AND TENURE

Currently 49% of Tasmania is forested, which is about 66% of the original forest cover. In meeting the State's comprehensive, adequate and representative (CAR) reserve targets:

- 40% of forest is reserved;
- 69% of old-growth is reserved; and
- 95% of wilderness is reserved.

This reservation level is equal to or better than the other Australian states (NFI, 2003).

3. SFM FRAMEWORK

In accord with the RFA, on State forest this framework consists of:

- Forestry Tasmania's SFM policy;
- An environmental management system (EMS) certified to the international standard (ISO 1401);
- Objectives, targets and indicators;
- Sound scientific research base to underpin continuous improvement; and
- Annual SFM reporting.

Forestry Tasmania is seeking certification under the Australian Forestry Standard to further demonstrate its SFM performance in the forest. The implementation of the EMS over the last few years has greatly assisted in improving field performance to a level where certification is attainable.

4. SFM REPORTING

Tasmania is in a position to report more completely on the sustainable management of its forests across all tenures than any other Australian State. This has been facilitated by the Tasmanian RFA covering the entire State. The *Tasmanian State of the Forests Report* is produced every five years at the time of the RFA five yearly review and uses the same sustainability indicators. These same indicators have contributed to national Montreal Process reporting (NFI, 2003) and *Australia's State of the Forests Report* (NFI, 2003).

On State forest SFM reporting is via Forestry Tasmania's annual *Sustainable Forest Management Report.* The first of these reports was provided as a public report for the year 2000-2001.

5. SFM OBJECTIVES AND INDICATORS

Forestry Tasmania has identified 9 objectives and 24 indicators against which to publicly report its annual SFM performance. The SFM objectives reported against and some results of indicator monitoring for 2002-2003 are reported below. The full details are available in the *Sustainable Forest Management Report* (Forestry Tasmania, 2004).

- *Legal compliance:*

The Forest Practices Board both investigates complaints and routinely audits operations on State forest against the Forest Practices Code. For 2002-2003 there were 1563 Forest Practices Plans in operation on State forest. Forestry Tasmania's performance against the Forest Practices Board audits has consistently exceeded the Board's 85% compliance benchmark. During the year nine notices to carry out works on State forest to comply with the Code were issued by Forest Practices Officers and the Board issued one fine to Forestry Tasmania. This is a similar result to 2001-2002.

There were no identified breaches of other State or Commonwealth legislation.

- *Biological diversity:*

Tasmania's Permanent Forest Estate Policy provides native forest retention thresholds both at a State and a bioregional level. At least 80% of the native forest mapped in 1996 for the RFA must be retained at the State level. Any clearing or conversion of native forest for plantation is monitored to ensure thresholds are not approached. The RFA also identified 20 priority vegetation communities for protection. These priority communities are not harvested on State forest.

Forestry Tasmania is also working to reduce reliance on the clearfelling of old growth forest. Since monitoring commenced two years ago 0.22% of the total old growth in Tasmania (1.239 million hectares) has been clearfelled (wet eucalypt forest) on State forest.

Compliance with the threatened species provisions of the Forest Practices Code has been maintained and work towards strategic management of threatened species continues with the regulatory authorities.

- *Productive capacity:*

The sustainable yield of high quality sawlogs from State forest is periodically reviewed with the most recent review in 2002 (Forestry Tasmania, 2002). Experts from the Australian National University externally audited this review process. They concluded with some qualifications that *'datasets, models, systems and methodology used in calculating the sustainable yields from Tasmania's native forests are suitable and adequate for the purpose and are mostly as good as, or better than, contemporary Australian best practice'.* The high quality sawlog cut, which is consistent with the sustainable sawlog yield, is monitored and reported in annual management plan reports that are publicly available.

To ensure ongoing productivity, regeneration of native forest following harvesting is monitored. It is reported upon 3-5 years after harvesting in Forestry Tasmania's Annual Report. The objective is to ensure 95% of the regenerated area meets the stocking standard (Forestry Tasmania, 2003). The standard for eucalypt regeneration was achieved. However for rainforest and blackwood swamps 85% of the regenerated area met the standard.

- *Ecosystem health and vitality:*

This indicator monitors animal browsing control (1080 usage) and fire management.

The use of 1080 to control browsing in newly established plantations and natural regeneration in native forest has progressively decreased over the last three years by 38%.

The area of State forest burnt by unplanned fire has remained below the ten-year average over the last three years despite the severe 2002-2003 fire season. The area severely burnt was above the ten-year average for 2002-2003.

- *Soil and water resources:*

The area of State forest managed primarily for protection purposes has remained constant at 49% of the total estate over the last three years.

Compared to 2001-2002 the use of chemicals for plantation establishment in 2002-2003 reduced with an associated move to the use of more benign chemicals.

Sampling of water catchments for chemicals is undertaken wherever agricultural herbicides, insecticides or aerial fertilising is carried out. In 2002-2003, 337 water samples were collected and analysed with nine instances of chemicals being detected but no instances exceeding Australian Drinking Water Guidelines.

- *Cultural values:*

The intention is to protect all identified cultural heritage sites. This was achieved during 2002-2003 however the requirements of the Forest Practices Code while remaining at a high level were not at the same standard as the previous year. This was due to three instances of inadequate post operational survey and two instances of failure to consult with the Forest Practices Board archaeologist.

- *Waste and pollution:*

Post logging residue assessments resulted in a mean residue of 3.2 tonnes per hectare, which was the same as for 2001-2002. The standard of less than 5 tonnes of merchantable wood residue per hectare was achieved for 86% of coupes sampled. The standard used represents about 2% of the average volume harvested from a clearfelled coupe.

- *Sound science:*

Total research expenditure for 2002-2003 remained similar to 2001-2002 although research expenditure as a percentage of forest management costs slightly decreased. This was due to increased operational costs and some vacancies in research staffing that have now been filled. Examples of research contributing to SFM appear under section 6 below.

- *Multiple community benefits.*

As a result of major tourism developments such as the Tahune Airwalk, annual visitor numbers to State forest have increased from less than 200,000 people in 2000-2001 to more than 350,000 people in 2002-2003. Apart from additional direct employment this has resulted in flow on social and economic benefits to local communities in towns such as Geevston and Huonville.

Across the State the mill door log value of wood products sold increased by 12% to $252 million indicating a buoyant industry.

6. RESEARCH

Much of the forest research to underpin SFM in Tasmania is being conducted at the Warra long-term ecological research site (LTER) (Brown *et al.*, 2001). The Warra LTER site covers 15,900 ha of State forest and adjoining conservation reserve. Warra is supported by seven site partners from Tasmanian and national research agencies. The four current priority research programs are:

- Investigating alternative silvicultural systems for wet eucalypt forest;
- Forest catchment hydrology;
- Developing and testing indicators for SFM; and
- Inventorying biodiversity, geodiversity and natural resources.

The first two of these programs are outlined in more detail below.

Clearfell, burn and sow (CBS) is the routine silviculture applied in lowland wet eucalypt forest in Tasmania. Hickey *et al.,* (2001) have reported concerns about the initial aesthetics, the likely impact on biodiversity, and the potential decline in rainforest understorey species if the forest continues to be silviculturally managed in this manner for the production of eucalypt sawlogs. This study is investigating five alternatives to CBS ranging from current practice to variable retention regimes (aggregated and dispersed tree retention) and finally single tree/small group selection. The advantages and disadvantages of each silvicultural regime have been determined and performance criteria and indicators identified for assessment. The first report against the following criteria is expected in 2007 (Hickey, pers.com.):

- Biodiversity – species richness and abundance;
- Productivity – includes regeneration, browsing impacts and growth of retained trees;
- Social – worker safety and public perceptions; and
- Economic – timber production, operational costs, fire/fuel management and financial evaluation of each treatment.

The Warra hydrology study (Ringrose *et al.*, 2001) has weirs constructed on three streams with a further 13 streams sampled fortnightly. The study, which commenced in 1998, has 3 objectives:

- Characterise the variability of water quality in a pristine stream;
- Determine the impact of logging on stream hydrology; and
- Obtain an overview of the water quality in the Warra LTER area.

7. THE FUTURE

Forestry Tasmania has a sound framework for SFM based on research and continuous improvement. Planning and operations are subject to external audit and Forestry Tasmania implements annual performance reporting. Independent certification against the Australian Forestry Standard can only strengthen the organisation's SFM performance.

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The Effect of Afforestation on Flow Duration Curves

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Summary: A method to assess the impact of plantation establishment on flow regime was developed using flow duration curves, FDC's. The starting point for the analyses was the assumption that rainfall and vegetation age are the principal drivers of evapotranspiration. The objective was to remove the variability in the rainfall signal, leaving changes in streamflow solely attributable to the evapotranspiration of the plantation trees. Data from Australian, South African and New Zealand catchments were used. A model was developed to firstly characterise changes in each flow decile over time, and then to construct FDCs adjusted for average rainfall. The model returned encouraging results, with satisfactory fits found for the majority of deciles at the 8 catchments, particularly the high flow (10th - 50th) percentiles. While the adjusted FDCs revealed peak flows were least impacted in all catchments, the responses over the remaining deciles split into two groups; a set of catchments where there was a marked decrease in median and low flows, and a fairly uniform reduction in flows across the flow regime. The modelled reductions are in accord with those from paired-catchment experiments and indicate this methodology may be of use where paired-catchment data is not available.

1. INTRODUCTION

Widespread afforestation through plantation establishment on agricultural land represents a potentially significant alteration of hydrologic regime, and is an emerging issue in Australia (eg. O'Loughlin and Nambiar, 2001, Vertessy *et al.*, 2003, Gerrand *et al.*, 2003). Experiments at the plot and catchment scale, and hydrologic analyses relating streamflow quantity to catchment vegetative cover, have established that streamflow from forested catchments is lower than that from a grassland or cropland (Zhang *et al.*, 1999, Holmes and Sinclair, 1986). Research from South Africa in particular has demonstrated flow reduction following afforestation with both softwoods and hardwoods (Van Lill *et al.*, 1980; van Wyk, 1987; Bosch and Van Gadow, 1990; Scott and Smith, 1997; Scott *et al.*, 2000). Zhang *et al.,* (1999, 2001) and Munday *et al.,* (2001) have developed simple and easily parameterised models to predict changes in mean annual flows following afforestation. However there is a need to consider the annual flow regime. The relative changes in the distribution of flows may have considerable site specific and downstream impacts, which ultimately have implications for the plantation industry. For example Sikka *et al.,* (2003) and Scott and Smith (1997) report proportionally greater reductions in low flows. The maintenance of environmental flows and dry season water supplies for agriculture and domestic consumption may be adversely affected under such a scenario.

This paper presents the results of a project aimed at quantifying changes in annual flow regime of catchments following plantation establishment. The flow regime has been represented by the flow duration curve (FDC). The key assumption was that rainfall and forest age are the principal drivers of evapotranspiration. Accounting for annual climate variability is central to the successful quantification of impact of land use change on the FDC. The observed impacts of the land use change may be exaggerated or understated depending on the prevailing climate. For any generalisation of response of the FDC to vegetation change, the climate signal must be removed. Consequently, the aims of this project were to (1) develop and fit a simple model to the observed deciles of the FDC that incorporated rainfall variability and plantation age, and, (2) remove the climate signal from the FDC by adjusting the FDC for average rainfall over the period of record. If the climate signal, represented by rainfall, could be successfully removed, the resulting changes in the FDC would be solely attributable to the vegetation. This technique is similar to the Generalised Additive Model (GAM) approach of Nathan *et al.,* (1999).

2. METHODS

Flow duration curves (FDCs) display the relationship between streamflow and the percentage of time it is exceeded as a cumulative density function. For the consideration of annual flow regime, daily flows are an appropriate time step for FDC construction. FDCs were computed from the distribution of daily flows for each year of record based on water years (May-April). Each decile over the period of record was then extracted from the FDC to form the data sets for analysis. The appropriate rainfall statistic (P) to represent the climate term was obtained from correlations with flow deciles. Generally mean daily rainfall returned the best overall correlations with all flow percentiles.

The choice of model form is dependent on selecting a function that describes relationship between forest age and ET. Scott and Smith (1997) demonstrated reductions in annual and low flows resulting from afforestation fitted a sigmoidal function, similar to forest growth functions. Consequently, we used a sigmoidal function to characterise the impact of growth. Thus the model took the form:

$$Q\% = a + b(\Delta P) + \frac{A_{sig}}{1 + \exp\left(\frac{T - T_{half}}{N_{sig}}\right)}$$

where $Q_{\%}$ is the percentile flow (i.e. Q_{50} is the 50th percentile flow), A_{sig}, and N_{sig} are coefficients of the sigmoidal term, ΔP is the deviation of annual rainfall from the period of record average, and T_{half} is the time in years at which half of the reduction in $Q_{\%}$ due to afforestation has taken place (Figure 1). Assuming that the rainfall term accounts for the climate signal the deciles of the FDC can be adjusted for climate by setting ΔP to zero. For this condition, a becomes the value of $Q_{\%}$ when the new hydrologic equilibrium under afforestation is reached. A_{sig} then gives the magnitude of change due to afforestation, and N_{sig} describes the shape of the response as shown in Figure 1. For the average pre-treatment condition $\Delta P = 0$ and T = 0, $Q_{\%}$ approximately equals $a + A_{sig}$.

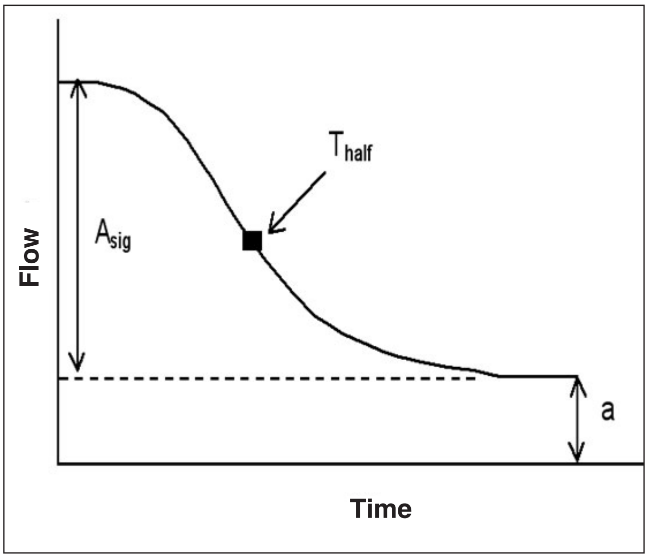

Figure 1. Generic form of Equation 1 and definition of model parameters.

3. DATA SETS

Streamflow data was obtained from ten catchment studies from southeastern Australia, New Zealand and South Africa. The initial criteria for selection of these catchments were known vegetation history and streamflow records of good quality. Full details of these catchments are given in Lane *et al*., (2003). All catchments excepting Traralgon Creek were afforested with pine species, predominantly *Pinus radiata*, with *P. patula* planted at the two Cathedral Peak catchments. Traralgon Creek has only 6% pine, with the remainder eucalypts species, most of which is *Eucalyptus regnans.* Soil depths vary from 1.0 m or less at Redhill, Stewarts Ck. Pine Ck, and Glendhu, 1.5-2.0 m for the South African catchments and 2.0 for Traralgon Creek.

4. RESULTS

4.1 Model Evaluation

The fit of the model (Equation 1) to each flow decile over the period of record was evaluated by the coefficient of efficiency (E) as shown in Table 2. Values of E > 0.7 were considered to be acceptable model fits. In general, the model performed well, with the poorest fits at the 2 Lambrechtsbos catchments, particularly for higher deciles (lower flows). The likely reason for the poorer model fits at Lambrechtsbos A is an annual decrease in stand water use after 12 years (Scott *et al.*, 2000) which does not conform to the sigmoidal form over the full 19 years of record.

Table 1. Study catchments. BFI is baseflow index

Catchment	Area (ha)	% aff.	Mean ann.	P (mm)	age (y) BFI
Traralgon Ck (Vic)	8700	~ 70	1472	19	0.37
Redhill (NSW)	195	78	866	9	0.39
Pine Ck (Vic)	320	100	775	11	0.26
Stewarts Ck 5 (Vic)	18	100	1156	20	0.28
Glendhu 2 (NZ)	310	67	1282	17	0.64
Cathedral Pk 2	190	75	1436	20	0.66
Cathedral Pk 3	139	86	1504	17	0.75
Lambrechtsbos A	31	82	1134	19	0.78
Lambrechtsbos B	66	89	1088	20	0.87
Biesievlei	27	98	1332	20	0.72

The significance of the climate term b and the time term Asig were tested and found to be satisfactory for most deciles (see Lane *et al.*, 2003 for details).

4.1 Adjusted FDCs – Magnitude of Flow Reductions

Given the satisfactory results of the model fitting and significance tests, the FDCs were adjusted for climate by setting $\Delta P = 0$. The climate adjusted FDC's produce an estimation of the change in flow percentiles over time for each catchment due to afforestation. The relative net proportional change due to afforestation is given by $A_{sig}/(A_{sig} + a)$. This quantity is plotted for all catchments in Fig 2. Two types of responses (groups) were identified. Group 1 catchments show a substantial increase in the number of zero flow days, with a greater proportional reduction in low flows than high flows. Group 2 catchments show a more uniform proportional reduction in flows across all percentiles. Catchments in group 2 can be broken up into group 2a and group 2b depending on the magnitude of the response reduction. The catchments in each group are:

- *Group 1:* Stewarts Creek, Pine Creek, and Redhill
- *Group 2a:* Cathedral Peak 2 and 3, and Lambrechtsbos B
- *Group 2b:* Lambrechtsbos A, Glendhu 2, Biesievlei and Traralgon Creek

Group 1 exhibit both the highest reduction of flows overall, and they show the largest proportional reduction at lower flows leading to a complete cessation of flow. Responses for the group 2a catchments are the most variable through the flow regime. Group 2b reductions are reasonably uniform through the percentile range.

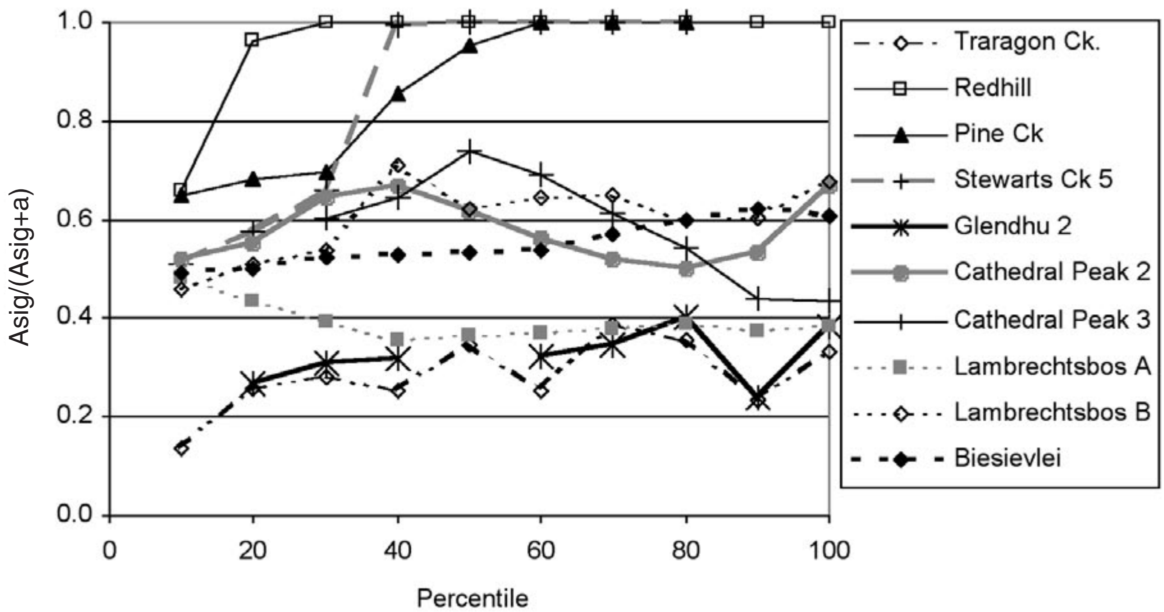

Figure 2. Net flow reductions Asig/(Asig+a) for all catchments.

Table 3. Coefficient of efficiency, E. ns indicates that no solution was found, and na denotes deciles with too few data points for analysis.

Site	Percentile									
	10	20	30	40	50	60	70	80	90	100
Traralgon Ck	0.82	0.85	0.81	0.81	0.83	0.78	0.80	0.77	0.65	0.56
Redhill	0.90	0.95	0.82	0.80	0.92	0.88	0.89	0.77	0.65	0.42
Pine Ck	0.56	0.76	0.88	0.99	0.99	0.99	0.71	0.99	na	na
Stewarts Ck 5	0.82	0.85	0.81	0.88	0.87	0.88	0.88	na	na	na
Glendhu 2	0.76	0.77	0.82	0.84	ns	0.87	0.89	0.90	0.86	0.76
Cathedral Peak 2	0.83	0.91	0.93	0.92	0.81	0.73	0.89	0.95	0.96	0.95
Cathedral Peak 3	0.68	0.75	0.78	0.91	0.96	0.95	0.94	0.81	0.84	0.79
Lambrechtsbos A	0.71	0.60	0.57	0.47	0.47	0.47	0.47	0.50	0.49	0.51
Lambrechtsbos B	0.82	0.76	0.70	0.66	0.65	0.65	0.62	0.59	0.58	0.58
Biesievlei	0.96	0.96	0.90	0.82	0.81	0.88	0.92	0.94	0.91	0.81

5. DISCUSSION

The general characterisation of FDCs and adjustment for climate has been very encouraging given the task of fitting our model to ten flow percentiles, for ten different catchments (resulting in 100 model fits) with substantially varying spatial scales, soils and geology, species planted and climatic environments. Although there were poor results for individual deciles, the FDCs at eight of the ten catchments were adequately described by Equation 2. When compared with available paired-catchment studies at 8 of the catchments (Lane *et al*., 2003), the results are in reasonable accord, further evidence that the method is robust and the rainfall variability has been accounted for. The estimated changes in flow regime for average rainfall $A_{sig}/(A_{sig} + a)$, reveal that while there are differences in responses between the analysed catchments as a whole, there are recognisable trends. Perhaps the most general finding was that in seven of the ten catchments the high flows (10th percentile) were less affected than median and lower flows. The three small Australian catchments converted to pine do not adhere to this pattern. They have similar shallow soils, potential evapotranspiration, rainfall distribution (relatively uniform) and low baseflow index, although Stewarts Creek is significantly wetter. The combination of small catchment area and the increased transpirative demand that exceeds summer and autumn rainfall and stored water results in the large impact on lower flows, compared to high flows. The higher BFI and deeper soils at the remaining catchments suggest greater storages, which would mitigate against flow cessation. Traralgon Creek would be expected to have the most subdued flow reductions because of the large area of *E. regnans* forest, and uncertain vegetation record. Peak stand water use of a natural stand of this species is around 30 years. Additionally in this large, “real world” catchment, there is a continuous cycle of forest management, which includes harvesting. A mixture of pasture and “scrub”, which could represent significant understorey stands, were replaced by plantation species. Consequently the difference between pre and post treatment ET may be less than at other catchments. Reductions of this magnitude could be more readily expected in larger, multi land use catchments than the very high impacts estimated at the smaller Australian catchments.

The overall total flow reductions presented here probably represent a maximum as there are very high percentages of afforestation, and most sites are small headwater catchments. Linear scaling of the flow reductions by more typical percentages of area afforested would reduce the estimated impacts.

However even diminished median-low flow reductions may be important. Scarcity of water in dry seasons is frequently problematic for water security and maintenance of environmental flows in many areas. While consideration of the positioning and area of catchment plantings may minimise annual flow reductions (Vertessy *et al.*, 2003) any diminution of low flows may be significant.

The results of this project are part of a complimentary set of studies carried out by the CRC for Catchment Hydrology on the impact of landuse change on annual and intra-annual flow streamflows that will assist in building a sound knowledge base required for management of land use change, in this case afforestration.

6. ACKNOWLEDGEMENTS

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Runoff Connections From Roads in a Steep Forested Catchment in Victoria

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Summary: Limiting connectivity between runoff sources and stream networks is crucial for preservation of water quality in forested environments. Hairsine *et al.,* (2002) proposed a probabilistic model of diffuse overland flow that predicted the hillslope lengths required to infiltrate road discharge, based on the concept of volume to breakthrough (Vbt). This paper extends this analysis to two different forest environments with the aim of testing the portability of the Hairsine *et al.,* (2002) model. Statistical analysis revealed the population of Vbt5 of the Victorian catchment in its normal state to be indistinguishable from that observed for the previous study areas, indicating the model is valid for a range of forest soils, and that there was no significant correlation of sediment plume length with site characteristics such as slope, width of flow, or existence of incised pathways. However, experiments on hillslopes burnt by bushfires show significantly lower Vbt5 and Vbt10 than unburnt slopes, highlighting changed post-fire drainage conditions.

1. INTRODUCTION

Connectivity between runoff sources and stream networks at the catchment scale has been a focus for several studies concerned with forest road drainage (Montgomery, 1994; Jones and Grant, 1996; Wemple *et al.*, 1996; Croke and Mockler 2001; La Marche and Lettenmaier 2001). Connection may occur through the formation of a gully below a relief culvert, entrance to a stream via a stream crossing culvert or table drain, or by overland flow. Minimising connectivity is the key to maintaining water quality associated with roads and tracks. There is a need for simple but robust design criteria for road and track drainage that considers both eroding and non-eroding discharges. Croke and Mockler (2001) demonstrated such a scheme for minimising gully initiation. Where flow is non-eroding (i.e. overland flow), the length of hillslope available to accommodate volumes of discharged water is the key to restricting connectivity. Hairsine *et al.,* (2002) proposed a probabilistic model of diffuse overland flow from roads to streams that predicts the length of hillslope required to accommodate volumes of water discharged from a drainage outlet. This analysis was based on the statistical distribution of measured volumes of water entering an area of hillslope before discharge is observed at the downslope boundary of that area, termed the volume to breakthrough (Vbt). Hairsine *et al.,* (2002) considered a 5 m length of hillslope (Vbt5) receiving redirected runoff from large scale rainfall simulation experiments on 9 logged hillslopes in southern NSW and East Gippsland, (Croke *et al.*, 1999a, b). These experimental areas are characterised by a temperate climate with 600-900 mm mean annual rainfall. The vegetation is dominated by open mixed-species eucalypt forests with shallow-moderate rooting depths. Soils are granite derived with depths of 1-2 m, or shallower, gravelly metasediments (< 0.75 m depth).

As the model was developed using the experimental data from the above sites, its utility in other environments is open to question without validation. In this paper we extend the previous analysis to the wet forest environments of the mountainous area of Central Victoria where the predominant species is deep-rooting mountain ash (*Eucalyptus regnans*). By this extension we examine the portability of the predictive capability of the model proposed by Hairsine *et al.,* (2002).

Secondly, we investigated the impact of wildfire on Vbt in the North East of Victoria. Development of soil hydrophobicity may occur following fire, and surface conditions are altered through loss of litter and grass and shrubs. Changes in the ability of hillslopes to infiltrate concentrated flows could result in longer sediment plume lengths compared with undisturbed slopes, posing an increased threat to water quality.

2. Methods

2.1 Study Sites

The unburnt experiments were carried out in the Upper Tyers catchment of the Victorian Central Highlands (Fig. 1). The area forms part of the timber resource of Victoria and the dissected terrain is traversed by numerous forest roads. The granodiorite derived soils in the Upper Tyers are very deep to giant acidic clay loam red or brown ferrosols or dermosols (Isbell, 1996), characterised by a sandy clay loam A horizon with an average depth of 0.4 m, overlying a thicker sandy clay loam B horizon (2-4 m), and a loamy sand C horizon that grades into saprolite at 4-8 m. Most B horizon depths are < 3 m, with a deepening of the profile from upper to lower slopes. While the A and B horizons have a low soil erodibility rating, the C horizon is susceptible to breakdown due to slaking (Sheridan *et al.*, 2001). In their undisturbed state, the soils have very high saturated hydraulic conductivities (Ks); B horizon Ks has been measured as > 5 m d^{-1} (Lane *et al.*, 2004) and A horizon values most likely to exceed that figure. Rainfall simulation in an adjacent catchment on the same soil type with intensities up to 110 mm h^{-1} did not produce surface runoff (Lane *et al.,* 2004). By comparison, Ks measured by Croke *et al.,* (1999a) on granite derived soils was < 1.5 m d^{-1}. The hydraulic properties and storage capacities of the Victorian Central Highland soils represent an extreme for forest soils in southeastern Australia, and consequently this study is a tough evaluation of the Hairsine *et al.,* (2002) model. Average annual rainfall is estimated to be 1800 mm with a uniform monthly distribution. Rainfall intensities are moderate, the 1:50 year 1 hour duration intensity is 45 mm per hour.

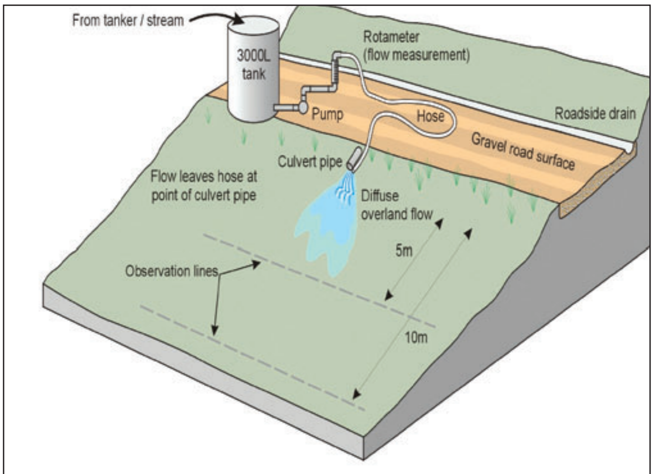

Figure 1. Experimental setup for Vbt study.

The North East burnt sites are situated in the Kiewa Valley, near Mt Beauty. The granodiorite and gneiss derived soils are generally <1.5 m deep, characterised by clay-loams and sandy clay-loams grading to coarse sand C horizons. Infiltration studies on unburnt slope returned values > 130 mm hr-1 (Sheridan *et al.*, 2004). Mean annual rainfall is around 1800 mm.

2.2 Vbt Experiments

The experiments in the Upper Tyers were performed at culvert outfalls so as to simulate actual road drainage. Sites were selected to be representative of the range of cutslope gradients and surface conditions (leaf litter, vegetative obstacles) in the area. This included steep (> 20°) and low (<10°) outfall gradients (Table 1). The 19 sites were located along four short road (< 5 km) lengths roads. Access and traffic safety considerations restricted the sample population and the outfalls chosen represented those that could be used.

Water was applied at the rate of 3 l sec-1, pumped from a 2400 litre reservoir and metered by a rotameter (Fig. 2). A water truck with 4000 litre capacity refilled the tank as it drained, giving a total volume of 6200 litres available for the experiments. On two occasions where it was possible to pump directly from a stream the available volumes was increased. Volume to breakthrough (Vbt) was measured for two lengths, 5 and 10 metres, denoted as Vbt5 and Vbt10, respectively. The width of flow was measured at several locations, and an estimate of the discharge rate was made at 10 metres following breakthrough. This was achieved by vacuuming all overland flow into a volumetric flask over a period of one minute. The vacuum was rested on the mineral soil surface so as to capture all overland flow crossing the 10 m transect.

The Kiewa experiments were performed on hillslopes without drain outlets. Measurements were made on burnt slopes approximately 1 month after the fires, and repeated at 6 and 12 months, with a set of experiments on unburnt slopes at 6 months.

3. RESULTS

To compare the Vbt5 results of Hairsine *et al.,* (2002) with the Upper Tyers data, a Mann-Whitney U test was performed. This non-parametric test was required as both data sets contained runs where there was no recorded breakthrough. These were not zero values but "greater than" values where there was too little water to achieve breakthrough. These are known as "censored data" and cannot be analysed parametrically.

For the unburnt cases where Vbt5 was recorded, the mean (μ_{vbt5} = 543 L) and variance (σ_{vbt5}^2 = 545481) of the Upper Tyers data exceeded those of the Hairsine *et al.,* (2002) data, (336 L, and 35600, respectively). When the censored data was considered the Mann-Whitney U test did not reveal a significant difference in the means of the two populations ($p = 0.314$). If an extreme value from the Upper Tyers is removed, the means are almost identical. These results indicate, despite the differences in experimental design, the Vbt5 distribution obtained for the Hairsine *et al.*, (2002) data could be used as an input for modelling in the very different soils of the Upper Tyers.

Relationships between both Vbt5 and Vbt 10 and the following variables were investigated; slope, flow width, contributing area to the drain, existing plume length and presence/absence, and existing channel length and presence/absence. There were no significant correlations between any of the variables and Vbt5 or Vbt10 for the whole data set. For example, slope and flow width returned coefficients of determination of Vbt5 (r^2 = 0.0, 0.11) or Vbt10 (r^2 = 0.23, 0.01). However there are some small groupings of sites where particular variables appeared to be

Table 1. Volume to breakthrough, flow measurements and site details for Upper Tyers.

Site	Vbt5 (L)	Vbt10 (L)	Surface gradient° 5m	Surface gradient° 10m	Flow width 5m	Flow width 10m	Existing plume length (m)	Length channel incision (m)	Discharge rate (l min^{-1})	Catchment area (m^2)
1	864	2280	28	27	0.4	0.2	0	0	7.00	175
2	177	2979	16	15	0.3	0.1	1.1	0.6	0.08	232
3	330	NF	4	9	0.5	NF	0	0		103
4	258	1620	8	8	0.5	0.25	0	0	39.00	160
5	735	7755	29	22	0.15	0.4	1.0	0	0.04	205
6	3360	NF	22	20	0.15	NF	0.6	0		75
7	NF	NF	3	18	NF	NF	0	0		200
8	156	672	25	20	0.3	0.2	10.0	0	2.20	300
9	147	507	29	25	0.4	0.5	10.0	2.5	8.40	142
10	156	NF	21	22	0.5	NF	1.0	1.5		102
11	123	660	30	30	0.35	0.15	1.8	0	29.50	281
12	240	5760	28	28	1.0	0.2	1.5	0	7.20	186
13	1020	3690	17	17	0.2	0.1	0	0.7	4.40	240
14	261	1221	13	13	1.0	0.3	4.2	0	23.00	581
15	135	2250	22	22	0.15	0.15	1.8	2.2	0.02	325
16	540	1350	11	24	0.15	0.2	1.0	0	8.70	180
17	135	930	12	12	0.5	0.5	2.2	1.9	10.00	300
18	666	2160	4	12	0.3	0.2	1.8	0	unknown	264
19	240	4320	23	23	0.5	0.15	0	0.7	1.25	405

important. The lowest combination of Vbt5 and Vbt10 were sites 8, 9 and 11. The hillslope surface at sites 8 and 9 were armoured by a very fine sediment skin over leaf litter, most likely a result of accumulated road drainage deposition. Site 11 appeared to exhibit noticeably wetter antecedent conditions than the other sites. The other sites with Vbt5 < 200 L (2, 10, 15 and 17) maintained concentrated flow to 5 m. The flow split into two or more plumes at 9 of the remaining 12 sites prior to 5 m. Of the other 3 experimental runs (3, 7 and 18), slope gradient may have been a factor (Table 1). There was no breakthrough at site 7 where slope was only 5° and the surface was particularly dry and friable. The variability in Vbt10 was largely dependent on the influence of obstacles (logs, leaf litter, plants) and depression storage that acted to slow and disperse concentrated flow, and losses to large macropores. Table 1 shows a low Vbt5 did not strongly correlate with a low Vbt10. The mean ratio of Vbt10 to Vbt5 was 8.4.

The results from the Kiewa cannot be directly compared with the other two data sets because of the experimental design (no antecedent plumes) and smaller sample population. However it is clear that the fire had a significant impact on slope hydraulic characteristics. Mean Vbt5 was 63 L immediately post-fire, 50 L at 6 months and 108 L at 12 months. The mean unburnt value from completed runs was 3612 L, with no breakthrough in 2 out of 6 runs. The Vbt10 values for those time periods were 390, 171 and 321. The latter is the mean of completed runs, there was one run where water did not breakthrough to 5 m. There was no breakthrough at 10 m in the unburnt runs. The decrease in Vbt5 and Vbt10 over the first 6 months following the fires was mirrored by observations of decreasing infiltration (Sheridan *et al.*, 2004).

4. DISCUSSION

The key finding of this study is that the volume to breakthrough over 5 metres for the mountain ash environment was indistinguishable from that of the mixed species eucalypt environment measured by Hairsine *et al.,* (2002). By extension, the model proposed by Hairsine *et al.,* (2002) could be employed to predict plume lengths in this environment, indicating the model is portable to a range of forest and types. Hairsine *et al.,* (2002) provided an example of a design application in the form of a probability graph, in which a minimum hillslope length required to maintain a 95% probability of plumes not reaching a stream as function of runoff generated from tracks under 2, 10 and 100 year storm recurrence intervals. A similar application can be provided for individual catchments using the appropriate storm intensity-duration curves. These predictive tools can be used in two ways; altering the drain spacings (and therefore catchment area) for existing roads to maintain hillslope availability, and in the siting of new roads and tracks where hillslope lengths can be maximised.

The results from the Kiewa catchment strongly indicate that such design applications and existing prescriptions for road and track and clogging coupe drainage may not be adequate under fire disturbance. The hillslopes in the experimental area received medium to severe burning in January 2003. Mortality of *E delegatensis* stands has resulted in salvage logging in this, and several other, catchments. The Vbt results suggest plume lengths would have increased substantially per unit of runoff in the first 6 months after the fires, and, in the light of widespread destruction of streamside buffers, increased the probability of connection between drainage points and streams. The 12 month observations indicate a degree of recovery, but the mean values are still < 30% of the other data sets, and only a fraction of the unburnt case. Although there has been a recovery of low vegetation, particularly bracken and some grasses and shrubs, the depletion of normal litter loads may hinder detention of flows.

The volume to breakthrough concept has proven valuable in quantifying hillslope drainage characteristics in disturbed and undisturbed conditions and demonstrate the utility of the Hairsine *et al.,* (2002) approach to road and track drainage design.

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Investigating the Effects of Bushfire on Catchment Water, Sediment, and Nutrient Yield from Forested Catchments

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Summary: The 2003 fires in NE Victoria burned 1.3 million ha of forest. Short term changes to water quality and short to long term alterations to catchment yields and flow regimes may be expected. Two previously instrumented research catchments in the East Kiewa Valley have been recommissioned to provide observations of discharge and sediment and nutrient loads. Rainfall simulation, unconfined flow experiments and soil hydrophobicity surveys have been undertaken to yield insights into runoff and sediment and nutrient generation processes. Observations will be used to improve the parameterisation of models representing post-fire sediment and nutrient production. These intensive small catchment observations will be combined with analyses and modelling of water quality, and modelling of yield at larger spatial scales to quantify and predict the hydrologic impact of the fires over much of the burnt area.

1. INTRODUCTION

The largest bushfires since 1939 burned 1.3 million ha of forests in NE Victoria during January and February 2003. Included in this area are the Upper-Murray, Kiewa, Ovens, Snowy, Tambo and Mitchell river basins. Although elevation of sediment and nutrient loads are widely recognised as consequences of intense wildfire, relatively few studies have been carried out Australia (eg. Prosser and Williams, 1998, Leitch *et al.,* 1984; Chessman 1986; Burgess *et al.*, 1981). Consequently the spatial and temporal distribution of impacts, and the processes driving mobilisation and transport are poorly known. Models exist for the prediction of sediment and nutrient transport, but there is little high quality data from post-fire forested ecosystems with which models can be correctly parameterised and evaluated. Such information is critical for predicting the consequences of wildfire for treatment and allocation of water supplies, "end of valley" sediment and nutrient loads, and river health targets. Likewise, the impact on catchment water yield is of critical importance to catchment managers. Increased peak flows are often thought to occur in the period prior to recovery of evapotranspiration, but the longer term impact may be considerable. Approximately 20% of the burnt forest was comprised of *Eucalyptus delegatensis* (Alpine Ash). This species has similar regeneration response to fire disturbance as *E. regnans* (Mountain Ash), which results in decreased yield from regrowth stands (Langford, 1976, Vertessy *et al.,* 2001). To address some of the foregoing issues, two de-commissioned research catchments in the East Kiewa Valley were re-instrumented immediately following the fires.

2. PROJECT AIMS

A. **Quantify the fluxes of sediment and nutrients from small forested catchments following bushfire,** including the immediate post-fire "flush" and the longevity of impacts:

- This entails the measurement of catchment discharge and sediment and nutrient concentrations

B. **Identify the physical processes and properties of the system driving changes in pollutant exports.** These processes fall into three categories: hydrologic processes, sediment detachment and transport, and nutrient mobilisation and transport.

- *Hydrologic processes* – quantifying the temporal changes in infiltration, hydrophobicity, canopy interception and evapotranspiration that will drive changes in sediment and nutrient exports.
- *Sediment detachment and transport* – identify and quantify the hillslope and in-channel processes that

generate sediment and nutrients, including the impact of changes in infiltration on runoff generation, rill and inter-rill erodibility and consequent mobilisation and transport of sediment and nutrients through time.

- *Nutrient mobilisation and transport* - Quantify nutrient mobilisation and immobilisation processes that contribute to increases or reductions in nutrient loads, identify the transport mechanism of nutrients by proportioning nutrient attachment to suspended sediment and to bedload, and quantify the production and export of both sediment transport phases at the hillslope and catchment scales. The transport mechanism is important in establishing in-stream residence times of pollutants and potential storages of nutrients.

C. **Improve the representation of processes within, and provide parameters for, hillslope erosion and catchment water quality models.** Numerous water quality models exist for the prediction of land-use change or disturbance on catchment exports of pollutants. These models range in spatial scale from hillslope to large catchment. Appropriate parameterisation is hindering the testing and application of these models. In particular, it is unknown whether existing models can adequately represent fire-induced pollutant generation and transport. The pollutant load and process data captured in this project, combined with water quality observations from larger burnt and unburnt catchments and fire severity and vegetation recovery mapping will allow model parameterisation at a hierarchy of spatial scales.

D. **Quantify the short term impact on flow regime**

- The short term impact on flow peaks and distribution, and persistence of any flow increased will be measured. Changes to flows will be examined in relation to observed runoff generation processes and vegetation recovery.

3. THE CATCHMENTS

The catchments are located in the East Kiewa River Valley, 20 km south of Mt Beauty in the Victorian North East alpine region. They were initially instrumented in 1978 to investigate the effect of logging on water and sediment yield. Slippery Rock catchment (136 ha.) was retained as the control, and approximately 30% of the Springs Creek catchment (244 ha.) was logged following a 3 year calibration period. The study concluded in 1987. The topography is steep (elevation 620-1520 m), mean annual rainfall is 1800 mm, and the geology is gneiss and granodiorite. Both catchments have friable brown gradational soils generally < 1.5 m deep comprised of clay-loams and sandy clay-loams grading to coarse sand C horizons.

4. METHODS

The previous research period at the catchments yielded discharge and sediment concentration observations over 10 years. These data will be compared to the current data set. Although an unburnt control catchment would be experimentally ideal, the before-and -after design is considered to be robust as the rainfall-runoff relationships from mature mixed-species and *E delegatensis* should not have changed markedly and there is no evidence of other disturbance that would influence sediment and nutrient mobilisation.

4.1 Stream Gauging, Sediment and Sampling

Discharge at Slippery Rock is measured through a 135° triangular broad-crested weir, and at Springs Creek by a Parshall flume attached to a culvert. *Turbidity* is measured on a 15 minute timestep by a Min-data backscatter turbidity probes with a range of 0-1000 NTU. Storm events are sampled by Sigma multistage autosamplers triggered by stage rise. *Bedload* is sampled continuously using 1.0 m wide flow control structures with five 5x50 mm slots in the bed delivering water and sediment to a 200 L settling tank. *Rainfall* is measured by tipping bucket gauges at high and low elevations in the catchments.

4.2 Sediment and Nutrient Analyses

Raw water samples immediately refrigerated and analysed for EC, pH and turbidity. Samples then filtered through 0.45μm nitrocellulose membrane filter, to separate filtrate and TSS. TSS and soil survey samples analysed for total Phosphorus by digest with Perchloric Acid, analysed using ICP. Filtrate analysed using HPLC-IC for; F, Cl, NO_3-,NO_2-, Br, PO_4- and

S. Filtrate analysed using ICP for; Fe, Ca, K, Mg, Na, P, S and Sr, and colorimetrically for TN using persulphate oxidation and a cadmium reduction step. Bedload analysed as per the TSS, plus OC the CHN Analyser (LECO CHN). Good relationships have been established between TSS and turbidity and between TSS and TP and TN, yielding sediment and nutrient rating curves constructed from continuously measured turbidity in the field. Estimations of exports can be made on a 15 minute timestep, ensuring hysteresis effects are fully accounted for, and problems with autosampler malfunction or excedence of capacity during long events are minimised.

4.3 Hillslope Processes

- Surface runoff, sediment and nutrient generation - rainfall simulation on 3.0m^2 plots at 100 mm hr^{-1} intensity for 30 minutes on burnt and unburnt hillslopes at 6 monthly intervals
- Infiltration and hydrophobicity – rainfall simulation for infiltration rates and field (water and ethanol method) and laboratory (MED test) hydrophobicity droplet tests performed at 6 monthly intervals.

Soils are samples at 4 depths to 25 cm at permanent soil sampling points on 9 transects with 5 sample points.

- Sediment detachment and transport by flow – confined overland flow experiments (rill) on bounded 3.0 m channels at incrementally increased flow rates for rill erodibility and for critical shear stress parameters.
- Unconfined overland flow experiments using the "Volume to breakthrough" method of Hairsine *et al.,* (2002) to examine the impact of fire on concentrated sediment plume movement.
- Ground surface and canopy vegetation surveys – these surveys are combined with the permanent soil sampling points to gauge fire severity and vegetation recovery rates.

5. HILLSLOPE AND CATCHMENT MODELLING

Objective C has two components; to provide post-fire parameters for existing models; and, to improve the representation of post-fire processes within future models.

The objective of parameterisation of existing models will be considered first. A critical consideration in the parameterisation of any model is that parameters correctly represent the spatial and temporal scales that are either explicit or implicit within the model structure. As an example, a number of possible models are listed below, approximately in order of increasing complexity and decreasing spatial and temporal scale of application;

1. Those that represent generation and export rates as a concentration as function of land-use (mass per volume runoff) eg. EMSS, CMSS and often include simple bucket model hydrological balances.
2. Those that include a hillslope erodibility parameter (mass per unit energy), often the RUSLE K factor, to generate sediment, and link this empirically with nutrient loads eg. ANSWERS, SedNet.
3. Empirical hillslope-scale annual-average (eg. USLE, RUSLE) and event-based (eg. MUSLE) erosion models.
4. Those that include a more process-based representation of runoff, and sediment detachment and transport at the hillslope scale, yet are still applicable at the broader scale eg. SWAT, HSPF.
5. Dedicated hillslope-scale, event-based erosion and water quality models with explicit dynamic representation of processes of infiltration, overland flow, detachment and transport eg. CREAMS, GLEAMS, WEPP, GUESS.

Parameters within the above models effectively "lump" processes at a spectrum of specific spatial and temporal scales. The critical point to note is that the scale of application of these models is often not at the scale at which the primary data for parameterisation can, or is, collected. To resolve this issue, some intermediate modelling steps are often required to derive parameterising datasets at the correct scale for the parameter and model in question; eg. the USLE K value. However for other models parameters may be determined directly from measured catchment export data, (eg. EMSS, CMSS) from hillslope process studies (eg. WEPP, CREAMS, GLEAMS, GUESS), and from a combination of these data sources and modelling (SedNet, USLE, RUSLE, ANSWERS, EMSS, CMSS). The proposed modelling approaches will allow the calculation of long term average parameters for models such as EMSS, CMSS, and the USLE and its derivatives (eg. RUSLE).

The second objective, to improve the representation of post-fire processes within future models, will be achieved via data from the process studies described in Method B. An iterative processes whereby models are selected, parameterised, tested, and modified depending on the capacity of the model to represent the processes that have been identified will be implemented. An example of this may be the temporal change in hydrophobicity, and how this influences runoff generation at the hillslope scale.

6. OTHER DATA SETS AND UPSCALING

In addition to the high resolution data sets from the East Kiewa there are a number of other data sets from larger catchments with water yield and quality observations. These include the West Kiewa (10,000 ha), and eight larger catchments with post-fire autosampler data. These data will be used to quantify and model broad-scale water quality impacts. For example the monthly time-step Gippsland Lakes Model (Grayson and Argent, 2002) will be updated with post fire sediment and nutrient parameters, with a similar analysis performed for the North-East.

7. YIELD IMPACTS

- *Short term impacts:* streamflows over the first three years post-fire will be compared to the previous record using analysis of flow duration curves and General Additive Modelling
- *Long term impact:* It is highly likely the most persistent, and possibly most deleterious, impact of the fires will be on water yield over a timescale of decades. Modelling is required to provide catchment managers and water authorities with estimates of the magnitude and duration of changes in yield. A modelling program with the CRC for Catchment Hydrology, Sinclair Knight Merz and the Forest Science Centre will address this issue in a three-step process: firstly the SKM ForestImpact model (Daamen *et al.,* 2003) will be run to provide first-cut estimates of annual yield changes on a regional basis. Subsequently the CRC for Catchment Hydrology Macaque model will be used on selected catchments for a more rigorous, process based analysis. The Macaque results will then be used to refine the broader-scale ForestImpact predictions, and feed into the regional water quality modelling.

8. SUMMARY

This paper sets out the rationale and experimental design for a substantial research program into the impact of bushfires on catchment hydrology and sediment and nutrient export. The program includes high resolution measurement of small catchment outputs and fundamental processes that will feed into modelling at a range of spatial and temporal scales, and compliment broader scale quality and quantity modelling for catchment management and water allocation.

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New Guidelines for the Protection of Headwater Streams in Tasmania

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Summary: The Tasmanian Forest Practices Code in Tasmania at present applies a standard 10 m machinery exclusion zone to most Class 4 (headwater) streams in the forestry estate. A new procedure for assessing erosion risks in and adjacent to streams is described. It is based on defined erosion hazard classes based on riparian slope and soil erodibility, and the identification of erosion features. Flow diagrams taking account of erosion hazard class, erosion features and proposed land use lead to five alternative prescriptions for Class 4 stream protection.

1. INTRODUCTION

At present the standard prescription for the protection of Class 4 streams in the Tasmanian forestry estate (headwater streams with a catchment of 50 ha or less) is a 10 m machinery exclusion zone (Forest Practices Board 2000), although this can be upgraded for biological conservation reasons or by a specialist's recommendation.

Davies *et al.,* (1999), in their review of soil and water provisions in the Forest Practices Code, recommended further research into forestry operations in Class 4 catchments and increased protection for Class 4 streams in some situations. Observations by McIntosh and Laffan (in press) on 112 Class 4 streams indicated that, within the stream or the 0–10 m riparian zone, the incidence of seven defined *'erosion features'* (landslides, sheet erosion, tunnel gully and rill erosion, sediment accumulation, near-vertical stream banks >1 m high, recent boulder movement, and a channel >4 m wide) was correlated with *'erosion hazard'*, defined on the basis of riparian slope and soil erodibility. For 74% of streams in coupes in which environmental risks were apparent, and which were therefore notified according to Forest Practices Board procedures, greater protection than the standard 0-10 m machinery exclusion zone was recommended.

This paper formalises the process for Forest Practices Officers to derive such prescriptions.

2. PROCEDURES

To assess actual and potential risk to a Class 4 stream it is necessary to know:

(1) the maximum slope in the 0–10 m riparian slope;

(2) the soil erodibility;

(3) the erosion hazard class; and

(4) whether erosion features are present.

Slope

The measured maximum slope should apply to at least 5–10% of the riparian zone.

Soil erodibility

Soil erodibility in the riparian zone is determined by correlating soils with those described in publications, including 'Forest Soils of Tasmania' (Grant *et al.,* 1995), and the Forest Soil Fact Sheets (www.fpb.tas.gov.au). Alternatively, soil erodibility may be determined by Laffan's field method (Laffan 2000) or by checking with a soil scientist.

Erosion hazard class

From the riparian slope and soil erodibility, erosion hazard in the riparian zone is determined from Table 1.

Erosion features

The presence of these above-mentioned features, alone or in combination indicates that a Class 4 stream and its riparian zone is at greater risk of erosion than a stream and riparian zone without these features.

Working out protection prescriptions

From the determination of erosion hazard, and the record of erosion features, the appropriate prescriptions are worked out from flow diagrams. An example is given in Figure 1. The assessment leads to five alternative prescriptions (Figure 2).

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Table 1. Erosion hazard in relation to soil erodibility class and slope angle.

Maximum slope in riparian zone*	Soil Erodibility Class				
	Low	Moderate	Moderate - High	High	Very High
<3º	A	A	A	C	C
3 - 8º	B	B	B	C	D
9 - 11º	B	B	C	C	D
12 - 14º	B	C	C	D	E
15 - 19º	C	C	D	E	E
20 - 26º	D	D	E	E	E
>26º	E	E	E	E	E

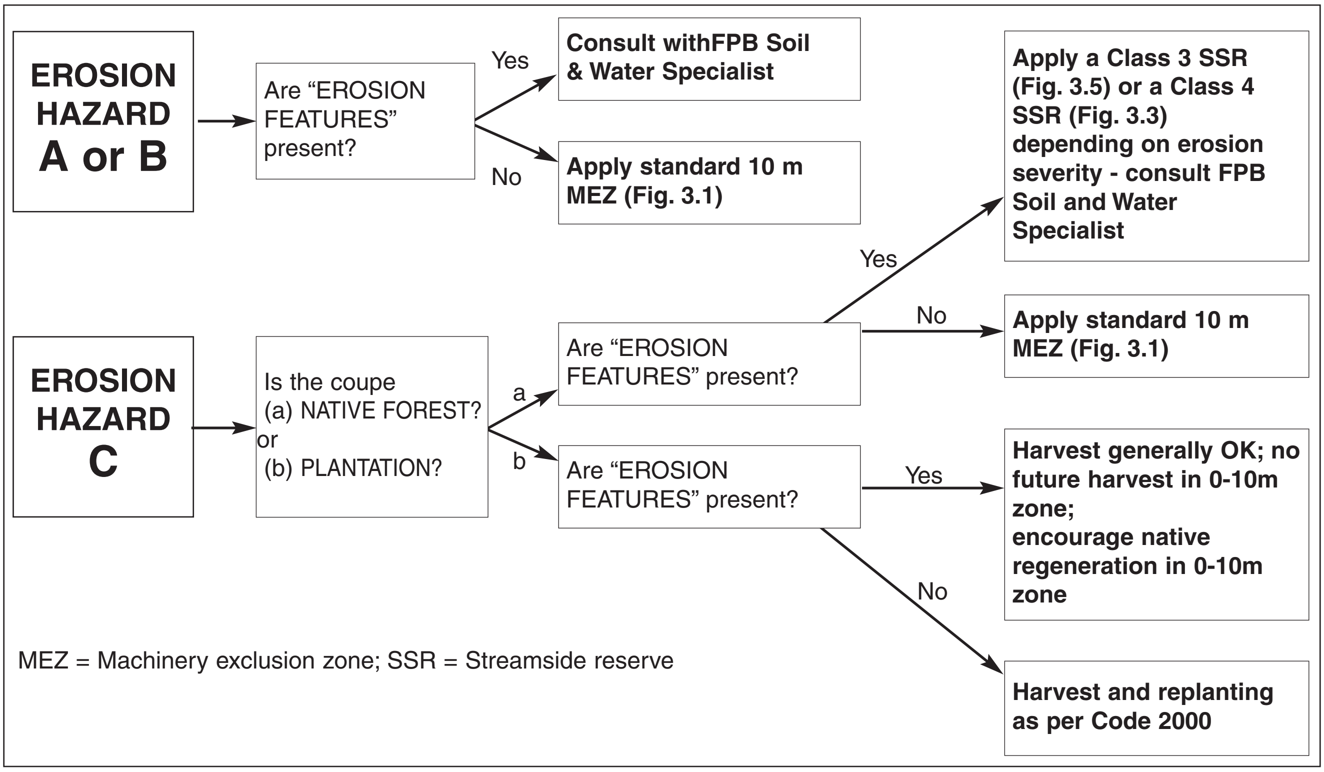

Figure 1. Example of a flow diagram for working out the appropriate prescription to apply to a Class 4 (headwater) stream.

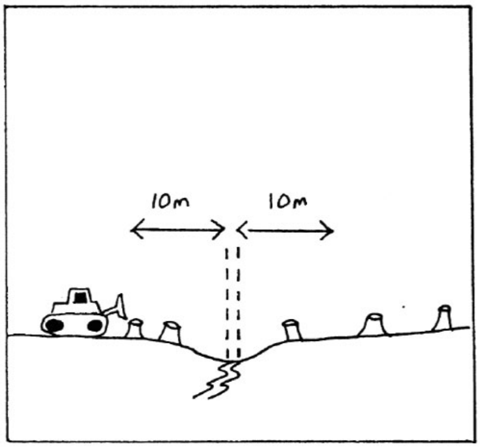

Figure 2.1. Standard Class 4 Machinery Exclusion Zone (MEZ) as defined in the Forest Practices Code (2000).

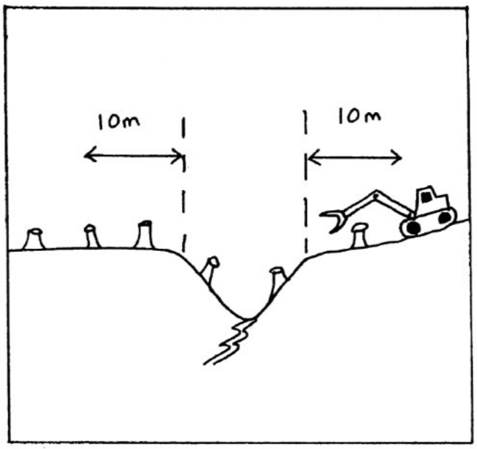

Figure 2.2. Extended Class 4 Machinery Exclusion Zone (MEZ). The 10m wide zone is measured from where the steep (>19º) gully sides begin.

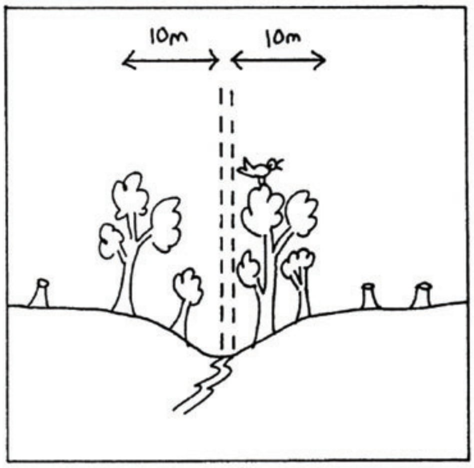

Figure 2.3. Class 4 Streamside Reserve (SSR). A 10m no-machinery and no-harvest zone is applied.

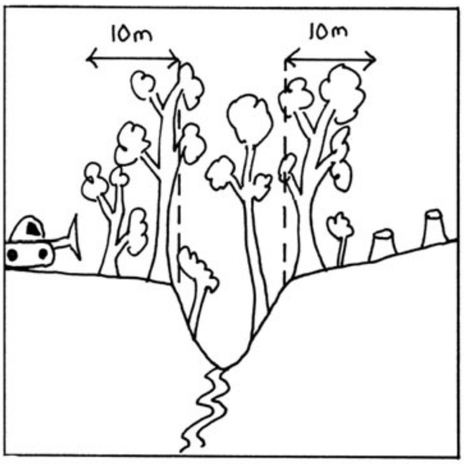

Figure 2.4. Extended Class 4 Streamside Reserve (SSR). The 10m wide SSR is measured from where the steep (>19º) gully sides begin.

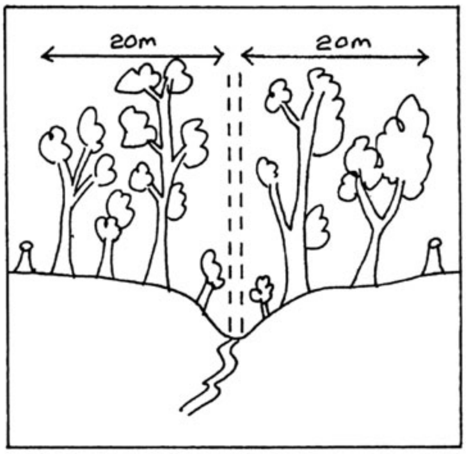

Figure 2.5. Class 3 Streamside Reserve (SSR). The Class 4 stream is upgraded to Class 3 status, and a normal Class 3 SSR as defined in the Forest Practices Code (2000) is applied.

Figure 2. Prescriptions to be applied to Class 4 streams, based on erosion hazard and erosion features assessment.

Rehabilitation Success on Landings: Are Physical Properties or Nutrient Status the Limiting Factor?

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Summary: Landings are one of the most difficult parts of a harvested coupe to rehabilitate, and conventional wisdom attributes this largely to compaction and heavy soil disturbance. In recent years, ripping and respreading of topsoil have been practiced, in accordance with the Code of Forest Practice, to attempt to improve regeneration on landings and other compacted and disturbed areas of the coupe, but problems continue. This project was initiated after several small operational trials showed a strong response to fertiliser on landings and extraction tracks. Two newly-harvested coupes were chosen at each of two locations (Otways and Central Highlands). After standard rehabilitation work and seeding had been done at each site, three repetitions of a matrix of nine different fertiliser treatment plots were set up on landings at four sites. Fertilisers were applied at rates of 0, 100 and 200 kg/ha N and 0, 50 and 100 kg/ha P. Similar matrices were set up in the general harvest area (GHA) of each site. Preliminary results collected six and eighteen months after establishment show a significant response to both N and P on landings. This is seen as an increased height of eucalypts, and reduction in the area of bare ground. There was some response to P on the GHA, but not to N, and the response by eucalypts in particular was weaker than on the landings. The results from this study indicate that regeneration on landings used in forest harvesting may be limited as much by soil nutrition, as it is by soil physical constraints. It is indicated that better eucalypt height growth and bare ground cover can be obtained at 18 months by the application of NP fertiliser at sowing, with optimum rates indicated as being 200kg/ha N and 100 kg/ha P. This response is comparable or better than that of the GHA at this age. Continued monitoring will be required to determine whether this is a lasting response. These findings have lead to a revision of landing establishment and rehabilitation practices in Victoria for use on landings that cannot be corded and matted.

1. INTRODUCTION

A landing, as part of a harvested coupe, is a place where trees or parts of trees are extracted for sorting and processing and loaded for transport from the forest. Rehabilitation is the restoration and revegetation of a site of disturbance usually associated with forest road works and landings (EPA Victoria 2003). During routine harvesting operations, the establishment and use of landings can result in impacts to the soil profile and soil structure. Without rehabilitation these impacts can significantly effect subsequent revegetation. Studies by Jacobsen (1983), King *et al.,* (1993) and Rab and Kelly (2002) have established that in the absence of rehabilitation impacts on soils and vegetation establishment and development can last for many years. A recent EPA audit of 30 Victorian coupes harvested during the 2002-03 operational year has indicated that the average area used for landings was just 0.05 ha, which is well below the 0.5 ha permitted in ash-type coupes (EPA Victoria 2003). Nonetheless with usually more than one landing per coupe and over 450 coupes harvested per year the area of impact could be in the order of 50 ha. In Victoria, the Code of Practices for Timber Production – Revision No. 2, 1996 (NRE 1996) requires that surface soil be stockpiled prior to the construction of the landing for use during rehabilitation, as well as a requiring that landings be located on an area where minimal soil disturbance is likely. Rehabilitation of landings as well as including the replacement of topsoil, also includes bark removal, levelling, draining and ripping to loosen compacted soil to a depth of at least 0.4 m with ripping lines no more than 2 m apart. The main goal of the ripping is to disturb the compacted subsoil, and this is usually achieved by either bulldozer-mounted tine(s) or the tines on excavators used for loading logs onto trucks. Routinely there is no fertiliser applied as part of landing rehabilitation.

Over the last few years in Victoria's more productive and moderately sloped forests landings (and primary extraction tracks) have been 'corded and matted' to reduce soil damage during the wetter times of the year. This technique generally relies on using cut-to-length pulpwood to cord the landing and bark to mat it. Where this technique is used soil compaction and profile disturbance is minimised, but unfortunately this technique can only be routinely used on a relatively small proportion of landings.

This study was initiated after several small operational trials showed a strong revegetation response to fertiliser on landings and extraction tracks, indicating that there was scope to improve conventional approaches to landing rehabilitation. The study has the general aim to improve regeneration on the landing to a level that is equal to or better than regeneration in the general harvest area. More specific objectives are to determine an appropriate fertiliser regime (nutrients and application rate), and whether the effects of fertiliser are sustained in the long-term.

2. METHODS

Four recent coupes were chosen as study sites, three in the Otway Ranges (named SSP and Egan's Track) and two near Powelltown (Big Creek and Syd Creek) in the Central Highlands. Both these sites are mountainous areas, with elevation varying between 290-800 m, and slopes between 5-30º. Soils at all sites were friable brown gradational, acidic, with an A horizon of 10-30 cm and a slightly heavier B horizon 100-200 cm deep. The soils were generally fertile, well structured and permeable. Both sites experience high rainfall, with an annual median rainfall of 1930 mm at Weeaproinah (near the Otways sites) and 1475 mm at Powelltown (Bureau of Meteorology, 2002). Powelltown is generally warmer than Weeaproinah, with annual mean daily temperature ranges of 14.4ºC - 7.7ºC at Weeaproinah and 18.3ºC - 6.7ºC at Powelltown. Forest types supported by the study sites were generally tall open forest, with closed forest in nearby gullies. All sites were in EVC 30 (Wet Forest) except for Syd Creek which was in EVC 29 (Damp Forest) (NRE 1996b). The SSP and Big Creek sites were located in pure *Eucalyptus regnans* stands, but the other study sites were in mixed species stands. All sites were harvested between March 2000 and May 2001. Harvesting was by clearfall, the standard method in both locations. Following harvesting, all sites were slash-burnt to some extent except for SSP site, but burning was much more intense at the Powelltown sites. The Otways sites were all mechanically disturbed before sowing. Landing areas were ripped, and topsoil spread on landings to a greater or lesser extent. Sites (including landings) were regenerated by direct seeding, either aerially (Powelltown) or by hand (Otways). Seeding occurred between March and May 2001, and fertiliser was applied to the trial plots in April/May 2001. While there is limited published data relating to the use of fertiliser on landings, in natural systems nitrogen and phosphorus are the most likely limiting nutrients (Wang *et al.,* 1996). This study involved nine treatments with three different application rates of nitrogen, as ammonium sulfate (0 kg/ha, 100 kg/ha and 200 kg/ha) and phosphorus, as triple superphosphate (0 kg/ha, 50 kg/ha and 100 kg/ha) assigned to a matrix of 5x5m plots in a randomised block design. At each coupe there were three blocks on the designated landing and three blocks in the general harvest area (GHA). Measurements were made on 4x4m plots with 0.5m buffers.

3. RESULTS

At establishment of the study the surface cover of each rehabilitated landing was visually assessed either as, topsoil, subsoil, bark or wood (other), as outlined in Table 1.

Table 1. Surface cover of rehabilitated landing at establishment of study for each site.

	Surface Cover (%)		
Site	**Topsoil**	**Subsoil**	**Other**
Syd Creek	70	23	7
Big Creek	43	40	17
Egan's Track	19	57	24
SSP	3	97	0

Preliminary results collected at 6 and 18 months after establishment, show that the most useful dependent variables to describe differences between the plots were 'Eucalypt height' and 'Bare ground'. Using these two variables, a comparison of the rehabilitated landing (no fertiliser) with the GHA (no fertiliser) at 18 months indicated some significant differences, as shown in Figure 1.

The response to fertiliser in the GHAs at 6 months was much weaker than on the landings, which is not surprising given the better performance of the GHA controls compared to the landing controls. On GHA eucalypt height and bare ground generally responded to P, but only very weakly to N (not shown).

On landings, eucalypt height and bare ground indicated a response to both N and P at 18 months. This is seen as an increased height of eucalypts, and reduction in the area of bare ground, as shown in Figure 2. A comparison of Figures 1 and 2 indicates that eucalypt height and bare ground responses on the better performing landing treatments are at least comparable to performance on the GHA (Controls) at each site.

4. DISCUSSION

This study found that in the absence of fertiliser there were notable differences between the landings and the GHA in terms of bare ground and eucalypt height. Across all sites there was more bare ground on the landing and this was significantly greater for SSP (Fig. 1b). Also the average eucalypt height was greater in the GHA for all sites and this was significant at Big Creek and Syd Creek (Fig. 1a). Additional support for this finding has been reported for a coupe study in *Eucalyptus regnans* forest monitoring soil physical properties and eucalypt growth for 10 years after harvesting (including landing rehabilitation) (Rab 2004, and Rab and Kelly 2002). The study found that for a 10-year old landing soil physical properties (organic matter content, bulk density and macroporosity) values were not significantly different from that of undisturbed areas in the 0-300 mm soil depth. For macroporosity values had actually significantly increased after 10 years, and for bulk density values had shown significant recovery. However, in relation to tree growth, the undisturbed area produced saplings with DBHOB growth significantly higher than that observed on the landing (by a factor of 2.1). When this was combined with height, the height to DBHOB ratio, a measure of sapling vigour, was found to be highest on the landing (indicating reduced vigour). This supports the proposition that even though routine rehabilitation is effective in restoring soil physical properties there is still a significant impact on eucalypt growth.

The trends in treatment response are similar between sites, with generally a stronger response to N and P in combination and a more variable response to P or N alone (Figure 2). At Big Creek and Syd Creek there was a greater response to P than N. Whilst at SSP and Egan's Track this was reversed with a better response to N than P. Overall, at 18 months the optimum responses were obtained from either 200N50P or 200N100P, with the later marginally giving a better 'bare ground' response. This response of the optimum performing treatments is clearly comparable or better than that of the GHA at this age (Figures 1 and 2). Continued monitoring will be required to determine whether this is a lasting response. The impact of topsoil spreading is unclear, with controls indicating little response to this. However, treatment response trends do follow topsoil surface cover, as outlined in Table 1.

The weaker response to fertiliser on the GHAs than on the landings is not surprising and probably reflects the generally better nutritional status of the GHAs, where soil mixing is on average lesser than on landings.

5. CONCLUSION

The results from this study indicate that regeneration on landings used in forest harvesting may be limited as much by soil nutrition, as it is by soil physical constraints. It is indicated that eucalypt height growth and bare ground cover at least comparable to that on the GHA can be obtained at 18 months by the application of NP fertiliser at sowing, with optimum rates indicated as being 200 kg/ha N and 100 kg/ha P. Even though these findings are preliminary, they have lead to a revision of landing establishment and rehabilitation practices in Victoria for use on landings that cannot be corded and matted.

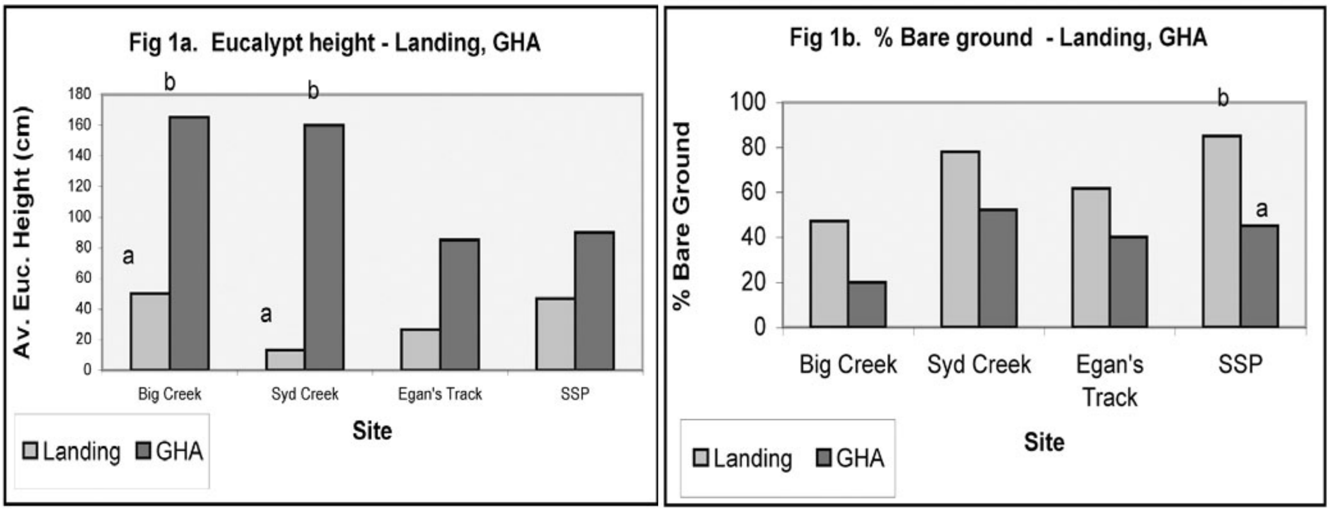

Figure 1. Comparisons between landing control and GHA control at 18 months for each site in terms of 1a. eucalypt height and 1b. bare ground. Letters indicate significant differences (T-test, P<0.05).

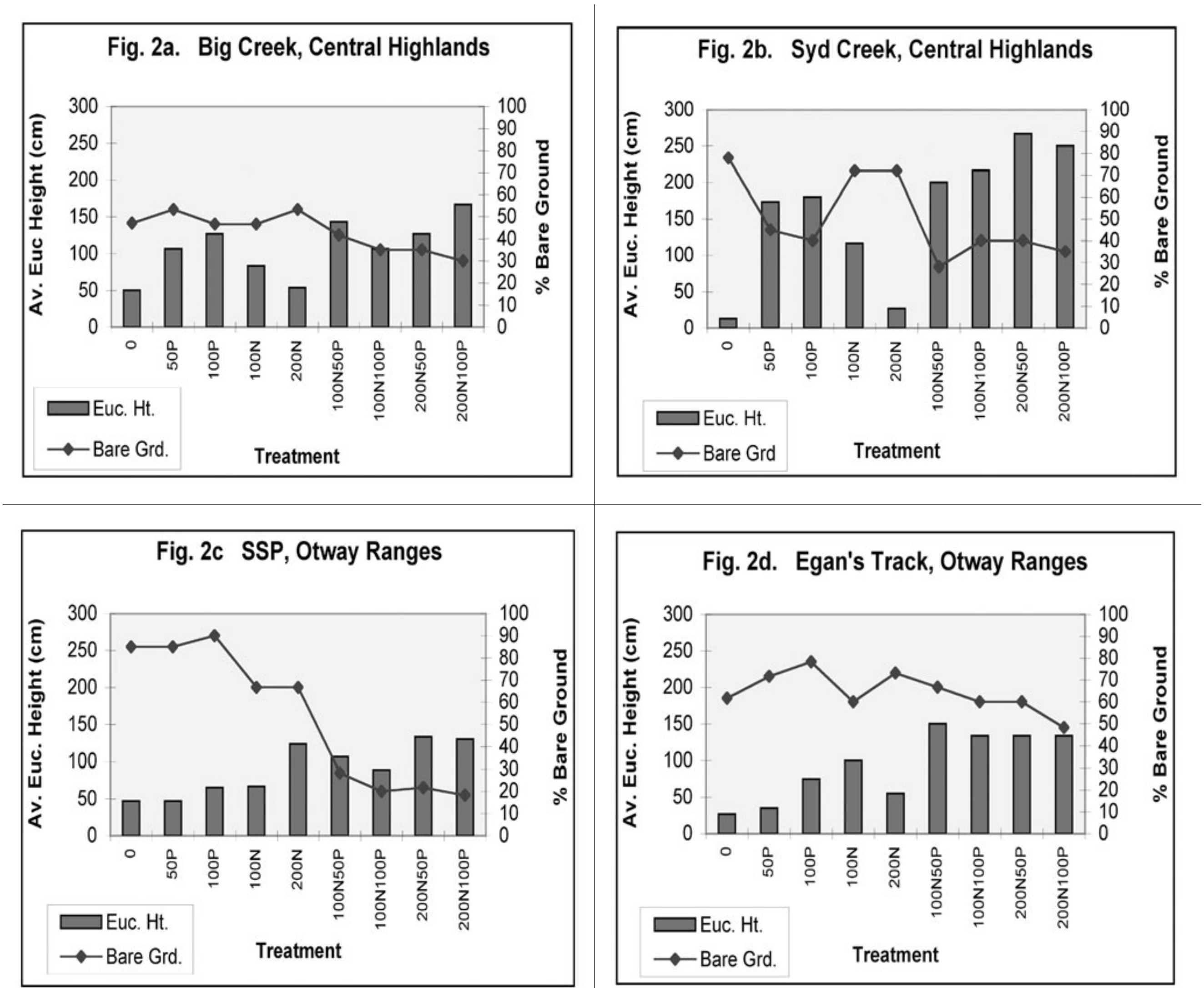

Figure 2. Effects of fertiliser and rate on landings for each site in terms of eucalypt height and bare ground at 18 months.

Plate 1. Syd Creek response on landing at 18 months.

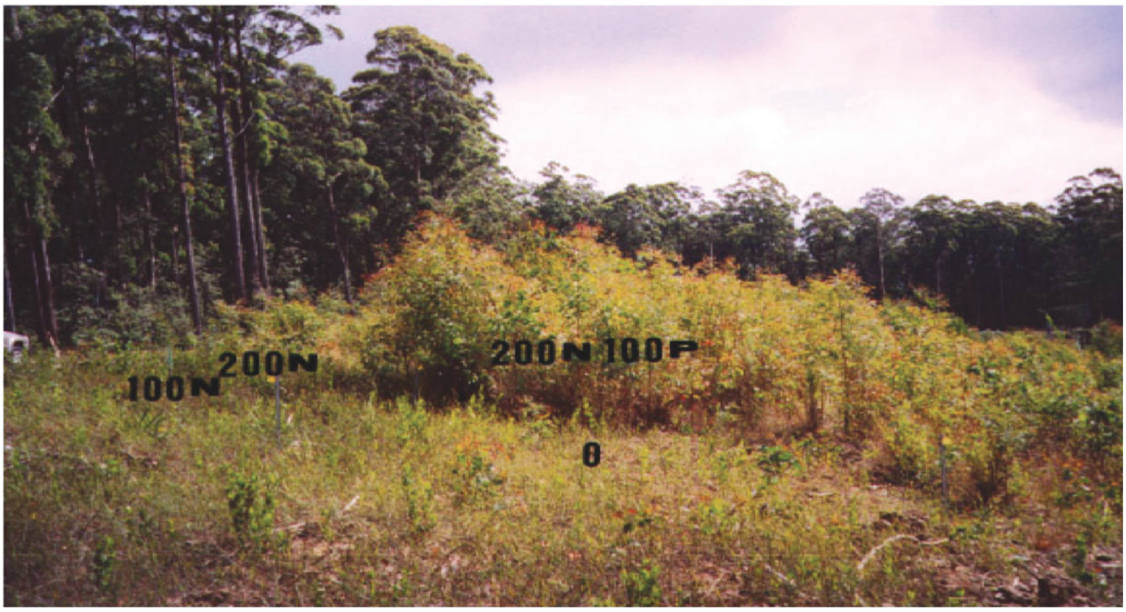

Plate 2. Egan's Track fertiliser response on landing at 18 months.

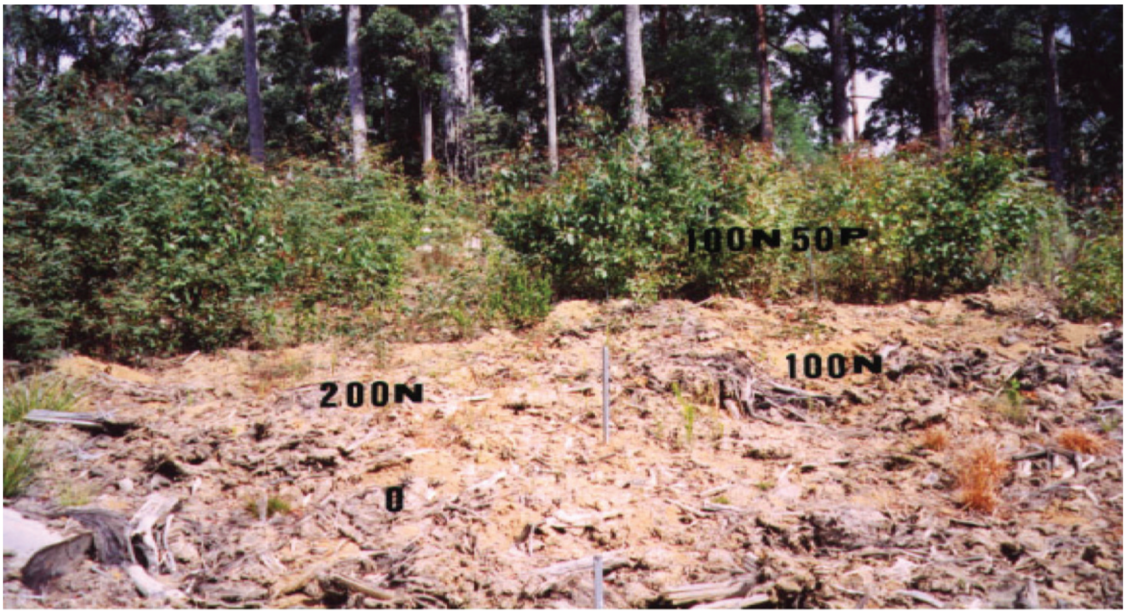

Plate 3. Egan's Track fertiliser response on landing at 18 months.

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Experimental Measurement of Water Quality and Erosion Parameters in Forests: Some Experimental Methods

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Summary: Turbidity probes have been utilised throughout world as a relatively inexpensive and easy method of measuring impact of catchment disturbances on water quality. A relationship can be built between turbidity and total suspended solids for a particular catchment. However continuous monitoring of streams with turbidity probes is difficult because of interference from debris, turbulence, shifting steam bed, fouling of probe lens, shallow streams and damage during high flow, all potentially causing erroneous results. Methods and equipment have been developed to improve in-situ turbidity probe resolution. These include new ideas in the mounting of probes, the fabrication of off line stilling tanks, and the use of new data logging and turbidity probe technology.

1. INTRODUCTION

Accurate and continuous measuring of stream turbidity is central to determining the impact that forestry practices and bush fires have on water quality. In the monitoring of water quality it is desirable to measure the turbidity of a stream as a surrogate for total suspended solids. It is advantageous to measure turbidity to construct total suspended solids (TSS) rating curves as it has been proven superior to the traditional methods of relating stream discharge to TSS (Lewis 1996). In particular, problems with hysteresis in the discharge-TSS relationship are avoided. Turbidity probes have latterly been marketed as a “set and forget” instrument that will provide good quality data with little effort. However the measurement of turbidity is often more difficult than expected, especially in flashy headwater streams that are turbulent, often changing in depth, and containing debris, all of which potentially interrupt continuous monitoring and produce erroneous results. These difficulties usually occur during peak flow events, this is when large increases in turbidity occurs, and is when a majority of the sediment is displaced within a catchment. Outlined in this paper are some of the difficulties encountered in the measuring of stream turbidity and the methods used to overcome some problems.

2. INFRA-RED TURBIDITY PROBES

The turbidity of a solution can be determined by using optical electronic devices. Most field based turbidity probes manufactured use a cone of infra-red light that is emitted into the surrounding solution, and the amount of light reflected back is detected by sensors that are positioned on the probe close to the light source. The in-situ turbidity probes that we have employed measure change the amount of infra-red light transmitted to the amount of light that is reflected, which is proportional to water turbidity. This measurement is then converted into 4-20 mA signal output to the data logger, and the probe is scaled to cover a turbidity range of either 0-10, 0-100, 0-1000, 0-10000 NTU. However at very high turbidity (>1000 NTU) the linearity of the relationship between the probe turbidity reading and total suspended solids is lost.

3. ISSUES

We have used turbidity probes in several long-term studies involving the impact of forest road erosion (Lane and Sheridan, 2002) and fire (Sheridan *et al.*, 2004). These and other unpublished studies have yielded considerable insights into the utility and caveats associated with turbidity measurement.

The major issues that we have identified include:

- Exceedence of range;
- Calibration;
- Time averaging;
- Lens fouling;
- Probe mounting (overcoming interference from turbulence and debris).

4. EXCEEDENCE OF RANGE

In the heavily fire disturbed catchment study (Sheridan *et al.,* 2004) the monitored turbidity in the streams range from <10 NTU at baseflow, to >30000 NTU at peak flow. If the baseflow turbidity is doubled (ie. a 100% error) the annual sediment load discharging increases by only 14%. However if the turbidity is doubled during storm events where the turbidity is >1000 NTU, the annual sediment discharge increases by >70%. This illustrates the importance of accurately capturing turbidity at peak flow storm events, rather than small inaccuracies at baseflow. Increasing the range leads to a corresponding decrease in the resolution. Range selection depends on the objective of the study, however it is important to note that one instrument will not provide both high resolution and high range. One solution to this would be to install two probes, one for baseflow measurement the other for peak flows.

5. CALIBRATION

Probes require calibration to check for, and correct, component drift over time. Calibration is difficult and costly because the solutions for calibration are not commercially available and therefore require time consuming laboratory procedure's to produce. Two point calibration of probes require the immersion of the probe into two Formazin solutions of a known turbidity, then assigning appropriate values to the logger. Two point calibration uses a full scale e.g. 1000 NTU solution and 0 NTU solution. Formazin solutions (especially the dilute lower NTU solutions) also have a short shelf life (days) and therefore need to be prepared at the time of calibration. Another method of 2 point calibration assumes correct factory calibration of the probe, the logger channel is then calibrated using a loop calibrator which produces a 4 mA and 20 mA currents to the turbidity probe logger channel, and a value of 0 and 1000 NTU is assigned respectively. However, we have found that despite careful calibration by the above methods, erroneous values of turbidity are obtained from the probes.

It is important to determine the relationship between turbidity and the actual TSS of the stream, because different shaped and sized particles reflect light differently. Naturally occurring sediment being transported in a stream will differ in size and shape from the Formazin white polymer solution, and therefore reflect light differently (Gippel 1989). The installation of an autosampler allows simultaneous turbidity readings and sampling, these samples can be analysed for turbidity and TSS in the laboratory. From these measurements a relationship can be built between field and laboratory turbidity, and laboratory turbidity and TSS (Figure 1 and Figure 2).

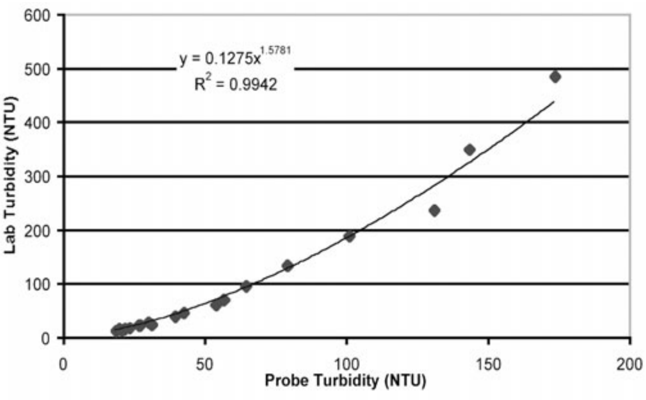

Figure 1. Relationship between lab turbidity (autosampler) and field turbidity.

6. TIME AVERAGING

Single point measurement of turbidity has the potential to be erroneous due to debris or turbulence obstructing the light path at that time of reading. Time averaging involves taking multiple readings eg. six readings every twenty seconds then averaging these values to produce one reading. By this method an erroneous measurement is averaged out before the result is written to data to the logger, and is in effect equivalent to taking systematic replicates. The disadvantage is that considerable more power is used.

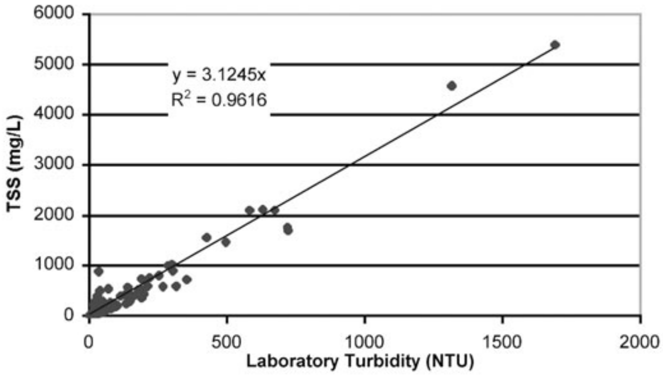

Figure 2. Relationship between field turbidity and laboratory turbidity.

7. LENS FOULING

One of the technical limitations of in-situ continuous turbidity measurement is the fouling of the lens with fine material and algae, causing a steady increase in turbidity readings over times (Figure 3). This phenomenon seems to occur in both clean and turbid water and in cold and warm conditions (Gippel 1994). Several manufacturers supply lens washing kits that periodically spray the lens, however a recent advancement involves the addition of a solenoid activated lens cover that only opens while the reading is being taken. Experience in a number of mountain locations indicates that the drift associated with algae growth is practically eliminated, and very little drift is detected over long periods of non-cleaning.

8. PROBE MOUNTING (INTERFERENCE)

The use of probes in small headwater mountain streams produce two common problems: maintaining sufficient depth of water column, and turbulence during peak flow.

Stream beds often fill in with sediment, not allowing a deep enough water column for accurate turbidity readings. A pool at least 50 cm in depth is required. The method developed is to encase the probe within a pipe, and hang the pipe into the stream, either hung by stretching a cable across the stream using star pickets, or by attaching a hinge type mechanism to an existing structure eg. a large log. This method allows for ideal positioning of the probe and control over the depth at which the probe reads. The hanging and hinged methods also overcome the common problem of floating debris hitting and damaging the probe.

To overcome the issue of turbulence we have designed offline stilling tanks that subsample the stream. Off points from the stream are fitted with stainless steel suction sieves with 1 mm perforations, to prevent large stones and other debris becoming lodged in the tank. Inflow and outflow is controlled via a ball valve to approximately 100 ml/s, and the probe is positioned in the top of the tank which has a suitable depth for accurate readings. These tanks can also be used as autosampler intakes. Figure 4 shows a comparison of the hanging probe and an offline mounted probe in turbulent conditions.

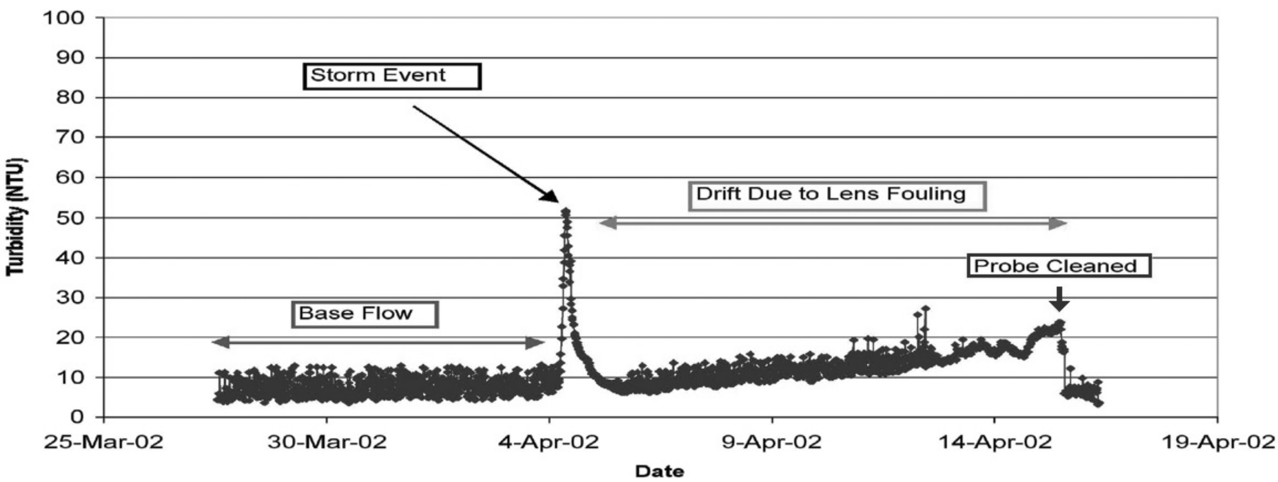

Figure 3. Effect of fouling on turbidity readings during constant baseflow conditions at Tyers Junction.

The added advantage of this approach is that all the electronic equipment can be mounted a considerable distance from the stream, which eliminates the danger of equipment damage during large events. Limitations of this approach are that a suitable fall is required (streams must be reasonably steep), so that the take off line can pipe the water to the apparatus with sufficient head as to allow flow through the apparatus.

9. CONCLUSION

There have been some advances in the technology used in the continuous in-situ measurement of stream turbidity in the last 20 years. However there are few reports on in-situ mounting and housing of probes to enable the collection of accurate results. This paper has illustrated some of the problems we have encountered in the monitoring of water quality, and methods we have employed to produce a more reliable measurement of turbidity with off the shelf items. Improving methodology of measuring water turbidity is of huge importance, as it is a relatively cheap surrogate for total suspended solids, however it is important to determine the relationship between turbidity and TSS as it is catchment specific.

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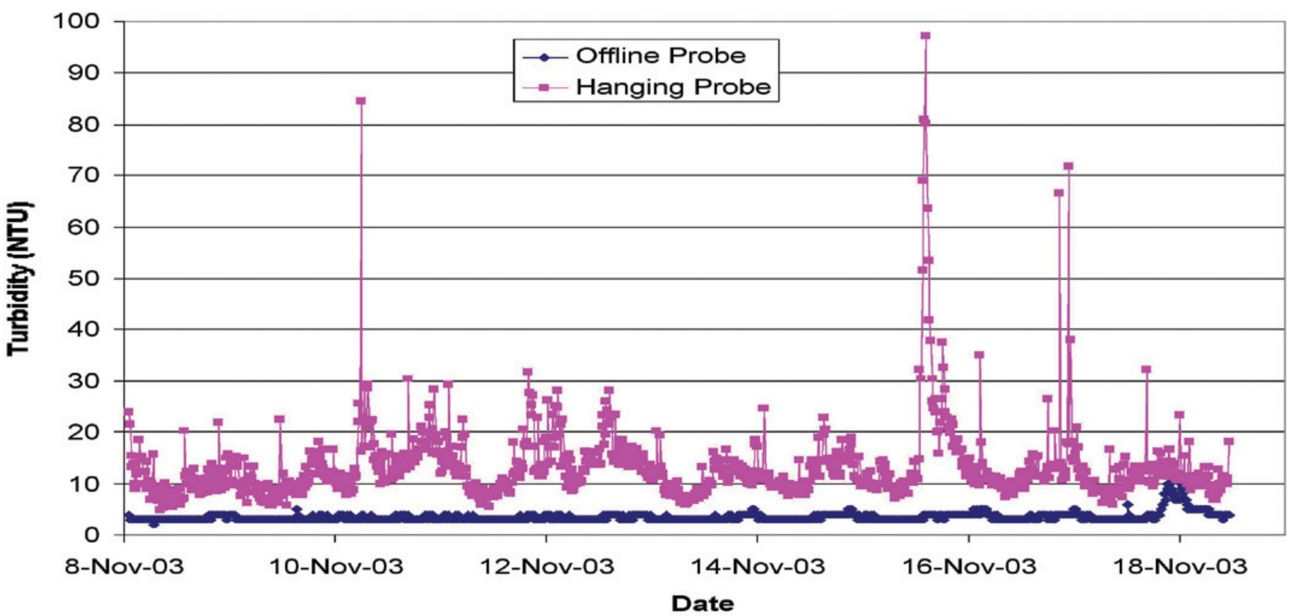

Figure 4: Raw data from springs creek culvert.

Long-term Impact of Timber Harvesting on Soil Bulk Density and Tree Growth in Victorian Central Highlands

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Summary: Soil profile disturbance and compaction can have a long-term detrimental impact on soil properties. This study was undertaken to quantify changes in soil bulk density and determine *E. regnans* sapling growth 10-yrs after logging in the Victorian Central Highlands forest. No significant differences in bulk density were found between 1989 and 1999 for most of the sampling sites. The rate of recovery of bulk density varied depending on the degree of soil disturbance and soil depth. The subsoil disturbance (HA-S3) had more influence on soil bulk density than any other type of disturbance. In general, the effect of soil profile disturbance and compaction on soil properties was greater at surface soil than deeper soil. The *E. regnans* sapling height and diameter growth was significantly lower in the HA-S3 and primary snig tracks (PST).

1. INTRODUCTION

In the Victorian Central Highlands forest, soil profile disturbance and soil compaction and their effect on Eucalypt regeneration was studied for the first three years at four clearfelled coupes (logged in 1989) at Tanjil Bren site in the Noojee forest district. Soil physical properties and its effect on seedling establishment and seedling growth were measured on permanent plots one and three years after logging (Rab, 1994, 1996; King *et al.,* 1993a, 1993b). It was concluded that density and growth of *Eucalyptus regnans* seedlings was strongly related to the level of soil profile disturbance and soil compaction and fire intensity. Minimum growth was observed on heavily compacted soils. Subsoil profile disturbance was detrimental to the early growth of seedlings as well. There is no information available (but see Jakobsen, 1983) on the long-term impact of timber harvesting operation on soil properties and their consequent effect on tree growth in this forest. The objectives of this current study was to quantify the changes in bulk density and determine tree growth characteristics 10-yrs after logging in this forest.

2. METHODS

2.1 Study area

One clearfelling logging coupe, Old Mill (logged during 1989), was selected for this study. The study area is located at 150 km East of Melbourne (30º 2' S, 146º 15' E), situated near Tanjil Bren, in the Victorian Central Highlands. The parent material of the coupe is granitic rock. The landform is mainly steep hills to moderately steep rolling hills (McDonald *et al.,* 1990). The major soil type of the area is red gradational with a small percentage of yellow brown gradational soil (Northcote *et al.,* 1975). The average depth of the topsoil is 250 mm with up to 400 mm along drainage lines. The depth of subsoil varies between 650 mm and 1200 mm. The climate of the study area is cool temperate and average annual rainfall is approximately 1900 mm. Average monthly temperature ranges from 0º in July to 20º in February. Elevation ranges from 615 to 950 m. The slopes vary from 0º to 20º. The vegetation of the forest is predominantly 1939 regrowth Mountain Ash *(Eucalyptus regnans)*, with occasional overmature and regrowth Grey Gum *(Eucalyptus cypellocarpa)*.

2.2 Logging and seed-bed preparation

The coupe was logged during 13th February to 27th March 1989. The seed-bed was prepared on 20th May 1989. The seed-bed preparation involved redistribution of logging slash and scarification of surface soils over the entire coupe using a D7G CAT fixed track crawler tractor. The log landing was also ripped using a D7G CAT crawler tractor.

2.3 Soil bulk density

Sampling for soil bulk density was carried out in February 1999, about ten years after timber harvesting and seed-bed preparation. The intact core samples were collected from 0-100 mm soil depth using a brass core (63 mm long, 72 mm inside diameter). The cores were driven into the soil with a falling weight hand corer. The intact core samples were trimmed and volume of any small gaps in the samples were determined by filling the gaps with known volume of sand (Samples with large gaps were discarded). The samples were oven-dried at 105ºC for 24 hrs and ground to determine gravel content. The mass of oven dry soil was determined by subtracting the mass of the gravel from the total mass of oven dry soil and gravel. The bulk density of soil was determined by dividing the oven-dried mass of soil with the volume of soil.

2.4 Tree growth

Tree growth characteristics were measured in February 1999. Three growth plots (replications) were randomly located on each of the eight soil disturbance categories (24 plots in total). Each growth plot was 25 m^2 in area; generally 5 x 5^2 m plots with the exception of the snig track disturbance categories where the plots were 10 x 2.5 m^2 in order to accommodate the dimensions of those particular disturbance areas. Within each growth plot, the diameter at breast height over bark (DBHOB) of every *E. regnans* sapling was recorded. The top height of the two largest DBHOB E. regnans saplings were also recorded.

3. RESULTS

3.1. Bulk density

The values of bulk density in the 0-100 mm soil depth for the 1999 measurement were compared with those of the 1989 measurement (Figure 1). The bulk density decreased from 1989 (immediately after logging)

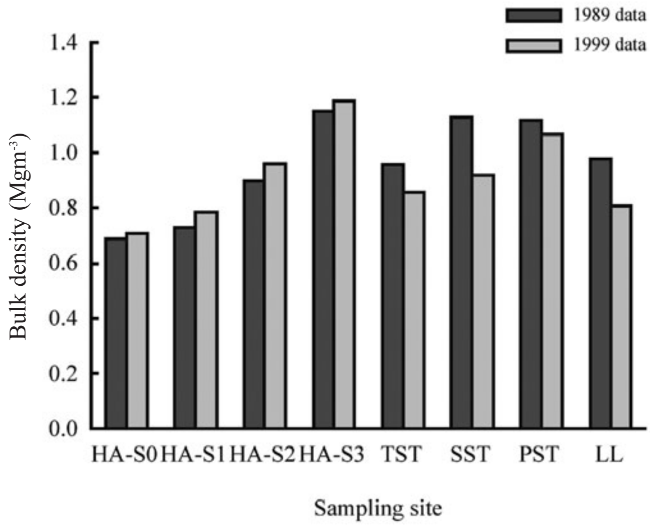

Figure 1. Bulk density (%) on various soil disturbance classes and operation categories compared to undisturbed areas in the 0-100 mm soil depth during 1989 and 1999 in the Old Mill coupe in the Victorian Central Highlands forest.

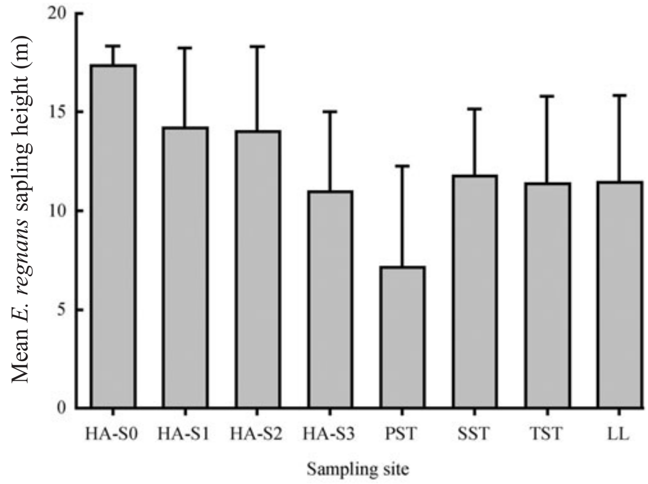

Figure 2. The mean and 95% confidence interval of 10-yr old *E. regnans* sapling DBHOB for various sampling sites in the Old Mill coupe.

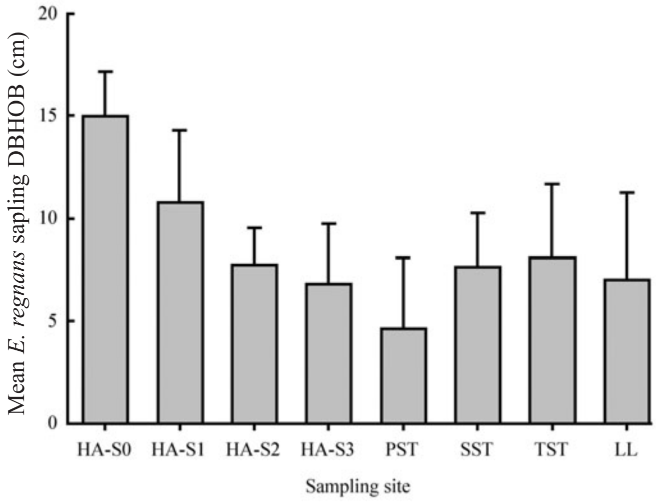

Figure 3. The mean and 95% confidence interval of 10-yr old *E. regnans* sapling height at various sampling sites in the Old Mill coupe.

measurement in all three types of snig tracks and in the log landing. However, statistically, no significant differences in bulk densities between the 1989 and 1999 measurements were found for all sampling sites except for SST. In the SST, bulk density decreased significantly, about 21%, compared to the 1989 data.

3.2 Tree growth

3.2.1 Diameter at breast height over bark, DBHOB

The undisturbed area produced saplings with DBHOB growth significantly higher than that observed on the log landing (by a factor of 2.1), primary snig tracks (by a factor of 3.2) and subsoil disturbed general harvested area (by a factor of 2.2) (Figure 2).

3.2.2 Height growth

Whilst the soil disturbance category mean sapling height was highest for the undisturbed area when compared to all other categories, no significant differences between category means were detected within either the compacted or uncompacted analyses ($p = 0.0819$ and 0.2463 respectively), (Figure 3).

ACKNOWLEDGEMENTS

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Meteorological Aspects of the Devastating Bushfires in Southwest Western Australia on 27 December 2003

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Summary: Two devastating bushfires occurred in the southwest of WA on the afternoon of 27th December 2003 during a period of extreme fire weather with temperatures in the low 40s, northerly winds averaging 30-40 kilometres per hour and low humidity. The fire weather conditions for this event were comparable in severity to some of the major fire weather events that have occurred in the region in the past 40 to 50 years. These conditions happen only rarely, only once every five years in the Tenterden region and have not previously been recorded at Bridgetown. The synoptic pattern on the day, with a deep pre-frontal trough was very characteristic of severe fire weather days in the region.

Interpretation of guidance from global and regional scale Numerical Weather Prediction (NWP) models enabled forecasters to alert fire agencies to the extreme fire weather conditions several days in advance. Steadily improving skill in NWP models in both the medium and short-term should mean that Bureau of Meteorology warnings will be a key and increasingly reliable element of mitigation strategies in the lead up to days with high potential for bushfire disasters.

1. INTRODUCTION

The southwest forest region of Western Australia, which includes the towns of Tenterden and Bridgetown, has a typically Mediterranean climate, characterised by mild winters, a pronounced winter rainfall maximum, and a reliable and prolonged seasonal drought during the summer and early autumn period (McCaw and Hanstrum 2003). The reliability of the summer drought after the heavy winter rainfall leaves the southwest forest region vulnerable to bushfires during its long, dry summer, and on occasions, strong, hot winds whip up uncontrollable fires.

Two devastating bushfires occurred in this region during the afternoon of Saturday 27th December 2003. The fires occurred during a period of extreme fire weather with temperatures in the low 40's, northerly winds averaging 30-40 kilometres per hour and low humidity. In the most devastating fire two women died tragically at around 4 pm near the small town of Tenterden, about 80 kilometres north of Albany. The fire burnt a total of 13000 ha of crops and pastures, and took a heavy toll on livestock with 15000 sheep and hundreds of cattle lost. The second fire occurred near Bridgetown where one house was destroyed and over 4500 ha of timber plantations and pasture were burnt.

The fire weather conditions for this event were very severe and prolonged and comparable in severity to some of the most extreme fire days that have occurred in the region in the past 40 to 50 years. The major fire weather events to affect this region over this period have been April 1978 (associated with TC *Alby*), January 1981, December 2000 (Mt Barker fire) and December 2003 (Tenterden fire).

The synoptic pattern on the day consisted of a front approaching the west coast with a deep pre-frontal trough near the west coast, which moved eastwards during the day. This pattern is very characteristic of extreme fire weather days on the south coast. The trough brings not only extreme fire weather but also an abrupt and marked wind shift. This wind shift can be dangerous as it can quickly broaden an existing fire front.

Interpretation of guidance from global and regional scale Numerical Weather Prediction (NWP) models enabled forecasters to alert fire agencies to the extreme fire weather conditions several days in advance.

2. EXTREME FIRE WEATHER CLIMATOLOGY IN THE SOUTHWEST FOREST REGION

The most significant meteorological influences combined with the fuel amount and degree of curing can be combined into a single parameter, known as the 'Fire Danger Index' (FDI). Forecast values of the index provide fire agencies with an indication of the likely difficulty in suppressing fire. An extreme fire danger rating requires an FDI value of greater than 49. When the FDI reaches 100, fire becomes nearly uncontrollable.

A study of extreme fire weather events in the southwest forest region over the warmer months (December to April), assuming 100% curing and using 3-hourly synoptic data and more frequent automatic weather station data for Albany, Bridgetown, Rocky Gully and Katanning was performed. Synoptic 3-hourly data were available for the period 1957-2003 for Bridgetown and Katanning and from 1965 onwards for Albany. Higher frequency automatic weather station data, usually hourly or half-hourly were available for all four sites from 1994 onwards. November was excluded on the assumption that curing would be less than 100% in these southern districts at this time of year.

The analysis reveals that extreme fire weather days in the southwest forest region can be expected to occur roughly twice each season at Tenterden but only about once every four seasons at Bridgetown. Strong northerly or northwesterly winds are associated with the majority of extreme fire weather days at all locations. The average duration is likely to be around three to four hours. 'Very severe' fire weather events, comparable to the conditions at Tenterden on 27th December 2003, happen only rarely, probably around one in five years in the Tenterden region and have not been recorded at Bridgetown.

3. EXTREME FIRE WEATHER PATTERNS

The severity of the weather conditions plays a major role in the occurrence of severe bushfires in the southwest of WA. Typically, during the summer months the synoptic pattern comprises a ridge of high pressure that persists south of the southern coast, with a cell of high pressure often forming in the Great Australian Bight. An inland trough is also a regular feature of the weather pattern over southern W.A. during the warmer months. In severe fire weather situations though the synoptic pattern is greatly changed from this climatological pattern.

Two major synoptic patterns that produce the strong northerly winds and associated extreme fire weather days about the southwest forest districts are known. One situation, which is usually the most severe, is when a tropical cyclone is travelling and accelerating southwards into the mid latitudes off the west coast. Major fire events associated with tropical cyclones occurred in 1937 and 1978 (TC *Alby*). Hanstrum (1990) reported that during the period 1956 to 1989 tropical cyclones capable of producing extreme fire weather conditions over the southwest occurred at an average of one year out of every five.

The second situation, which is typical of extreme fire weather conditions near the south coast, is associated with a pre-frontal trough. The development of a pre-frontal trough and associated wind change line is associated with the approach of cold fronts to the southwest of WA, especially in spring and early summer. With the approach of the front the trough generally aligns itself more north/south and intensifies as it migrates eastwards (Hanstrum *et al.,* 1990). The trough is characterized by strong, hot and dry northerly winds ahead of an abrupt cooler and moister southwesterly wind change. If a mid level trough is also present, gusty winds are likely to develop during the afternoon as strong mid level winds mix down to the surface.

The surface synoptic situation at 0900 WST on 27th December 2003 showed a high located in the eastern Great Australian Bight with a strengthening cold front approaching the southwest of the state. The front was particularly strong and displaced well northwards for this time of year. Ahead of the front a pre-frontal trough extended northwards from this low along the west coast to Geraldton. To the east of the trough strong northerly winds (77 km/h at 600 m) in the warm air above the inversion were present at Albany.

The pre-frontal trough intensified during the day. As the surface temperature rose the inversion broke and the strong northerly winds began to blow near the ground, allowing the extremely hot and dry air (in excess of 42 degrees) located to the northeast of Perth on 26th December to move southwards to the south coast on 27th December. The time series graph (Figure 1) for Rocky Gully, situated over 100 kilometres to the east of Bridgetown and about 35 kilometres to the southwest of Tenterden shows the progression of the relevant meteorological conditions through the day. From 0900 to 1400 WST the winds backed and increased markedly to a consistent northerly at 40 km/h, with the temperature rising to 40 degrees and relative humidity declining to 13%. This resulted in the FDI rating reaching extreme around 1030 and remaining in that range until nearly 1500, resulting in dangerous fire weather conditions for nearly five hours. The maximum FDI recorded was 91 at 1300. The trough moved to the east of Rocky Gully around 1430-1500 with a subsequent sharp wind change around to the cooler, moister west to southwesterly by 1600.

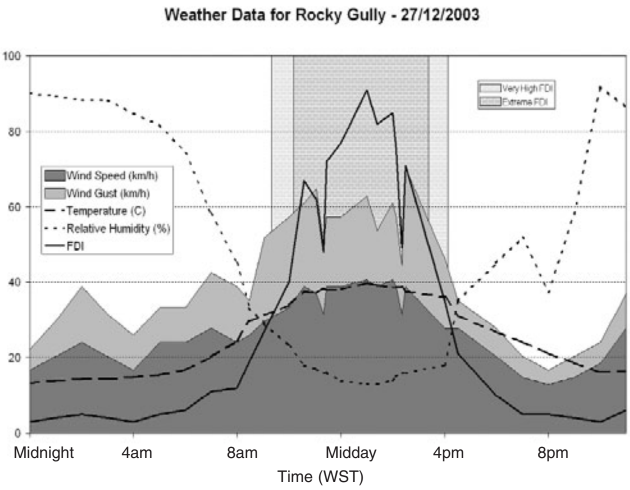

Figure 1. Time series of Fire Danger Index and its constituent weather parameters (wind, temperature and relative humidity) at Rocky Gully on 27th December 2003.

As there is no Bureau of Meteorology meteorological recording equipment at Tenterden, weather conditions at Tenterden have been estimated. The wind change line on the pre-frontal trough moved eastwards through Tenterden shortly after 1500 WST, moving at 40-45 km/h. Tenterden would have experienced the worst fire conditions approaching 1500 WST. It is estimated that extreme FDI conditions were experienced from shortly after 1000 WST until around 1600-1700 WST. Peak values of FDI would be expected to be in the range 100-110.

4. FORECASTING PERFORMANCE

The fire weather services that the Perth Regional Forecasting Centre provides are part of a national framework. The broad objectives of the fire weather service are:

1. To provide the public with routine forecasts of fire danger during the fire season and with fire weather warnings when the fire danger is expected to exceed a critical level.
2. To provide fire management authorities and other emergency services with detailed routine forecast during the fire season; fire weather warnings when the fire danger is expected to exceed a critical level; operational forecasts to assist in combating ongoing fires; special forecast for hazard reduction burns; and climate information (Johnstone, 2002. s 19.3.22, p 490).

Interpretation of guidance from global and regional scale Numerical Weather Prediction (NWP) models enabled forecasters to alert fire agencies to the extreme fire weather conditions several days in advance.

The normal lead time for the issue of a "potential blow up day" is 48 hours but, mindful of the fact that effective communications on Christmas Day could be difficult, forecasters issued a "preliminary advice of a potential blow up day" on Christmas Eve. The written product was accompanied by phone call contact with key personnel at FESA - Bushfires Service and the Department of Conservation and Land Management [CALM]. This advice was reissued on Christmas Day and followed up with forecasts of extreme fire danger in many of the inland parts of the Southwest Land Division with the routine issue of the fire weather forecasts during the afternoon of 26th December 2003. The early morning [5am] issue of the fire weather forecasts on the 27th extended the area of extreme fire danger to include the inland parts of the Southwest District where the Bridgetown fire occurred.

The actual conditions observed at Tenterden were very close to those that were forecast. The temperature was estimated to be around 41 degrees with relative humidity near 10-15%. The winds in the morning were estimated to be north to northeast around 25-30 km/h, then north to northwest at 40 km/h til 1500 WST. The west to southwest wind change was estimated to move through the area just after 1500 WST in close agreement with forecast conditions.

5. FOREST MANAGEMENT IMPLICATIONS

A region-specific climatology of extreme and 'very severe' fire weather events similar to the one that is presented here should allow a modern risk assessment analysis of fire events to be performed. This should provide an indication of the level of threat and emergency response preparedness that may be required, **before** the fire season begins.

In the lead up to days with high potential for bushfire disasters, steadily improving skill in Numerical Weather Prediction models in both the medium and short term, should mean that Bureau of Meteorology warnings will be a key and increasingly reliable element of mitigation strategies.

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Sediment Generation Rates from Different Forest Road and Track Types

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Summary: The impact of unsealed forest roads on stream water quality has long been appreciated. However there remains considerable uncertainty regarding the absolute levels and the range of concentrations and loads of sediment generated from forest roads. This paper aims to answer the question: What is the range of sediment generation rates that should be reasonably expected from forest roads (concentrations and total loads)? Ten sections of forest road were specifically selected to incorporate a wide range of the key physical site factors that are likely to affect the rate of sediment generation. Each road section was mapped for all relevant characteristics and permanently instrumented for one year each to measure rainfall, runoff, and traffic continuously. Suspended and bedload sediment were integrated measurements over two-three week site-service intervals. When sediment delivery rates from the ten sites are considered taking into account differences in rainfall volume, the slope of the sites (all data normalised to 9% slope), and small variations in the monitoring period, rates are found to vary 26 fold from 216 $mg/mm_{rain.}m^2$ for a Class 2 gravel road with minimal traffic, to 5373 $mg/mm_{rain.}m^2$ for an unsurfaced erodible track with light vehicle traffic. Given the range in rainfall, road catchment area, traffic, surfacing quality and road slope that exist for roads and tracks within the forest estate, it is likely that the true range of sediment delivery rates from roads is several orders of magnitude. These data have significant implications for prioritisation of road-stream crossings for remediation for water quality protection. Understanding the erosion processes and site properties that determine this variation in sediment delivery rates is the key to better predictions of the impact of specific road-stream crossings on water quality.

1. INTRODUCTION

Government agencies responsible for land management and water quality are increasingly interested in understanding the sources of sediment and nutrients within catchments so as to set end-of-valley and river health targets. Forest roads have been identified as a likely source of sediment within well managed production forests. This study aims to quantify the range of sediment generation rates and total loads that should reasonably be expected from a broad range of forest road and track types. This information will assist in the development of strategies for the reduction of sediment and nutrient loads in streams and water bodies.

2. METHODS

Site Selection

The ten permanently instrumented experimental road segments are identified in this paper by the codes R1, R2…R10, and are distributed within three regions; Upper Tyers (R1-R3), Marysville (R4-R6), and Lower Tyers (R7-R10). These three regions are all within State Forest in the Central Highlands area of the Great Dividing Range, Victoria. The codes and coordinates of each road site are given in Table 1.

The climate at all sites is temperate, with winter dominant, predominantly orographic rainfall in the 800-1500 mm/y range, and minor winter snowfall at the higher elevation sites. Rainfall erosivity is in the 1500-2000 MJ.mm/ha.h.y range (Sheridan and Rosewell 2003), which is relatively erosive in a Victorian context, though at the continent scale erosivity is not high.

Road sites were selected to represent the spectrum of roads and tracks common within Victorian forested areas. At one end of the spectrum are the well surfaced, well maintained timber production roads, built to a high engineering standard (eg. slope limitations and drainage requirements) to carry large volumes of logging traffic.

For example, the Upper Tyers sites (R1, R2, and R3) are all located on the South Face Road, a major 2-lane, Class 1, crowned and gravel surfaced timber and tourist road traversing the south face of Mt Baw Baw. Of slightly lesser road quality are the Marysville sites (R4, R5, and R6) all established on a Class 2 logging access road. At the other end of the spectrum are the fire access tracks (eg. R10), and recreational 4WD tracks (eg. R8), that receive very little design or maintenance, and carry only occasional light vehicle traffic. Specific properties of each experimental road section are given in Table 1.

Site Instrumentation

Methods of site instrumentation at each of the experimental sites (R1-R10) are as described in Sheridan *et al.,* (this volume), except the KBDI (drought index) was not determined for all the instrumented sites in this paper.

3. RESULTS

Net Annual Loads

A total of 235 data intervals were recorded from the ten instrumented road segments over a two year study period, providing 10 plot/years of continuous data. The net aggregated annual data for rainfall, runoff, and sediment loads, for the experimental period for the 10 instrumented sites is shown in Table 2.

The ratio of total annual runoff to total annual rainfall varied from a low value of 0.32 at R6 to a high value of 1.01 at R1. The unreasonably high value at the R1 site (runoff should not exceed rainfall) reflects the fact that an intermittent spring adds water to this road catchment during rainfall events via the base of the table drain on the cutslope side of the road. Excluding the R1 site, the range is 0.36 to 0.79. The lowest runoff ratios appear to be associated with roads with high exposure to radiation and wind, either due to a northerly aspect, roadside clearing or adjacent low forest height (Table 1). The average value for the unsurfaced tracks was 0.66, while for the surfaced roads was 0.55.

Raw, measured sediment loads varied approximately 11 fold across the ten instrumented roads from 126 kg at R2 to 1441kg at R7. However conclusions in terms of ranking types of roads with respect to sediment delivery rates should not be drawn from this raw data. Each instrumented road site differed in important characteristics such as catchment area, road slope, and rainfall (see Table 1). When sediment delivery rates from the ten sites are considered taking into account differences in rainfall volume, the slope of the sites (all data normalised to 9% slope), and small variations in the monitoring period, rates are found to vary 26 fold from 216 $mg/mm_{rain.}m^2$ at R6 (Pooleys) to 5373 $mg/mm_{rain.}m^2$ at R7 (Delpretes).

Annual sediment loads ($mg/mm_{rain.}m^2$) from the 7 gravel surfaced roads were strongly related ($r^2 = 0.97$) to truck traffic axles (Figure 1). The data is presented for comparison with Bilby *et al.,* (1989), who used robust techniques to investigate the effect of traffic on forest road sediment generation in the US (though with a much greater range in traffic volume than in the current study). The relationship given in Figure 1 extends the results presented by Sheridan *et al.,* (this volume) to enable annual sediment loads from gravel roads to be estimated given information on road slope, rainfall and traffic.

4. CONCLUSION

Given the range in rainfall, road catchment area, traffic, surfacing quality and road slope that exist for roads and tracks within the forest estate, it is likely that the true range of sediment delivery rates from roads is several orders of magnitude. Ongoing modelling efforts are using the data from the current study, combined with catchment wide survey of stream crossings, to better estimate the magnitude of this variation, so as to enable the total load of sediment and nutrients from roads at the catchment scale to be estimated.

Table 1. Specific characteristics of the instrumented road segments.

Site Property										
Region	**Upper Tyers**			**Marysville**			**Lower Tyers**			
Site Code	**R1**	**R2**	**R3**	**R4**	**R5**	**R6**	**R7**	**R8**	**R9**	**R10**
Site Name	*7k mark*	*Growlers*	*Triple B*	*Low gear*	*W station*	*Pooleys*	*Delpretes*	*4WD tk*	*Tanjil 7*	*Firebreak*
Road Class	1	1	1	2	2	2	3	5	1	4
Wearing Surface	blue metal	blue metal	blue metal	gravel	gravel	gravel	Silty Clay	Medium	Medium	Silty Clay
Road Segment Length (m)	97	105	98	198	105	172	148	179	157	179
Road Slope (%)	4	3	6	7	7	11	4	12	4	11
Total Catchment Area (m^2) *	1127	1031	840	676	500	1007	711	558	1000	415
Runoff Producing Area (ha)*	612	649	621	601	406	879	598	558	600	415
Altitude (m)	615	770	790	461	480	950	430	497	497	490
Aspect of Road Surface	80	130	10	315	205	330	81	43	174	37
Aspect of Cutslope	170	220	280	225	295	240	351	--	84	--
Forest Height (m)	40	40	0	40	<10	<8	30	30	30	15
Forest Type	M Ash	M Ash	M Ash	M Ash	M Ash	M Ash	Mixed Sp.	Mixed Sp.	Mixed Sp.	Mixed Sp.
Latitude	S-37.9217	S-37.9078	S-37.8845	S -37 36.322	S -37 35.954	S -37 34.725	S -37.9820	S -37.9457	S -38.0079	S - 37.9000
Longitude	146.3443	146.2937	146.2731	E 145 42.417	E 145 42.482	E 145 43.815	E 146.2939	E 146.5039	E 146.2874	E 146.2400
Traffic – trucks*	1685	389	0	1020	727	129	0	0	3369	0
Traffic – light vehicles	--	--	--	--	--	--	5276	218	5734	205
Date Monitoring Started	07/08/01	07/08/01	07/08/01	01/08/01	01/08/01	01/08/01	09/10/02	09/10/02	09/10/02	09/10/02
Date Monitoring Finished	14/08/02	14/08/02	14/08/02	09/08/02	09/08/02	09/08/02	21/10/03	21/10/03	21/10/03	21/10/03
Downloads	26	26	26	27	27	27	19	19	19	19
Monitoring Period (weeks)	53	53	53	53	53	53	54	54	54	54

Aspect is degrees from north. Trucks are defined as a return pass of a semi trailer. Light vehicle is defined as a return pass. Runoff producing area (mm) is based on the area of low hydraulic conductivity, which may include the wearing surface, the verge and the table drain. Total catchment area is the sum of the vertical projection of all road elements.

Table 2. Summary of aggregated annual data for the ten instrumented road segments.

Site Code	Total Precip. (mm)	Total Runoff (mm)*	Ratio runoff/ precip.	Total measured sediment (kg)	Total. Sed. Load adj for slope load (kg)	Sediment Load (t/ha.y)*	Mean Sediment Conc. (g/L)	Normalised Sediment Load $mg/mm_{rain}.m^2$*
R1	1517	1535	1.01k	428	698	11.2	0.75	755
R2	1270	987	0.78	126	246	3.7	0.38	298
R3	1385	587	0.42	181	265	4.2	0.73	309
R4	1173	594	0.51	316	376	6.1	1.05	534
R5	1413	511	0.36	131	154	3.7	0.73	266
R6	1523	481	0.32	336	290	3.2	0.69	216
R7	723	574	0.79	1441	2322	37.4	6.82	5373
R8	892	649	0.73	690	525	9.1	1.45	1054
R9	855	482	0.56	454	854	13.7	3.09	1665
R10	1161	543	0.47	1126	868	19.9	3.80	1781

*area is the Hortonian runoff producing (ie. compacted) area of the road catchment; light vehicle axles not included in totals for gravel surfaced roads; [k] spring added unknown quantity of water to the mapped road catchment therefore ratio greater than 1

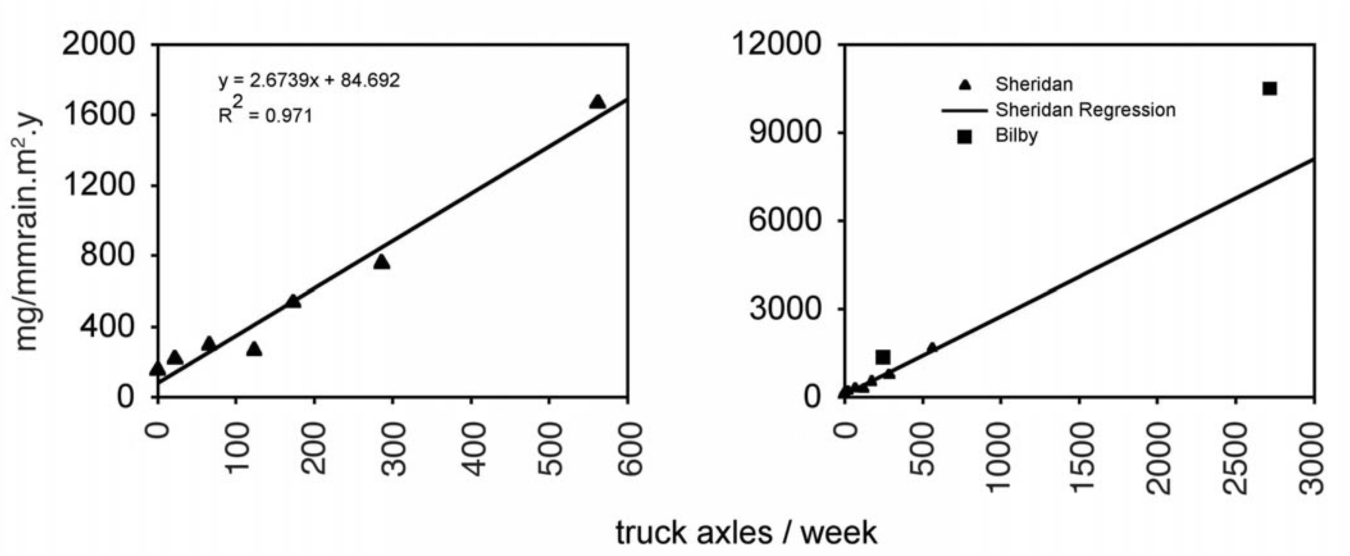

Figure 1. The relationship between traffic level (truck axles per week) and net annual sediment delivery ($mg/mm_{rain}.m^2$) for seven gravel surfaced forest roads instrumented in this study. Data compared to data from Bilby *et al.,* (1989), also for gravel surfaced forest roads.

5. ACKNOWLEDGEMENTS

Thanks to regional Department of Primary Industries staff Neil Allen and Steve Griffiths, for their assistance with site management. Thanks to the Victorian Forest Service, DSE, for financial assistance for this research.

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Impacts of Bushfire on Catchment Water Quality: First Year Results

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Summary: Results are described from a research project (see Lane *et al.,* this volume) that aims to measure sediment and nutrient loads from two instrumented 100-200 ha burnt catchments, and to relate these loads to the fire-altered physical processes that generate sediment and nutrients; specifically; runoff generation, and sediment/nutrient availability, mobilisation and transport. Initial analysis of early data indicates that total catchment exports of suspended sediment have increased by about eight times over the pre-fire rates (from 35 t/y pre-fire to 280 t/y post-fire, for equivalent rainfall) and increases in phosphorous (associated with the suspended solids) export rates are likely to be of the same order. The post-fire export rate for phosphorous on TSS is 0.48 t/y. Exports of bedload have also probably increased ten fold, though the levels of phosphorous associated with this fraction are low. These results are preliminary and ongoing analysis of early results from both the instrumented catchments will refine these estimates further. At the hillslope scale, considerable temporal variation in soil saturated hydraulic conductivity has been measured in both burnt (annual variation 40 mm/h) and unburnt (annual variation >100mm/h) areas, apparently resulting from measured seasonal oscillations in soil water repellency. Water repellency in burnt areas peaked in Spring following the fire (saturated hydraulic conductivity of 26 mm/h), coinciding with a loss of water repellency in the unburnt areas (saturated hydraulic conductivity of > 150 mm/h). In late summer of 2003 and 2004, both the burnt and unburnt sites had similar hydraulic conductivity values of around 60-70 mm/h. The early data presented in this paper, and ongoing experimental work, will improve our fundamental understanding of fire-related sediment and nutrient dynamics, and provide parameters to maximise the predictive and planning capacity afforded by modelling advances.

1. INTRODUCTION

High sediment loads and reduced stream water quality are common observations following forest fires. The possible mechanisms for these changes include; reduced evapotranspiration, resulting in increased water storage and more frequent saturated overland flow (especially around drainage lines); reduced land surface roughness and surface cover due to loss of litter and surface vegetation allowing increased surface flow velocities on unprotected soils; reduced interception and storage of rainfall in vegetation and litter, increasing the available rainfall for peak and total runoff generation; increased water repellency due to soil heating, and; increased nutrient mobility due to fire.

This paper presents results from a research project aimed at quantifying the magnitude of changes to these mechanisms and relating these changes to measured catchment exports of water, sediment and nutrients.

2. METHODS

Detailed methods are described in Lane *et al.,* this volume.

3. RESULTS

These results presented in this paper should be considered as preliminary only. In addition, the data presented forms only a small subset of the results to date. The focus here has been given to the available analysed data where broad post-fire trends and changes have been observed. As more early data are analysed, conclusions may change.

4. HILLSLOPE PROPERTIES AND PROCESSES

Soil water repellence in the <u>unburnt</u> areas oscillates seasonally, resulting in measured plot-scale hydraulic conductivities reducing from wet season values of greater than 150 mm/h down to around 50-70 mm/h in the dry season (Figure 1).

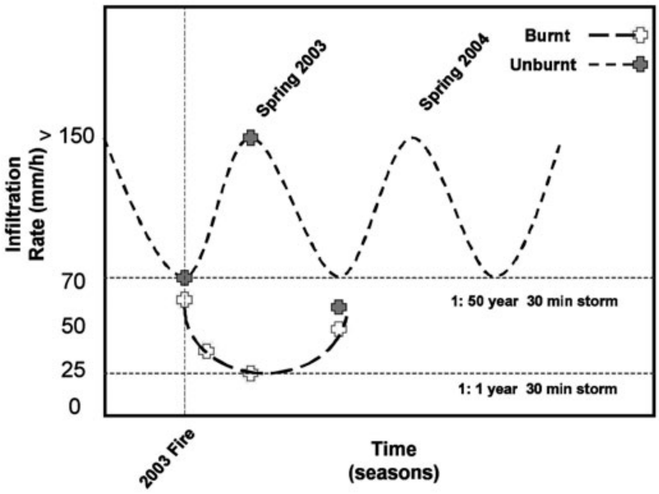

Figure 1. Temporal oscillations in saturated hydraulic conductivity (measured under simulated rainfall) for the burnt and unburnt sites compared (note that the peak unburnt value exceeded 150 mm/h and was beyond the range of measurement).

Wet season values were beyond the range of the measurement equipment and are likely to be considerably higher than 150 mm/h. Immediately after the fire the burnt and the unburnt sites displayed similar levels of water repellence, and this was reflected in similar measured hydraulic conductivities between 60-70 mm/h. However, the burnt sites retained water repellence through winter, and hydraulic conductivities reduced to a low value of 27 mm/h in Spring, before increasing again to similar values to the unburnt sites (50-60mm/h) in the dry season.

Putting these conductivity values in the context of local rainfall intensity-frequency-duration data, it indicates that six months after the fire while it would take a greater than 1:50 yr, 30-min storm to generate runoff in the unburnt sites, it would take only a 1:1 yr 30-min storm to generate runoff in the burnt sites. It is important to note that these hydrologic differences between burnt and unburnt sites appear to be confined to the wetter months, while in the drier part of the year water repellence affects the hydraulic conductivity of the burnt and unburnt sites equally.

Another important consideration is that all the measured saturated hydraulic conductivity values are relatively high (relative to local rainfall intensities) meaning that surface-runoff producing rainfall events are still relatively uncommon, even in the burnt areas.

5. CATCHMENT EXPORTS

The relationship between continuously (15 min) measured turbidity (NTU) and infrequently measured total suspended solids (TSS) is shown in Figure 2a. In turn the relationship between TSS and phosphorous on the TSS is used to calculate phosphorous concentrations from the NTU-TSS relationship (Figure 2b). These data are combined with 15-min discharge data to calculate TSS and phosphorous exports.

TSS concentrations (mg/L) were found to be approximately 3.1 times the turbidity (NTU), while Phosphorous on TSS (mg/L) is about 0.0017 times the TSS (mg/L). Relationships between total nitrogen (TN) and continuously measured parameters are less clear and are yet to be established. The relationships between the parameters turbidity, TSS and phosphorus have remained relatively stable for the post-fire period.

Catchment exports of suspended solids for the year following the fire (Mar 2003-Mar 2004) are shown in Figure 3 as a function of annual rainfall. Expressing the annual sediment load results as a function of annual rainfall normalises the data for the effect of different rainfall amounts in the different experimental years. While this is an improvement over reporting of loads alone, this approach still does not account for differences that may exist in the intensity of rainfall and runoff events in the different experimental periods. Accounting for these differences forms part of the future analysis of this raw data. The current analysis indicates that the sediment load from the catchment is about eight times greater (per unit of rainfall) than for the unburnt condition.

Catchment sediment and nutrient production rates can also be considered in terms of long term average concentrations. The average TSS concentration from Slippery Rock Weir for the year post fire was 57 mg/L. This value can be compared to commonly assumed sediment concentrations used in water quality modelling exercises for unburnt forests. For example, Grayson and Argent (2002) adopted a value of 5 mg/L for unburnt SE Australian forests. Chiew and Scanlon (2002) applied a dry weather value of 7 mg/L and an event value of 32 mg/L for forests in SE Queensland. These comparisons indicate that the increase in the

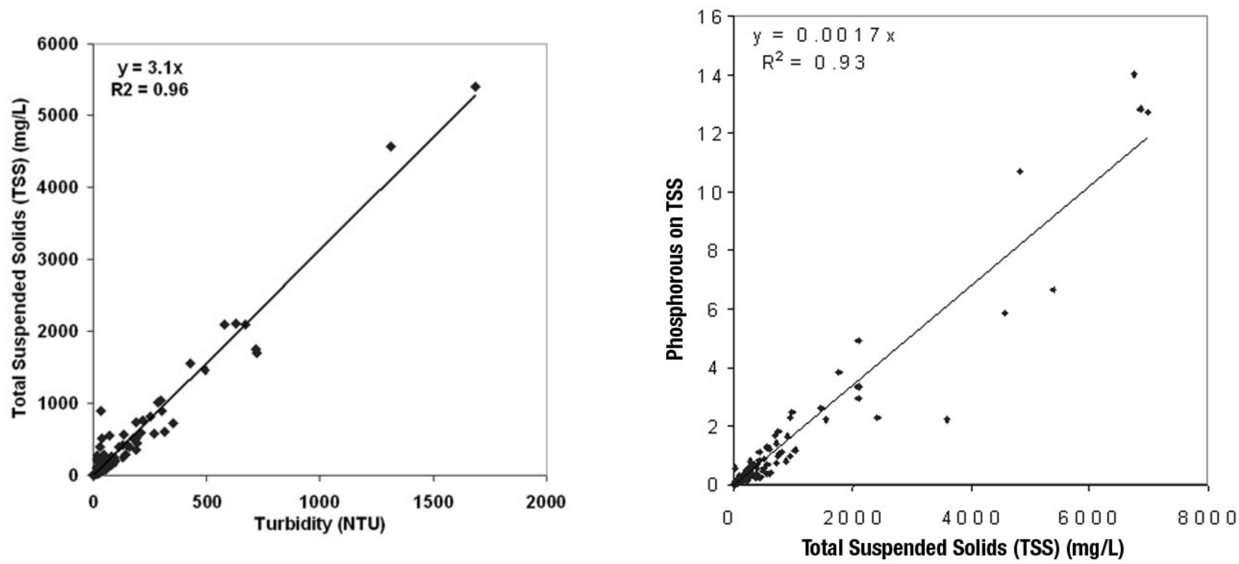

Figure 2. Relationship between, (a) turbidity and TSS, and (b) TSS and Phosphorous on TSS, for data from the Slippery Rock experimental catchment.

mean suspended sediment concentration due to the fire may be about one order of magnitude. This conclusion is of similar magnitude to the direct comparison of burnt and unburnt values from the experimental catchments (Figure 3).

Similarly, long-term mean measured phosphorous concentrations post fire can be compared to literature values used for water quality modelling. The average phosphorous on TSS concentration from Slippery Rock Weir for the year post-fire was 0.10 mg/L. Grayson and Argent (2002) used a value of 0.02 mg/L for unburnt SE Australian forests and Chiew and Scanlon (2002) applied a dry weather value of 0.030 mg/L and an event value of 0.10 mg/L in the study noted above. These comparisons indicate that the mean increase in phosphorous concentration over the

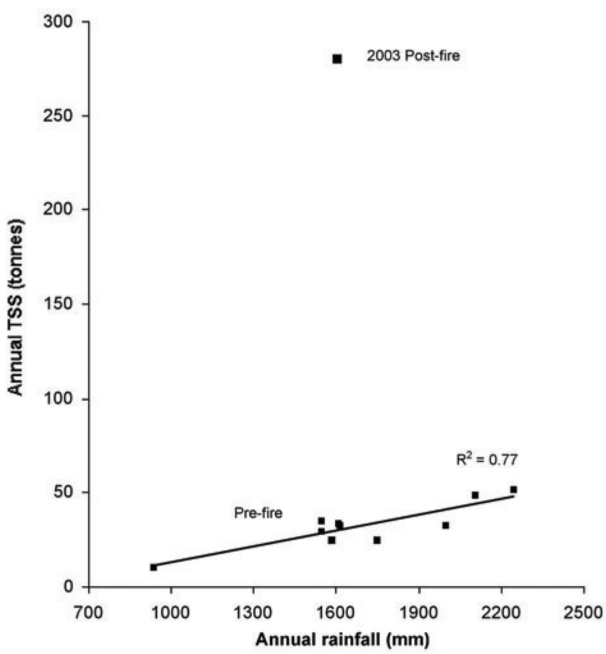

Figure 3. Relationship between annual rainfall (mm) and annual TSS (t) for the unburnt condition 10 years pre-fire and for the burnt condition the year immediately following the fire. Data is for the Slippery Rock Weir catchment.

unburnt condition may be about three fold. Analysis of samples from nearby unburnt catchments will enable a better comparison of burnt and unburnt phosphorous values in the future.

6. CONCLUSION

Post-fire increases in sediment and phosphorous loads from the experimental catchments are in the order of eight times. Large seasonal variation in soil saturated hydraulic conductivity have been measured in both burnt and unburnt areas due to seasonal oscillations in water repellency. The persistence of water repellency into the wetter months was found to be far greater at the burnt sites, resulting in low hydraulic conductivity values at the burnt sites of around 26 mm/h in Spring following the fire.

7. ACKNOWLEDGEMENTS

The authors would like to acknowledge funding assistance for this research from the Victorian Bushfire Recovery Program, Catchments and Water Division, Department of Sustainability and Environment, Land and Water Australia, and the Victorian Forest Service, Department of Sustainability and Environment.

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Erosion Hazard and its Assessment in Southeast Australian Forests

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Summary: A simple field-based erosion hazard assessment scheme was developed to provide a transparent process for erosion hazard assessment in Victorian forestry areas. The scheme involves the field assessment of a range of indicators of soil and site properties likely to affect runoff generation and soil erosion. This project evaluated the assessment scheme by comparing the predictions of erodibility, saturated hydraulic conductivity, and erosion hazard (using the scheme) against measured values of erosion and infiltration under simulated rainfall and runoff experiments on nine soils from native forest production areas in Victoria. The results indicate that while the scheme successfully provided an indication of the end members in the erodibility spectrum, it was unable to predict the hydraulic behaviour of the soils tested. These results suggest that the erosion hazard assessment scheme provides an effective first approximation of erosion hazard, however further work is required in some areas of the scheme. In particular, conceptual development of the scheme has identified the need to develop methods to assess both *erosion hazard* and *water pollution* hazard. For example, many of the most erodible soils measured during this research produced very coarse sediment that is unlikely to reach streams in a forest environment, where streams are "disconnected" from coarse sediment supply by vegetated buffer strips. In Victorian forest environments it is likely that water pollution hazard is more strongly linked to the *spatial arrangement* of coupe features (i.e. streams, buffers, tracks, roads, landings, gullies) than to inherent soil properties such as erodibility. These spatial characteristics can be modified through planning and through the selection of harvesting systems. In addition, the potential risk associated with the off-coupe roading should be included in future hazard assessments. The findings presented here relate to field experiments and conceptual development of an erosion and water pollution hazard scheme for Victorian Native Forests.

1. INTRODUCTION

The 1996 Victorian Forestry Code of Practice requires that soil permeability and areas at risk of soil erosion are identified prior to the initiation of forestry activities. Until recently, a suitable methodology for the assessment of erosion hazard in Victorian forestry regions was not available. The Soil Erosion Hazard and Soil Permeability Assessment and Classification Instruction (herein referred to as the SEHSPAC Instruction) was subsequently developed by Forest Science Centre staff (Murphy and Lane, 1999) to provide this function. The objective of the SEHSPAC Instruction is to alert forest managers to the potential for significant erosion from logging coupes and to provide a method for estimating soil permeability to allow the determination of appropriate buffer widths around drainage lines. To date, a prototype of the scheme has been released for a trial period within the Forest Service. This project aims to test the performance of this scheme, and provide for its improvement and further development.

2. EROSION HAZARD ASSESSMENT

The SEHSPAC Instruction involves the field assessment of a range of indicators of soil properties likely to affect runoff generation, sediment detachment, and sediment transport (eg. texture, aggregate stability, etc). Scores for these site-specific factors are given a weighting based on the importance of a particular factor for erosion processes. Through a series of tables, the user determines a final category for erodibility, erosion hazard, and permeability. These tables are accompanied by guidelines for the assessment of each factor. The soil and site properties, and factor weightings, were selected based on a qualitative evaluation of the literature, keeping in mind the requirement that the assessment is intended for use by field staff with minimal training in soil science.

3. VALIDATION OF THE HAZARD ASSESSMENT

Results from *individual* components of the SEHSPAC Instruction (eg. permeability, erodibility) were compared to measured erodibility and soil hydraulic parameters from simulated rainfall and runoff experiments. To test the performance of the SEHSPAC Instruction for the *overall* prediction of erosion hazard, modelling approaches were used based on the parameters measured during experimentation. To determine interrill erodibility, replicates of rainfall simulation plots measuring 0.5 m wide by 1 m long were conducted on nine soil types representative of the erodibility spectrum of soils common in Victorian forestry areas. A rainfall intensity of 100 mm/h for 30 min was applied to plots free of vegetation with a uniform slope. Timed runoff samples were collected at approximately 3, 5, 8, 12, 16, 20 and 30 minutes from the start of rainfall. To determine rill erodibility, water was applied at a range of flow rates (500 to 6500 L/h) to a confined 3 m long channel. Between 4 and 6 flow rates were applied to each rill. Flows at a given rate were applied for 1.5 min and samples of the runoff collected at 0.5, 1 and 1.5 min from the start of the application of runoff. Sediment concentration of rainfall and runoff simulation samples were determined gravimetrically. The layout and operation of typical rainfall and runoff simulation plots is shown in Figure 2.

The infiltration rate was calculated as the difference between the measured rainfall and runoff rate. The *interrill* erosion rate E_i (kg/m^2.s) is considered a function of the following factors (Kinnell 1993):

$$E_i=K_i*I*Q*S_f*C_f \qquad (1)$$

where:

I (m/s) is the steady-state rainfall rate, Q (m/s) is the steady-state runoff rate, and K_i (kg.s/m^4) is an interrill erodibility coefficient related to soil properties. S_f and C_f are non-dimensional slope and cover adjustment factors respectively calculated from (NSERL 1995):

$$S_f\ C_f = [1.05 - 0.85\ \exp^{(-4 \sin S)}\ ]\ [^{e-2.5C}] \qquad (2)$$

where:

S is the slope angle, and C is assigned a value equal to the % surface cover. For these experiments on bare surfaces, rock was considered as a part of the media and thus $C_f = 1$. Interrill erodibility was calculated by measurement of erosion rates under rainfall and transposing Eq 1 to solve for K_i. The *rill* sediment delivery rate E_r (g/s) is considered as a power function of discharge Q_r (L/min), and the tangent of the rill slope S_1:

$$E_r=K^{r2}*Q_r^{\ a}*S_1^{\ 1.5a} \qquad (3)$$

Eq 3 is transposed to determine the rill erodibility coefficients K^{r2} (g.min^a/L^a.s) and a empirically using non-linear regression from the measured erosion rates at a range of flow rates during field runoff simulation experiments (Kemper *et al.,* 1985). The field determined erosion parameters were; a) compared directly with scores from the SEHSPAC kit, and; b) used in a process-based erosion model (not described) to predict the relative levels of erosion for the different soils tested at the hillslope scale

4. RESULTS

The results from the SEHSPAC Instruction and the field rainfall and runoff simulation experiments are given in columns 2 to 9 in Table 1. Figure 1*a* compares the predicted susceptibility to breakdown rating (from the SEHSPAC kit, Table 1) with the measured interrill erodibility, indicating that the soil parameters successfully provide an indication of the end members in the spectrum. Figure 1*b* indicates that the predicted permeability (from the SEHSPAC kit Table 2) was poorly related to the measured infiltration rate. All soils were classified as having high or very high permeability, though infiltration under rainfall was less than 10 mm/h for three of the nine soils tested. This is partly attributed to the absence of a test for hydrophobicity in the field assessment, as some of the soils in the test were clearly hydrophobic.

All soils were ranked from lowest to highest erosion hazard based on the SEHSPAC erosion hazard rating. Results are given in the 2nd last column of Table 6. Using an erosion prediction model parameterised from the field rainfall and runoff simulation data, the *predicted* relative rank is shown in the far right column. Comparison of these ranks indicates reasonable agreement for the end members of the set.

5. FURTHER CONCEPTUAL DEVELOPMENT

The next step in the development of erosion hazard assessment in Victorian forests is to improve the assessment of the risk of *water quality pollution*. The current system is focused more towards the assessment of on-site erosion hazard. However, an erosion hazard does not necessarily equate to a *water pollution hazard*. For example, many of the most erodible soils in this study produced very coarse sediment that is unlikely to reach streams in a forest environment. One can also imagine a case where the erosion hazard is *high* because the soil is erodible, yet the water pollution hazard is *low* because there are no nearby streams for sediment laden water to be delivered to.

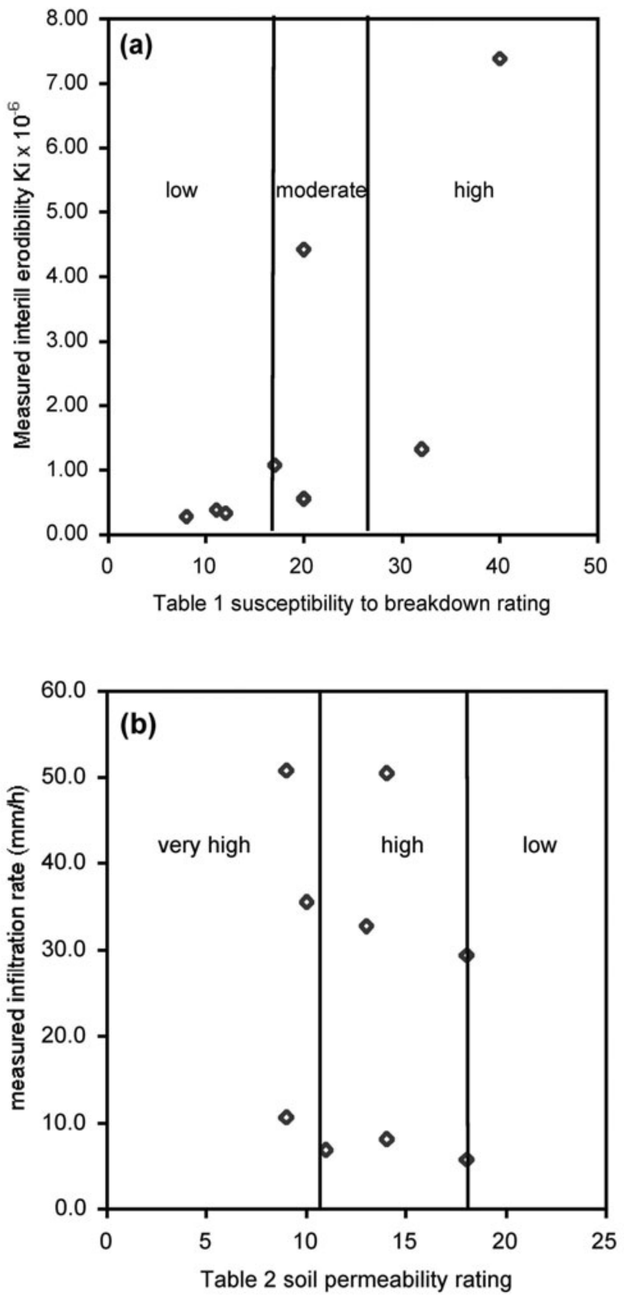

Figure 1. The relationship between; a) measured interrill erodibility and predicted susceptibility to breakdown from the SEHSPAC kit, and; b) measured infiltration rate and predicted soil permeability from the SEHSPAC kit. Vertical lines define the rating categories given in the SEHSPAC kit.

In Victorian forest environments it is likely that water pollution hazard is more strongly linked to the *spatial arrangement* of coupe features (ie. streams, buffers, tracks, roads, landings, gullies), and to topographic properties rather than to inherent soil properties such as erodibility. For example, Dignan (1999) surveyed 10 km of coupe buffer and found eight instances of sediment intrusion into buffers, all of which originated on roads, snig tracks and compacted surfaces.

The types of factors that are likely to dominate the risk of water pollution hazard due to on-coupe forest harvesting activities are illustrated in Figure 3. Future hazard assessment systems must take these factors into account in the assessment process if the true risk of water pollution is to be determined. It is important to note that some of the spatial characteristics of coupes that contribute to water pollution risk (illustrated in Figure 3) can be modified through planning and through the selection of appropriate harvesting systems.

Figure 2. The runoff (left) and rainfall (right) simulation methods used to determine rill erodibility, interrill erodibility, and infiltration parameters.

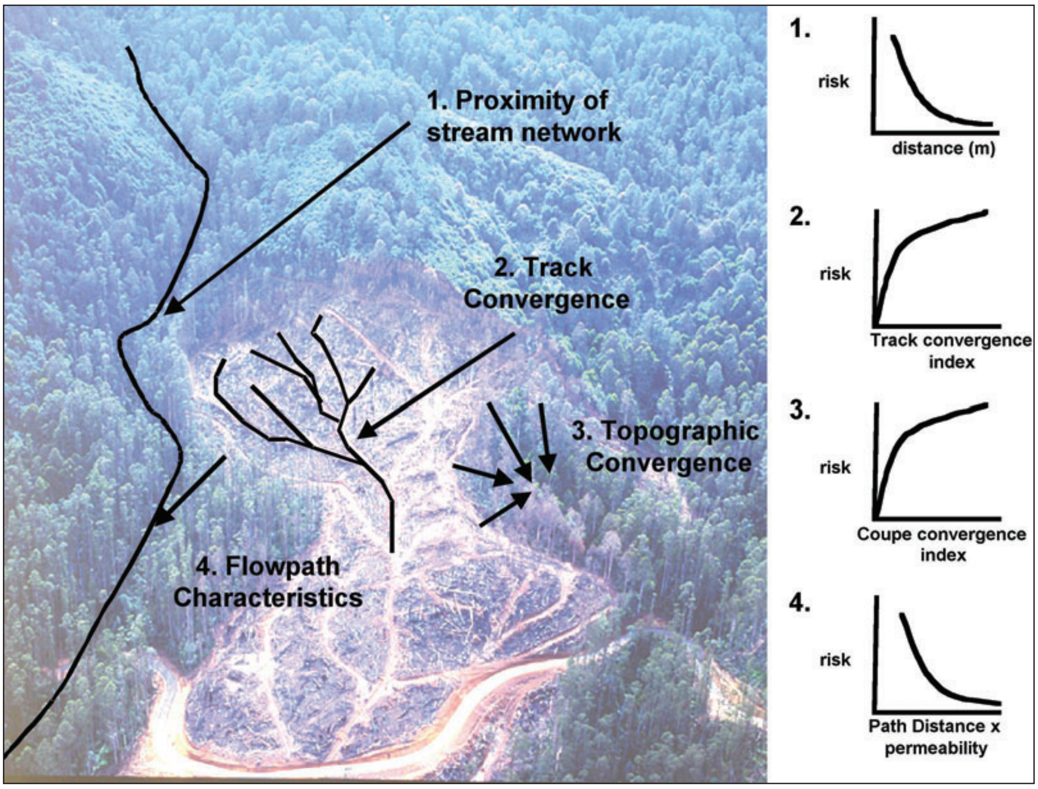

Figure 3. The spatial arrangement of on-coupe features that are likely to impact on the risk of water pollution during harvesting.

Table 1. Erosion parameter values from rainfall and runoff simulation, and scores from the SEHSPAC Instruction. Soils are ranked from lowest to highest (2nd last column) based on the erosion hazard rating (from Table 5 of the SEHSPAC kit). The predicted relative rank using an erosion prediction model parameterised from the field rainfall and runoff simulation is shown in the far right column.

Site location, soil texture, and horizon[A]	Simulation data					Score from SEHSPAC Table;			Rank	
	K_{sat} (mm/h)		K_I x 10^{-6} (kg.s/m^{-4})		Rill[B] K_{r2} (g.min/ L.s)	1	2	4	Rank SEHSPAC	Rank-Model
Otways clay A_1	8.1	(2.2)	0.28	(0.0)	0.32	8	14	22	1	2
Erica clay A_1	29.3	(10.9)	0.38	(0.1)	0.31	11	18	29	2	1
Mt Cole sand A_1	10.6	(8.8)	0.56	(0.2)	2.0	20	9	29	3	5
Otways sand A_1	50.8	(6.3)	4.41	(0.9)	3.0	20	9	29	4	4
Wombat clay A_1	5.7	(4.3)	0.32	(0.0)	0.37	12	18	30	5	3
Noojee clay A	50.4	(6.6)	1.07	(0.2)	--	17	14	31	6	7
Noojee clay loam B/C	32.7	(4.6)	0.55	(0.0)	6.41	20	13	33	7	6
Erica sand C	35.5	(7.8)	1.32	(0.5)	14.8	32	10	42	8	8
Otways sandy loam B	6.8	(4.0)	7.39	(2.2)	17.1	40	11	51	9	9

NB. Standard Deviation shown in brackets; [A] letter refers to soil horizon; [B] K_{r2} determined when *a* in Eq 2 = 0.7

Finally, the contribution of the off-coupe roading network to water pollution hazard is rarely (with few exceptions) taken into account in erosion hazard assessment schemes for forestry activities. Given that many studies have indicated that the greatest impact on water quality in well managed forests is due to roads, it is surprising that this aspect of forestry is not included in assessment schemes. Future development of water pollution hazard assessment should include the risk of the construction, operation, and maintenance of the roading network impacting on water quality.

6. CONCLUSION

The soil-breakdown component (ie, erodibility) of the SEHSPAC Instruction worked reasonably well, while the permeability component of the kit did not work. Overall rankings of erosion hazard for nine soils using the SEHSPAC Instruction were similar to erosion model predictions. These results suggest that the erosion hazard assessment scheme provides an effective first approximation of *erosion hazard*, however further work is required in some areas of the scheme. In particular, conceptual development of the scheme has identified the need to develop methods to assess both erosion hazard and *water pollution hazard*. This will require the assessment of on-site indicators of water pollution risk that are more related to the spatial arrangement of the coupe features than to the intrinsic soil properties of the site.

7. ACKNOWLEDGEMENTS

Thanks to the Victorian Forest Service for their support for this research. Thanks also to the regional DSE staff for their kind assistance in the selection and establishment of experimental sites.

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The Effect of Traffic Quantity and Road Water-Status on Water Quality from Forest Roads

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Summary: A study investigated how runoff water-quality from unsealed forest roads is affected by a) the quantity of truck traffic, and b) the road water-status, at the time of trafficking. Three sections of a gravel surfaced forest road were instrumented for more than a year and exposed to low and high levels of truck traffic during wet winter conditions and dry summer conditions. Rainfall, runoff, road moisture, and traffic were measured continuously, while suspended and bedload sediment were integrated measurements over two week site-service intervals. The median event sediment concentration from the three road segments under low truck-traffic conditions (<9 return truck passes) was 256 mg/L, increasing 3.7 fold to a median of 942 mg/L under high truck-traffic conditions (>9 return truck passes). When these data are expressed on a slope-adjusted, per-unit-rainfall basis accounting for differences in rainfall and site characteristics, a similar median increase of 4.2 fold is found due to high truck traffic. These concentrations, and increases due to traffic, are substantially less than many previously reported values. Initial analysis of the data indicates adjusted sediment delivery rates when trafficked in low water-status conditions may be higher than when trafficked in high water-status conditions. This study shows that for a good quality, well maintained gravel forest road, heavy truck traffic increases median sediment loads about 4 fold, and trafficking during wetter conditions does not appear to generate more sediment than trafficking during dry conditions. Careful interpretation is required regarding the meaning of these results in management terms.

1. INTRODUCTION

The significance of forest roads with respect to forest water quality has been long recognised e.g. Hoover 1952) although it has only generated a limited amount of focused research (e.g. Bilby 1989, Burroughs and King 1989, Luce and Black 1999, Ziegler *et al.*, 2000). The influence of traffic on increased sediment loads has been reported by several authors; Grayson *et al.*, (1993) found a 2 fold increase, Croke *et al.*, (1999) report a 4-5 fold increase with traffic, Foltz (1996) an 8-12 fold increase depending on gravel quality, Bilby *et al.*, (1989) a 20 fold increase, and Reid and Dunne (1984) a 100 fold increase. From these diverse results, (many using different methodologies) it is difficult to conclude what effect traffic has on sediment delivery from forest roads.

There have been no studies *specifically* addressing the issue of the effect of road water-status at the time of use on water quality. However, it is known that the water content of the road affects the strength of the road, and one of the most commonly observed negative effects of road use under wet conditions is deformation of the cross-sectional profile resulting in longitudinal rutting. This often compromises lateral drainage of the road, concentrating flows along the road surface, causing by-passing of drainage structures leading to rilling of the road surface and associated high sediment generation rates.

A trial by Foltz and Elliot (1997) investigated the impact of tyre pressure on sediment generation. The results showed an average 80% reduction in sediment generation, largely attributable to a reduction in rutting depth when the tyres were inflated to lower pressures. Deeper, longer ruts promoted concentrated flow and increased erosion rates. These roads would also require more frequent grading, which also increases sediment generation. The conditions of increased rutting and grading are akin to the road surface conditions that could be expected under wet weather traffic. Therefore, the magnitude of changes in sediment loads associated with the tyre pressure treatments may give an indication of the magnitude of changes expected due to wet use of roads. However, this does not account for the fact that highway tyre pressures also crushed aggregates to a much greater extent, maintaining a supply of sediment for transport. Bilby *et al.,* (1989) found that traffic *prior* to storm events also resulted in elevated sediment generation rates (in comparison to traffic *during* storm events), suggesting that eliminating traffic on roads during wet weather may not reduce the amount of sediment delivered to the stream to the extent that might be expected.

This study aims to quantify the effect of traffic volume and road water-status on sediment generation rates. Better understanding of the processes of sediment generation from unsealed roads will enable more targeted and effective management prescriptions for the protection of stream water quality in forests.

2. METHODS

Site Selection

The experimental site (Feglins Rd) is a gravel-surfaced, unsealed forest road near Marysville, Victoria. Elevation ranges from 450 m to 950 m, annual rainfall is around 1200 mm/y and average annual erosivity is estimated at 1700 MJ.mm/ha.h.y. Three, 100-200 m long sections of road were selected for monitoring with similar characteristics of road slope, cut-batter slope, geology, climate, soil type, catchment size, gravel type, table drain properties, and ratio of batter to road surface. The drainage area at each experimental site includes approximately _ the crowned-road wearing surface, the road verge on one side, the table drain (cut-ditch) and the cut-slope. Specific site characteristics are given in Table 1. The three road segments were all monitored for at least a full year.

Data Collection Methods

At each experimental road section road-runoff discharging from the under-road culvert was carried to the measurement apparatus (Figure 1) via a pipe. Coarse sediment (termed bedload in this paper) was deposited in a bedload trap fitted with baffles. Water discharged from the bedload trap into a tipping bucket, which was logged electronically to measure discharge. Each alternate tip of the bucket directed a 20-mL sub-sample of runoff to a storage container. Rainfall intensity was measured using a pluviometer and road traffic was recorded using pressure pulses generated as vehicle wheels traversed a single rubber tube fitted across the roadway. Daily road water-status was estimated for the area using the Keetch-Byram Drought Index (KBDI) (Keetch and Byram 1968) based on daily precipitation and maximum temperature. Instrumentation was serviced every 2-3 weeks and samples analysed to allow the calculation of runoff volume, suspended and bedload concentration and load, and rainfall, traffic and soil water characteristics. A standard traffic unit is defined as a return pass of a 6 axle, 22-wheel semi-trailer (3 axles when unloaded and piggy-backing the trailer on the return pass). An average of the daily KBDI values was calculated for each data interval. A KBDI value of 40 was used to separate "wet" and "dry" road water-status categories, based on a visual assessment of the seasonal oscillation of the KBDI (Figure 2). An accumulated traffic volume of 80 truck axles (< 9 return truck passes) prior to a defined storm (Wischmeir and Smith 1978) was used to separate Low and High traffic categories. This separation criteria approximates less than one truck per day.

3. RESULTS

Summary statistics (Table 2)

From the monthly rainfall, and the oscillation of the KBDI shown in Fig 2, it is concluded the climatic conditions during the experimental period were typical of an average year. Cut off values of the KBDI index used to define "wet" and "dry" categories for analysis are shown in Figure 2.

High traffic volumes resulted in increases in median suspended sediment concentrations, increasing by 3.7 times from 256 to 942 mg/L (Table 2). When expressed in adjusted terms, taking account of differences in rainfall in low and high traffic periods, and the properties of the individual sites, the median increase is 4.2 times.

In absolute terms, applying high traffic during low road water-status conditions (ie. summer) resulted in sediment concentrations 106% higher (1436 vs. 696 mg/L) than applying high traffic in periods of high road water-status (i.e. winter). However, this simple comparison ignores the confounding effects of differences in both traffic and rainfall characteristics for these two categories. The traffic volume during dry high-traffic conditions was 43% higher, while rainfall events were less frequent (14 vs. 21 storms), but with twice the median peak intensity (46 vs. 22 mm/h) (Table 2).

These confounding effects (and others) are, however, accounted for to some extent by expressing the erosion rate per unit of rainfall, and adjusting for slope gradient and the area of the individual sites. Under high traffic, dry conditions, median sediment delivery ($mg/mm^{rain}.m^2$) was 372 and for wet conditions was 219. These values indicate that sediment generation rates may be higher when trafficked under dry conditions, possibly due to increased abrasion of surface materials.

Figure 1. The experimental measurement apparatus during a storm event, showing delivery pipe from the road culvert, bedload trap, manifold, tipping bucket, and suspended load sampler (visible protruding from the discharging flow).

Table 1. Specific characteristics of the instrumented road segments.

Site Property	Lower Road Section (R4)	Middle Road Section (R5)	Upper Road Section (R6)
Static			
Road Length (m)	198	105	172
Road slope (%)	7	7	11
Catchment Area (m^2)	680	500	1010
Wearing surface area (ha)	330	210	620
Batter area (m^2)	120	146	197
Altitude (m)	461	480	950
Aspect of road surface	315	205	330
Aspect of cut batter	225	295	240
Forest Height (m)	40	<10	<8
Latitude	S 37º 36.322	S 37º 35.954	S 37º 34.725
Longitude	E 145º 42.417	E 145º 42.482	E 145º 43.815
Dynamic			
Rainfall (mm)	1173 (1524)	1413 (1645)	1513 (1513)
Traffic (Return truck passes)	1020 (1552)	727 (845)	129 (129)
Date monitoring started	23/07/2001	01/08/2001	01/08/2001
Date monitoring finished	29/11/2002	04/10/2002	09/08/2002
Data intervals in each category			
Low traffic – High water-status	6	10	11
Low traffic – Low water-status	1	1	4
High traffic – High water-status	9	2	0
High traffic – Low water-status	4	5	4

* Note that the measurements at the three sites were for different periods. Values in parenthesis sums for all the data intervals within the study period. The value outside the parenthesis is the sum for all the data intervals within the common period 01/08/2001 to 09/08/2002. Aspect is degrees from north

Table 2. Summary experimental data for traffic-volume and road water-status categories.

		traffic volume-water-status category				traffic volume		water-status		All
		Low Wet	**Low Dry**	**High Wet**	**High Dry**	**Low**	**High**	**Wet**	**Dry**	
	N	27	6	11	13	33	24	38	19	57
Precip. (mm)	Σ	2439	382	687	502	2820	1189	3126	884	4010
	av.	76	65	55	43	76	43	70	43	55
Peak Precip. (mm/h)	av.	22	18	22	46	20	32	22	32	22
Runoff (mm)	Σ	798	127	337	140	925	477	1135	266	1401
	av.	26	21	22	8	26	13	25	8	21
No. storms	Σ	52	6	21	14	58	35	73	20	93
Truck Traffic[A]	Σ	39	9	1017	1455	48	2473	1057	1464	2521
	av.	0	1	70	87	0	74	1	39	4
Susp. sed. (mg/L)	av.	253	281	696	1436	256	942	383	986	450
Total sed. (mg/L)	av.	289	288	721	1528	289	969	395	1016	506
KBDI	av.	1.7	56.1	4.8	123.6	2.8	59.7	2.8	95.7	4.8
Erosion rate (t/ha)	av.	0.087	0.091	0.133	0.117	0.087	0.125	0.090	0.106	0.104
Sed del rate ($mg/mm^{rain}.m^2$)	av.	84	116	219	372	86	365	122	241	144

av. = median value for the data intervals in this category, N = number of data intervals, Σ = sum, [A] number of return passes of a standard semi trailer log truck

In overview, *the summary statistics* indicate that for a good quality, well-maintained gravel forest road, high traffic volumes result in about a 4-fold increase in sediment delivery rates. Trafficking during wetter periods does not appear to lead to an increase in sediment delivery rates.

Time series plots-Figure 3

Inspection of Figure 3 indicates that peaks and troughs in suspended sediment concentration (only for data intervals with defined storms) generally conincide with peaks and troughs in traffic volumes, often with a slight time lag. Low-traffic sediment concentrations of between 200-400 mg/L are typical at each of the three instrumented sites, indicating that the sites are suitable for direct comparison, despite differences in static site conditions (Table 1). Baseline values at the Upper site (R6) are slightly higher than the other two sites, proably reflecting the steeper road slope and greater area (and therefore higher peak flows) of this catchment. This catchment also appears to be more sensitive to traffic, possibly because it is steeper and traffic is applying greater torque to the road surface during acent and decent, therefore detaching more sediment.

Peak values of 3500-4500 mg/L in Feb-Apr 2001 at all sites are associated with both/either high traffic and high peak precipitation intensities. These high sediment concentration values should be contrasted with periods with high precipitation volumes, (eg. Aug and Oct 2001, July 2002) where peaks in sediment concentration do not occur, illustrating the key role of precipitation *intensity* as opposed to rainfall *volume*, in the generation of suspended sediment.

The similar range of the maximum and minimum sediment concentration values recorded at each site indicates there is about an order of magnitude variation in sediment concentration that can be expected under a wide range of rainfall and traffic conditions. Inspection and comparison of sediment concentrations generated under high water-status (Aug to Nov 2001, and Jun to Oct 2002) and low water-status (Dec 2001 to May 2002) conditions suggests there is no obvious relationship between this parameter and the sediment concentration of runoff water. Following the cessation of traffic, generation rates generally decline exponentially to pre-traffic levels after 50-70 mm of runoff.

In summary, inspection of the *raw time series data* for the three road segments indicates that peak traffic volumes result in increases of up to ten times in erosion rates from roads, with the greatest increases occurring during periods of high intensity precipitation.

Other observations

Suspended sediment comprised about 90% of the total sediment load (Table 2), which has significant implications for the selection of erosion and sediment control methods.

4. CONCLUSION

Inspection of the data from the current study shows that median suspended sediment concentrations from well gravelled forest roads of moderate slope and length, and under low traffic conditions, are around 200-400 mg/L, increasing up to 10 fold during intense rainfall events following heavy traffic. The median increase due to high traffic was about 4 fold. These sediment generation values and traffic related increases are substantially less than generally reported previously. The lower values found in this study are largely attributed to improved methods for continuously monitoring erosion rates from roads under field conditions. The results also show that if roads are surfaced well and maintained correctly, their use in moderately wet periods should not result in greater impacts on water quality than summer usage.

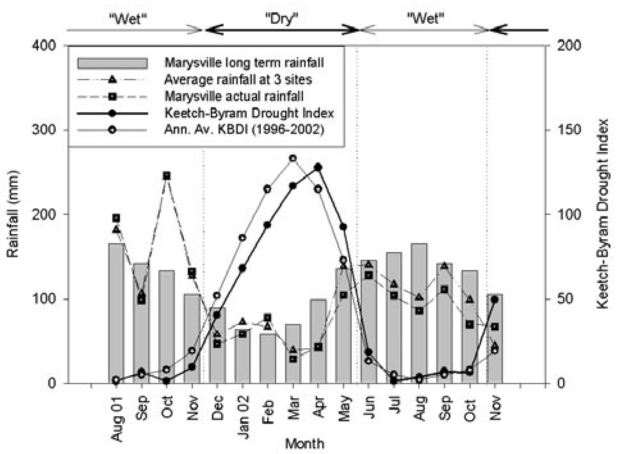

Figure 2. Rainfall and drought-index (KBDI) values compared for nearby Marysville and the study site for the study period.

The results from this study will assist forest managers and regulators to better understand and manage the impact of forest roads on water quality.

5. ACKNOWLEDGEMENTS

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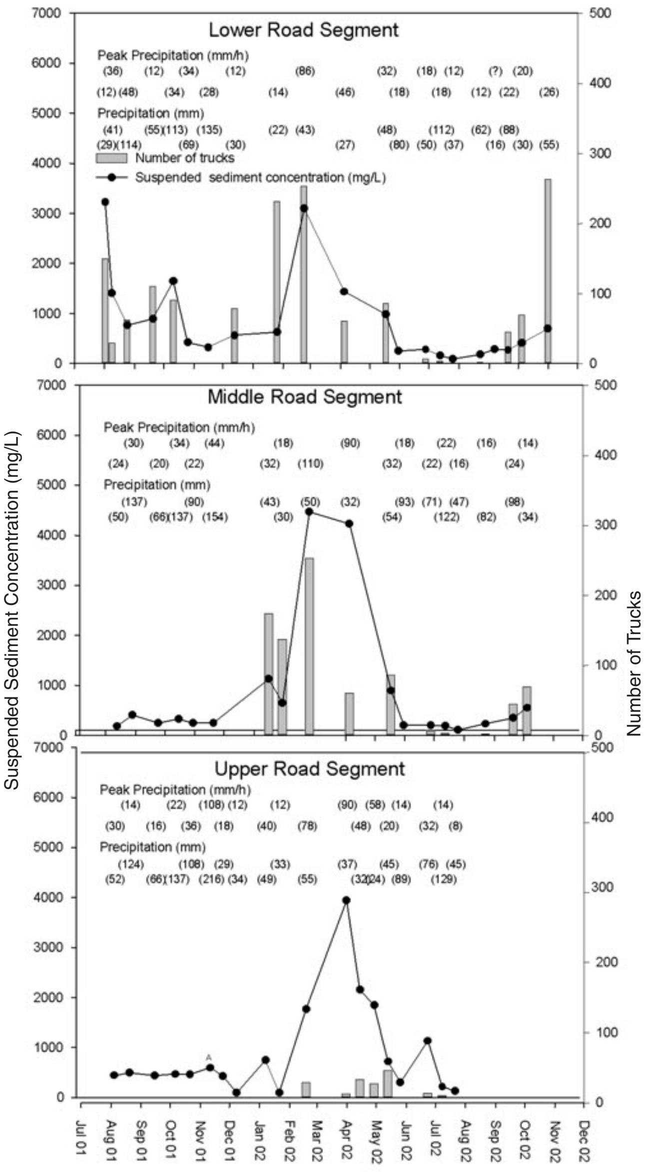

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Figure 3. The relationship for the 3 instrumented road segments between traffic and the suspended sediment concentration of runoff for data intervals containing at least one storm event. Values shown in parenthesis are the peak and total precipitation values for the data interval.

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Modelling Road to Stream Connectivity: The Development of a Cost-Effective Framework using GIS

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Summary: Many researchers are now exploring the possibilities of physically-modelling the interaction between road surfaces and the natural drainage system. Many of the existing hydrological models, however, require detailed and often expensive parameterisation and it remains unclear whether this is reflected in a greater accuracy of model predictions. We propose the modelling of road runoff and sediment using the connectivity concept, whereby we predict the probability of runoff reaching a stream via a gully or overland flow pathway. We derive simple empirical relationships to predict the occurrence of these two pathways and use the data in the development of a risk assessment map. The paper describes this process and some of the problems related to the accuracy of input parameters derived from standard 20 m DEMs. The procedure we outline here is currently being finalised for use by the forest industry as a cost-effective methodology to manage road networks in forests for water quality protection.

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 many of these are not well equipped to investigate the potential impact of road surfaces on hillslope or catchment hydrology.

In a previous paper (Takken and Croke, 2004a), a conceptual framework was presented whereby road to stream linkage is assessed through the evaluation of different connectivity pathways. Field observations were used to identify the type of delivery pathway found at the outlet of a road drain. A gullied pathway can directly route runoff from the road to the stream and therefore, the initiation of gullies at drain outlets should be avoided. In the absence of a gully, runoff is dispersed on the hillslope, where it may infiltrate. If any flow reaches the stream through diffuse overland flow, it depends on the volume of runoff discharged on the hillslope, the hillslope length between the outlet and the stream, and the infiltration characteristics of the soil.

In this paper we discuss a modelling approach that can be used to create a catchment map highlighting the best locations for road construction to limit sediment delivery. The output includes a practical guide on acceptable drain spacings to limit runoff delivery to streams. The approach is based on the identification of the two different delivery pathways, ie. gullies and diffuse overland flow. Firstly, we discuss how the two types of delivery pathways can be distinguished. Secondly, we look into the parameters that are required as input to our model and how they can be derived from available input data. In the final part we show a map that presents the maximum permitted drain spacing in a catchment.

The Albert River catchment (described previously in Takken and Croke, 2004a) is used as example catchment in this paper. For a description of the site and field survey methodology see Takken and Croke (2004a).

2. IDENTIFICATION OF PATHWAYS

Croke and Mockler (2001) found that the occurrence of gullies at drain outlets could be predicted by road contributing area and hillslope gradient. They presented a gully-threshold curve that can be used to predict the maximum drain spacing along the roads to minimise the risk of gullies being formed:

$$RCA = 70 / \sin(\alpha) \qquad (1)$$

Where RCA = road contributing area (m) and α is the hillslope gradient at the drain outlet (degrees).

For 91.5% of the observed gullies, the threshold curve correctly predicts a gully. However, the overall accuracy is lower (55.6%), because there are many drains for which a gully is predicted, where there was no gully found at the outlet.

A logistic regression analysis was applied to the field data using the same two variables of RCA and slope and a weighting factor was applied to compensate for the fact that the Albert River data set contained less gullies than non gullied sites. The resulting logistic regression equation predicts the probability of gully occurrence at the drain outlet.

$$\text{logit}(p) = -0.43/\sin(\alpha) + 0.0031 \cdot RCA + 0.31 \qquad (2)$$

In Figure 2 three threshold lines are drawn based on a probability of gully occurrence of 0.1, 0.5 and 0.9 respectively. All points that plot above the line have a higher probability of gully occurrence than the probability value represented by the line.

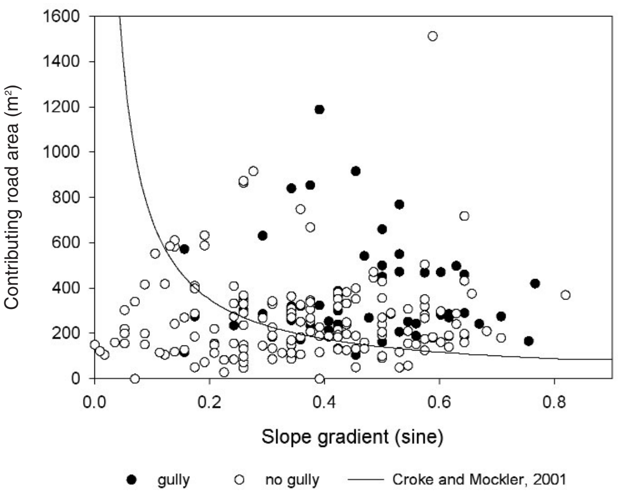

Figure 1. Contributing road area versus hillslope gradient for surveyed drains in the Albert River catchment and the threshold curve of Croke and Mockler (2001) (Eq 1). The threshold line predicts that a gully would form at all road drainage locations plotted above the line.

The predicted probability of gully occurrence was higher than 0.1 for 96.6% of the gullies, higher than 0.5 for 74.6% of the gullies and higher than 0.9 for 25.4% of the gullies. Using the line for a probability of 0.5, the overall accuracy of the predictions is 67.7%, which is only slightly better than the overall accuracy obtained with the threshold equation of Croke and Mockler (2001). Clearly, there are additional parameters than the two selected in this equation that affect gully occurrence in the catchment. However, the threshold equation using these area slope parameters provides a methodology that highlights sites where the probability of gully occurrence is largest.

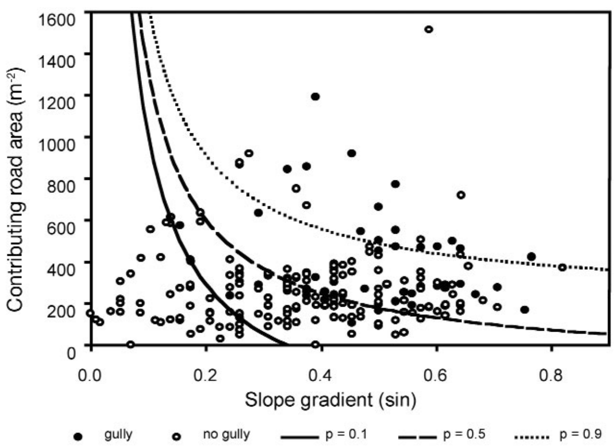

Figure 2. Road contributing area versus slope gradient for surveyed drains in the Albert river catchment, with threshold curves based on a logistic regression equation fitted to the data, for a probability of 0.1, 0.5 and 0.9.

3. MODEL INPUT PARAMETERS

To use these gully threshold curves to distinguish between gullied and diffuse pathways, we need hillslope gradients and runoff-contributing areas as input parameters to our model. The Vbt5 model of Hairsine *et al.,* (2001) is applied to calculate the probability of runoff reaching the stream through a diffuse delivery pathway. To apply the Vbt5 model to define connectivity through diffuse pathways, we need road contributing area, and the available hillslope length between the drain outlet and the stream.

In our study of the Albert River catchment, we collected field data on slope gradients and road contributing areas to drains. The distance from the drain to the stream was measured by manually drawing flowlines from the drain outlet to the stream perpendicular to 10 m interval contour lines. Generally, this detailed information is not readily available and we need to extract these input parameters from available data such as the DEM and the map of known streams in the catchment.

3.1 Slope

The derivation of slope gradients from a DEM is a standard GIS procedure. However, the accuracy of the calculated values depends on the accuracy and resolution of the DEM. Slope gradients are calculated from a DEM with a 20 m resolution for all surveyed drains in the Albert River catchment and are plotted against measured values in Figure 3. Slopes calculated from the DEM generally underestimate the actual slope gradient.

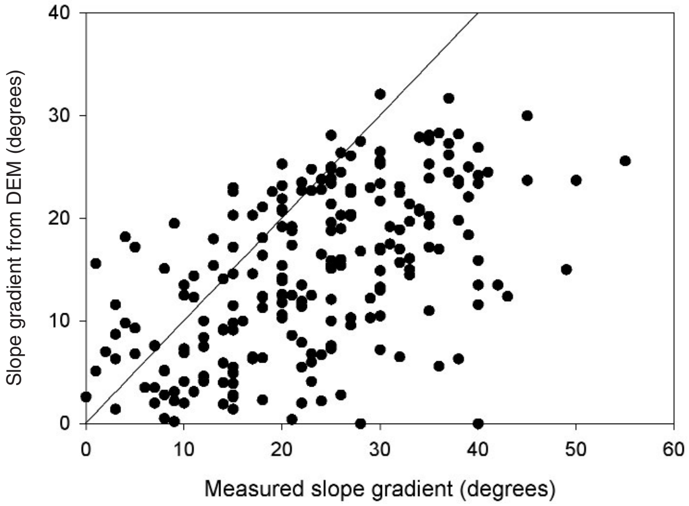

Figure 3. Slope gradients calculated from the DEM versus slope gradients measured in the field, for the surveyed drains in the Albert River catchment. The line in the graph presents the 1:1-line.

3.2 Distance

For all surveyed drains with a diffuse pathway at the outlet the Vbt5 model was applied to calculate the probability of runoff reaching the stream for a simulated 1 in 10-yr event in the Albert River catchment (Takken and Croke, 2004a). The distance from the drain to the stream is an important factor determining this connectivity. Figure 4 shows the cumulative frequency (%) of the distance to the stream for all drains with a diffuse pathway and for those drains that the model predicts would contribute runoff to the stream.

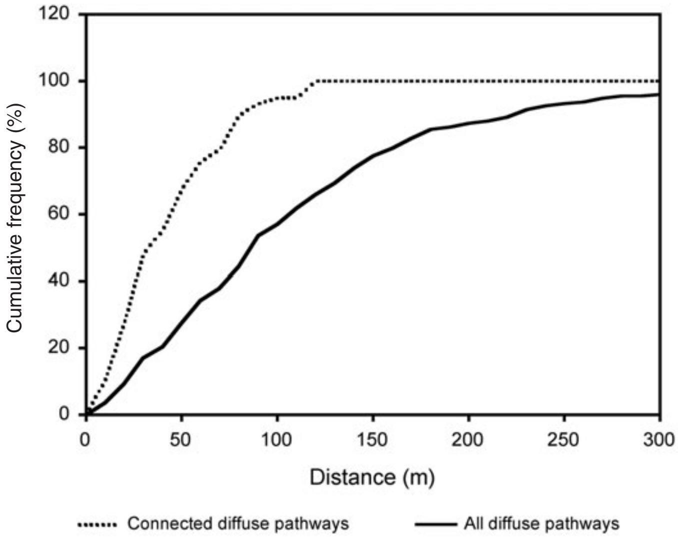

Figure 4. Cumulative frequency of the distance to the stream for all drains with a diffuse pathway and for the part of the drains that are predicted to delivery runoff to the stream during a 1 in 10 yr event.

Clearly, the drains that contribute runoff to the streams are drains located relatively close to the streams. The maximum distance from a drain to the nearest stream amongst the connected drains is 119 meters.

Based on a map of streams, the straight-line distance (i.e. the Euclidean distance) from any point in the catchment to the nearest stream can be calculated within the GIS. A vector map of the Albert River catchment was rasterised and used to produce a map of Euclidean distances to the streams. This gives a measure of the minimum distance to the nearest stream. However, water flows in the direction of the steepest descent, which is frequently not the same as the straight-line distance. To be able to calculate the distance in flow direction, the DEM was used to define flow directions for each grid cell. This flow direction grid was then used to define for each grid cell the flowpath down to the stream and the length of this flowpath is calculated (Figure 5).

The Euclidean distance under-predicts the manually measured distance to the stream. The difference between the two measures becomes larger with larger distances. On the other hand, the distance defined by following the flowdirections predicted from the DEM generally over-predicts the distance to the stream. Large errors in both calculated distance values relate to drains positioned at ridgetops, whereby the distance was automatically identified on the wrong side of the road. Although more sophisticated methods to define

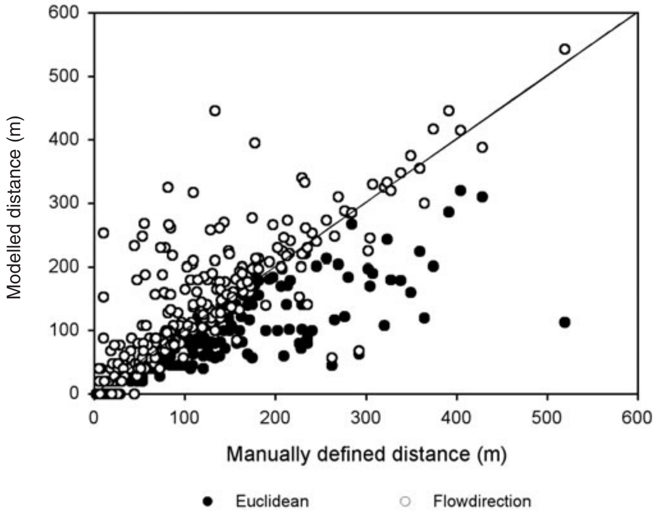

Figure 5. Calculated distances to the stream versus the manually derived distance to the stream.

the length of the flowpath from a drain to the stream are under development, again the accuracy of the DEM and the mapped streams greatly affect the quality of the modelled distance to the stream. If a DEM is used with a 20 m resolution, it is impossible to accurately define the flow distance to the stream. Errors of +/- 50m will be very common. The use of the Euclidean distance would be the most conservative method and the most accurate for areas close to the streams. It is in these proximal areas where an accurate measure is most important, and as such, this may be the best measure to use as it does not make use of a DEM.

3.3 Road contributing area

Algorithms to calculate the road length or area contributing to a drain from the road network, the DEM and the drain locations are under development. The procedure defines the location of tops and saddles along the road and uses these points to define flow direction along the road. Subsequently drains are superimposed on the road network to calculate the length of road contributing to the drain. Results so far have shown that it is extremely difficult to define the position of saddles and tops along the roads accurately using a 20 m DEM. In addition, field observations have highlighted that the actual area of road contributing to a drain is often not the same as the length of road between two drains. In many observed cases, at least part of the runoff by-passes the drain either through failure of the road crown or the development of road rills which cause flow to continue down-road. This limits our capability to precisely define road contributing areas to the existing drain network. However, our model can be applied in such a way that we define the maximum length of road that should be allowed to contribute to a drain to limit runoff delivery.

4. MODELLING MAXIMUM CONTRIBUTING ROAD LENGTH

Based on the slope gradient map derived from the DEM, the maximum length of road that can be allowed to drain to a particular location before gully initiation may happen was calculated using equation 1 and assuming a (contributing) road width of 5 meters.

The Vbt5 model predicts the length of the plume of runoff based on the total volume of runoff discharged by the drain during an event. A mean plume length equal to the distance to the stream corresponds to a 50% probability that runoff will reach the stream. The runoff volume is calculated from the amount of simulated rainfall, the infiltration rate of the road surface and the area of road contributing to the drain. A one in 10-yr event was simulated with a road infiltration rate of 11.7 mm/hr (Takken and Croke, 2004a). A contributing road width of 5 meters was assumed. Application of the Vbt5 model in this way allowed us to relate the distance to the stream to a maximum drain spacing that should be applied to obtain a probability of less than 50% that runoff would

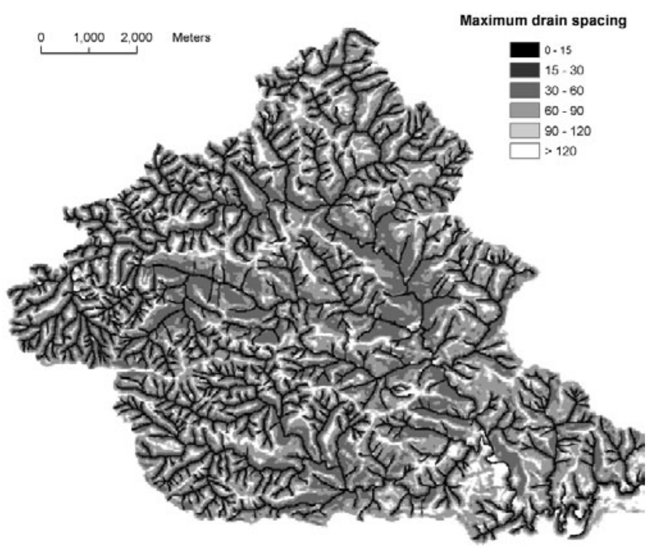

Figure 6. Map showing the maximum drain spacing that should be allowed along a road in order to limit gully formation and runoff delivery through diffuse pathways.

reach the stream through a diffuse pathway. The flowpath distances were then used to create a map of maximum drain spacing.

The two maps of maximum drain spacing were overlain and the minimum values of the two was taken to create a final map of maximum drain spacing, which is shown in Figure 6.

5. THE WAY FORWARD

Based on relatively simple models, a risk assessment of road to stream connectivity has been outlined here. Areas in the catchment that are unsuitable for road construction, or where the spacing between the drains should be very small can be highlighted. In general, areas with steep slopes and areas close to streams need to be avoided. This means that the number of direct stream crossings should be minimised and ridge-top roads are the best location in terms of limiting runoff and sediment delivery. The methodology applied here is relatively easy and will produce a risk map that is conservative, which is best for management purposes.

The results obtained by any model depend, on the one hand, on the accuracy and completeness of the model description, but also on the quality of the input data. Including more detailed process descriptions in models, increases the need for input parameters that often cannot be measured at the scale that is required within the model. Recent developments in remote sensing allow us to acquire more accurate elevation data by laser altimetry. This can result in DEM's with much smaller resolution. This type of data may help to improve model results. However, the cost of this type of data is high and therefore a relatively simple risk assessment map as the one presented in this study, may be the most cost-effective option available to forest managers to assess road to stream connectivity.

6. CONCLUSIONS

The type of predictive tools that could be applied to the management of harvesting/roading practices in forested catchments has been an on-going issue for debate in catchment modelling. The decision to apply simple field-based empirical relationships rather than highly parameterised models is likely to be made on the basis of both the objectives and cost-effectiveness of the project. In this study, we developed a process based framework for road runoff delivery via two specific pathways (gullies and diffuse overland flow) and described the procedure for the development of a catchment-scale risk assessment for road drainage. The occurrence of gullies at road drains can be adequately described in a predictive sense using two parameters of contributing area and slope. The probability of runoff reaching terms via overland flow pathways can be predicted using the Vbt5 model of Hairsine et al., (2002). The input parameters for this model are relatively few and easily collected in either the field or via a DEM. The current resolution of DEM's will limit the accuracy of our predictions but given the huge cost in obtaining more accurate data, we believe the current methodology represents the most cost-effective approach that can be incorporated into existing forest roading and harvesting databases. The common use of GIS by many of the state agencies and forest industry should mean that it is now possible to collate available topographic and road infrastructure data and develop catchment-scale risk assessment maps that will limit the delivery of road described runoff and sediment. We will continue to develop this approach and provide a working interface for application in the forest industry.

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Road to Stream Connectivity in Forestry Catchments

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Summary: Unsealed forest roads are the most important source for runoff and sediments in forested catchments. The probability of runoff and sediments generated on these roads being delivered to the streams depends to a large extent on the degree of connectivity between the road and stream networks. This paper describes a method to characterise the delivery pathways, which can help to identify connectivity hotpots and thereby minimise potential impacts from road-derived runoff and sediment.

1. INTRODUCTION

Sediment and nutrient delivery to streams depends on the degree of connectivity between the sediment sources and the stream network. The characteristics and spatial distribution of the sediment delivery pathways strongly affects connectivity and hence sediment delivery. In this paper we apply a methodology to characterise the delivery pathways and compare their spatial distributions in two catchments in southeast Australia.

2. DELIVERY PATHWAYS

Several studies have examined the mechanisms by which sediment delivery from roads to streams may occur (Montgomery, 1994; Croke and Mockler, 2001; La Marche *et al.,* 2001; Wemple *et al.,* 2001). In these studies the degree of connectivity between roads and streams is assessed based on a classification of the different types of delivery pathways. Here, we distinguish three types of delivery pathways (Figure 1). The most direct delivery of runoff from roads to streams occurs at stream crossings. Secondly, gullies may form at the outlet of the road drain, through which runoff is routed efficiently to the streams as concentrated channel flow. If the gully is continuous from the drain outlet to the stream, it acts as an extension of the stream network (Croke and Mockler, 2001). Finally, runoff and sediment can be delivered to the streams by diffuse overland flow.

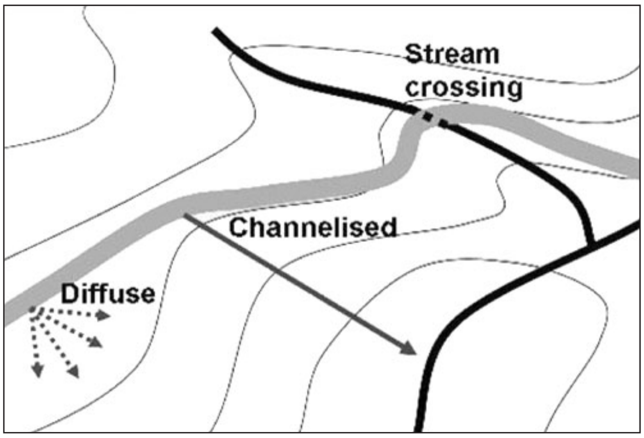

Figure 1. Types of sediment delivery pathways.

3. STUDY SITES

In this paper we compare two forested catchments in southeastern Australia. The first is located in the upper part of Cuttagee Creek catchment approximately 7 km south of Bermagui in NSW. It is used for hardwood forestry and is part of the Eden Management Area. It is situated within coastal foothill ranges with a relief range from near sea level up to 360 m. Slope gradients within the catchment range from 3 to 57%. Soils have developed from Ordovician metasediments and are predominantly gravelly-duplex loams 0.3 to 1 m deep.

The second study site is the upper part of the Albert River catchment in Victoria, located ca 15 km west from Yarram. This area is part of the Strezlecki Ranges and is managed for both hard - and softwood plantations. The terrain is more extreme, with slope gradients ranging from 2 to 63% and elevations range from 50 to 570 m ASL. A soil cover of predominantly brown dermosols has developed on steeply bedded Cretaceous mudstones and fine-grained sandstones. The soils are characterised by a weakly structured clay loam grading to a light clay in which structure is more developed. Soils are generally less than 1.5 m deep. Summary characteristics of the two catchments are presented in Table 1.

Table 1. Catchment characteristics.

	Albert	Cuttagee
Catchment area (km²)	82	38
Annual rainfall (mm)	1450	850
Type of Forest	Pine & Eucalypt	Eucalypt
Type of Forestry	Plantation	Native
Stream density (km/km²)	4.5	4.4
Road density (km/km²)	4.2	2.0

4. METHODOLOGY

Field survey

In both catchments data relating to road-to-stream connectivity were collected. In the Cuttagee Creek catchment representative road segments were selected by stratified sampling and all drains along these segments were surveyed (Croke and Mockler, 2001). In the Albert River catchment, all roads with regular drainage structures were surveyed, whereby drains were randomly selected by surveying every third or sixth drain along the road segments. Small roads without any drainage structures were not surveyed (Table 2).

Table 2. Total and surveyed lengths of roads in the two catchments in km.

	Albert		Cuttagee	
	Total	**Surveyed**	**Total**	**Surveyed**
All roads	345	113	75	24
Major gravel road	83	75	15	4
Minor gravel road	91	27	20	9
Natural surface road	172	11	30	11
Small Track	-	-	11	0

At each surveyed drain measurements were made of the area of road contributing to the drain and of the slope gradients of the road surface and the discharge hillslope. The hillslope below the outlet was investigated and notes were made on the presence or absence of a gully and on any other evidence of erosion and deposition.

GIS analysis

The study areas were classified in three slope position classes: ridgetops, midslopes and valley bottoms (using the slope position model of Hatfield, 2003). This resulted in classified slope position maps, whereby ca 25% of the area was defined as valleybottom, ca 25% as ridgetop and ca 50% as midslope for both catchments. The road networks were overlain with the slope position maps to calculate the percentages of the road network located in these three hillslope positions.

For each surveyed drain, the distance from the drain to the stream was measured by manually drawing a flowline from the drain location to the stream perpendicular to contour lines (10 m interval) within the GIS.

Assessment of connectivity for diffuse pathways

For drains with a diffuse pathway at the outlet, the Vbt5 model of Hairsine *et al.,* (2002) was applied to calculate the volume of road-derived runoff reaching the stream by dispersive overland flow. Hereto, a rainfall event with a 10-year recurrence interval was simulated, corresponding to an event with duration of 30 minutes and an intensity of 45 and 86 mm/hr for Albert and Cuttagee respectively. For the road surfaces, a steady-state infiltration rate of 11.7 mm/hr was used, corresponding to the mean value found during rainfall simulations on unsealed road surfaces (Croke *et al.,* submitted). These rainfall and infiltration data were used in combination with the road contributing area to calculate the volume of runoff to each road drain. The Vbt5 model was then applied to define the volume of runoff that would reach the stream.

5. RESULTS

Road position and drain characteristics

In terms of road location, there is a clear difference between the two catchments. In the Cuttagee

Table 3. Surveyed road length classified according to slope position.

	Albert		Cuttagee	
	km	%	km	%
Survey. road length	113	100	24	100
Valleybottom	12	11	1	2
Midslope	42	37	2	9
Ridgetop	59	59	22	89

Catchment the majority of the roads (89%) are located on ridgetops, while in the Albert River catchment only 59% of the roads are ridgetop roads. In the Albert catchment, 37 and 11% of the roads are located in midslope and valley bottom positions respectively (Table 3).

Table 4. Drain characteristics.

	Albert	Cuttagee
Number of surveyed drains*	246	217
Culverts	210	21
Mitres	27	174
push outs	5	20
cross banks	4	2
Median slope at outlet (degrees)	23	13
Median road contributing area (m^2)	252	109
Median distance to streams (m)	92	125

*Excluding drains at stream crossings

In the Albert River catchment the major drainage structure is the culvert drain, while in Cuttagee mitre drains are more common (Table 4). The contributing area to a culvert is generally larger than to a mitre drain, which is reflected in the higher median road contributing area found for the Albert River catchment. In the Albert catchment the median slope gradient at the outlet is higher and the median distance to the stream is smaller than in the Cuttagee catchment (Table 4). This can also be explained by the larger percentage of roads in midslope and valley bottom positions in the Albert River catchment.

Connectivity pathways

In the Albert River catchment 37 of the surveyed drainage structures (14%) drained directly to a stream at a stream crossing (Table 5). In total more than 4.5 km of road is draining towards these drains. It should be noted that this figure only includes the surveyed drainage structures and that there are about three times as many stream crossings along the surveyed road segments. Moreover, the Albert River catchment has a large number of small natural surface roads without any drainage structures that were not included in the survey. Overlaying the complete road network with the streams in the GIS shows that the total number of stream crossings in the catchment is 255.

At 64 of the surveyed drains (23%) a gully was found at the outlet. Of the remaining 173 drains, the Vbt5 model predicts that 58 contribute runoff to the streams by diffuse overland flow during a 10 yr event (Table 5).

In the Cuttagee catchment only one stream crossing is present amongst the surveyed drains. This is a bridge where 113 m of road length drains directly into the stream. Overlaying road and stream maps of the Cuttagee catchment reveals that there are only 2 stream crossings in the catchment. At 41 drains (19% of surveyed drains) a gully was found at the outlet. Only 12% of the surveyed drains are predicted to contribute diffuse overland flow to the streams during a 10-yr event (Table 5).

Table 5. Connectivity pathways.

	Albert		Cuttagee	
	number	%	number	%
Surveyed drains*	273	100	218	100
Stream crossings	37	14	1	0.5
Gully	64	23	41	19
Diffuse connected	58	21	26	12
No connection	114	42	150	69

*Excluding some drains with missing values

Spatial patterns

In the Albert River catchment, the majority of the surveyed stream crossings are located along the Albert River road in the centre of the catchment (Fig 2). All other drains along the lower part of this road are connected through diffuse overland flow, due to their close proximity to the stream. Another area where diffuse connected pathways are concentrated is in the eastern part of the catchment (Fig. 2). These drain locations are characterised by a relatively long contributing road area (ranging from 90 to 748 m^2 with a mean of 337 m^2) and/or a small distance to a first order stream (4 to 143 m with a mean of 57 m). Gullies are more widely distributed over the catchment, but the highest concentration of gullies is found along the road at the southern end of the catchment. This area has pine plantations and a combination of long contributing road areas and steep slope gradients may explain the occurrence of the gullies here.

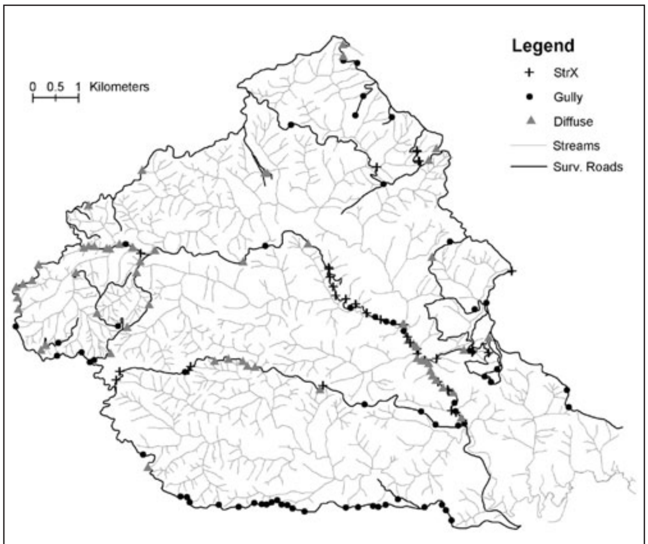

Figure 2. Spatial distribution of the different types of delivery pathways in the Albert River catchment. (Only the surveyed roads are shown.)

In the Cuttagee Creek catchment, a high concentration of gullies is found in the upper part of the catchment, where 19 gullies are formed along a cut-and-fill road. Road drains along this road are 30 to 248 m apart, resulting in a mean contributing area of 361 m^2 and runoff is discharged onto hillslopes with mean gradients of 22.4 degrees. This, in addition to the occurrence of a large rainfall event in the catchment immediately post road construction is likely to explain the predominance of gullied pathways along this road. Isolated gullied pathways are also present in the rest of the catchment, where again the coincidence of long road lengths and steep discharge gradients has contributed to gully development. The only stream crossing along the surveyed road segments is located on the same cut-and-fill road mentioned above. Drains that may contribute diffuse overland flow to the streams are widely distributed over the catchment. They are generally related to large contributing areas (ranging from 67 to 985 m^2, with a mean of 275 m^2) and/or short distances to first order streams (10 to 80 metres, with a mean of 145 m).

Figure 3. Spatial distribution of the different types of delivery pathways in the Cuttagee Creek catchment. (All roads are shown).

6. DISCUSSION

Stream crossings, where runoff with associated sediments and pollutants enters the stream, are a direct form of connectivity where there is a high risk for disturbing water quality and aquatic biota. The number and position of stream crossings is planned as part of the Forest Roading Plan within any catchment. Frequently, however, their association with stream disturbance and water has gone unnoticed. Previous work in the United States, however, clearly states that in general, it is suspended sediment discharged directly into a defined channel, which has greatest impact on the streams water quality (Bilby, 1989). Both catchments outlined in this paper present a very different spatial pattern of road-to-stream connectivity in part because of the differences in the number and position of stream crossings. In the Cuttagee Creek catchment there are very few stream crossings, due largely to the ridgetop position of most of the roads. In the Albert River catchment the number of stream crossings is much higher, due to the lower slope positions of the roads. In this situation the construction of drainage structures upstream from the stream crossings would greatly limit the length of road directly draining to the stream crossing. Presently, most stream crossings in the catchment are located on roads without any drainage structures.

In an early study of road related sediment production, Haupt (1959) suggests that drains discharging onto hillslopes where infiltration can occur, will have little impact on stream sedimentation. When runoff is dispersed onto the adjacent hillslope below a drain outlet, infiltration occurs and certainly should be promoted as part of any road-drainage practices. However, there are several examples of road placement where infiltration of road runoff is limited by either, or a combination of, the relatively short distance between drain outlet and stream edge and the large volume of road-derived runoff. As a result, plumes of runoff are commonly observed to reach the stream in non-channelised or diffuse pathways. The Vbt5 model used in this study predicts the volume (or probability) of runoff reaching the stream through diffuse delivery pathways, taking into account infiltration and detention storage. In the catchment examples used here, the model predicts some runoff delivered to the streams via this pathway. Prevention of this sort of connectivity can be best achieved by the location of roads away from streams, thus maximising potential for infiltration, and minimising road contributing area through drain location.

The final form of road-to-stream connectivity investigated here describes the development of channelised or gullied pathways between the drain outlet and the stream. In the Cuttagee catchment 41 gullies were found. Croke and Mockler (2001) report that for 93% of these gullies their occurrence could be explained by the road contributing area and slope gradient. This could also explain the even larger percentage of gullied pathways in the Albert River catchment, as discharge hillslope gradients and road contributing areas are generally higher in this catchment (Table 4). Prevention of this form of connectivity pathway is paramount to water quality protection. Once formed, these gullies are extremely difficult to rehabilitate and can form extensions of the stream network. Consideration of the controlling factors, slope gradient and road contributing area, and is essential in road planning and drainage guidelines.

7. CONCLUSIONS

The degree of road-to-stream connectivity in a catchment can be highly variable and depends to a large extent on catchment characteristics such as topography, slope, and position of the roads, the type of drainage structure and the length of road contributing to the drains.

By focussing on the location of drains and the type of delivery pathway at their outlet, an evaluation can be made of the degree of connectivity in the catchment. The methodology applied in this study is designed to highlight connectivity hotspots in the catchment, which provides important information to forest managers for water quality protection.

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Channel Types in Steep Forestry Catchments of Southeast Australia and Potential Responses to Forest Roads

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Summary: A channel type, such as a pool-riffle reach, can respond to changes in sediment supply and discharge in various ways based on the magnitude of change and the resistance of the channel to change. The resistance depends on channel gradient, sediment size and organisation, which are both reflected in the longitudinal profile. Channel response to increased sediment supply ranges from infilling of the bed framework material to the reduction of pool residual volume. This paper looks at characterising the channel types that occur on steep gradients based on a North American reach-scale classification scheme. Topographic and surface sediment surveys were conducted and analysed using spatial autocorrelation and grain size analysis. Channel reach types identified are discussed in relation to North American channel types and the potential response of channel type to change based on energy-resistance relationships. Future work on determining the effect of forest roads on channel bed composition, benthic and hyporheic invertebrate communities, based on channel type is outlined.

1. INTRODUCTION

It is known that roads generate sediment (Reid and Dunne 1984; Croke, Hairsine, *et al.,* 1999) and if connected to streams (Croke and Mockler 2001) lead to increased turbidity or suspended sediment loads (Grayson, Haydon *et al.,* 1993; Cornish 2001; Lane and Sheridan 2002). Few Australian studies have looked at the effect of road sediment on channel beds and the aquatic biota residing on, and within, the substrate (Richardson 1985; Davies and Nelson 1993). Most information has been drawn from different countries. Such studies have not acknowledged the diversity in channel morphology that may affect in-stream processes (Lisle 1987; Whittaker 1987) and ecological communities (Olsen, Townsend *et al.,* 2001). There is great variation in channel morphology of steep upland streams and rivers which are often simply referred to as either ‘pool and riffle bedforms’ or ‘pool-riffle reaches’.

Channel morphology present in rivers of the northern hemisphere mountainous regions show varying levels of coarse sediment arrangements that reflect specific form resistance to flow. (Montgomery and Buffington 1997) proposed a reach-scale channel morphology classification system for mountain rivers that relies on aspects of channel form that reflect channel processes. The model proposes that bed roughness configuration evolves to reflect the interaction between sediment availability and stream energy (Figure 1). Hence, for the length of a reach in which the combination of controlling variables are held constant, then the resultant bedform should be replicated downstream until a change in one or more of the controlling variables eventuates and results in a bedform change and therefore a different reach type. Similarly, any temporal change in the controlling variables in the reach may effect bedform composition. Responses may range from the complete destruction of bedform (Harvey 1987; Harvey 2001) to a change in channel bed composition such as infilling of the interstitial space of the channel bed framework (Lisle and Hilton 1999). If reach types equate to specific interactions between stream energy, substrate resistance or size and sediment availability, then reach type may also provide a template for aquatic habitat types. A reach-scale classification system may, therefore, be the most appropriate scale to evaluate the effects of forest catchment disturbance on in-stream processes, both geomorphological and ecological.

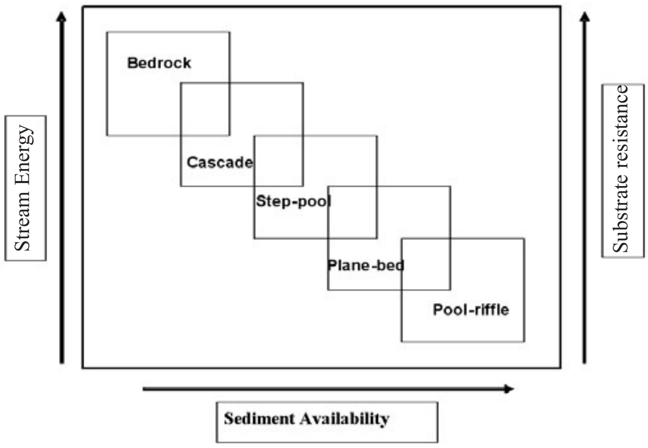

Figure 1. A conceptual model of interaction between stream energy, substrate resistance and sediment availability that relate to specific bedforms which are continuous over a reach. Modified after Montgomery and Buffington (1997).

It is acknowledged that there is a significant gap in knowledge of process based classification (Knighton 1998) which is partly due to the lack of data and testing of recently proposed models (Madej 1999; Wohl 2000). This paper reports on measures and evaluation of bed roughness configuration in SE Australian steep upland streams with respect to the conceptual models of reach scale channel morphologies along energy gradients as described for Northern Hemisphere mountain rivers. A template of channel reach morphologies based on form (bed roughness configuration) and processes (interacting flow hydraulics) will be developed upon which channel response to change can be evaluated. Specific questions are: 1) Does the Northern Hemisphere model of reach morphologies apply to SE Australian channel types? 2) What are the characteristics of SE Australian steep upland streams? 3) Does lithology affect reach type? and 4) Does forest road density affect reach type?

2. METHODS

Surveys sites were located within the Lachlan Fold Belt including coastal mountains and eastern slopes of SE NSW and ACT. To quantify longitudinal channel bed profiles, 30 reaches encompassing a range in gradients, two lithologies (either metasediment or granite) and two forestry types (plantations or native harvest forests) were surveyed systematically using a Total Station and stadia to measure channel bed elevation at 1 or 2 m intervals along the centre line of the channel. Surface water height was not included because channels were predominantly dry due to drought conditions. Minimum channel length surveyed was 10 times the channel bankfull width. A reach contained no tributary confluences and uniform channel confinement and gradient. Reach grain size was surveyed systematically with a minimum of 100 clasts selected and intermediate or b-axes measured (Wolman 1954).

Longitudinal reach profiles were analysed by Moran's I spatial autocorrelation analysis (Legrande and Fortin 1989) to determine departures from spatial randomness. The longitudinal profiles were de-trended using either linear or quadratic regression analysis, and the residuals analysed by Moran's I statistic. The statistic is interpreted using correlograms which show lag distances of spatially dependent points (bars or pools) and spatially independent points (random) along a reach (Figure 2). The spacing and length of channel units (i.e. bars) and bedforms (i.e. bar-pool couplet) can be interpreted at the reach-scale. Reach grain size data were used to determine median (D50) and D84 grain sizes.

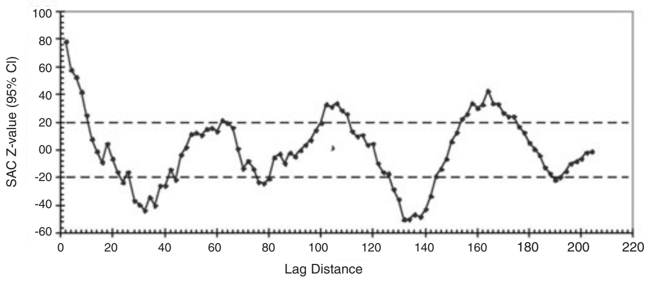

Figure 2. A correlogram of Moran's I spatial autocorrelation analysis on a de-trended longitudinal profile.

3. RESULTS AND DISCUSSION

Reach grain size data illustrate two different distributions based on catchment lithology. Metasediment reaches illustrate a unimodal distribution with a mode of pebble to cobble (Figure 3). Granite reaches illustrate a bimodal distribution with modes of sand to fine gravel and boulders (Figure 3). These different modal distributions can be explained in terms of rock weathering processes. Reaches from both lithologies contain bedrock outcrops.

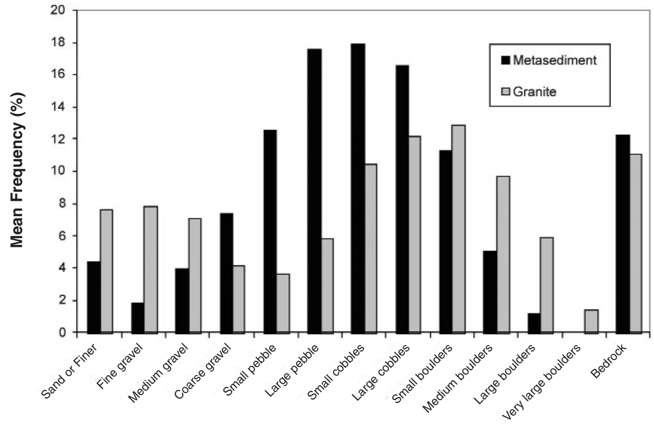

Figure 3. Mean grain size distributions for metasediment (black) and granite (grey) reaches.

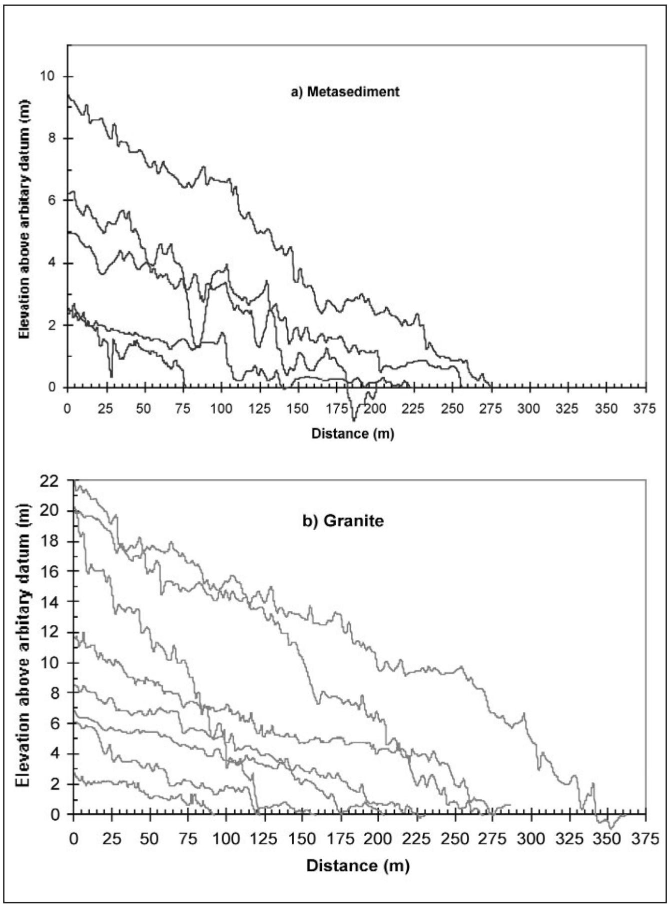

Figure 4. Sample of reach gradients from a) metasediment catchments and b) granite catchments.

Reach longitudinal profiles varied considerably between metasediment catchments and granite catchments. Granite catchments exhibited a far greater range in reach gradients, up to 14%, compared with reaches of metasediment catchments which ranged up to 3.5% (Figure 4). A feature of metasediment reaches not evident in the granite reaches is the occurrence of deep scour pools along the profile (Figure 4a). Differences in reach gradients between granite catchments and metasediment is not due to bias in site selection, as relief and mean hillslope gradients are greater for metasediment catchments (Figure 5).

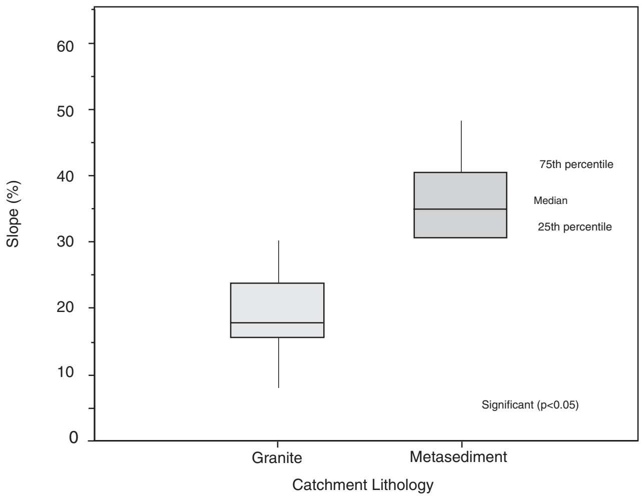

Figure 5. Box plots of mean hillslope for surveyed metasediment and granite catchments in SE NSW.

Spatial autocorrelation analysis and reach grain size data describe five main reach types, (1) Bedrock step-pool, (2) Core-boulder cascade, (3) Forced bar-pool, (4) Irregular step-pool and (5) Forced plane-bed (Figure 6).

Bedrock step-pool reaches occurred at 26% of sites in both metasediment and granite lithologies. Reach gradients ranged from 3% to 14% and intervals between steps ranged from 1 to 4 times the channel widths in length, with step length less than one channel width.

Core-boulder cascade reaches occurred in 9% of sites and only in granite lithologies. Gradients ranged from 7% to 10%. Core-boulder cascade reaches are non-alluvial because the D_{84} grain size (2.0 m) exceed the competence of the stream. Spatial autocorrelation analysis indicates randomly positioned boulders along the reach with no significant pools.

Forced bar-pool reaches occurred at 18% of sites, and only in granite lithologies. Reach gradients ranged from 2.5 to 4.5% and D_{84} grain sizes from 0.5 m to 1.35m. As for core-boulder cascades, the forcing element in the reaches exceed stream competence and therefore, bars are very stable. This stability may result in long-term (greater than 50-100 yrs as reported by (Grant, Swanson *et al.,* 1990) sediment trapping and storage with consequences for benthic and hyporheic communities. Bar spacing ranges between 6-13 channel widths in length and bar lengths of between 1-3 channel widths.

Irregular Step-pool reaches occurred at 12% of sites in both metasediment and granite lithologies. Reach gradients ranged from 2.5 to 4.0% and D_{84} ranged from 0.25m – 0.45m. Spatial autocorrelation analysis describe irregular step and bar features, with bar spacing ranging from 1-7 channel widths in length. Bar mobilisation may require floods with greater than 20 average return interval (ARI) (Chin 1998).

Forced plane-bed reaches occurred at 35% of sites, and only in metasediment lithologies. Reach gradients ranged from 1 to 3% and D84 ranged from 0.04m – 0.25m. Spatial autocorrelation analysis describes long distances (up to 15 channel widths) of random or uniform positioned cobbles along the profile interrupted by significant scour pools forced by bedrock bars, bedrock enforced bends or lateral bedrock outcrops at spacings of between 7 to 15 channel widths.

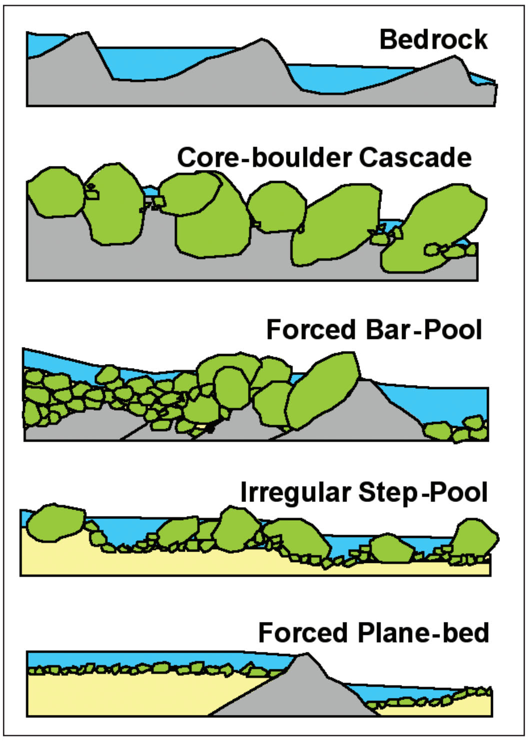

Figure 6. SE Australian steep upland bedforms making up reach types.

4. CONCLUSIONS

In general, the reaches surveyed do not conform to previously reported North American classification model. Instead, five reach morphologies were found and classified as bedrock step-pool, core-boulder cascade, forced bar-pool, irregular step-pool and forced plane-bed reaches. Catchment lithology does affect channel reach morphology with granite reaches having much greater ranges in gradient than metasediment reaches. Forced plane-beds reaches dominated metasediment steep upland streams while forced bar-pools reaches were the dominant form in granite steep upland streams. The five reach morphologies vary in sediment calibre and resistance as well as sediment organisation patterns. This provides a physical habitat template for conducting studies on ecological processes occurring in steep upland streams.

5. FUTURE WORK

Eight streams in SE NSW have been instrumented with stage monitoring and bedload sampling devices to yield information on annual bedload fluxes in relation to reach morphology type and unsealed forest road crossings. Samples of the channel bed will be taken at varying distances from road crossings and will be analysed for porosity, percentage of fines and abundance and diversity of biota living on and within the channel bed leading to a quantification of the in-stream effects of road crossings.

6. ACKNOWLEDGEMENTS

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Utilising Climate Variability Insights to Improve Forest Fire Management

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Summary: Recent research has revealed that the risk of forest fire across New South Wales (NSW) varies significantly from year to year. Importantly, the variability in forest fire risk is strongly related to known large-scale climate variability phenomena, namely the El Niño/Southern Oscillation (ENSO) and the Inter-decadal Pacific Oscillation (IPO). The risk of forest fire is significantly enhanced during the El Niño phase of ENSO and the chance of fire is even further elevated during El Niño events that occur when the IPO is negative. An example of this enhancement of El Niño impacts is the severe forest fires experienced in NSW during the spring/summer of 2002/03 which was the first El Niño event in the current negative IPO phase. It is now possible to predict ENSO events up to nine months in advance while the IPO is a feature of the climate system with decadal to multi-decadal persistence. This study reveals the potential to use long term climate variability insight to predict seasons with significantly elevated forest fire risk with sufficient lead time to enable systematic and targeted preparation.

1. INTRODUCTION

The weather variables that generally increase the risk of forest fires are low precipitation and relative humidity combined with high temperature and wind speed (Luke and McArthur 1986). The high variability of rainfall and temperature in northern and eastern Australia has been strongly associated with the regional influence of the El Niño/Southern Oscillation or ENSO (eg. Allan 1988; Stone and Auliciems 1992; Kiem and Franks 2001). These studies indicate that large-scale climate variability on annual/inter-annual timescale influences at least two (rainfall and temperature) of the four weather variables that contribute to forest fire risk in Australia.

In addition to the annual/inter-annual effects of ENSO, a number of studies have also examined decadal and longer scale climate variability. It has been demonstrated that the Inter-decadal Pacific Oscillation (IPO) modulates the strength and nature of the ENSO cycle (Power et al. 1998, 1999; Folland *et al.,* 1999; Allan 2000). The IPO is the coherent pattern of sea surface temperature variability occurring on inter-decadal time scales over the Pacific Ocean and is similar to the Pacific Decadal Oscillation or PDO (Mantua *et al.,* 1997; Franks 2002a). Importantly, Power *et al.,* (1999) showed that individual ENSO events (i.e. El Niño and La Niña) had stronger impact across Australia during the negative phase of the IPO. Furthermore, it has recently been shown that in addition to influencing the magnitude of ENSO impacts the IPO also modulates the frequency of extreme ENSO events, leading to multi-decadal periods of elevated flood or drought risk depending on the phase of the IPO (Franks 2002b; Franks and Kuczera 2002; Kiem and Franks 2004; Kiem *et al.,* 2003).

Recent research by Verdon *et al.,* (2004) has shown that the risk of forest fire across New South Wales (NSW) varies significantly from year to year. This variability was shown to be strongly related to both ENSO and the IPO. This paper summarises the results found by Verdon *et al.,* (2004) and highlights the potential to use this insight to improve bushfire management practices.

2. CLIMATE VARIABILITY AND FIRE RISK

The daily risk of forest fire was assessed using the Forest Fire Danger Index (FFDI), developed by McArthur (1967). The FFDI is designed for general fire danger forecasting purposes and is in common use throughout eastern Australia. Verdon *et al.,* (2004) adopted the term "high" fire danger to mean high, very high or extreme fire danger (i.e. FFDI>12).

The average proportion of days between September and February with "high" fire risk during El Niño and non-El Niño years (i.e. La Niña or Neutral) were compared to determine whether there is any difference between the two phases. A simple test of proportions was applied to determine if the number of days at "high" fire risk in El Niño and non-El Niño years is significantly different. Verdon *et al.,* (2004) found that the probability of daily "high" fire danger during El Niño years is significantly different to non-El Niño years at the <1% level for 19 of the 22 stations studied.

To illustrate the affect of ENSO on fire risk, the ratio of the probability of "high" fire risk during El Niño years to non-El Niño years was also calculated. The percentage increase in the probability of daily "high" fire risk during an El Niño is displayed in Figure 1.

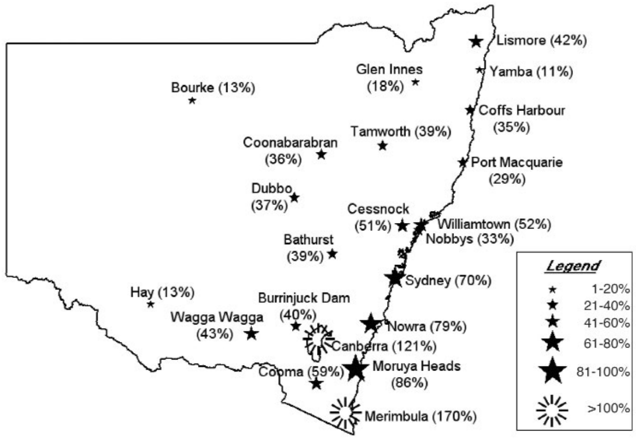

Figure 1. Percentage increase in the probability of daily "high" fire risk during El Niño years when compared to non-El Niño years (Verdon *et al.,* 2004).

The probability of daily "high" fire danger is shown to be markedly increased during El Niño years at all stations investigated. The increase in the proportion of days classified as "high" fire danger during El Niño years is 51% when averaged across the 22 study sites. The influence of El Niño appears to be strongest in the south-eastern corner of NSW with Canberra and Merimbula displaying an increase in risk of 121% and 170% respectively.

The probability of daily "high" fire danger in El Niño years occurring in the negative IPO periods, when ENSO impacts are enhanced, was also compared with the probability of "high" fire danger in all other El Niño years to determine whether there is a significant difference.

The proportion of days with "high" fire risk during El Niño years, that occur when the IPO is negative was found to be significantly different (at the <1% level) to the proportion during all other El Niño years at 11 of the 14 study sites whose 'fire weather' data records spanned both negative and non-negative IPO phases.

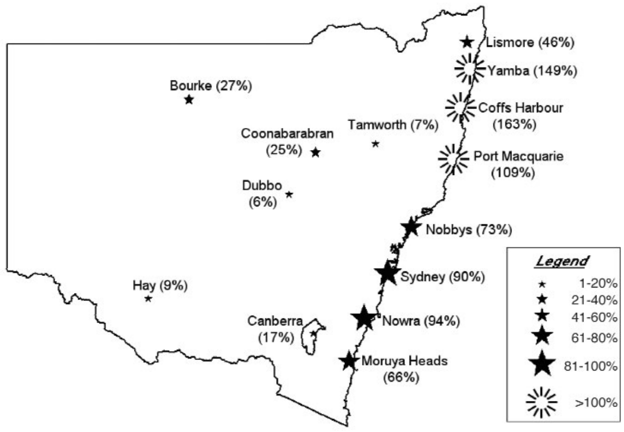

Figure 2. Percentage increase in the occurrence of daily "high" fire risk associated with El Niño and IPO negative years compared to all other El Niño years (Verdon *et al.,* 2004).

Figure 2 displays the percentage increase in the probability of daily "high" fire risk during IPO negative El Niño years when compared with all other El Niño years.

Figure 2 demonstrates that the probability of "high" fire risk markedly increases during IPO negative El Niño years at all 14 study sites.

To further investigate the magnitude of fire risk associated with IPO negative El Niño years, due to the IPO induced amplification of El Niño impacts, the probability of daily "high" fire danger in IPO negative El Niño years and all non-El Niño years was also compared.

The proportion of days with "high" fire risk during IPO negative El Niño years was found to be significantly different (at the <1% level) to the proportion during non-El Niño years at all 14 stations studied.

Figure 3 displays the percentage increase in the probability of daily "high" fire risk during IPO negative El Niño years when compared with non-El Niño events.

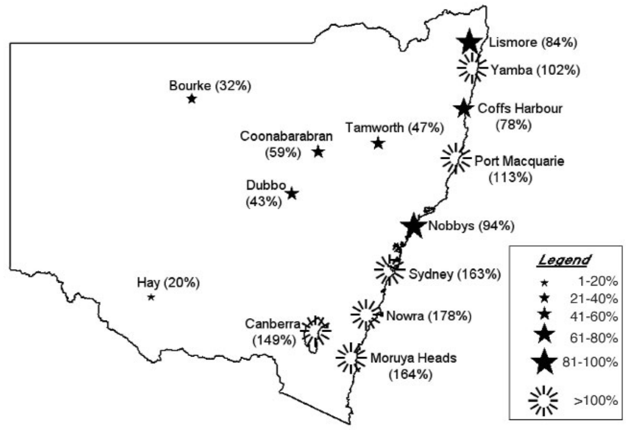

Figure 3. Percentage increase in the occurrence of daily "high" fire risk associated with El Niño and IPO negative years compared to non- El Niño years (Verdon *et al.,* 2004).

Figure 3 shows that the increase in the probability of a "high" fire danger day occurring during IPO negative El Niño events is even greater than the increase observed when all El Niño events are compared to non-El Niño events. The average increase in the risk of "high" fire danger across the 14 study sites during IPO negative El Niño events is 95% (which is much greater than the average increase of 51% obtained when all El Niño and non-El Niño events were compared). The greatest impact of the IPO and El Niño on fire risk appears to be along the east coast and the south-eastern corner of NSW. Yamba, Port Macquarie, Sydney, Canberra, Nowra and Moruya Heads all display an increase in forest fire risk of greater than 100% during IPO negative El Niño years.

3. MANAGEMENT IMPLICATIONS

This paper shows how observed multi-temporal scale climate variability influences the risk of forest fire in NSW, Australia. In particular, during the El Niño phase of ENSO the proportion of days with a "high" fire danger rating are significantly increased and this risk is even further enhanced when the IPO is negative. A recent example of the enhancement of El Niño impacts can be found in the severe forest fires experienced across much of NSW during the spring and summer of 2002/03, which was the first El Niño event in the current negative IPO phase. Despite the 2002/03 event being of only 'moderate' strength, with respect to sea surface temperature and atmospheric pressure anomalies, the impacts experienced in NSW were far worse than other 'stronger' El Niño events that occurred when the IPO was not negative.

The study by Verdon et al 2004 reveals the significant potential to use long term climate variability insight to predict forest fire risk in NSW. It is now possible to accurately predict ENSO events up to nine months in advance (Kiem and Franks 2001). The implications of this are particularly relevant to the Australian climate due to the fact that NSW has been plagued with devastating forest fires since colonisation, and whilst it is impossible to eradicate fires of this nature, it is possible to be prepared.

A possible management plan to improve bushfire preparedness would be to maximise hazard reduction burning and clearing of fuels prior to fire seasons occurring in an El Niño year and in particular, El Niño years within the IPO negative phase. In the event of a prediction indicating an upcoming season of high fire danger additional targeted hazard reduction during the autumn and winter prior to the fire season could be conducted. Implementing systematic and targeted fuel reduction in periods prior to an elevated risk season may in the long term save money, property and possibly lives.

Whilst hazard reduction burning would prove beneficial in preventing devastating bushfires in the wake of an El Niño event, simple management techniques could also be adopted for the management and suppression of fires that do occur during these years. It is imperative that sufficient personnel and volunteers as well as fire suppression equipment are available to manage fires that do break out during the fire season. Therefore additional training and recruitment prior to the fire season in El Niño years is a simple measure that could be taken to prevent, manage and suppress serious bushfires that are likely to occur.

Bushfire preparedness is an important aspect of bushfire management. This preparation is made possible with the advantage of up to a nine-month lead-time on an approaching serious fire season. A number of general management practices that could be implemented during El Niño years, especially those occurring in the IPO negative phase have been outlined.

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Monitoring the Potential Impacts of Forestry Activities on Water Quality in NSW: Preliminary Results from the Middle Brother Paired Catchment Program

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Summary: Water quality monitoring was conducted in a paired catchment native forest program over an eight year period to assess the possible impacts of forestry activities on low- and high-flow turbidity and total suspended sediment (TSS) concentration. Preliminary results of analysis suggest that forestry activities resulted in slightly elevated turbidity at low flows in the impact catchment for a period of at least twelve months after the commencement of harvesting. After this time there was a return towards pre-harvest levels. Further analyses are required to determine any possible impacts on high flow turbidity, as well as equivalent analyses of TSS data. Given that the majority of sediment is transported during high-flow events, it is imperative that these data are analysed to assess, where possible, any changes in turbidity and TSS response to storms in the post-harvest period.

1. INTRODUCTION

Forestry activities, including timber harvesting and road construction, in the absence of mitigating practices, have the potential to impact upon stream water quality, mainly in the form of increased suspended sediment concentrations and/or in-stream turbidity. Elevated stream turbidity and suspended sediment concentrations are dependent upon an increase in soil erosion and its subsequent delivery to the stream channel network.

In the event that soil is disturbed, the potential for water pollution via higher sediment concentrations is mainly dependent upon: the timing, magnitude, intensity and frequency of rainfall events experienced post-disturbance (eg. Prosser and Williams, 1998; Croke *et al.,* 2001); the presence of uninterrupted linkages between the source of sediment and a receiving drainage feature (eg. Croke and Mockler, 2001); the occurrence of streamflow in the receiving drainage feature; and connectivity throughout the drainage network.

In a forest environment managers cannot control hydrological events, which are often unpredictable. However, it is possible to employ forest management and soil conservation measures that aim to either prevent or reduce soil erosion. Soil conservation measures introduce discontinuities in the landscape that aim to prevent direct linkages between sites of disturbance and drainage features. Accordingly, State Forests of NSW applied for, and has been issued with, five Environment Protection Licences (EPLs) by the Environment Protection Authority (EPA) for the carrying out of forestry activities on State forests and Crown timber lands. The object of each licence is to require practical measures to be taken to protect the aquatic environment from water pollution potentially caused by these activities, and to ensure that monitoring of the effectiveness of the licence conditions in achieving the relevant goals is undertaken.

The conditions and practical measures contained within the EPLs are many and varied. These include, but are not limited to: soil conservation measures for bridges, culverts and causeways; appropriate drainage spacings on roads and snig tracks; seasonal harvesting restrictions; slope restrictions for harvesting and road construction activities; wet weather restrictions on the use of roads and log dumps; mass movement hazard conditions; soil dispersibility conditions; and protection of drainage features by the use of filter strips and/or buffer strips

While some of the individual conditions of the EPLs have been tested to some extent (eg. Lacey, 2000; Walsh and Lacey, 2003), the overall effectiveness of the conditions in meeting the objectives of the licence has not been tested at a catchment scale. In an attempt to provide some feedback on the effectiveness of the EPL conditions, State Forests is required to implement a water quality monitoring (WQM) program. The current program was devised in 1999 and came into effect in February 2000. It replaced a previous program that operated between 1994 and 1999. The objective of the WQM program is to determine if there is an identifiable impact on water quality from licensed forestry activities and if so, to quantify the level of that impact. To meet these objectives, State Forests conducts water quality monitoring at a number of representative locations in both native forests and softwoods plantations.

The scope of the revised monitoring program incorporates both native hardwood forests and softwood plantations at 21 sites. These include two sites in Middle Brother SF (Mid North Coast Region, Native Forests), five sites in Yambulla SF (South East Region, Native Forests), five sites in Kangaroo River SF (North East Region, Native Forests), five sites in Canobolas SF (Macquarie Region, Softwoods) and four sites in Bago SF (Hume Region, Softwoods). The native forests program consists of a series of control (not harvested) and impact (harvested or to be harvested) catchments in each monitored State forest. The softwoods program consists of upstream native forest control (not harvested), internal softwoods impact (harvested or to be harvested) and downstream integrated impact sites in each monitored State forest.

The aim of this paper is to present preliminary results from the paired catchment native forest program that was implemented in Middle Brother State Forest between November 1994 and May 2003.

2. STUDY AREA: MIDDLE BROTHER STATE FOREST

Middle Brother State Forest (31º38'S, 152º42'E) is located within State Forests' Native Forests Division Mid North Coast management region, and comprises the impact catchment (MBRO01). The control catchment (MBRO02) is adjacent and was formally in Middle Brother State Forest but on 1st January 1999 was transferred to the National park estate (Pjurrigan National Park). The MBRO01 catchment is 124.9 ha in area and drains into Batar Creek, while the MBRO02 catchment is 236.0 ha in area and drains into Stoney Creek. Both Batar Creek and Stoney Creek are tributaries of the Camden Haven River.

The Mid North Coast region of NSW experiences a mild mid-latitude climate featuring hot summers in the absence of a distinct dry season. There is a distinct summer rainfall maximum at Middle Brother, with a winter/spring minimum. Mean annual rainfall for the period 1994-2003 was 1740mm. The Middle Brother catchments are situated on Late Triassic granitoid rocks that have intruded the Lorne Basin (Brunker et al., 1970; Major *et al.,* 2002). Soils in the Middle Brother catchments are generally podzolics and have been classified as predominantly soil regolith stability class 1, which are high coherence soils with low delivery potential (Murphy *et al.,* 1998). Both catchments were assigned an Inherent Hazard Level (IHL) of 2, which equates to a "high soil erosion and water pollution hazard", according to the EPL (on a scale of 1 to 4).

The majority of both the impact and control catchments comprise Moist Blackbutt or Dry Blackbutt forest, according to the revised Baur classification (Forestry Commission of NSW, 1989). These forests have been intensively managed for multiple uses, including timber production for over 70 years and have been thinned on an average 10-year cycle for the last 30 years. The most recent harvesting to occur in the MBRO01 catchment was in 1990. The forests have a variable age-structure but are predominantly either maturing stands or areas of younger regrowth (Major *et al.,* 2002).

3. HARVESTING OF THE IMPACT CATCHMENT

Harvesting of the MBRO01 catchment commenced on 13th December 1999 and was completed by March 2000. It is estimated that 52 ha (41.6%) of the catchment was harvested with approximately 25.8 ha of the catchment being sufficiently disturbed to be identified from aerial photographs taken in February 2001, providing an indication of the degree of canopy removal (ie, ~20.7%). Parts of the catchment not harvested were too steep, inaccessible, pre-merchantable or excluded from harvesting due to

IFOA conditions. Approximately 80% of the harvested area was harvested using Single Tree Selection (STS). The remaining 20% was harvested using mainly STS with some Australian Group Selection (AGS). Logs were extracted using a tracked vehicle and the estimated total length of snig tracks was 2.5km.

Post-harvest burning took place in March and July 2000 and was of low intensity resulting in negligible soil exposure. The areas affected by burning were patchily distributed within the harvested area.

4. INSTRUMENTATION AND METHODS

Stream gauging stations were located at the outlet of each catchment while a tipping bucket rain gauge (pluviometer) and manual rain gauge were located at the summit of Middle Brother Mountain (MBRO03) in the headwaters of the MBRO02 catchment. Each gauging station was equipped with: a Pressure Transducer; an Automatic Pump Water Sampler; Data Logger; and Staff Gauge.

Stream height was logged at six-minute intervals. Routine water samples were collected at low flows, while event samples were pumped from each stream on the rising and falling limbs of flood events. Samples collected were then analysed in the laboratory for Turbidity (for the period 1995-2003) and Total Suspended Sediment (TSS) concentration (for the period 1998-2003).

5. PRELIMINARY RESULTS: LOW FLOW TURBIDITY

For the purposes of analysing results from the Middle Brother program, data were divided into the following two ranges:

(i) High Flow data – these data consisted of results from samples collected during flow events. Events were defined as having occurred when the stream rose by an amount equal to or greater than 0.12m and 0.16m at the MBRO01 and MBRO02 sites, respectively.

(ii) Low Flow data – these data consisted of results from all other samples that were taken during low flow periods, ie they were not taken during flow "events".

The remainder of this paper will be confined to a discussion of changes pre- and post-harvesting in low-flow turbidity. Analysis of low-flow turbidity involved the initial compilation of paired (between the control and impact sites) data points representing the period 18/5/1995 to 30/4/2003. Mean low-flow turbidity in the impact catchment was higher than in the control catchment for the 12-month period following the commencement of harvesting (Table 1). Subsequently mean low-flow turbidity levels decreased and were similar to the control catchment. While the low-flow turbidity was elevated due to the impact of harvesting, absolute values of turbidity were still low.

Logarithms were taken of the paired data and differences calculated for each observation, ie IVC=I-C (where I is Impact site and C is Control site). These IVC values were then plotted as a time series (Fig. 1). A range of smoothing methods was examined to reduce trend and seasonal effects, and to remove serial correlation in the residuals. Four loess smoothers with varying spans (of 2.5, 5, 10, 40) were examined, and based on the data and auto-correlation functions, a span of 10 was chosen. In Splus the loess smoother was fitted with model IVC~lo(day,span0.10) within the generalised additive model (GAM) function (Fig. 1).

Table 1. Summary of low flow turbidity for various periods at the impact (MBRO01) and control (MBRO02) sites. Values reported are means ± 2 standard errors.

Period	MBRO01 Low flow Turbidity (NTU)	MBRO02 Low flow Turbidity (NTU)	No. Samples
Pre-harvest	3.3 ± 0.7	3.4 ± 0.7	193
14/12/99 to 13/12/00	10.7 ± 5.6	4.2 ± 0.7	18
14/12/00 to 13/12/01	7.7 ± 7.0	2.9 ± 1.0	9
14/12/01 to 30/4/03	5.2 + 1.5	4.3 + 1.5	10

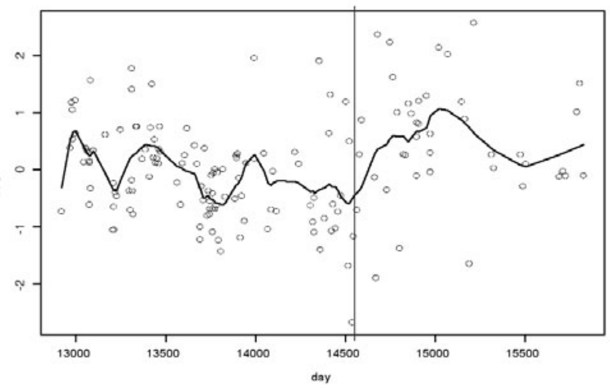

Figure. 1. Time series plot of IVC (log(turb impact) – log(turb control)) and loess fit for low-flow turbidity data. The vertical line represents the commencement of harvesting.

The loess fit of IVC differences is variable in both the pre-harvest and post-harvest periods (Fig. 1). However, the loess fit suggests that low-flow turbidity IVC values were consistently positive for at least 12 months after the commencement of harvesting, after which the trend suggests a steady return towards zero IVC. These trends are confirmed by observations summarised in Table 1.

6. DISCUSSION AND CONCLUSIONS

The data presented suggest that forestry activities impacted upon low flow turbidity in the impact catchment for a period of at least 12 months, after which there was a return towards pre-harvest levels. A limitation of the study is that there is a limited number of paired low-flow observations in the post-harvest period (n=37) when compared to the pre-harvest period (n=193), making statistical comparisons difficult.

While these preliminary analyses have provided an indication of impacts at low flows, an analysis of high-flow turbidity data needs to be conducted, as well as equivalent analyses of TSS data. Given that the majority of sediment is transported during high-flow events, it is imperative that these data are analysed to assess, where possible, any changes in turbidity and TSS response to storms in the post-harvest period.

7. ACKNOWLEDGEMENTS

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Hydrologic Effects of Forest Roads in a Steep Mountain Landscape of the Pacific Northwest USA

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Summary: Unpaved roads constructed to access forest resources are a ubiquitous feature of the mountain landscape in the Pacific Northwest U.S. Studies across the region indicate that forest roads alter runoff production, increase erosion rates, and negatively impact fisheries habitat. This paper describes on-going research at the H.J. Andrews Experimental Forest in western Oregon to examine the effects of forest roads on hydrologic processes. Intensive field measurements are used to develop models for predicting the interception of subsurface flow along road cuts. These models range from simple analytical solutions to numerical simulations and demonstrate that in this landscape, where roads are often constructed in midslope positions on shallow soils, the potential for subsurface flow interception and redistribution of hydrologic flowpaths is substantial. These findings offer some potential for planning new road networks or designing restoration strategies to remove roads.

Appendix A

PROGRAM

8:30 am	Depart for field site from CSIRO Discovery Centre, Black Mountain Site, Clunies Ross St, Acton
9:15 am	Kowen Forest: General management issues in unburnt pine plantation on stable soils, located approximately 15 km to the east of Canberra
10:00 am	Pine plantation harvesting issues, active plantation harvesting site, Kowen Forest
10:30 am	Morning tea, Old Homestead picnic area, general discussions
11:00 am	Depart for Cotter River Catchment
11:45 am	Lookout over burnt out Cotter Catchment from Winslade, on Cotter Road
12:15 pm	Lunch at Cotter River Picnic Area
13:15 pm	Depart for Pierces Creek Forest
13:30 pm	Pipeline Road site, divide into 3 discussion groups and rotate through 3 sites focusing on • hillside erosion processes (group 1) • roading management (group 2) • creek crossings and riparian management (group 3)
14:45 pm	Edge of Namadgi National Park to inspect erosion impacts in natural environment
15:30 pm	Erosion and sediment control practices in post fire environment
16:30 pm	Track closure and rehabilitation
17:00 pm	Cotter Picnic area for refreshments and general discussion
17:30 pm	Return to Canberra

Introduction

Today we will visit two contrasting environments of a working plantation forest east of Canberra and the severely fire affected Pierces Creek forest to the west of Canberra. ACT Forests have kindly given us unfettered access to all areas, and we will be accompanied by Andrew Winter, Senior Harvesting Forester to guide and explain operations as we go. We encourage everyone to contribute as much as possible, and hope that the tour will provide a great basis for some stimulating and positive discussion.

Stop 1: Kowen Forest

The first stop will be 15 km to the east of the City, at Kowen Forest, a 5800ha area of pine plantation which was not affected by the Canberra Fires. The plantation sits atop the Queanbeyan scarp, on shallow gravely soils derived from Ordovician metasediments. The terrain is gently undulating with only short sections of steeper slopes. The hillslope soils are stable and have low inherent erodibility. The drainage network has been incised extensively by gully erosion which developed after the post European clearing for agriculture. Plantations were established in the ACT in the 1930s and 40s to rehabilitate degraded lands and enhance the landscape, and it is only since the 1980s that a fully fledged forest industry has been established. We will be viewing the plantation from an elevated site on Fearnside Rd, to discuss a range of aspects of plantation management including establishment and maintenance, management of the road network and codes of practice. We will then travel to a site for a first hand viewing of plantation thinning. From here we travel to a picnic area in the forest for morning tea and to continue discussions.

Stop 2: Pierces Creek

After lunch we travel to Pierces Creek Forest, an area of 3800 ha, which was totally destroyed in the Canberra fires. There has been extensive salvage logging, and the site is now in preparation for replanting of pines. The terrain and soils contrast sharply with the Kowen plantation and contribute to challenging conditions in which to maintain a forest roading network. There is a far greater proportion of steeper and longer slope sections and this has resulted in a road network along lower slopes and drainage lines. The soils have developed on coarse-grained acid volcanic geology, and display a coarse textured massive topsoil overlying a weakly-structured clay subsoil. Soil drainage is impeded by the very low permeability of the subsoil, and they develop perched watertables on lower slope situations, which are extensive across the plantation. The drainage lines are extensively incised into the unconsolidated valley fills, forming a highly connected drainage system. The area has been severely affected by post-fire storms which have caused extensive erosion of slopes and drainage lines, and sediment export to the Cotter catchment.

The road system in Pierces Creek forest is presently being upgraded and rationalised, as part of the fire recovery process. The program includes realignment of roads away from drainage lines, road closures and rehabilitation, improving road drainage and construction of a range of soil conservation measures to stabilise degraded areas and retain sediment.

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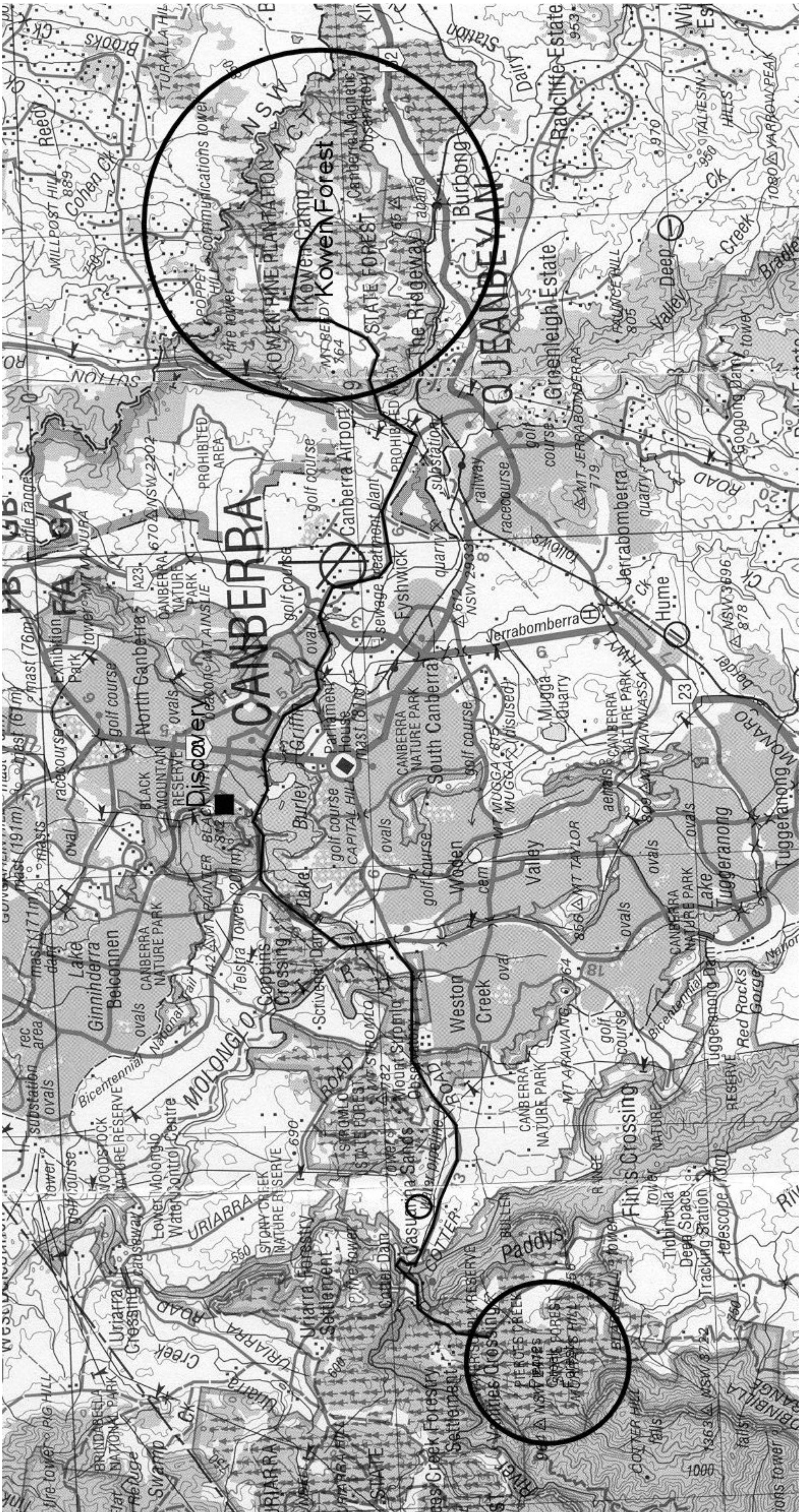
Notes


Kowen Forest
Discovery
CANBERRA
QUEANBEYAN
Canberra Airport
Jerrabomberra
Hume
Fyshwick
MOLONGLO
Weston Creek
Tuggeranong

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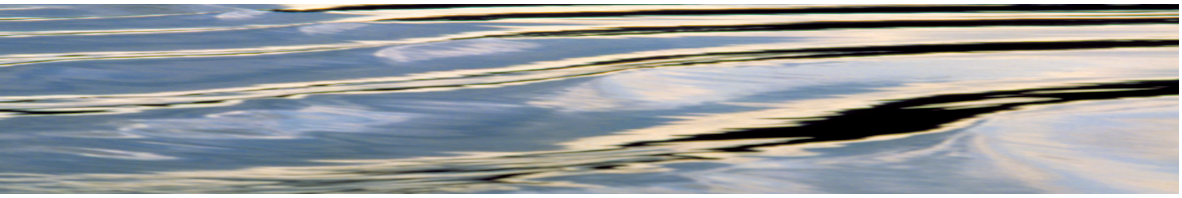

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