Revegetation of water supply catchments following bushfire: A review of the scientific literature relevant to the Lower Cotter catchment

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Executive Summary

This report synthesises the scientific literature pertaining to the impacts of different landuse and vegetation options upon key environmental parameters identified by the ACT government for Lower Cotter catchment. These environmental parameters are:

- water yield;
- water quality;
- controlling sediment yield to streams;
- biodiversity;
- bushfire management;
- weed control;
- road network requirements;
- ongoing maintenance; and
- fire sensitivity and regenerative capacity after fire.

The brief for this report is set out in Appendix A.

Many studies have shown that landuse or broad vegetation type (such as pine or native forest or pasture land) is not the sole determinant of water quality within a catchment. Rather, individual pollutant sources need to be considered. These may include roading and specific erosion processes that may have developed over a catchment's history, such as the incision of gullies into valley fills. For these reasons the scope of this investigation includes these potential threats to water quality.

By killing a large portion of the pine vegetation in the Lower Cotter catchment, the 2003 bushfires will have altered the catchment's hydrologic balance. The immediate reduction in live vegetation post-fire is likely to result in a short term (3-7 year) increase in catchment water yield. Beyond this and under re-establishment of either native forest or introduced pine species, water yields are likely to decline in the medium term (7–50+ years) as this vegetation grows and its water use increases. Studies show

similar trends in water yield over time between catchments planted to pines and native eucalypt vegetation. Revegetation of the former pine plantation area using grassy vegetation would result in the highest stream flow yield as evapotranspiration (water use) by grassy vegetation is markedly less than for trees.

The loss of vegetative cover from the Lower Cotter catchment after the 2003 bushfires has resulted in an increased rate of soil erosion from both hillslope and gully sources, impacting negatively upon water quality. The delivery of eroded sediments to waterways is a major source of pollution as undesirable nutrients (particularly phosphorus) are bound to fine sediment particles. Elevated nutrient levels in water storages promote algal blooms which can then cause oxygen depletion in standing water bodies such as reservoirs. This may trigger a series of chemical reactions involving the release of metals including manganese and iron into the water from bed sediments. Studies show that export rates of total nitrogen from native forests, pine plantations and pasture lands are quite variable with no clear differences in nitrogen export rates between these landuses. Total phosphorus export rates for the two forest types are similar. Total phosphorus export rates from pastured catchments are noted in some cases to be higher than for forested landuses.

A general principle for the control of pollution levels is that reductions in soil erosion rates will reduce the pollutant loads in streams. Investigations from the Southern Tablelands of New South Wales found that pre-European or near-natural catchment conditions have sediment yields approximately 20 times lower than landscapes modified for agricultural purposes with their associated erosion problems. This illustrates there is potentially a large reduction in sediment yield that could be achieved for the Lower Cotter catchment by restoring the catchment towards near-natural conditions.

Studies of rural catchment management world wide have found that establishing persistent and undisturbed vegetation in riparian (stream side) zones is an effective and efficient approach to improving water quality. Riparian zones reduce the ingress of pollution from hillslopes, stabilise streambanks, shade the stream environment and are recognised as a high priority area for habitat conservation and restoration. A further universal conclusion is the importance of roads and tracks as a source of sediment

and related erosion. In forested catchments road-sourced sediment has been found to be a large portion of excess sediment in the rivers. For this reason the management of roads should be viewed as a priority in the Lower Cotter catchment, irrespective of the revegetation option selected. Both forest establishment and harvesting activities can elevate soil erosion rates, though some of this eroded sediment may be deposited before reaching the stream network. Similar conclusions regarding soil disturbance will also apply to salvage logging, though no specific studies relating to this have been conducted. Through careful plantation establishment and harvesting operations with well-defined sediment management guidelines, both activities can have a relatively benign impact on water quality.

Three general themes emerge from the scientific literature regarding potential biodiversity outcomes of the three revegetation options. First, native forests and woodlands are associated with higher levels of habitat and consequently support greater biodiversity than either pine plantations or pasture lands. Secondly, there is clear evidence of biodiversity benefits from preserving remnant native vegetation within modified landscapes such as pine plantations or agricultural landscapes. Research indicates that riparian zones are important areas within which to preserve native vegetation and establishing corridors linking otherwise isolated stands of remnant vegetation will be beneficial from a biodiversity perspective. Studies of minesite rehabilitation works offer positive examples of the success of restoring natural ecosystems in formerly degraded landscapes, though on a much more restricted spatial extent. Within several decades of replanting former minesites, both plant and animal species abundance can return to near predisturbance levels.

A number of studies concerning weed management from the ACT have shown that pine wildlings spread from plantation areas into adjacent native forests. This is largely an edge effect confined to several kilometres from the plantation boundary. Pine plantations are associated with a significantly higher level of blackberry occurrence than continuous areas of native forest.

Bushfire intensities for native forest and pine plantations are broadly similar and under extreme weather conditions are higher than for grassland fires. All three vegetation

types can readily attain fire intensities well beyond the effective suppression limits of firefighters. Fuel reduction burning has been shown to have little impact upon the ability of fire fighters to suppress wildfires in either Pine or Eucalypt forest or grasslands under severe fire weather conditions. Fuel reduction burning would however increase the number of days per year that successful suppression operations could be undertaken. There is a suggestion in the literature that under likely future climate change the frequency of bushfires in southern Australia (including the ACT) will increase and once ignited, future bushfires will spread faster, be more intense and more difficult to suppress. Such conditions would elevate the risk that a commercial pine plantation may be damaged by bushfire prior to reaching maturity. Native woodland and Eucalypt forest vegetation is well adapted to fire and periodic burning often stimulates regeneration in many species. Following the 2003 bushfires, regeneration of native species has been shown to be markedly better than for pines and over the long term would represent the most resilient vegetation option with regards to bushfire.

A combined appraisal. Each of the above environmental components needs to be considered with economic and social factors to achieve a combined appraisal. Table 1 summarises the land and natural resource factors associated with each of the possible vegetation types.

There are limitations to the transfer of experience from one catchment to another due to each catchment's combination of biophysical attributes and land use history. To address this a process of on-going monitoring and research would facilitate adaptive management within the Lower Cotter catchment. Specific items in this process are described in Section 5.

The Lower Cotter Catchment provides a major research opportunity for investigating the behaviour of a water supply catchment in a post fire period. These opportunities include linking wildlife and fuel reduction management investigations, sub-catchment scale vegetation rehabilitation trials and the monitoring of the movement of water and pollutants within the catchment associated with various rehabilitation strategies

Vegetation type	Water quantity	Water quality	Controlling sediment yield to streams	Biodiversity	Bushfire management	Weed control	Road network impacts	On going maintenance	Fire sensitivity
Native vegetation of similar or different genetic material	✓✓ [1]	✓✓ to ✓✓✓ [2]	✓✓ to ✓✓✓ [4]	✓✓✓	✓ to ✓✓	✓✓ [10]	✓✓ to ✓✓✓ [11,12,13]	✓✓ [10,12]	✓✓✓
Radiata pine or similar species	✓✓ [1]	✓ to ✓✓✓ [2]	✓ to ✓✓✓ [5]	✓✓ [7]	✓ to ✓✓ [9]	✓✓ [10]	✓ to ✓✓ [11,12,13]	✓✓✓ [10,12]	✓
Grassland (native and/or exotic)	✓✓✓	✓ to ✓✓✓ [3]	✓ to ✓✓✓ [6]	✓ to ✓✓ [8]	✓✓ to ✓✓✓	✓✓ [10]	✓✓ to ✓✓✓ [11,12,13]	✓✓ [10,12]	✓✓✓

Table 1: A summary table of vegetation types and the corresponding land and natural resource factors. Note that ✓✓✓ means most favourable, ✓✓ intermediate and ✓ least favourable.

1. Tall perennial vegetation uses more water than grassland. This may not be an issue if the inflows to the Cotter reservoir regularly exceed the outflows.
2. Water quality from the land is normally good but the density of roads and time to recover from fire or harvest mitigates this.
3. Water quality from grassland is normally good but grazing, access of grazing animals to streams and the density of roads mitigates this.
4. Sediment yield is normally low but roads and tracks may increase this.
5. Sediment yield is normally low but roads, tracks and harvest activities increase this.
6. Grassland can have very low sediment delivery but heavy grazing and roads can increase this.
7. Plantations normally have native vegetation as part of the design. This combination is superior to open grazed grasslands.
8. Native grasslands are an under represented vegetation type in Australia's reserves and can be very important additions to conservation efforts. Grazed grasslands have generally low biodiversity.
9. There are no definitive studies describing the effect of pine plantations on the propagation of bushfires.
10. Weed control is a major issue for all land uses. While prescribed approaches are available for plantations, the establishment of native vegetation may require adaptive approaches.
11. Roads are required for access to infrastructure and for fire management. In plantations the existing road network will require upgrading to bring it to national standards.
12. Roads are a major trigger for soil erosion and reduced water quality. Road density, road design and maintenance regimes determine the degree of this impact.
13. The star rating here reflects the amount of effort required to change the current road layout to an acceptable level for the given land use. The impact of roads is also reflected in the water quality and controlling sediment yield to streams columns.

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1 Introduction

1.1 Background

The Cotter River catchment has been the main water supply catchment for the national capital since its inception. The Cotter River rises within the mountains of Namadgi National Park and has a catchment area of 482 km^2 at its confluence with the Murrumbidgee River. The Cotter catchment, particularly above Bendora Dam, is valued for its relatively high rainfall and the good quality of runoff that it generates.

The Cotter River is impounded by three water storage reservoirs constructed to supply drinking water to Canberra. Details of these reservoirs are listed in Table 2. Prior to 1961, Canberra's water was sourced from Cotter Dam and hence the entire Cotter catchment had a role as a water supply catchment. With the completion of Bendora Dam in 1961, Canberra's water abstractions from the Cotter River (up until 2005) were sourced entirely from Bendora Dam, with releases into Bendora occurring from Corin Dam upon its completion in 1968. Thus from 1961 until 2005, the Cotter catchment below Bendora Dam played no role as a water supply catchment.

In 2004, the need for additional flexibility in the ACT's water supply system was recognised (ACT Government, 2004a). One of the outcomes was the reactivation of the Cotter Reservoir as a source for abstractions to Canberra's urban water supply system. Since 2005, the infrastructure has been in place to pump water from Cotter Reservoir into Googong Dam via the established water reticulation system, and the option of enlarging the capacity of Cotter Dam to 76000 ML has been noted as a possible further modification to the storage infrastructure (ACT Government, 2004c). Consequently, the quality and quantity of runoff from the Cotter River catchment below Bendora Dam has become an issue of much greater concern to the ACT community.

On the 8th January 2003, bushfires were lit by lightning along the Brindabella Ranges north and West of Canberra and also amidst the Snowy Mountains to the south. These fires eventually burnt a large portion of the Australian Capital Territory including almost all the Cotter River catchment between Bendora and Cotter dams (Figure 1).

Prior to the 2003 bushfires, much of the Lower Cotter catchment was managed by ACT Forests as a commercial pine plantation. The extensive destruction of the pine plantations in the Lower Cotter catchment by the 2003 bushfires along with the reincorporation of the Lower Cotter catchment into the Territory's water supply catchment has seen community debate about the appropriate land management strategy for the Lower Cotter catchment (Kanowski, 2005; White, 2005), particularly in light of its future role as a water supply catchment.

1.2 Catchment History

Starr (2004), Dovers et al. (2005) and White et al. (submitted) present landscape histories of the Cotter catchment and their major findings relating to the Lower Cotter are summarised here. Pastoral settlement of the catchment occurred from the 1830's and the Lower Cotter catchment was cleared of its native vegetation around 1890. Major soil erosion within the Lower Cotter catchment is reported following a rabbit plague in 1925 and in a subsequent attempt to stabilise soil erosion, pine plantations were established. Continued clearance of native vegetation to make way for commercial pine plantations continued until 1961. As noted by Dovers et al. (2005), there has been a history of concern over turbidity levels in Cotter Dam dating from at least 1931. These high turbidity levels became of lesser importance following the construction of Bendora Dam in 1961, enabling supply to Canberra of water from the near-pristine upper Cotter catchment.

This landscape history is likely to be influencing contemporary patterns of sediment generation and movement within the Lower Cotter catchment. Gully networks are important sources of sediment and nutrients within the Murrumbidgee catchment in general (Wallbrink et al., 1996; Wasson et al., 1998; Olley and Scott, 2002) and are recognised as important sediment generation sites within the Lower Cotter catchment. It is likely that many gullies present within the Lower Cotter catchment were formed during the early phases of catchment clearing and the introduction of livestock (Eyles, 1977; Prosser, 1991; 1996; Scott, 2001). Additionally, the road network established within the catchment for commercial forestry purposes was recognised in 1962 as a

Reservoir	Catchment (km^2)	Constructed	Capacity (ML)
Corin Dam	197	1968	75400
Bendora Dam	290	1961	10700
Cotter Dam	482	1912	4700

Table 2: Reservoir characteristics for the Cotter River catchment.

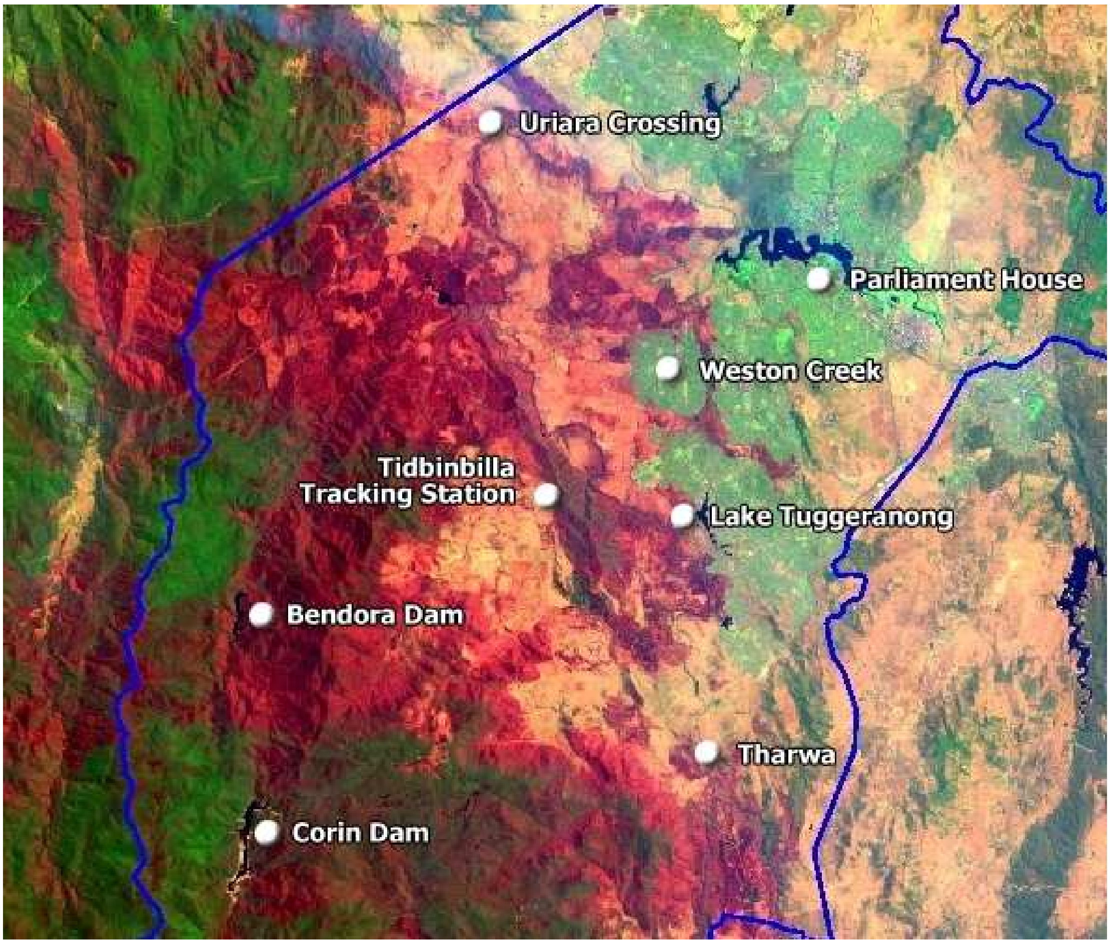

Figure 1: Landsat ETM+ satellite image of the northern ACT taken on 26 January 2003, after the 18 January bushfires. Burnt areas appear red, and almost the entire Lower Cotter catchment can be seen to have been burnt. Satellite image provided by Geoscience Australia.

major source of the elevated turbidity levels in Cotter Dam (White et al., submitted). More recent land management has seen rationalisation of the road network within the Lower Cotter catchment and this process has been accelerated in the catchment management works since the January 2003 bushfires (pers com. A. Winter ACT Forests 2005).

1.3 Fire History

White et al. (submitted) record five major bushfires occurring within the Cotter catchment over the last century with the January 2003 fires being the most extensive. The timing of these major bushfires are suggested to be related to the occurrence of climatic and oceanographic changes in the Pacific Ocean, including the El Niño/Southern Oscillation and longer cycle Pacific Decadal Oscillation.

The January 2003 bushfire created suitable conditions for major hillslope soil erosion to occur. During intense rains of February 2003, there are documented examples of large pulses of sediment being delivered to the Cotter River and its reservoirs from hillslope erosion. These include 19,300 tonnes of poorly sorted inorganic sediment delivered to Bendora Dam from tributary creeks sourced mainly from erosion of the adjacent hillslopes (Wasson et al., 2004). White et al. (submitted) observe unprecedented increases in a number of pollutants within Cotter Dam following the 2003 bushfires, including turbidity (30 fold increase), iron and manganese (up to an order of magnitude), though two years after the fires, turbidity levels in the upper Cotter reservoirs have largely returned to pre-fire levels due to natural revegetation stabilising the hillslopes. In contrast. elevated turbidity levels have persisted in Cotter Dam since the 2003 bushfires (White et al., submitted).

Intense bushfires are widely recognised as causing catchment sediment yields to increase by up to several orders of magnitude for a limited duration following the fires (Brown, 1972; Leitch et al., 1983; Diaz-Fierros et al., 1987; Scott and Van Wyk, 1990). This increase is attributed to changes in soil hydrophobicity, reduced vegetative cover and the low cohesion of ash and desiccated soil. However, in many environments, particularly those in a more natural state, sediment yields are observed to return to

pre-fire levels within five to ten years (Brown, 1972) as vegetation regrows and leaf litter accumulates on the ground (Fox et al., 1979).

1.4 Other investigations

Since the 2003 bushfires, the Cotter catchment has been the focus of a number of research activities. Studies of post-fire sediment delivery to Bendora Dam and water quality within the reservoir have been conducted by Wasson et al. (2004) and Govinnage-Wijesekera et al. (2005). White et al. (submitted) have assembled an analysis of the long term fire history of the Cotter catchment and discussed the impacts of the fires upon water quality in Cotter Dam. Croke and Takken (2005) provide methodology for priority setting for the control of soil erosion associated with roads within the Lower Cotter Catchment. The method involves a rapid field survey of the road network, topographic analysis of the catchment and the position of roads and associated drains, and prediction of the likely areas of gullying and diffuse delivery of sediment to the stream network. The field survey was done in a sub-section of the Lower Cotter Catchment and predictions used the spatial data from the catchment and parameters derived from studies outside of the catchment (but within south east Australia). The method is especially useful given the importance of roads as a sediment source, the high density of roads in the Lower Cotter Catchment and the prohibitive costs of treating all roads without prioritisation.

1.5 Vegetation options considered in this study

The Lower Cotter catchment is a rural catchment that now forms part of the ACT water supply. The catchment contains significant areas of native forest and land that supported a pine plantation prior to the 2003 bushfires. The catchment also contains the Pierce's creek forestry settlement.

The terms of reference for this project (see Appendix A) set out the possible options for revegetation and associated land use that are considered in this review. The options for revegetation of the Lower Cotter catchment can be broadly described as:

- Re-establishing a commercial pine (or other similar commercial species) plantation.
- Re-establishing the forest and woodland vegetation characteristic of the catchment prior to European settlement, as a reserve.
- establishing a grassland ecosystem.

Note that this report makes no assumptions concerning the preferences of various advocates and attempts to provide an objective appraisal of these various options to inform policy development.

1.6 Objectives of this study

The primary objective of this study is to synthesise prior scientific research concerning the outcomes of establishing a range of different vegetation types and land uses in a rural catchment following a major bushfire. The report focusses upon the likely impacts of different landuse and vegetation options upon key environmental attributes identified by the ACT government for the Lower Cotter Catchment (see Appendix A). These attributes include:

- water yield;
- water quality;
- controlling sediment yield to streams;
- biodiversity;
- bushfire management;
- weed control;
- road network requirements;
- ongoing maintenance; and
- fire sensitivity and regenerative capacity after fire.

Many studies have shown that landuse (such as pine or native forest or pasture land) is not the sole determinant of down stream water quality and that land management measures relating to particular land uses, including roading and the treatment and consideration of a catchment's landuse history are important. For these reasons we have expanded the scope of this investigation to include the potential impacts of these associated features upon water quality in particular.

The study focuses on research conducted in landscapes similar to the Lower Cotter catchment. It strictly considers biophysical aspects, with the economic and social consequences of the proposed land use options not considered.

2 Catchment Description

The Lower Cotter catchment, defined as the catchment between Bendora and Cotter Dams is 192 km^2 in size. Of this, 107 km^2 is national park and 84 km^2 was classified as plantation or production forest in 1996/7 by the Bureau of Rural Sciences. A small proportion of the Lower Cotter catchment extends across the ACT border into New South Wales (Figure 2).

2.1 Hydrology

There are 23 years of stream gauging observations that record both dam spills and releases below Bendora (station number 410747) and Cotter (station number 410700) dams, spanning the period 1976 to the present. Annual runoff generated in the Lower Cotter catchment (ie. below Bendora Dam), denoted Q_{LC}, can be calculated from these records as:

$$Q_{LC} = Q_{410700} - Q_{410747} + \Delta CD \tag{1}$$

Where Q_{410700} is annual total streamflow at gauging station 410700 (below Cotter Dam), Q_{410747} is annual total streamflow at gauging station 410747 (below Bendora Dam) and ΔCD is the annual change in storage in Cotter Dam, constrained to be $\leq$4700 ML. Over the period 1964 to 2004, Q_{410700} has averaged 86411 ML per year and in only three years (1980,1982 and 1994) has it fallen below 10000 ML per year. Thus for the overwhelming majority of years, the term ΔCD can be assumed to be a relatively minor component of the Lower Cotter runoff budget. For the purposes of estimating Q_{LC}, we therefore assume ΔCD to be zero. Figure 3 shows the calculated Q_{LC} values on an annual basis (sorted by runoff volume) and their mean. The annual Q_{LC} values are close to normally distributed, have a mean value of approximately 30000 ML [1] and range from a maximum of 74000 ML in 1983 to a minimum of 2900 ML in 1982. This 25 fold range in annual runoff is due to enhanced conversion of rainfall into runoff in

[1] An independent estimate of the volume of runoff for the Lower Cotter catchment of 36000 ML per year was derived by the ACT Government (2004b)

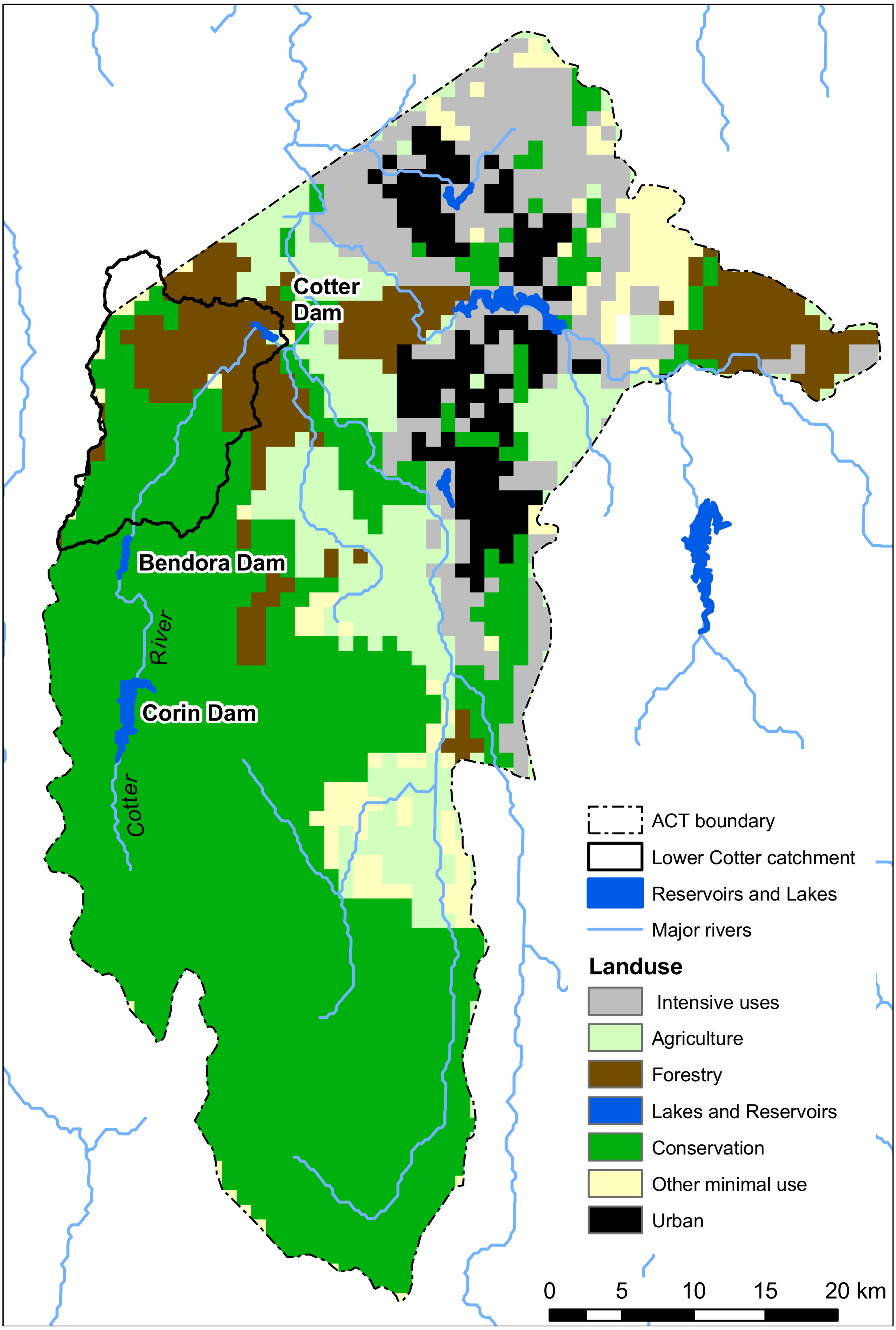

Figure 2: Landuse map of the ACT showing major rivers, reservoirs and the Lower Cotter catchment. Landuse classification performed by the Bureau of Rural Sciences in 1996/7.

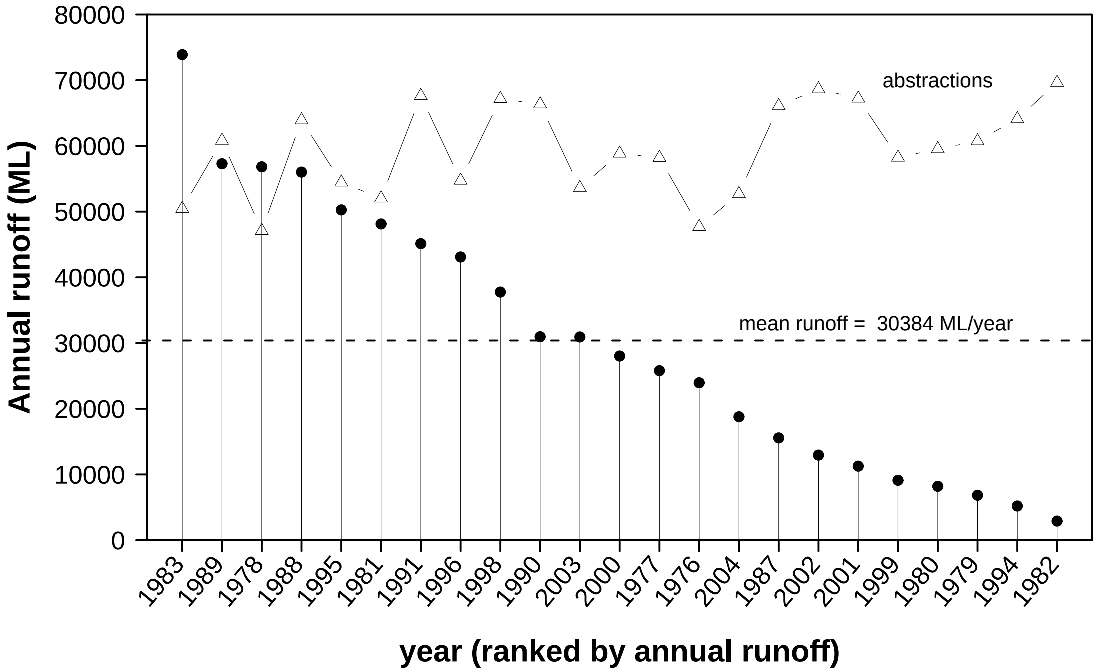

Figure 3: Annual runoff generated within the Lower Cotter catchment and corresponding abstractions from Bendora Dam for the period 1976 to 2004 (six years of missing data). Note that the years have been sorted and displayed according to total runoff generated in the Lower Cotter catchment.

high rainfall years relative to low rainfall years. To put these numbers into perspective, the average quantity of runoff generated from the Lower Cotter catchment is equivalent to approximately half the annual abstractions from Bendora Dam to Canberra's water supply (Figure 3). Thus while somewhat variable, the runoff from the Lower Cotter catchment is potentially an important resource for the ACT.

2.2 Soils

There are three main soil groups within the Lower Cotter catchment and their distribution is controlled by the three main bedrock types with the catchment. In the immediate vicinity of Cotter dam are yellow and red earth soils with a moderate gravel component developed on Silurian volcanic bedrock. These soils have a low water retention ca-

pacity, low permeability and the yellow earths in particular are erosion prone and their high density is detrimental to plant root growth (Talsma, 1983). High gravel contents are also characteristic of the red earth soils developed on the metamorphosed sediments found in the upstream sections of the Lower Cotter catchment and on the soils developed upon the Adamellite bedrock in the south east. The red earths are the most erosion resistant soils in the catchment.

Bushfires have a range of effects on soil characteristics, depending in part on the nature of the soil and the organic content. Wild forest fires typically heat the uppermost 100 mm of soil (Walker et al., 1986; DeBano, 2000) and vapourise organic compounds which subsequently condense and cover the soil particles in a hydrophobic layer (Prosser and Williams, 1998; DeBano, 2000; Pierson et al., 2001). Through this process, runoff generation in the post-fire period can be increased substantially in localised areas, though subsequent infiltration at sites further downslope may mean no noticeable change in catchment water yield (Prosser and Williams, 1998). Hydrophobic conditions may persist for several weeks or months following a mild fire and for several years following an intense fire burning a large accumulation of fuel (DeBano, 2000; Pierson et al., 2001). Hydrophobicity may also be a feature of unburnt forest soils.

2.3 Geomorphology

The Lower Cotter catchment lies in a transition zone between the Brindabella Ranges and the Southern Tablelands. The catchment generally comprises soil- mantled hills of variable gradient with colluvial deposits upon footslopes and alluvial valley fills several meters deep present upon the valley floors. Episodic incision and re-aggradation of these valley fill deposits occurred across the Southern Tablelands prior to European settlement (Prosser, 1991; Gillespie et al., 1992; Prosser et al., 1994). Localised bedrock outcrops occur, particularly along incised reaches of the Cotter River. Floodplains are restricted in area, occurring along some of the larger streams and sections of the Cotter River.

2.4 Road Network

There is an extensive road network within the Lower Cotter catchment that has been constructed to facilitate forestry operations and to service the water supply infrastructure. The majority of road length is unsealed and road densities are generally high, exceeding 1.3 km/km^2 in some plantation areas (Croke and Takken, 2005).

3 Review of catchment responses to post fire vegetation management

3.1 Water Yield

3.1.1 General Principles

All plants evaporate water through their leaves. This water is extracted from the soil adjacent to the plant roots and the rate of evaporation depends on the weather, the available soil moisture, and the total area of leaves upon the vegetation (trees and understorey). When a fire damages vegetation this triggers a sequence of changes in the way the vegetation uses water though evaporation. Detailed mechanistic descriptions of these processes for Mountain Ash forest can be found in Vertessy et al. (2001) and for mixed species Eucalypt forests in Roberts et al. (2001).

Damage to vegetation can range from mild scorching of the understorey to death of all above ground vegetation. Where damage is mild, vegetation rapidly recovers on a time scale of weeks to months with ground cover rapidly recovering and water use patterns of the vegetation being little affected by the fire. At the other extreme, tree death or clear fell harvesting results in the re-initialisation of the vegetation with a succession of vegetation filling the habitat void left in the wake of the fire. In this instance, the period immediately after the fire (typically 2 to 8 years) is associated with lesser water use by vegetation than occurred in the pre-fire forest (Watson et al., 1999). Thus the post-fire period is associated with increased stream flows and recharge of groundwater systems compared with a mature forest in the pre-fire period for the same rainfall. The vegetation then enters a phase of rapid growth and extraordinary competition between component plants, so that water use is higher than that which occurs for a mature forest. This phase can persist for periods ranging from 20 to 200 years depending upon the tree species (Watson et al., 1999; Cornish and Vertessy, 2001; Roberts et al., 2001). During this phase catchment flows are generally noted to be below those coming from mature forests.

The magnitude of the changes to the water cycle triggered by fire are dependant

(a) (b) (c)

Figure 4: The Goodradigbee River and its riparian vegetation. Pre-fire original site photograph (a) and post-fire photographs (b) and (c). Site reference: Brindabella 072 (source: Michael Doherty, CSIRO Sustainable Ecosystems)

on the severity of the fire and the consequent ecosystem succession, the species of trees, including the susceptibility of the tree to tree death during a bushfire and the environmental setting including the rainfall zone (see further details below).

3.1.2 Native Forests

During the January/February 2003 fires in south eastern Australia, some 1,390,000 hectares of largely native forest were burnt. This area corresponded with the upland portion of many of the sub-catchments of the Murray-Darling basin and some coastal catchments. Figure 4 shows some typical burnt riparian vegetation from the Goodradigbee River. These areas have high rainfall relative to other parts of these catchments, meaning that the potential consequences for runoff generation and stream flow are significant.

Mitigating the effect of the 2003 fire on the stream flow are a number of factors. Firstly, the severity of the burn was highly variable across the total burnt area with only relatively small areas experiencing tree death (Sinclair Knight Mertz, 2004). In comparison with the rest of the Murray Basin the Lower Cotter catchment had a relatively

high proportion of forest killed by the fire. These tree-killed areas were primarily areas of pine plantation. Much of the Murray Basin burnt in 2003 had previously experienced major wildfire and was already in a phase of physiological recovery and associated increased water use.

Two estimates of the impact of the 2003 fires on stream flow in the Murray-Darling Basin have been completed and complement some Victorian research that is in progress at the time of writing. In the first study Vertessy (2003) considered initial estimates of the fire extent and severity and its impacts on water use in the Upper Murray catchments of Victoria. This study suggested that the inflows to the Murray would be reduced by approximately 430 GL/year in 20 years time. The study further concluded that better estimates of the severity of the fire and associated tree damage and the age of the burnt trees were required to refine estimates further.

In a second, more comprehensive investigation, Sinclair Knight Mertz (2004) used satellite imagery of the fire affected area as input to a spatial model of vegetation combustion and tree water usage. The different types of forest (mountain ash, mixed species Eucalypt and Snow Gum forests) and the age of the vegetation prior to the 2003 fire were also considered. This approach resulted in a far more rigorous and spatially explicit forecast of the magnitude and persistence of the effects on streamflow in the studied catchments. The study predicted an initial, large increase in streamflow that ranged from 14 to 106%, expected to persist for approximately 7 years. Subsequently it is estimated that a small reduction in the total inflows to the Murray River (compared with the no fire scenario) will occur. The maximum reduction in the total stream inflow to the Murray varies between -129 GL per year and +4 GL per year depending on the assumptions made concerning the relationship between fire severity and tree death.

The assessments described above are based on long term catchment studies and an understanding of the relationship between plant leaf areas and rates of transpiration. The long term studies of stream flow with changing vegetation have been investigated over the past century for many catchments and a review of this field is presented by Andreassian (2004). These studies generally use major vegetation change, either man made (e.g. clear felling of forests) or natural (e.g. severe tree-killing fires) as a treat-

ment and then evaluate the subsequent streamflow. Typically two or more catchments are studied with one kept as a control or reference site with no change in vegetation. All of these studies conclude that the temporal pattern of evapotranspiration as the vegetation regrows is a major influence on the quantity of stream flow (Zhang et al., 2001). For forested catchments the amount of water used by trees as evapotranspiration varies on a time scale of years to decades (Vertessy et al., 2003) with the major variable being the forest age or time since the major disturbance.

In Australia the impact of forest age on streamflow has been investigated in detail for a wide range of catchments, ranging from the mountain ash forests in Melbourne's water supply catchments to relatively dry environments near Bathurst in New South Wales. For the near-Melbourne catchments, both Kuczera (1987) and Watson et al. (1999) developed equations that describe the mean annual streamflow as a function of forest age. The changes in streamflow for a 2000 mm/year rainfall catchment are shown in Figure 5 for both the Kuczera and Watson relationships. The major difference between these two curves is the initial increase in streamflow predicted by the Watson curve that is not reflected in the work of Kuczera (1987). This increase is due to the reduction in leaf area associated with disturbance such as logging or forest fire (Vertessy et al., 1996). For this reason the Watson curves provide a more realistic shape of the streamflow response in forested catchments subject to major disturbance.

The results of the studies described are of relevance to the Lower Cotter Catchment. To enable forecasts of the changes of steam flow to be predicted for the Lower Cotter catchment, a detailed knowledge of the spatial distribution of rainfall, the original forest age, estimates of the patterns of fire severity, the proposed vegetation and its management and scenarios of disturbance by fire over the coming decades would be required.

3.1.3 Pine Plantations

Fires that are of moderate to severe intensity (>2000 kilowatts per metre) kill *Pinus Radiata* vegetation (Burrows et al., 1989). Hydrologically, tree death is the equivalent of a clear fell logging operation in that it reduces the transpiration of the forest to the

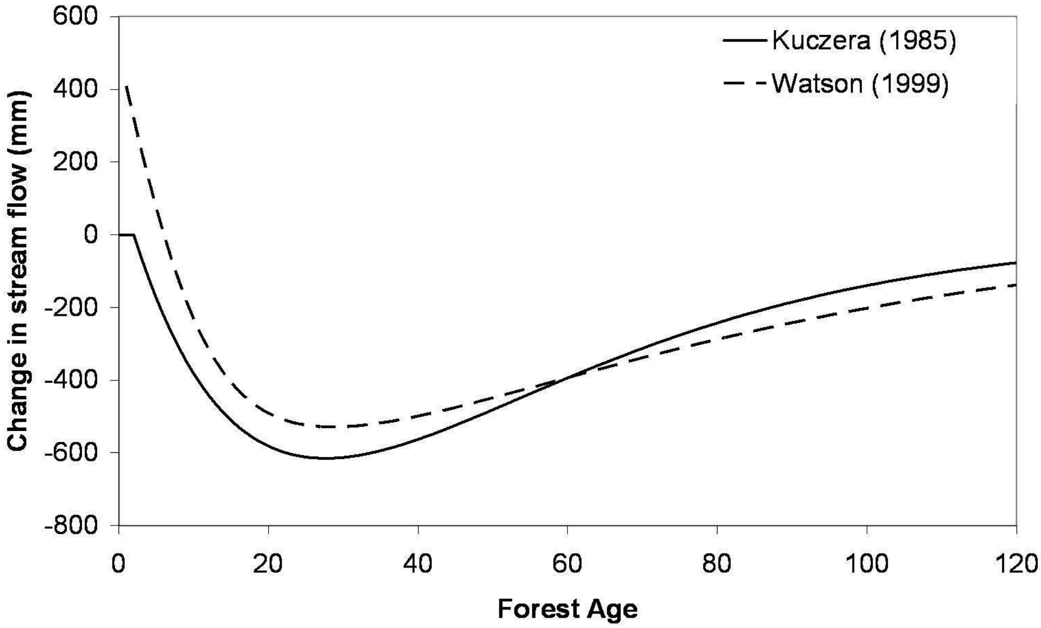

Figure 5: Streamflow response curves developed for Mountain Ash forests for a rainfall of 2000 mm/year as derived by Kuczera (1987) and Watson et al. (1999). Note that magnitude of the change is dependant on rainfall zone with greater differences expected for higher rainfall zones.

level provided by the remaining living ground cover.

Through analysis of paired catchment studies across a range of environments using native forest and plantation treatments, Brown et al. (2005) found that the impacts on streamflow of pine plantation growth and native forest regrowth are indistinguishable. Thus it is highly likely that for the areas of the Lower Cotter catchment previously planted to pines, but burnt in 2003, that the water yield after the fires will follow the temporal patterns as shown in Figure 5 under either re-establishment of a pine plantation or reversion back to native vegetation.

Many researchers have found forest age will impact on streamflow (see section 3.1.1). Brown et al. (2005) demonstrate that the effect of rotation lengths in forestry operations for plantations and native forestry is important as it determines the proportion of time that the forest is in the early growth and lower water use phase (ages 0 to $\sim$5 years) compared with the rapid growth and high water use phase ($\sim$5 years to 50 years).

3.1.4 Grasslands

Grasslands are universally found to yield more streamflow than for forests in the same environment (see detailed discussion in section 3.1.5). Grassland leaf levels recover faster after fire than forests (e.g. contrast the behaviour of grasslands described in Mark (1994) with tall Eucalypt forests described in Kuczera (1987)). On physiological grounds, we expect that stream flow quantity will vary little in the period immediately following fires. Small stream flow increases may occur during the recovery period due to reductions in leaf areas, though this assertion awaits experimental testing.

In many environments grasslands are subject to encroachment by shrubs and trees. Davis (1993), in a study of catchments which in their native state are covered in chaparral (shrubs), found that by chemically eliminating the regrowth of shrubs, so as to maintain grasslands, streamflow was significantly higher than untreated catchment over the 16-year period. Huxman et al. (2005) found that encroachment of woody plants on grassland had the effect of reducing the available quantity of streamflow. Fire is frequently used to control the encroachment of shrubs in such environments (see

example in Williamson et al., 2004).

The review of these studies and the comparative work described below in section 3.1.5 suggests that grasslands may be considered as a future vegetation in the Lower Cotter Catchment on the basis of increased streamflow.

3.1.5 Comparisons of long term water uses by different land uses

Decades after major fires there will be clear differences between catchments of differing land uses in terms of the streamflows they yield. These differences have been evaluated in summaries of many catchment measurements to produce global relationships which are useful in predicting the behaviour of non-experimental catchments.

Zhang et al. (2001; 2004) analysed catchment runoff measurements for more than 200 catchments around the world, and concluded that broad vegetation type (i.e. forest or non-forest) together with annual rainfall and potential evaporation is a key factor determining annual catchment stream flow. They captured these relationships in what have been referred to as the 'Zhang curves', expressing the relationship between long-term average catchment climate and water use, respectively (Figure 6).

Figure 6 shows that for Australian catchments of moderate rainfall there is normally more streamflow for dominantly grassed catchments than there is for dominantly forested catchments. This conclusion is consistent with the world wide analysis presented in Zhang et al. (2001). Note also there is a lack of data concerning streamflow from grassed, high rainfall (>1000 mm/year) catchments in the Australian data set presented in Figure 6.

3.1.6 Water Quantity Conclusions

By killing a large portion of the pine vegetation in the Lower Cotter catchment, the 2003 bushfires will have altered the catchment's hydrologic balance. The immediate reduction in live vegetation post-fire is likely to result in a short term (3-7 year) increase in catchment water yield. Beyond this and under re-establishment of either native forest or introduced pine, water yields are likely to decline in the medium term (7–50+ years) as this vegetation grows and its water use increases. Studies show similar water yield

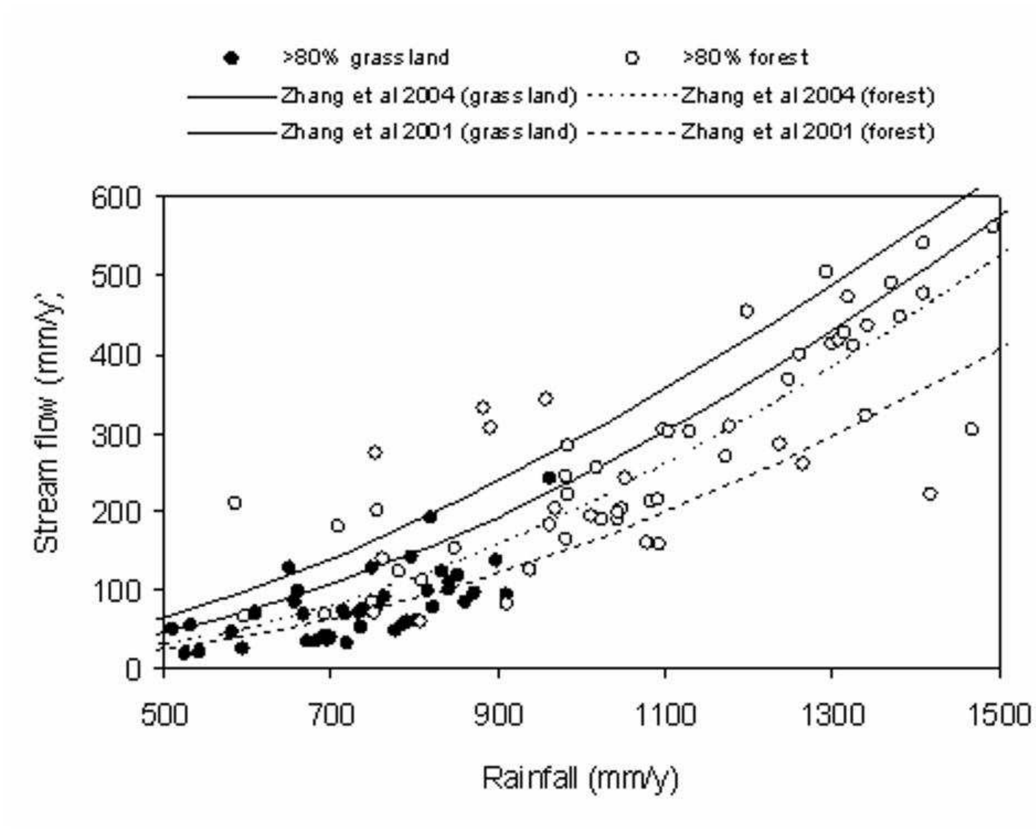

Figure 6: The relationship between average annual catchment rainfall and water yield for forested and grassed Australian catchments used in the analysis of Zhang et al. (2004). (Reproduced from van Dijk et al. (in review) with permission.)

between catchments planted to pines and native eucalypt vegetation. Revegetation of the former pine plantation area using grassy vegetation would result in the highest stream flow yield as evapotranspiration (water use) by grassy vegetation is markedly less than for trees.

3.2 Water Quality

The runoff generated within the Lower Cotter catchment will have two main uses. First, it may be abstracted from Cotter Dam into the ACTs urban water supply system. Second, as flows pass through to the catchment they will be available for the local ecosystem as environmental flows. Water quality guidelines for aquatic ecosystems and human recreational and consumptive uses can be found in ANZECC (2000). For both uses there are generic and catchment specific water quality issues of relevance in the Lower Cotter catchment.

Within an urban water supply context, the principle of adopting multiple "barriers" between potential pollutants and the consumer is standard practice (National Health and Medical Research Council, 2004). In the ACT's water supply system, one of the important barriers traditionally has been the near pristine catchment condition of the upper Cotter River, augmented by filtration infrastructure. The land use history of the Lower Cotter catchment, combined with the 2003 bushfires, means that this pristine catchment "barrier" does not presently exist for the Lower Cotter catchment.

As a general principle for management of water supply catchments, the minimisation of delivery of two important nutrients, nitrogen and phosphorus (Harris, 2001) as well as dissolved organic carbon (Flemming and Cox, 2001) to waterways is important. Both nitrogen and phosphorus are essential nutrients for plants and animals and are naturally present in varying degrees within soil. As it is the fine grained sediments to which many pollutants such as phosphorus are most strongly attached (Palis et al., 1990; Pettersson, 1998; Webster et al., 2001), delivery of sediments to a reservoir or water body can represent the main source of nutrient input. Both phosphorus and nitrogen are commonly added to soils in agricultural and forestry environments through application of fertilisers to enhance plant growth (Haygarth and Jarvis, 2002; Binkley

et al., 1999). In the following sections, the different landuse options are reviewed in terms of the generation of sediments, nitrogen and phosphorus and their delivery to the stream network. Whilst the negative impact of dissolved organic carbon deliveries to reservoirs has been noted in the literature, particularly in a number of studies from the Mt Lofty Ranges in South Australia (Stevens et al., 1999; Flemming and Cox, 2001; Cox and Pitman, 2001), information regarding its delivery rates is limited and no landuse based comparisons are able to be made. However, delivery of dissolved organic carbon after the 2003 bushfires does appear to be of importance in elevating manganese and iron concentrations in the Cotter reservoirs (White et al., submitted).

3.2.1 Suspended Sediment

The terms of reference for this review mentioned examining the effects of a range of land management options on turbidity. Turbidity pertains to the optical characteristics of water and has been shown to be strongly related to the concentration of fine grained ($<$63 micron) sediment in suspension (Olley and Scott, 2002; Moliere et al., 2005). Due to its ease of automatic measurement, turbidity is more commonly measured than actual suspended sediment concentrations. From a water quality perspective, managing turbidity levels as a proxy for suspended sediment loads is an appropriate goal as reductions in turbidity will generally be associated with reductions in fine grained suspended sediment and, as mentioned above, it is the fine grained sediments to which many pollutants are bound.

Suspended sediments in streams and their downstream delivery to a reservoir for example, are related to three processes. First, sediments must be eroded from their source and the three main erosion sources typically of concern in rural catchments are hillslope erosion, gully erosion and river bank erosion. In environments with a high density of unsealed roads, such as in a forestry plantation, substantial amounts of sediment can be generated from the road surfaces (Croke et al. 1999, Takken and Croke, 2004, Croke 2004). Second, the delivery of eroded sediments to a stream network is required. Delivery may be via overland flow reaching a channel, through eroding gullies directly joining a larger stream, or in the case of forestry roads, road drainage infras-

tructure feeding directly to the stream network. Finally, the downstream transport and potential storage of sediments on floodplains can attenuate the sediment flux through a catchment, particularly for coarse bedload sediments and to a lesser degree for fine grained sediments carried in suspension (Prosser et al., 2001; McKergow et al., 2005).

The complexities in sediment generation and downstream transport within river networks mean that there is usually a need to model the processes of interest in order to construct catchment sediment budgets. Models, such as Sednet (Prosser et al., 2001; McKergow et al., 2005) exist to predict end-of-catchment suspended and bedload sediment budgets and identify their spatial sources by integrating sediment generation based on landuse, gully erosion patterns, rainfall and topography with downstream transport and deposition. Application of a model such as SedNet is beyond the scope of this report. However, it is still useful to compare sediment generation rates from the specified landuses. Whilst this will not result in quantitative predictions of catchment outlet sediment loads for different landuses, the processes of downstream sediment transport and deposition are dependent largely upon catchment hydrology and channel morphology and are consequently more difficult aspects of the landscape to actively manage. The major “lever” available for managing sediment and associated nutrient loads in catchments is often that of managing the sediment generation, predominantly by managing the condition of the landscape (including the number and type of roads), which is partially related to landuse.

Lu et al. (2003) present the most recent analysis of hillslope soil erosion rates for Australia, including a collation of measured soil loss rates classified by landuse. A summary of soil loss rates from grazing lands, pasture and woodlands (including forests) is shown in Figure 7. Unfortunately no comparison of native and pine forests is given by Lu et al. (2003). It should be noted that these are plot scale measurements and cannot be automatically scaled up to a catchment. Soil loss rates within a landuse category vary by several orders of magnitude and considerable overlap exists between the three different landuse classes. Much of the within class variability will relate to different soil erodibilities and slopes between different catchments. Soil loss from pasture appears to be less in general than for the other two land uses. This may reflect greater

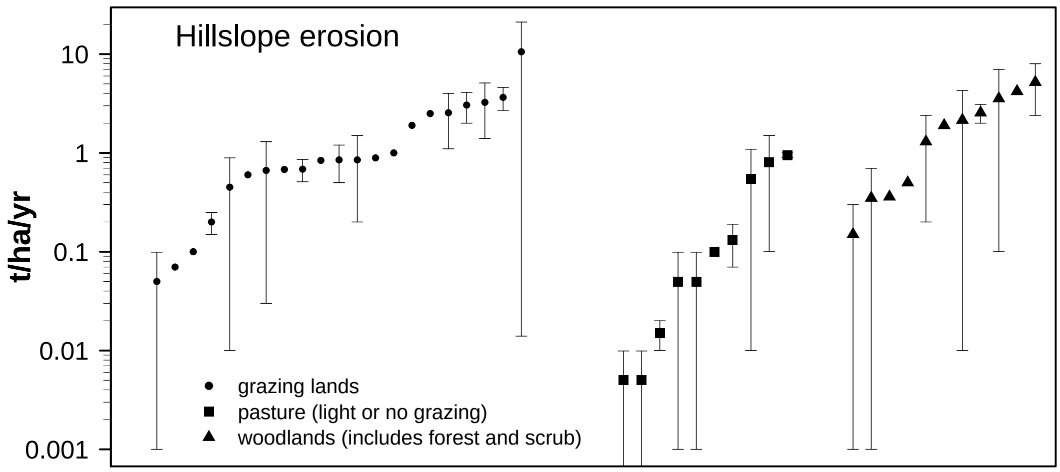

Figure 7: A synthesis of soil loss measurements collated by Lu et al. (2003) for Australia, classified by landuse and sorted in ascending order within each landuse class. Note that tropical grazing lands north of 20 degrees south have been omitted.

vegetation coverage compared to unimproved grazing lands and perhaps lower slopes relative to some of the forest data points. The variability in these data do however highlight the influences of local controls upon sediment yields.

In the absence of a detailed modelling study, we use long term measurements of catchment sediment yields published by Olley and Wasson (2003) to illustrate changes in sediment yields potentially achievable at the outlet of the Lower Cotter catchment under hypothetical "standard" and "near-natural" land management scenarios typical of the Southern Tablelands of NSW and the ACT. These empirical sediment yield relationships have been derived from dated sedimentary deposits and are integrative measures of sediment yield under the prevailing landscape, land management and climatic conditions of the Southern Tablelands for defined time periods. For the "standard" land management scenario, we adopt the sediment yield curve for the period AD1945–1994 of Olley and Wasson (2003). This period is associated with contemporary hillslope erosion rates, active gully incision (but at a rate below that of the main phase of gully expansion) and with relatively efficient downstream sediment transport.

Sediment yield for this period is estimated as:

$$SY = 38A^{0.91} \tag{2}$$

where SY is sediment yield in tonnes per year and A is catchment area (km^2). For the "near-natural" scenario, we use the pre-AD1820 sediment yield curve. This curve is characteristic of the low (pre-European) rates of gully and hillslope erosion of the Southern Tablelands and efficient sediment trapping along generally unincised valley floors and is expressed as:

$$SY = 1.6A^{0.79} \tag{3}$$

In one sense this "near-natural" scenario represents the minimum sediment yield likely from the Lower Cotter catchment under the least disturbed catchment conditions. This may not be a state to which the catchment could be feasibly returned, but it is useful to examine for comparative purposes.

To construct these sediment budgets, we divide the Lower Cotter catchment into two sub-catchments. The first is a 107 km^2 sub-catchment corresponding to the existing nature reserve and native forest areas within the Lower Cotter catchment. This area is assumed to have a long term sediment yield corresponding to natural, pre-European conditions and no management control will be exerted (ie. it is assumed that natural revegetation will occur following the 2003 bushfires). The second block of 84 km^2 corresponds to the area of former pine plantation for which two potential land management scenarios are considered. The first scenario is a reversion to land management practices and associated sediment yields typical of the Southern Tablelands of NSW over the last half century, and the second scenario is a return of this sub-catchment to near-natural conditions. The total Lower Cotter catchment sediment yield is calculated as the sum of the two sub-catchment yields. Whilst area specific sediment yields decrease non-linearly with area due to downstream storage and a simple summation of nested sub-catchment yields would technically be inappropriate, the error resulting from a direct summation is likely to be minor in relation to the uncertainties in the data used to derive the sediment yield models.

	Natural conditions (t/yr)	Standard Southern Tablelands (t/yr)
sub-catchment 1	64	64
sub-catchment 2	53	2142
Total sediment yield	**117**	**2206**

Table 3: Estimated catchment sediment yields from the Lower Cotter catchment under two management scenarios: (1) a return to near-natural catchment conditions typical of the pre-European period, or (2) a return to “standard Southern Tablelands” land management over the area of former pine plantation.

From Table 3, it can be seen that there is almost a twenty fold difference in sediment yield between the two management scenarios. A reversion to near-natural catchment conditions would yield approximately 117 t/yr of sediment from the Lower Cotter catchment whilst “standard” management over the former pine plantation area would see a catchment yield of approximately 2200 t/yr. Whilst this analysis highlights the potential value in terms of reduced sediment yields of directing management actions towards restoring the catchment to a natural condition, there are several caveats that apply. Firstly, the 107 km^2 nature reserve sub-catchment may well have a sediment yield higher than the pre-European model of Olley and Wasson (2003) on account of its relatively steep terrain. Secondly, roads within these native forests may also influence contemporary rates of sediment generation and delivery, thus elevating catchment sediment yields. Thirdly, no account has been made for any increases in sediment yield from either sub-catchment in relation to the recent 2003 bushfires. However, elevated sediment yields from the native forest section of the catchment could reasonably expected to decline to pre-fire background levels under natural regeneration within several years (Brown, 1972).

The data are clear in indicating that under standard Southern Tablelands land management practices, catchment sediment yields are likely to remain high relative to pre-European or near-natural catchment conditions. This indicates the need for additional sediment management actions beyond simply dealing with extant landuse if the aim is to minimise sediment delivery to the Lower Cotter River and Cotter Reservoir and

maximise the suitability of the catchment runoff for ecologic and human consumptive water uses.

3.2.2 Forest establishment and harvesting activities

The soil loss observations from forested land presented in Figure 7 do not include the possible effects of plantation establishment or harvesting activities in changing catchment sediment yields. The degree to which the establishment of pines within a plantation environment alters erosion rates and nutrient exports has been studied by Constantini and Loch (2002). They found that traditional replanting techniques involving stick raking and burning of post-harvest debris followed by mounding to improve drainage exposed soils to erosive processes and also elevated runoff and nutrient export rates from the harvest area. By contrast, site preparation works involving retention of post-harvest debris upon the ground surface as a trash blanket and minimal soil disturbance was associated with higher infiltration rates, greater stability with respect to erosion and the lowest nutrient export rates. This study showed that clear water quality benefits could be achieved from adopting minimum site disturbance practices during plantation establishment compared to traditional methods. Conversely, traditional high disturbance plantation establishment methods elevate the risk of pollutant delivery to waterways.

Forest harvesting activities also disturb the soil surface and the exposed sediments are potentially available to be eroded and transported by overland flow (Wilson, 1999). Furthermore, the use of heavy harvesting machinery results in soil compaction, increasing runoff generation, which in turn means more water is available to erode and transport any exposed sediment (Croke, Hairsine and Fogarty, 1999). Wallbrink et al. (2002) have quantified the effects of forest harvesting on hillslope sediment redistribution. Whilst they noted particularly high rates of soil loss from specific areas such as snig tracks (25$\pm$11 t/ha/yr) and log landings (101$\pm$15 t/ha/yr), much of this sediment was found to have accumulated at other sites within the general harvest area or within grass filter strips (Croke, Hairsine and Fogarty, 1999) such that little sediment left the site to be transported downstream. The implication of this is that through

careful harvesting operations with well defined sediment management guidelines, forest harvesting activities can have a relatively benign impact upon sediment delivery to streams and water quality (Nisbett, 2001). However, better definition of the effects of forest harvesting activities on sediment yields at a sub-catchment or catchment scale (as opposed to the hillslope scale) remains a largely unresolved issue (Croke, Hairsine and Fogarty, 1999).

The interaction of bushfires and forest harvesting operations was a topic studied by Wilson (1999). It was noted that whilst catchment sediment yields invariably rise in response to major bushfires (eg. Brown 1972), Wilson (1999) found that logging activities increase post-fire erosion rates above those associated with burnt but otherwise undisturbed bushland due to the enhanced runoff generation effect mentioned above. A similar conclusion of enhanced runoff generation from “disturbed” areas (including walking tracks and firebreaks) resulting in increased erosion was reached by Zierholz and Hairsine (1995) in their review of post-fire impacts on soil erosion rates. These studies show that landscape disturbances, including forest harvesting activities, exacerbate soil erosion in the event of bushfires.

3.2.3 Roads and tracks

Several studies have found roads to be a major, if not the dominant source of sediment to streams in forested catchments. Motha et al. (2003) found that gravelled and ungravelled roads in a mixed species native forest contributed a large portion of the sediment in the Tarago River that feeds the Tarago reservoir, part of Melbourne’s water supply.

The development of road and track networks as part of logging and grazing landuses exacerbates problems of surface erosion and runoff development (Reid and Dunne, 1984; Ziegler et al., 2001; LaMarche and Lettenmaier, 2001). Roads generate overland flow for a wide range of rainfall intensities and durations. This runoff may act to carry sediment to the natural stream network and/or erode sediment from the adjacent hillslope when it is discharged from the road in a concentrated form. A recent review of managing sediment delivery in managed forests (Croke and Hairsine, in

press) suggested that road networks and road drainage can be designed to disconnect road runoff from streams thereby eliminating the water quality impacts of roads. These authors suggest that a three step process of design needs to be applied:

1. Roads are located and drained so as to minimise erosion by water of the road surface and adjacent drains,

2. The combination of the distance between road drains and the slope of the hillslope below the road is limited so that gully erosion below the road drainage outfall is prevented, and

3. The combination of the distance between road drains and the distance to the stream from the road drainage outlet needs to be sufficiently small to cap the volumes of runoff discharged onto the hillslope below the road so that overland flow does not reach the stream.

Grayson et al. (1993) found that applying a strict enforcement of forest code prescriptions (e.g. suspension of logging during wet weather, protection of runoff producing areas with buffer strips, and the management of runoff from roads, skid trails and log landings) eliminated intrusion of sediments and pollutants into streams.

3.2.4 Nitrogen

Nitrogen is added to the soil naturally from atmospheric fallout (Holland et al., 1997), by biologic process that fix nitrogen from the atmosphere and through the addition of nitrogen rich fertilizers in agricultural systems. Through soil and in-stream processes, a range of nitrogen species are converted to nitrate, which is highly soluble and does not become fixed on clays or organic matter. In this form it is available for mobilisation from a catchment into waterways. Elevated nitrate levels in streams and standing water bodies (which may also result from nitrogen fixation by cyanobacteria) may contribute to eutrophication of surface waters. Eutrophication involves a reduction in the number and species of plankton and diatoms while the total mass of phytoplankton increases. Most significantly, cyanobacteria and unicellular blue green algae (both of

which are toxic to animals and humans) increase in abundance. As phytoplankton and macrophyte debris accumulates within a water column, oxygen concentrations can decrease with adverse impacts upon other oxygen dependent wildlife (such as fish) living in the waterways (Hatch et al., 1996). Within reservoirs, anoxic conditions also trigger a series of chemical reactions that liberate a range of pollutants that are otherwise bound to sediment upon the reservoir floor, such as manganese and iron (White et al., submitted). These pose particular challenges for water treatment for urban supply.

Nitrogen addition to waterways may originate from both point and diffuse sources. Point sources may include sewerage treatments plants or discharge points from high intensity agricultural activities such as cattle feed lots. Within the Lower Cotter catchment, potential point sources will likely be limited to any sewerage discharges from the Pierces Creek forestry settlement. These are likely to be very minor and entirely manageable with current best practice. Diffuse nitrogen sources from natural, forestry and agricultural land uses will be associated with the vast majority of nitrate entering waterways in solution. One approach to evaluating the effects of different landuse scenarios on nutrient delivery to streams is to compare annual average nutrient exports from catchments (or experimental plots) with different landuses. The critical issue becomes one of defining appropriate nutrient generation rates (mass of nutrient per unit area per year) for different landuses. Marston et al. (1995) and Young et al. (1996) present a review of the international scientific literature relating to nutrient generation rates from different landuses. Their results from pine plantations, native forest and pastures are summarised in Figure 8. In the case of total nitrogen (all waterborne types of nitrogen), nutrient export rates for pine, native forest and pasture can be seen to vary over several orders of magnitude. Exports from native (eucalypt) forests range from 0.9 to 5 kilograms per hectare per year, with the two lower data points being from the Mt Lofty Ranges in South Australia. There are two observations of total nitrogen export from pine plantations in New Zealand catchments. These two points span a wide range of values and overlap with the distribution of nitrogen export rates from Australian native forests. This suggests that there is no clear difference in nitrogen generation rates between native forest and pine plantations.

Nitrogen generation rates from pasture cluster into two groups: a high generation rate group with rates between 4 and 20 kg/ha/yr and a low nitrogen generation group with rates below 2 kg/ha/yr. Of the high nitrogen generation group, only one data point is from an Australian catchment and this was from the Mt Lofty Ranges in South Australia and had a value of 4.6 kg/ha/yr. By contrast, the lower group contains more Australian observations including a value of 0.62 kg/ha/yr for Ginninderra in the ACT, published by Costin (1980). Thus evidence suggests that nitrogen exports from Australian catchments are, in general, lower than those observed elsewhere, arguably due to low atmospheric deposition rates (Holland et al., 1997). It also suggests that the nitrogen rates associated with pasture are comparable with those of pine and eucalypt forests noted above. Given this, there is no conclusive evidence to support any major differences in nitrogen generation rates between the different landuse options of native forest, pine forest and pasture (grassland).

3.2.5 Phosphorus

Phosphorous concentrations in freshwater ecosystem also act as an important control upon biologic processes and biomass. The build up of soluble, inorganic phosphorus in surface water changes the species composition of algal and planktonic communities and allows for large phytoplankton blooms to develop (Harris, 1994). The decomposition of excess algae can result in anoxic conditions developing in the water column, as outlined in section 3.2.4. Similar to nitrogen, phosphorous additions to waterways can occur from point sources (such as sewerage treatment works or septic tank discharge points), which are likely to be negligible or absent within the Lower Cotter catchment, or from diffuse, non-point sources.

Young et al. (1996) and Marston et al. (1995) present reviews of diffuse total phosphorous generation rates for different landuses from Australian and international studies, and their data, classified into landuses of pine forest, eucalypt forest and pasture are summarised in Figure 8. Little difference is evident in the range of phosphorous generation rates between native and pine forests with both land uses typically yielding 0.05 to 0.3 kg/ha/yr. Phosphorous exports from pasture intersect this range but

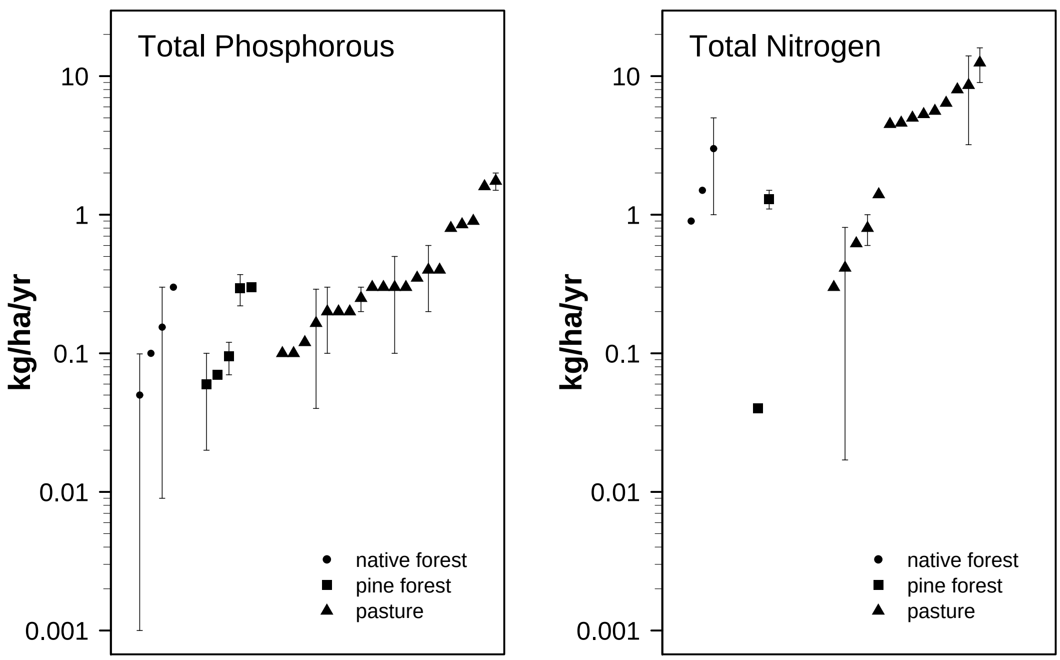

Figure 8: Nutrient export rates listed in Marston et al. (1995) for three different landuses. Note that export rates for each vegetation type have been ranked and are displayed in ascending order from left to right.

there are numerous observations of phosphorous generation rates many times higher than that of forested land. Some of these relatively high phosphorous generation rates are from Australia and near the ACT. Examples include a value of 0.35 kg/ha/yr obtained by Cullen (1991) for the Lake Burragorang catchment, 0.3 kg/ha/yr obtained by Cullen and Rosich (1979) from rural lands in the ACT and 0.12 kg/ha/yr obtained Costin (1980) from grazed catchments in Ginninderra. Thus there is evidence for potentially higher phosphorus generation rates from pasture lands relative to that of the two forest landuse options.

Phosphorus delivery to streams can originate from addition of fertilizers to the land to improve plant growth, but may also be sourced from native phosphorus incorporated into the soil. Caitcheon et al. (1995) demonstrate that erosion of naturally phosphorus-rich basaltic soils in the Chaffey Dam catchment located in northern NSW constituted the major source of phosphorus delivered to the Dam. Wallbrink et al. (2003) also demonstrated an important contribution of sediment attached native phosphorus (as opposed to fertiliser sourced phosphorus) in Bundella Creek (NSW). Similarly for the Murrumbidgee catchment, Wallbrink et al. (1996) show the Murrumbidgee's phosphorus load to be dominated by phosphorus attached to fine grained sediments eroded by gully erosion. The implications of these studies are clear: even with no phosphorus addition to a catchment from fertilisers, soil erosion can be a major source of phosphorus delivered to freshwater systems. This implies that controlling soil erosion within a catchment will have a positive impact upon in-stream and reservoir nutrient levels.

3.2.6 Riparian Buffer Zones and Wetlands

The potential of persistently vegetated riparian zones to reduce delivery of sediments from hillslopes to streams in agricultural landscapes has been demonstrated by many studies including McKergow et al. (2003). They found large, order of magnitude reductions in in-stream suspended sediment concentrations and sediment export rates from an agricultural catchment within which riparian buffer strips had been added. Riparian management had limited impact on total phosphorus and total nitrogen loads however. Similar conclusions were reached by Hairsine (1997) for forested environments.

Wallbrink and Croke (2002) present a more detailed analysis of sediment redistribution in response to harvesting activities. Despite considerable sediment generation on snig tracks and other harvesting related disturbance sites, all the eroded sediment was trapped within either the general harvest area or within stream side buffer strips, with the buffer zones trapping eight times the sediment of the general harvest area on a per unit basis. There is clear evidence therefore of the benefits that well managed riparian zones can have upon controlling sediment delivery to streams.

In addition to riparian filter strips, Zierholz et al. (2001) have demonstrated that in-channel wetlands that have naturally formed within incised channel systems in Jugiong Creek, NSW, have played an important role in trapping sediment within the catchment and preventing downstream delivery. This demonstrates that the formation of in-stream wetlands within the smaller drainage lines of the Lower Cotter catchment could potentially have a marked impact upon sediment movement within the catchment.

3.2.7 Water Quality Conclusions

The minimisation of the delivery of sediment and nutrients (which are often attached to fine sediment particles) is an important goal from a water quality perspective. Both nitrogen and phosphorus are critical elements controlling biologic production in aquatic ecosystems. However, elevated concentrations can promote toxic algal blooms and lead to oxygen depletion within waterways and storages. Thus their concentrations generally need to be minimised.

At the catchment scale, defining nutrient export rates for different land uses can be used as a guide for planning. Nitrogen export rates vary by several orders of magnitude for pine eucalypt and grassland land uses and the export rates for each landuse generally overlap with the others. Thus there is no evidence for any intrinsic difference in nitrogen generation rates between the three landuses. Native eucalypt and pine forests have variable yet broadly similar phosphorus export rates. Phosphorus export rates from pasture are also highly variable though in a number of cases are observed to be notably higher than for the two forest landuses.

Soil loss rates from pasture and woodland vegetation types are highly variable,

though the existing data suggest soil loss rates from pasture environments are lower on average than for woodlands. However, in stream sediment loads are affected by much more than land surface vegetation. Existing gully networks and roads are typically the major sediment sources and most likely dominate the instream sediment loads in the Lower Cotter catchment. Comparison of likely sediment yields from the Lower Cotter catchment under natural and standard Southern Tablelands land management practices suggest a 20 fold variation in sediment yield. Even if a return to natural conditions is either not sought or is deemed not feasible for the catchment, there is clearly scope to reduce contemporary sediment yields from the catchment.

3.3 Biodiversity

The vegetation of the Lower Cotter catchment prior to European settlement of the region was a mixture of Eucalypt forest amidst the higher elevation areas and Eucalypt woodlands around the lower elevation areas adjacent to Cotter Dam (Australian Natural Resources Data Atlas, 2003). Both forest types have under storeys comprising low trees, shrubs and grasses and occasionally in the woodlands, bare ground. Woodlands are characterised by widely spaced trees with gaps between crowns (projected foliage cover $< 30\%$) (Specht, 1970). At small spatial scales floristic diversity of woodlands is noted to be higher than for any other vegetation recorded in south-eastern Australia (Lunt, 1990; Yates and Hobbs, 1997).

Much of the scientific literature that relates to biodiversity issues in modified landscapes focusses upon comparisons of species diversity and abundance between areas of introduced vegetation and adjacent areas of native vegetation. There is a large amount of literature regarding the biodiversity implications and habitat changes associated with establishing pine plantations within forested landscapes. There is also literature regarding the habitat potential of remnant forest and woodlands within agricultural landscapes modified from their pre-European conditions.

Yates and Hobbs (1997) discuss approaches to restoring structural diversity and ecologic function in remnant woodlands, and note a limited amount of scientific data describing techniques and guidelines for successfully accomplishing this. Three ap-

proaches were however identified:

- Allowing recolonisation of a former woodland through natural regeneration processes by removing agents of degradation (such as livestock).
- Enlarging existing woodlands and introducing selected species.
- A combined approach of disturbance removal, species reintroduction and site amelioration.

With regards to the first and second options, it was concluded that where livestock grazing has been excluded, increases in plant species abundance have generally been restricted to annual and perennial species that have seeds which disperse long distances and to woody perennial species that maintain persistent soil seed banks or resprout from vegetative material. Many native grasses in particular do not have long lived seed banks (Morgan and Lunt, 1999). In many of the cases reviewed by Yates and Hobbs (1997), little recruitment of the Eucalypt species that dominate undisturbed woodland canopies was noted. It was concluded that this form of regeneration will at best be a very slow process and at worst allow the development of communities dominated by a subset of woodland and exotic species. Hence the second option. In this case, introduction of critical species which lacked natural regenerative capacity due to prior landscape degradation is undertaken. The third option is based on suggestions (Tongway, 1990; Ludwig et al., 1990) that prior landuse may have had a sufficiently detrimental effect on soil structure and resource regulatory processes that provide plants with water and nutrients that repair of these functions is essential for the restoration of plant species diversity.

Recently, Thrall et al. (2005) have illustrated dramatic improvements in the success rate of restoring of native vegetation in degraded landscapes through inoculating the roots of acacia species with native rhizobia as part of direct-seeding techniques. On average, inoculation led to a 118% increase in establishment of acacia seedlings, substantially reducing the seed requirements. At sites experiencing harsh climatic conditions, subsequent survival of inoculated seedlings was found to be significantly greater

than for uninoculated controls and the inoculated acacias grew faster during the critical early phase of establishment, although this varied among species and sites. This form of rhizobia inoculation is an example of re-establishing important plant-soil interactions in degraded landscapes, which can contribute to the development of biodiverse self-regenerating native ecosystems.

3.3.1 Species diversity and abundance

The availability of suitable habitat is an important influence upon the species composition of a landscape (Smith and Lindenmayer, 1988). There are general differences in both habitat potential and species diversity associated with native forest, pines and grassland. In a direct comparison of the habitat potential of pine plantations at Kowen Forest, ACT and nearby native forest, Davidson (1976) found that native forests supported a more diverse bird population. Habitat diversity and structural complexity in the native forests was identified as important for sustaining the higher levels of bird diversity, as was the presence of prunings on the ground in plantation areas. Schnell et al. (2003) observed consistent decreases in ant species richness in comparing remnant woodland, eucalypt plantation and pasture. Similar conclusions were reach by Sinclair and New (2004) for pine forests relative to eucalypt forests. Catling et al. (1998) note that small mammal abundance was always low on sites with low habitat complexity. Parris and Lindenmayer (2004) found that introduction of a pine plantation within a former eucalypt forest at Tumut lead to a lower species richness of frog communities in the pine plantation and in small and/or isolated remnant patches of native forest. In examining the bird species richness of grassland dominated pastoral landscapes in northern Victoria, Radford et al. (2005) show evidence of a positive relationship between species richness of woodland dependent birds and the extent of habitat cover. The number of woodland bird species present was found to decline markedly in pastoral landscapes with less than 10% tree cover (ie. less habitat complexity). It has also been shown that within two years of re-establishing woodland vegetation near the ACT, the number of bird species increases and continues to increase as habitat diversity increases (Greening Australia, 2001) over time. Thus for birds at least, there is clear evidence for

a greater species diversity associated with woodlands over pines (Lindenmayer, 2000) and pastoral grassland landscapes. Furthermore, species diversity is likely to increase consistently within several years under revegetation of native woodlands. Watson et al. (2001) identify the difficulties of recreating habitat for all bird species in the northern ACT due to existing habitat fragmentation. However, feasible woodland management options could provide habitat for 95% of species.

Re-establishing a pine plantation in the Lower Cotter catchment would likely entail areas of pines growing amidst remnant or revegetated native vegetation. There is a large volume of scientific literature pertaining to habitat fragmentation that is relevant to this landscape scenario. Lindenmayer et al. (2000) note that mammalian species assemblage are substantially impoverished in pine plantations relative to both Eucalypt forest remnants or larger areas of continuous eucalypt forest, leading these researchers to emphasise the ecologic value of remnant patches of native forest within plantation environments. Similar conclusions were reached by Fischer and Lindenmayer (2002) for birds and by Fischer et al. (2005) who concluded that while skink populations could survive in pine plantations near Tumut, a large amount of native forest within 1000 m of a pine area had a beneficial effect upon lizard species richness. Banks et al. (2005) note that habitat fragmentation due to establishment of a pine plantation amidst native vegetation near Tumut restricted the dispersal of a carnivorous mammal, *Antechinus agilis*, particularly the male as the animals were reluctant to cross the habitat interfaces from the native vegetation into the pines. Lindenmayer (2000) and Lindenmayer et al. (1999) present extensive evidence to show that larger patches (ie. $>$ 3 ha) of remnant native vegetation support more native animals than smaller patches, but small to intermediate-sized patches (0.5-3 ha) nevertheless supported important vertebrate populations and were worth conserving (Lindenmayer, 2000; Fischer and Lindenmayer, 2002). These results highlight the importance of conserving (where it remains) or potentially replanting (Greening Australia, 2001), native vegetation within a pine plantation landscape.

Intact areas of riparian vegetation appear to be important dispersal routes for small mammals such as the bush rat (*Rattus fuscipes*) and should arguably be targeted for

retention during softwood plantation development if it were to occur. Restoration of native vegetation should preferentially occur along gully lines to link existing patches of remnant native vegetation that occur within the softwood plantation estate (Lindenmayer, 2000).

There is evidence from some studies of a relatively benign impact of landscape scale habitat fragmentation upon certain species. For example, Major et al. (2003) found that arboreal insects were not adversely affected by the level of habitat fragmentation of woodlands in south eastern Australia. Similarly, Major et al. (1999) found that the species richness and total abundance of both ants and beetles did not differ significantly between remnant native vegetation along roadside strips within agricultural landscapes and larger native forest areas within State Forests, though there were differences in community composition. Some bird species were also observed to increase in abundance within pine dominated landscapes (Lindenmayer et al., 2003), though others decreased. What these studies illustrate is that any systematic landscape change, whether of anthropogenic or natural origin, will produce biologic “winners” and “losers”. That is, some species who are adapted to a new set of conditions, be they an introduced pine plantation or a post-bushfire environment in a native forest, may respond positively in terms of abundance, whilst others may be affected detrimentally.

Changes in habitat complexity over time are also a feature of natural systems, particularly in response to major disturbances such as bushfires. Catling et al. (2001) document consistent increases in habitat complexity over an eighteen year period following bushfires in 1980 at Nadgee Nature Reserve (NSW). Changes in relative abundance of different species over time were related to changes in habitat, with some species responding positively and others negatively to the longer term pattern of natural post-fire regrowth. Features such as tree canopy cover, shrub cover, ground cover, litter cover and wetness were significant in many models of species abundance (Catling and Burt, 1997; Catling et al., 2000) for the area. Smyth et al. (2002) similarly note a diverse range of responses of individual species to forestry operations, fire regimes and landscape change in Queensland forests.

3.3.2 Forest harvesting and biodiversity

Harvesting of timber may be an option within the Lower Cotter catchment under certain revegetation strategies. Here, the literature regarding the effects of harvesting activities on flora and fauna is reviewed.

Catling et al. (2000) note a general preference of ground-dwelling fauna for forests that were least disturbed by logging activities. However, some species were found to be more tolerant of disturbance than others. Lindenmayer et al. (1990) and Lindenmayer et al. (2000) note that clear fell timber harvesting of native forests removes important habitat for fauna dependent upon tree hollows. Hazell (2003) presents a summary of potential threats to frog survival in Australia and noted that frogs appeared particularly sensitive to many disturbances in or near their habitats. Gillespie (2002) show that tadpole growth and development in south eastern Australia was significantly reduced by higher sediment loads within streams. Forest harvesting activities or road infrastructure that leads to soil disturbance and delivery of sediments to streams, along with the associated forestry infrastructure (roads, drains etc) has the potential to impact negatively upon the growth and development of tadpoles and reduce recruitment of tadpoles to the terrestrial (frog) stage.

There is evidence that riparian buffer strips ameliorate some of the detrimental effects of timber harvesting. Quinn et al. (2004) note that clear felled New Zealand forests had the lowest species diversity and relative abundance of a range of insect species. Sites that had been logged leaving continuous buffers did not differ in terms of insect species richness from those in intact native or mature plantation forest, indicating that buffers greatly reduced disturbance associated with logging as far the studied insects were concerned. Regarding in stream effects, logging activities resulted in increases in periphyton biomass and water temperature, associated with changes in stream lighting, and increased channel instability and fine sediment load. Boothroyd et al. (2004) note that stream lighting was heavily influenced by the presence of riparian vegetation as well as stream size for small to moderate size streams and that mature radiata pine in riparian areas of pre-harvest sites provided shading which was similar but less variable than that recorded in native reference sites.

Three general themes emerge from the scientific literature regarding potential biodiversity outcomes. First, native forests and woodlands are associated with higher levels of habitat potential and greater biodiversity than either pine plantations or pasture lands. Secondly, there is clear evidence of the benefits to biodiversity of preserving remnant native vegetation within modified landscapes such as pine plantations or agricultural landscapes. Research indicates that riparian zones are important areas within which to preserve native vegetation and establishing corridors linking otherwise isolated stands of remnant vegetation will be beneficial from a biodiversity perspective. Thirdly, with any change to a landscape's vegetation, there will inevitably be some species that are well adapted to new vegetation conditions and may respond positively whilst the same vegetation change may have detrimental effects on the abundance of others.

3.3.3 Minesite Rehabilitation

There is a substantial body of literature, much of it from Australia, relating to the practices and success of mine site rehabilitation that is potentially of relevance to revegetation of the Lower Cotter catchment, particularly for any areas where native vegetation may be re-established. Minesite rehabilitation projects in Australia have generally aimed to restore the native ecosystems of an area (Bell, 2001), though examples of re-establishing pasture exist (Brown and Grant, 2000). Some characteristics of mine sites, such as being highly disturbed landscapes, often located in harsh climatic environments and with limited potential for natural resprouting of vegetation (Grant and Koch, 2003) make them similar to the challenges faced in the post-fire rehabilitation areas of former pine plantation in the Lower Cotter catchment.

Analyses of minesite rehabilitation projects, predominantly from Western Australia, generally show positive results against a number of important ecologic indicators. Infiltration rates of water into rehabilitated soils increased strongly with increasing vegetative cover, whilst erosion rates were greatly reduced, declining from 3035 t/ha at 0% vegetative cover to 0.5 t/ha at 47% cover (Loch, 2000). Soil chemical properties are also noted to recover to near pre-disturbance levels in under 10 years (Ward, 2000).

General plant selection advice for minesite rehabilitation in NSW can be found in Grant et al. (2002). Windsor and Clemments (2001) concluded that *Themeda australis*, a native grass common to the ACT, has many attributes that make it an ideal species for mine rehabilitation. They found that adequate populations of *T. australis* can be achieved by applying 12 month old seed bearing hay of *T. australis* directly onto rehabilitation areas.

Faunal responses to minesite rehabilitation typically vary, though results presented by Nichols and Nichols (2003) suggest that although the rates of faunal recolonization vary from species to species, with time most or all mammal, bird, reptile, and ant species should inhabit rehabilitated bauxite mines in Western Australia with populations similar to those in un-mined forest. Similarly, Grant and Koch (2003) note that the majority of orchid species identified in unmined areas were present in rehabilitated lands after 1 to 31 years, though plant densities were an order of magnitude lower on rehabilitated lands. More generally, Gardner (2001) found that rehabilitated jarrah forests in Western Australia had 60 to 97% of forest average species richness. It has also been found that on well developed vegetation communities on rehabilitated minesites that provide sufficient habitat diversity, bird populations can equal or exceed those recorded in healthy un-mined forest (Armstrong and Nichols, 2000; Ludwig et al., 2003).

Innovative revegetation techniques have also been developed for minesite revegetation. In addition to the seed inoculation technique of Thrall et al. (2005) noted earlier, Bell (2001) note that treating seed to be broadcast across a landscape with smoke, or water through which smoke has been bubbled, the germination rate of broadcast seed was doubled. This had resulted in reduced seed requirements. These studies show that even for highly degraded landscapes, re-establishment of native ecosystems can be successfully achieved. In addition, technology developed for mine site rehabilitation may benefit revegetation works in the Lower Cotter catchment if a return to native vegetation was selected for part or all the catchment

3.3.4 Weed management

One of the important aspects to consider in the revegetation of the Lower Cotter catchment is the proximity of the Lower Cotter catchment to Namadgi National Park and Canberra Nature Park. Any revegetation with non-indigenous species offers the adverse possibility for invasion of these species into these adjacent conservation reserves. The spread of pine wildlings from plantation areas into adjacent native forests is well documented. Burdon and Chilvers (1977), Foster (1979) and Lindenmayer and McCarthy (2001) provide examples of this from the Brindabella Ranges of the ACT and the Tumut region of NSW. Pine invasion is largely an edge effect confined to half (Burdon and Chilvers, 1977) or several kilometres of a plantation edge. Occasionally, pines are found deeper into native forest areas (Foster, 1979), perhaps growing from seed transported by birds. Burdon and Chilvers (1977) and Lindenmayer and McCarthy (2001) note greater occurrences of P. radiata wildlings in areas that were dominated by dry eucalypt forest types (such as any remnant woodlands that may be preserved in the Lower Cotter catchment) and in areas with a limited cover of ground vegetation. The relationship between pine invasion and prior site disturbance was also noted by Foster (1979). Pine plantations were also associated with a significantly higher level of blackberry (*R. fruticosus*) occurrence than continuous areas of native forest (Lindenmayer and McCarthy, 2001).

3.3.5 Biodiversity conclusions

The original woodland and forest vegetation of the Lower Cotter catchment most likely had high levels of floristic diversity. There is consistent evidence that native vegetation communities are associated with higher levels of faunal biodiversity than pine plantations, which in turn support higher levels than grass based agricultural landscapes. However, for both pine plantations and agricultural settings, pockets of native vegetation can provide important habitat to native animals. Pine wildlings are likely to leak from any plantation area into nature reserves in or near the Cotter catchment. Experience from mine site rehabilitation projects show that functioning native ecosystems

can successfully be re-established in highly degraded landscapes and a number of techniques exist to dramatically increase the success rates of native plant seedlings.

3.4 Bushfire management implications

Bushfires are a naturally occurring process within the Australian landscape. Fire intensity, defined as the heat release per metre of fire front, is calculated by the equation:

$$\mathrm{I} = \mathrm{HwR} \tag{4}$$

where I is the fire intensity (W/m), H is the heat yield of the fuel (J/g), w is the available fuel (kg/m^2) and R is the rate of spread (m/s) (Byram, 1959). Values of H for grasslands can be found in Cheney and Sullivan (1997) and typically vary from 11500 kJ/kg to 18600 kJ/kg. Heat densities for forest fuels are higher, varying from 16000 to 23500 kJ/k (Dickinson and Kirkpatrick, 1985). Fire intensity is a useful diagnostic of the suppressibility of bushfires once started. Ground crews typically have difficulty bringing fires under control once their intensity exceeds 4000 kW/m (Luke and McArthur, 1978) and fires with intensities exceeding 10000 kW/m are exceeding difficult to suppress by any means (Hirsch and Martell, 1996). Here, we investigate variations in the available fuel load, w, for the vegetation types of native forest, pine plantation and grassland, review their rates of spread and finally examine fire intensities under the different vegetation scenarios.

Fuel loads accumulate over time in forested and grassland environments due to the senescence of vegetation. Following on from a disturbance (such as a previous fire or establishment of a pine plantation) that removes any accumulated fuel, fuel loads within forests or grasslands are noted to increase over a time period of 5 to 20 years, before reaching a steady state level where natural breakdown of litter is balanced by new additions (Walker, 1981; Chaffey and Grant, 2000; Lunt and Morgan, 2002). As noted above, the amount of fuel determines the amount of heat that can be released whilst the rate of heat release is affected by fuel properties, weather, wind direction and slope of the land. Fuel loadings are generally expressed in terms of mass of available fuel per unit area, with the available fuel generally considered that less than 25 mm in

Vegetation type	Source	fuel load (t/ha)
Grassland (Themeda triandra)	Lunt and Morgan (2002)	10 - 12
Ungrazed natural grassland	Walker (1981)	10
Introduced temperate pasture	Walker (1981)	5 - 15
Pinus (ACT)	Walker (1981)	17
Coastal forest	Chaffey and Grant (2000)	11 - 17
Open forest(ACT)	Walker (1981)	17.5
Open forest(Snowy Mountains)	Walker (1981)	40
Forest (Blue Mountains)	van Loon (1977)	6 - 24
Eucalypt forest (ACT)	Raison et al. (1986)	17 - 26

Table 4: Fuel loads under different vegetation types.

diameter. Hence while Eucalypt forests may have a much greater total above ground biomass than grassland, only about 5% is available as fuel for a bushfire whereas for grasslands, 100% combustion is common (Walker, 1981). Table 4 shows steady state fuel loadings for a range of vegetation types. Grassland vegetation typically has lower fuel loadings than either Pinus plantations or native forests, though there is overlap.

Catchpole (2002) indicate that there are no major difference in what is termed extinction moisture, or the moisture of fuel above which a fire is not self sustaining, between grassland and forests (ie. both vegetation types are equally likely to ignite). There is general agreement in the literature that once ignited, the fire spread rate, R decreases with increasing bulk density of the fuel. Cheney (1981), Cheney and Sullivan (1997) and Catchpole (2002) present relationships indicating the rate of spread of grassfires is roughly double that in forested environments. Thus while the heat density of grassland fuels may be lower than that of forests, this can be compensated for by a higher spreading rate. In Table 5, fire intensities for a range of vegetation options area presented. Both native grassland and forest vegetation share similar fire intensities except at the highest values, where intense fires in Eucalypt forests may exceed the possible intensities for grassland fires. Both forested and grassland vegetation types are clearly capable of producing fire intensities well above those manageable by both ground crews and aircraft attack. This conclusion will also apply to pine plantations given their similar fuel loads to Eucalypt forests.

Vegetation type	Source	fire intensity (kW/m)
Grassland (Themeda triandra)	Lunt and Morgan (2002)	100 - 1,200
Savannah grasslands	Williams et al. (2002)	500 - 10,000
Savannah grasslands	Rossiter et al. (2003)	2,000 - 16,000
Grasslands	Cheney and Sullivan (1997)	10 - 60,000
Grasslands	Luke and McArthur (1978)	$<$ 30,000
Mallee woodlands	Bradstock and Cohn (2002)	20,000 - 30,000
Eucalypt forest	Chafer et al. (2004)	$>$70,000
Ground suppression limit	Luke and McArthur (1978)	4,000

Table 5: Fire intensity for different vegetation types.

Controlled burning for fuel reduction purposes is often raised as an option for enhancing bushfire suppression. The degree of fuel reduction burning in and around the ACT received considerable attention following the 2003 bushfires (Select committee into the recent Australian bushfires, 2003). In a review of prescribed burning effectiveness, Fernandes and Botelho (2003) note that as fire behaviour increases in a non-linear manner with the decrease in fuel moisture and the increase in wind speed, which additionally vary in a much wider range than fuel properties, the influence of these non-fuel factors will increasingly prevail over the effects of fuel characteristics under severe weather conditions. Locally, the effects of fuel reduction burning in aiding bushfire suppression have been examined by McCarthy and Tolhurst (2001) in Victoria. Consistent with the conclusion reaches by Fernandes and Botelho (2003), they found that as fire danger increases, the benefits of fuel reduction burning in terms of fire suppression are less pronounced. That is, at higher levels of fire danger (ie. more severe weather conditions), weather influences become more important than fuel conditions in terms of successful suppression operations. Fuel reduction burning has the highest probability of assisting suppression activities within the first four years, beyond 10 years the probability that a fuel reduction burn will assist suppression activities was found to decrease significantly (McCarthy and Tolhurst, 2001; Fernandes and Botelho, 2003). Consistent with this, Chaffey and Grant (2000) argue that for rehabilitated mine sites (that originally had no vegetation cover), fuel loads in sites older than 8 years are

sufficient to sustain a crown fire. Bradstock et al. (1998) demonstrated that a reduction in the number of days of uncontrollable fire from 10 to 1 per annum around the urban-bush interface of Sydney could only be achieved through burning up to 40% of the interface on an annual basis. The implications of these studies are that during severe fire weather conditions, fuel reduction burning has little impact upon the ability of firefighters to suppress wildfires once ignited in any of the three main vegetation types. Burning at the frequency required to make a substantial difference in fuel loads (at least every 10 years) would likely have a substantial influence upon the species composition of the burnt vegetation (Lunt, 1997; Gill and Williams, 1996). It would also be highly detrimental to a pine plantation, where trees less than 12 years of age are easily killed by burning. Fuel reduction burning is however likely to increase the number of days per year during which successful bushfire suppression operations can be undertaken (Gill et al., 1989; Fernandes and Botelho, 2003), thus increasing the chances of extinguishing a fire before more severe weather conditions make suppression impossible.

One important theme raised in the literature was the significance of future climate change for possible changes to the bushfire regime of the ACT. Climate modelling for conditions of doubled atmospheric carbon dioxide concentrations suggests that daily summer maximum temperatures in southern Australia will be higher by 4 to 4.5°C (Whetton et al., 2001; CSIRO, 2001; Pittock, 2005). Beer and Williams (1995), Cary (2002) and Hennessy et al. (2005) have all examined the consequences of likely future climate change for bushfire characteristics. Forest fire danger ratings for southern Australia were predicted by all studies to increase, meaning that there would be a greater chance of fires starting. Hennessy et al. (2005) estimate for Canberra that the number of days per year that the forest fire danger rating will be very high or extreme will increase from its present value of 23.1 to between 27.9 and 38.3 by 2050. Once ignited, fires will have higher rates of spread, be more intense and more difficult to suppress (Beer et al., 1988). Predictions of the change in fire frequency made specifically for the ACT under a range of climate change scenarios show that Lower Cotter catchment is a particularly sensitive area of the ACT and is likely to experience a marked increase in fire frequency under likely future climate change scenarios (Cary, 2002). ACT wide

inter-fire intervals were expected to decrease from around 42$\pm$24 years to potentially as low as 12$\pm$6 years under the most dramatic climate change scenario. Fire intensity was also expected to increase by approximately 25%. This suggestion of greater fire risk and severity may be an important consideration in assessing the revegetation options, particularly with regards to protecting fire sensitive commercial forestry plantations which may be at greater threat from bushfire (Hennessy et al., 2005).

3.4.1 Fire and Regenerative Capacity

Fire has been present in the Australian landscape for millennia (Singh et al., 1981) and periodic burning is a requirement for successful rejuvenation of many Australian plant communities. It is often argued as an important factor for stimulating plant regeneration particularly through seed germination in the woodlands that most likely prevailed in the Lower Cotter catchment prior to European settlement (Hobbs, 2002). Consequently, native vegetation has a particularly high resilience (or capacity to regenerate) following bushfire. Local examples of this capacity include the study by Purdie and Slatyer (1976) of post-fire regeneration of dry sclerophyll forests of ACT. They noted that all species represented either by living plants in the tree and shrub strata and/or by seed in the soil and litter prior to burning regenerated during the first year after a fire. Similarly, Freudenberger et al. (2004) studied the post-fire survival and regeneration of planted juvenile (less than 10 years old) native vegetation and pines around the ACT following the 2003 bushfires. The resilience of recently planted native vegetation was found to be variable at a species level, but generally at least some Eucalypt and Acacia species survived and resprouted at all the burnt sites they studied. In contrast, the resilience of pines in the ACT six months post-fire was found to be negligible and few seedlings were noted. Barker (1988) has argued for fire events being the cause of a number of past snowgum recruitment occurrences in Kosciuszko National Park, whilst Burrows et al. (1990) demonstrated that Eucalyptus seedlings growing on post-fire ash beds exhibited superior growth and lower mortality to those growing on undisturbed soil. Morgan and Lunt (1999) note for native tussock grasses in Victoria consistent declines in plant health and vigor when inter-fire intervals exceed 6 years and suggest a burning

interval of less than approximately five years is required to sustain the grasslands. Birds species composition in native forests of the ACT have been shown to remain unchanged before and after fire (Catling and Newsome, 1981).

Unlike many native species, *Pinus radiata* does not have the capacity to resprout following the death of above ground tissue after fire. However, fire initiates release of stored seed from cones 2-3 days after the fire and the heating of the fire has a minimal effect upon seed viability (Reyes and Casal, 2002). Consequently natural regeneration of *Pinus radiata* after fire occurs through this process. However, juvenile pine wildings are particularly vulnerable to fire. Burrows et al. (1989) have shown that young *Pinus radiata* wildlings (up to 10 m in height) are readily killed by low intensity fires (ie. $<$ 200 kW/m). Studies cited by Fernandes and Botelho (2004) and Burrows et al. (1989) suggest that mature pines can survive when up to half to two thirds of the crown is scorched. The fire intensities attained under wildfire conditions (see Table 5) will certainly lead to extensive if not complete crown scorch and death of mature trees, as is clearly demonstrated by the pine mortality following the 2003 bushfires.

The conclusions of Freudenberger et al. (2004) from their study of the post-2003 bushfire regeneration are worth noting at this point. They stated that "plantings of native vegetation should never need total replanting again. The next fire will remove some species from the plantings, depending on the type of species, fire intensity and age of the plantings. However, even plantings less than ten years of age will have the capacity to at least partially regenerate after the next fire". In contrast, pines were noted to have " negligible inherent post-fire resilience . . . and remain exposed to the high risks of the next fire that will inevitably break out somewhere within the region over the next few decades".

3.4.2 Bushfire Management Conclusions

Fire intensities under native or pine forests and grass vegetation types will, under severe fire weather conditions, burn at intensities that make suppression extremely difficult to impossible. Also, under severe fire weather conditions, fuel reduction burning is unlikely to assist fire suppression activities, though will extend the number of days per

year that suppression activities can be undertaken. Studies have demonstrated that under future climate change scenarios, the hazard posed by bushfires to the Lower Cotter catchment is likely to increase. One of the critical decisions to be made then relates to the capacity of the replanted vegetation of the Lower Cotter catchment to regenerate successfully from bushfire. Both native vegetation and *Pinus radiata* have this capacity and indeed fire may be a requirement for successful regeneration of many native species. Studies from the ACT have however shown a higher degree of post-fire resilience of native vegetation to bushfire than for pines. Moreover, a shift to a more hazardous fire regime caused by climate change may impact negatively upon the quality of harvested timber from a commercial pine plantation if it were to burn. This then implies that greater resources may be needed to prevent wildfires spreading into a commercial pine plantation in the Lower Cotter catchment that presently allowed for. Episodic burning by bushfire may be acceptable for regenerated native vegetation for which no commercial timber harvesting was anticipated.

4 Conclusions

4.1 Fires, cycles in vegetation and hydrological response

Recent research has highlighted the regular nature of bushfires within the Cotter catchment. Five major bushfires are recorded within the Cotter catchment over the last century with the January 2003 fires being the most extensive. The timing of these major bushfires is suggested to be related to the occurrence of larger scale climatic and oceanographic changes in the Pacific Ocean, including the El Nino/Southern Oscillation and longer cycle Pacific Decadal Oscillation.

It is widely recognised that fire, vegetation and hydrologic cycles are linked. For example, studies from the Melbourne water catchments have demonstrated that the quantity of streamflow is strongly coupled to the fire/vegetation regrowth cycles. When vegetation is vigorously regrowing after bushfires then evapotranspiration is increased and streamflow decreased as a consequence. The recovery of the Melbourne water catchments following the 1939 bushfires has enabled a landmark scientific research program, the conclusions of which are partially transferable to the ACT.

Beyond water quantity, it is known that bushfires, through changing the amounts of litter on the soil surface and understorey and overstorey vegetation, allow a range of other landscape changes to occur. These include changes in the movement of sediments and associated pollutants, growth of weeds and the structure of habitats, food sources for other biota, fuel loads for subsequent fires and altered chemistry of streamflow. There are documented examples of major inputs of inorganic sediment to the upper Cotter reservoirs following the 2003 fires, and elevated manganese and iron levels. These changes are strongly linked to the changes in vegetation cover and its role in stabilising soils in the catchments of these reservoirs. Elevated turbidity levels have persisted in Cotter Dam since the 2003 bushfires.

Plantations have a range of additional management issues following destructive bushfires. These include: the necessity of salvage logging and associated soil disturbance, the increased likelihood of the invasion of weeds in environments where bare soil is common, reduced natural seed stock and their comparatively dense road

networks. Because of these reasons and that plantations may not follow the natural recovery cycle of native forests; there are particular management needs for plantations above and beyond managing post fire revegetation in native forest environments.

4.2 Vegetation/land use options for water quantity

There is strong evidence that forested catchments use more water than grassed catchments and as a consequence forested catchments have less stream flow than catchment in pasture. The magnitude of this effect varies with rainfall.

For forested catchments that are utilised for forest products (timber and pulp) the above effect is mitigated by the effect of rotation lengths. Forest fires can change the pattern of water use by the forest and the streamflow - more streamflow in the period immediately after the fire and less streamflow in the period of rapid regrowth (8 to 50 years after the fires). In native forests these effects vary in magnitude according to rainfall and the severity of fires. For pine plantations, where the trees are normally killed by severe fires, the effects are significant.

The influence of changed land use on streamflow quantity may not be important in the Lower Cotter Catchment because the ratio of the reservoir volume (4700 ML capacity at full supply level) to catchment area is small compared with other similar water supply catchments. In short this reservoir is more frequently likely to be at full supply level. However this conclusion will change if there is change to the storage capacity of the Cotter Reservoir or extractions from the reservoir are significant.

4.3 Vegetation and land use options for water quality

Within an urban water supply context, the principle of adopting multiple “barriers” between potential pollutants and the consumer is standard practice. One important barrier can be a pristine water supply catchment, though this option is not available for a large portion of the Lower Cotter catchment. In water supply catchments worldwide, riparian zone management is recognised as an efficient means of protecting water quality where the land surface is used for productive purposes.

As a general principle for management of water supply catchments, the minimisation of nitrogen and phosphorus delivery to waterways is desirable. These are two nutrients whose concentrations typically limit biological process rates and productivity and elevated concentrations can lead to cyanobacteria and unicellular blue green algae (both of which are toxic to animals and humans) blooms. As phytoplankton and macrophyte debris accumulates then decomposes, aquatic oxygen concentrations can decrease with adverse impacts upon oxygen dependent wildlife living in the waterways and water chemistry. Elevated levels of organic carbon may be another significant issue for the management of reservoirs and waterways. The degree to which bushfires release organic carbon in forms of concern depends upon the severity of the fire and buffering capacity of the soil.

Nitrogen and phosphorus can originate from point sources such as discharge points from sewerage treatment works, biologic fixation and fertiliser application. They are also naturally occurring elements in soils and elevated nutrient levels in waterways can result from high rates of soil erosion. Research from the upper Murrumbidgee catchment in particular has demonstrated that the phosphorus load within the river originates from sediments eroded from gullies and stream banks. This finding is likely to apply to the Lower Cotter catchment in the post-fire period because the low levels of vegetation and high road density (through concentration of runoff generated on the roads) have triggered extensive gully and stream bank erosion.

Reviews of nutrient generation rates from different forms of vegetation and landuse show that there is a very large range of nitrogen and phosphorus generation rates from each of the potential land uses. The pollutant generation rates of catchments with plantations, native forests and pasture land are not clearly separable in a statistical sense. It is widely recognised that land areas with undisturbed vegetation have relatively low rates of sediment, and nutrient movement. However, disturbance from roads, grazing and fire breaks are major factors in producing higher levels of the movement of these potential pollutants.

4.4 Management of roads

Roads are universally recognised as a major source of sediment generation and often trigger other soil erosion features in forested environments. Some studies suggest they are also significant contributors to stream sediment and nutrient loads in pasture and cropping environments. Native forests and pasture lands generally have lower road and track density than plantations and as a consequence have lower overall sediment generation rates.

The Lower Cotter catchment has a high density of roads especially in the plantation areas. There is strong evidence that roads are triggering accelerated erosion in the Lower Cotter Catchment for both the plantation and native forest areas. Recent research establishes that, as with many other catchments, the Lower Catchment contains many new gullies that have been triggered by inadequate drainage of roads.The removal and better drainage of roads is highly likely to have a beneficial impact upon water quality in the Lower Cotter Catchment in the short and long term. This conclusion holds independent of the vegetation cover

4.5 Riparian Vegetation as a priority

Many studies have demonstrated the value of riparian vegetation in providing useful functions within water supply catchments. These functions include:

- Reducing sediment and nutrient ingress to streams by both surface and subsurface pathways.
- The stabilisation of stream banks.
- The provision of habitat with in the stream channel and in the adjacent banks.
- The moderation of stream temperature variation through shading.

There are two main types of riparian vegetation: filter strips which provide a permanent ground cover and riparian forests which combine litter as ground cover with woody vegetation. The design of riparian zones varies from local to location with many local

guidelines existing. Some riparian zones extend to cover ephemeral streams while other are confined to perennial channels. Also some local designs combine grass filter strips with riparian forest immediately adjacent to the stream channel.

Returning riparian vegetation to the landscape is also likely to be highly beneficial for water quality and biodiversity in the Lower Cotter catchment. The relatively small areas of riparian zones and the controlling influence that riparian vegetation has on sediment generation and in-stream ecology make rehabilitation of the riparian zone an efficient path to improving some key catchment attributes.

4.6 Dealing with the consequences of soil erosion that has already occurred with in stream management measures

There is a widely held view that an extremely large pulse of sediment has entered the streams of the Lower Cotter Catchment since the 2003 bushfires, though this requires formal investigation. This sediment has partly moved into the Cotter Reservoir and is partly in temporary sediment stores within the stream network. Vegetation on the land will have little or no impact on the movement of these large sediment stores already in the waterways to the reservoir. Constructed wetlands are a management measure potentially able to control the delivery of these key sources of sediment and related pollutants to the reservoir. As a consequence of the above observations three major conclusions are made:

1. Constructed wetland are a priority for management in the short term.
2. The Cotter Reservoir could be managed to encourage the formation of a stable wetland at its backwater zone.
3. A budget of sediment within the stream and reservoir is an urgent priority for monitoring and research.

4.7 Biodiversity

There is consistent evidence that native woodland or forest ecosystems support higher levels of biodiversity than do either pine plantations or pasture dominated landscapes.

For fauna, this is primarily due to the greater habitat provided in native woodlands. Evidence from minesite rehabilitation studies shows that native ecosystems can successfully be restored in highly degraded landscapes.

Preservation of areas of remnant native vegetation within pine plantations has a positive effect upon biodiversity levels of pine plantations. Riparian zones in particular are recognised as high priority areas for potentially re-establishing native vegetation within a pine plantation landscape.

4.8 Bushfire Management

Fire intensities under native or pine forests and grass vegetation types will, under severe fire weather conditions, readily attain intensities that make suppression extremely difficult or impossible. Also, under severe fire weather conditions, fuel reduction burning is unlikely to be of substantial assistance to fire suppression, though will increase the number of days per year that suppression activities can be conducted. Studies have demonstrated that under future climate change scenarios, the hazard posed by bushfires to the Lower Cotter catchment is likely to increase. One of the critical decisions to be made in light of this prediction relates to the capacity of the replanted vegetation of the Lower Cotter catchment to regenerate successfully from bushfire. Native vegetation appears to be more resilient in this regard than *Pinus radiata*. A shift to a more hazardous fire regime may impact negatively upon the quality of harvested timber from a commercial pine plantation if it were to be burnt. Alternatively, greater resources may be needed to prevent wildfires spreading into a commercial pine plantation in the Lower Cotter catchment from adjacent bushland, though this may be acceptable outcome for native vegetation within the catchment for which no commercial timber harvesting was anticipated.

4.9 Limitations

This report synthesizes the relevant scientific literature regarding water quality and quantity outcomes from a range of landuses and vegetation options, focussing on Australian catchments in particular. The degree to which the scientific conclusions from

other studies are transferrable to the Lower Cotter catchment are summarised below:

- water yield (the quantity of stream flow) - conclusions from other studies are relatively robust and transferable;
- water quality - there is a very high level of variability in the results for land use/water quality studies. Results concerning the importance of roads in the sediment generation in forested catchments are relatively robust. The importance of this factor justifies an on-going monitoring program in the Lower Cotter Catchment;
- controlling sediment yield to streams - as with water quality, the variability of published studies makes transferring conclusions between catchments difficult. The importance of riparian zone management is a universal conclusion.
- biodiversity - the importance of habitat for sustaining biodiversity is a widespread conclusion, as is the generally negative influence of forestry operations. Expert opinion on this issue is strongly in favour of returning endemic vegetation where possible;
- bushfire management implications - bushfire characteristics can be highly variable on an event basis, though the conclusions regarding peak fire intensities exceeding suppression limits are robust;
- ongoing maintenance the maintenance of road drainage to a high standard is a major priority for the management of these catchment - this conclusion is widely accepted.

5 Opportunities for learning and community participation

The rehabilitation of the Lower Cotter Catchment provides considerable opportunities for community learning, participation of the public in natural resource management, education (at all levels) and research. Bushfires and the recovery of catchments are an important part of the Australian landscape. Having the Lower Cotter Catchment so

close to Canberra offers major opportunities for learning through participation in the process of catchment recovery.

In this section we set out the ideas for learning that have been recorded from the many discussions that occurred in the process of this project. The ideas are presented here as suggestions, though some of the research-oriented ideas have a purpose of reducing the uncertainty of the predictions of outcomes based on the limited experience from catchments elsewhere.

5.1 A catchment rehabilitation centre

The Cotter catchment is an ideal location for a multi purpose centre that focuses on the processes of catchment recovery. Such a centre could serve as a base for community groups involved in special replanting projects, monitoring of wildlife recovery and recreation groups. It could also serve as a visitors centre for public education about the process of recovery and the associated eco-hydrological cycles. Finally it could serve as a base for field research concerning fire, water, vegetation and wildlife. There are many innovative approaches to such research that could involve the community including: the recording of bird and frog calls and their subsequent analysis via the internet by amateur interest groups; the uploading of water quality data from water watch groups to databases interpreted and shared by professional agencies; and the coordination of school projects concerning revegetation and monitoring so that shared learning approach and field data is available to all schools.

5.2 Research opportunities

The researchers who have contributed to this report have identified the following research opportunities that are specific to the Lower Cotter Catchment. These activities will substantially improve the basis for managing the post-fire recovery of this environment:

- Monitoring of water quality, water quantity and in-stream health of streams and the reservoir. In addition to the normal measured attributes, monitoring of the

downstream propagation of major stores of sediment and other pollutants that have mobilised following the fires and the effectiveness of mitigating this downstream transport through construction of wetlands would provide a greater understanding of the longer term catchment responses. The significance of these actions for the restoration of in-stream habitat is also an important question. This monitoring would be of benefit to the relevant agencies and broader community and would ideally be suited to be linked to CSIRO's Water Resources Observation Network (see http://www.wron.net.au)

- The development of methods to prioritise and target the management and removal of existing roads to protect water quality
- An assessment of which areas are regenerating successfully under natural conditions and which areas may, due to legacy landuse issues (loss of seed stock, poor soil quality) require more direct management or intervention.
- The establishment of sub-catchment scale trials of rehabilitation techniques. Subcatchment would receive differing rehabilitation management and the comparisons of the sub-catchments would form the basis of monitoring of water, wildlife and aesthetic attributes.
- Fire management research could also be included in the design of the catchment. Trails of different styles of control burning would be useful from a research and community education perspective.

References

ACT Government (2004a). Think water, act water, *Summary*.

ACT Government (2004b). Think water, act water: State of the ACT's water resources and catchments, *Volume 3*.

ACT Government (2004c). Think water, act water: Strategy for sustainable water resource management in the ACT, *Volume 1*.

Andreassian, V. (2004). Waters and forests: from historical controversy to scientific debate, *Journal of Hydrology* **291**: 1–27.

ANZECC (2000). Australian and New Zealand guidelines for fresh and marine water quality, *Technical report*, Australian and New Zealand Environment and Conservation Council.

Armstrong, K. and Nichols, O. (2000). Long-term trends in avifaunal recolonisation of rehabilitated bauxite mines in the jarrah forest of south-western Australia, *Forest Ecology and Management* **126**: 213–225.

Australian Natural Resources Data Atlas (2003). Pre-european and present native vegetation, *spatial dataset ANZCW0501009823*, National Vegetation Information System, Australian Government Department of the Environment and Heritage.

Banks, S. C., Lindenmayer, D. B., Ward, S. . J. . and Taylor, A. C. (2005). The effects of habitat fragmentation via forestry plantation establishment on spatial genotypic structure in the small marsupial carnivore, *Antechinus agilis*, *Molecular Ecology* **14**: 16671680.

Barker, S. (1988). Population structure of snow gum (*Eucalyptus pauciflora* Seib. ex Spreng) subalpine woodland in Kosciusko National Park, *Australian Journal of Botany* **36**: 483–501.

Beer, T., Gill, A. and Moore, P. (1988). Australian bushfire danger under changing

climatic regimes, *in* G. Pearman (ed.), *Greenhouse: planning for climate change*, CSIRO Australia.

Beer, T. and Williams, A. (1995). Estimating Australian forest fire danger under conditions of doubled carbon dioxide concentrations, *Climatic change* **29**: 169–188.

Bell, L. (2001). Establishment of native ecosystems after mining – Australian experience across diverse biogeographic zones, *Ecological Engineering* **17**: 179–186.

Binkley, D., Burnham, H. and Allen, H. (1999). Water quality impacts of forest fertilization with nitrogen and phosphorus, *Forest Ecology and Management* **121**: 191–213.

Boothroyd, I. K., Quinn, J. M., Langerb, E., Costleya, K. J. and Steward, G. (2004). Riparian buffers mitigate effects of pine plantation logging on New Zealand streams 1. Riparian vegetation structure, stream geomorphology and periphyton, *Forest ecology and management* **194**: 199–213.

Bradstock, R. and Cohn, J. (2002). Fire regimes and biodiversity in semi-arid mallee ecosystems., *in* R. Bradstock, J. williams and M. Gill (eds), *Flammable Australia*, Cambridge University Press.

Bradstock, R., Gill, A., Kenny, B. and Scott, J. (1998). Busfire risk at the urban interface estimated from historical weather records: consequences for the use of prescribed fire in the Sydney region of south-eastern Australia, *Journal of Environmental Management* **52**: 259–271.

Brown, A., Hairsine, P. and Freebairn, A. (2005). Tasmanian Land Use Change and Stream Flow (TasLUCaS), *Client Report*, CSIRO Land and Water.

Brown, G. and Grant, C. (2000). Nutrient status of pasture ecosystems established on rehabilitated overburden and topsoil sites in the Hunter Valley, New South Wales, *Australian Journal of Soil Research* **38**: 479–491.

Brown, J. (1972). Hydrologic effects of a bushfire in a catchment in south-eastern New South Wales, *Journal of Hydrology* **15**: 77–96.

Burdon, J. and Chilvers, G. (1977). Preliminary studies on a native Australian Eucalypt forest invaded by exotic species, *Oecologia* **31**: 1–12.

Burrows, N., Gardiner, G., Ward, B. and Robinson, A. (1990). Regeneration of *Eucalyptus wando* following fire, *Australian Forestry* **53**: 248–258.

Burrows, N., Woods, Y., Ward, B. and Robinson, A. (1989). Prescribing low intensity fire to kill wildlings in Pinus radiata plantations in Western Australia, *Australian Forestry* **52**: 45–52.

Byram, G. M. (1959). Combustion of forest fuels, *in* K. P. Davis (ed.), *Forest fire: control and use*, McGraw-Hill.

Caitcheon, G., Donnelly, T., Wallbrink, P. and Murray, A. (1995). Nutrient and sediment sources in chaffey reservoir catchment, *Australian journal of soil and water conservation* **8**: 41–49.

Cary, G. J. (2002). Importance of a changing climate for fire regimes in australia., *in* R. Bradstock, J. williams and M. Gill (eds), *Flammable Australia*, Cambridge University Press.

Catchpole, W. (2002). Fire properties and burn patterns in heterogenous landscapes., *in* R. Bradstock, J. williams and M. Gill (eds), *Flammable Australia*, Cambridge University Press.

Catling, P. and Burt, R. (1997). Studies of ground-dwelling mammals of Eucalypt forests in north-eastern New South Wales: the species, their abundance and distribution, *Wildlife Research* **24**: 1–19.

Catling, P., Burt, R. and Forrester, R. (1998). Models of the distribution and abundance of ground-dwelling mammals in the eucalypt forests of south-eastern New South Wales, *Wildlife Research* **25**: 449–466.

Catling, P., Burt, R. and Forrester, R. (2000). Models of the distribution and abundance of ground-dwelling mammals in the eucalypt forests of north-eastern New South Wales in relation to habitat variables, *Wildlife Research* **27**: 639–654.

Catling, P. C., Coops, N. and Burt, R. (2001). The distribution and abundance of ground-dwelling mammals in relation to time since wildfire and vegetation structure in south-eastern Australia, *Wildlife Research* **28**: 55–564.

Catling, P. and Newsome, A. (1981). Responses of the australian vertebrate fauna to fire: an evolutionary approach, *in* A. M. Gill, R. Groves and I. Noble (eds), *Fire and the Australian biota*, Australian Academy of Science.

Chafer, C., Noonan, M. and Macnaught, E. (2004). The post-fire measurement of severity and intensity in the Christmas 2001 Sydney wildfires, *International Journal of Wildland Fire* **13**: 227–240.

Chaffey, C. and Grant, C. (2000). Fire management implications of fuel loads and vegetation structure in rehabilitated sand mines near Newcastle, Australia, *Forest Ecology and Management* **129**: 269–278.

Cheney, N. P. (1981). Fire behaviour., *in* A. M. Gill, R. Groves and I. Noble (eds), *Fire and the Australian biota*, Australian Academy of Science.

Cheney, P. and Sullivan, A. (1997). *Grassfires: fuel, weather and fire behaviour*, CSIRO Publishing.

Constantini, A. and Loch, R. (2002). Effects of site preparation on runoff, erosion, and nutrient losses from *Pinus* plantations established on the coastal lowlands of south-east queensland, *Australian Journal of Soil Research* **40**: 1287–1302.

Cornish, P. and Vertessy, R. (2001). Forest age induced changes in evapotranspiration and water yield in a eucalypt forest, *Journal of Hydrology* **242**: 43–63.

Costin, A. B. (1980). Runoff and soil and nutrient losses from an improved pasture at Ginninderra, Southern Tablelands, New South Wales, *Australian Journal of Agricultural Research* **31**: 533–546.

Cox, J. W. and Pitman, A. (2001). Chemical concentrations of overland flow and throughflow from pastures on sloping texture-contrast soils, *Australian journal of agricultural research* **51**: 211–220.

Croke, J. (2004). Forest harvesting activities and water quality: a new approach to an old problem, *in* J. Croke, S. Mockler and I. Takken (eds), *Forestry management workshop proceedings, Canberra 23-25 March 2004*, Cooperative research centre for catchment hydrology.

Croke, J. and Hairsine, P. (in press). Sediment delivery in managed forests: A review, *Environmental Reviews* .

Croke, J., Hairsine, P. and Fogarty, P. (1999). Sediment transport, redistribution and storage on logged forest hillslopes in south-eastern Australia, *Hydrological Processees* **13**: 2705–2720.

Croke, J. and Takken, S. M. I. (2005). Soil Erosion Control and Mitigation on Forest Roads: Post-Bushfire Road Rehabilitation in the ACT Forest Estate, *Draft Final Report submitted to ACT Forests by the Forest Road Research Group*, University of New South Wales.

Croke, J., Wallbrink, P., Fogarty, P., Hairsine, P., Mockler, S., McCormack, B. and Brophy, J. (1999). Managing sediment sources and movement in forests: the forest industry and water quality, *Industry Report 99/11*, Cooperative research centre for catchment hydrology.

CSIRO (2001). Climate change projections for Australia, *Volume 1*, Division of Atmospheric Research.

Cullen, P. (1991). Land use and declining water quality., *Australian journal of soil and water conservation* **4**: 4–8.

Cullen, P. and Rosich, R. (1979). Effects of rural and urban sources of phosphorus of Lake Burley Griffin, *Progress in Water Technology* **11**: 219–230.

Davidson, P. (1976). Birds in pine forests, *Technical report*, Department of the Capital Territory.

Davis, E. A. (1993). Chaparral control in mosaic pattern increased stream flow and mitigate nitrate loss in Arizona, *Water Resources Bulletin* **29**: 391–399.

DeBano, L. F. (2000). The role of fire and soil heating on water repellency in wildland environments: a review, *Journal of Hydrology* **231-231**: 195–206.

Diaz-Fierros, F., Benito Rueda, E. and Perez Moreira, P. (1987). Evaluation of the U.S.L.E. for the prediction of erosion in burnt forest areas in Galicia (N.W. Spain), *Catena* **14**: 189–199.

Dickinson, K. and Kirkpatrick, J. (1985). The flammability and energy content of some important plant species and fuel components in the forests of southeastern Tasmania, *Journal of Biogeography* **12**: 121–134.

Dovers, S., Hutchinson, M., Lindenmayer, D., Manning, A., Mills, F., Perkins, P., Sharples, J. and White, I. (2005). Uncertainty and the environment, *Paper for Uncertainty Symposium, Goolabri 12-14 April*, Centre for Resource and Environmental Studies.

Eyles, R. J. (1977). Changes in drainge networks since 1820, Southern Tablelands, N.S.W., *Australian Geographer* **13**: 377–387.

Fernandes, P. and Botelho, H. (2003). A review of prescribed burning effectiveness in fire hazard reduction, *International Journal of Wildland Fire* **12**: 117–128.

Fernandes, P. and Botelho, H. (2004). Analysis of the prescribed burning practice in the pine forest of northwestern Portugal, *Journal of Environmental Management* **70**: 15–26.

Fischer, J. and Lindenmayer, D. (2002). Small patches can be valuable for biodiversity conservation: two case studies on birds in southeastern Australia, *Biological conservation* **106**: 129–136.

Fischer, J., Lindenmayer, D., Barry, S. and Flowers, E. (2005). Lizard distribution patterns in the Tumut fragmentation “Natural Experiment” in south-eastern Australia, *Biological conservation* **123**: 301–315.

Flemming, N. and Cox, J. (2001). Carbon and phosphorus losses from dairy pasture in South Australia, *Australian journal of soil research* **39**: 696–978.

Foster, M. B. (1979). *The ecology of Pinus Radiata D. Don invading dry schlerophyll Eucalypt forest in the A.C.T.*, Doctor of philosophy, Australian National University.

Fox, B., Fox, M. and McKay, G. (1979). Litter accumulation after fire in a Eucalypt forest, *Australian Journal of Botany* **27**: 157–165.

Freudenberger, D., Nicholls, N., Wilson, S. and Gold, L. (2004). Landscapes under fire: The survival of planted vegetatioin following the 2003 canberra fire storm, *Report to greening australia*, CSIRO Sustainable Ecosystems.

Gardner, J. (2001). Rehabilitating mines to meet land use objectives: bauxite mining in , *Unasylva* **207**: 3–8.

Gill, A. M. and Williams, J. (1996). Fire regimes and biodiversity: the effects of fragmentation of southeastern Australian eucalypt forests by urbanisation, agriculture and pine plantations, *Forest Ecology and Management* **85**: 261–278.

Gill, A., Moore, P., Donnelly, J. and Moore, A. (1989). Fire hazard assessement of farms and forests near Canberra, ACT, Australia, *Integrating research on hazards in fire-prone environments*, United States man and the biosphere program, pp. 53–57.

Gillespie, G. (2002). Impacts of sediment loads, tadpole density, and food type on the growth and development of tadpoles of the spotted tree frog *Litoria spenceri*: an in-stream experiment, *Biological conservation* **106**: 141–150.

Gillespie, R., Prosser, I. P., Dlugokencky, E., Sparks, R. J., Wallace, G. and Chappell, J. M. A. (1992). AMS dating of alluvial sediments on the Southern Tablelands of New South Wales, Australia, *Radiocarbon* **34**: 29–36.

Govinnage-Wijesekera, D., White, I., a. Wade and Mueller, N. (2005). Turbidity and contaminant loads in bendora reservoir, cotter river, act: relationship to fire and rainfall, *Preliminary report to actewagl*, Centre for Resource and Environmental Studies, Australian National University.

Grant, C. D., Campbell, C. J. and Charnock, N. R. (2002). Selection of species suitable

for derelict mine site rehabilitation in New South Wales, Australia, *Water, Air, and Soil Pollution* **139**: 215–235.

Grant, C. and Koch, J. (2003). Orchid species succession in rehabilitated bauxite mines in Western Australia, *Australian Journal of Botany* **51**: 453–457.

Grayson, R., Haydon, S., Jayasuriya, M. and Finlayson, B. (1993). Water quality in mountain ash forests- separating the impacts of roads from those of logging operations., *Journal of Hydrology* **150**: 459–480.

Greening Australia (2001). Bringing birds back: A globebox guide for bird identification and habitat restoration in ACT and SE NSW.

Hairsine, P. (1997). Buffer zones for managing sediment movement in forestry operations, *in* J. Croke and P. Fogarty (eds), *Erosion in Forests: Proceedings of the forest erosion workshop*, number 98/2, Cooperative Research Centre for Catchment Hydrology.

Harris, G. (1994). Nutrient loadings and algal blooms in australian waters - a discussion paper, *Occasional paper series 12/94*, Land and Water Resources Development Corporation.

Harris, G. (2001). Biogeochemistry of nitrogen and phosphorus in Australian catchments, rivers and estuaries: effects of land use and flow regulation and comparisons with global patterns, *Marine and freshwater research* **52**: 139–149.

Hatch, D., Goulding, K. and Murphy, D. (1996). Nitrogen, *Agriculture, hydrology and water quality*, CABI Publishing, chapter 1.

Haygarth, P. M. and Jarvis, S. C. (2002). *Agriculture, hydrology and water quality*, CABI publishers.

Hazell, D. (2003). Frog ecology in modified Australian landscapes: a review, *Wildlife research* **30**: 193–205.

Hennessy, K., Lucas, C., Nicholls, N., Bathols, J., Suppiah, R. and Ricketts, J. (2005). Climate change impacts on fire-weather in south-eastern australia, *Consultancy report*, CSIRO Marine and Atmospheric Research Climate Impacts and Risk Group.

Hirsch, K. and Martell, D. (1996). A review of initial attack fire crew productivity and effectiveness, *International Journal of Wildland Fire* **64**: 199–215.

Hobbs, R. (2002). Fire regimes and their effects in australian temperate woodlands., *in* R. Bradstock, J. williams and M. Gill (eds), *Flammable Australia*, Cambridge University Press.

Holland, E., Braswell, B., Lamarque, J., Townsend, A., Sulzman, J., Müller, J., Dentener, F., Brassuer, G., Levy II, H., Penner, J. and Roelofs, G. (1997). Variations in the predicted spatial distribution of atmospheric nitrogen deposition and their impact on carbon uptake by terrestrial ecosystems, *Journal of Geophysical Research* **102**: 15849–15866.

Huxman, T., Wilcox, B., Breshears, D., Scott, R., Snyder, K., Small, E., Hultine, K., Pockman, W. and Jackson, R. (2005). Ecohydrological implications of woody plant encroachment, *Ecology* **86**: 308–319.

Kanowski, P. (2005). Revegetation for a diversity of community objectives: why some pines are being replanted in the cotter catchment, Canberra Times.

Kuczera, G. (1987). Prediction of streamflow reductions following bushfire in ash-mixed species eucalypt forest, *Journal of Hydrology* **94**: 215–236.

LaMarche, J. and Lettenmaier, D. P. (2001). Effects of forest roads on flood flows in the Deschutes River Basin, Washington, *Earth Surface Processes and Landforms* **26**: 115–134.

Leitch, C. J., Flinn, D. and van de Graaff, R. (1983). Erosion and nutrient loss resulting from Ash Wednesday (February 1983) wildfires: a case study, *Australian Forestry* **46**: 173–180.

Lindenmayer, D. (2000). Islands of bush in a sea of pines: A summary of studies from the tumut fragmentation experiment, *Research Report 6/00*, Land and Water Resources Research and Development Corporation.

Lindenmayer, D. B. and McCarthy, M. A. (2001). The spatial distribution of non-native plant invaders in a pineeucalypt landscape mosaic in south-eastern Australia, *Biological conservation* **102**: 77–87.

Lindenmayer, D., Cunningham, R. and Pope, M. (1999). A large-scale "experiment" to examine the effects of landscape context and habitat fragmentation on mammals, *Biological conservation* **88**: 387–403.

Lindenmayer, D., McCarthy, M., Parris, K. and Pope, M. (2000). Habitat fragmentation, landscape context and mammalian assemblages in southeastern Australia, *Journal of Mammalogy* **81**: 787–797.

Lindenmayer, D., McIntyre, S. and Fischer, J. (2003). Birds in eucalypt and pine forests: landscape alteration and its implications for research models of faunal habitat use, *Biological conservation* **110**: 45–53.

Lindenmayer, D., Norton, T. and Tanton, M. (1990). Differences between wildfire and clearfelling on the structure of montane ash forests of Victoria and the implications for fauna dependent upon tree hollows, *Australian Forestry* **53**: 61–68.

Loch, R. (2000). Effects of vegetation cover on runoff and erosion under simulated rain and overland flow on a rehabilitated site on the Meandu Mine, Tarong, Queensland, *Australian Journal of Soil Research* **38**: 299–312.

Lu, H., Prosser, I., Moran, C., Gallant, J., Priestley, G. and Stevenson, J. (2003). Predicting sheetwash and rill erosion over the Australian continent, *Australian Journal of Soil Research* **41**: 1–26.

Ludwig, J. A., Hindley, N. and Barnett, G. (2003). Indicators for monitoring minesite rehabilitation: trends on waste-rock dumps, northern Australia, *Ecological Indicators* **3**: 143–153.

Ludwig, J., Hodgkinson, K. and Macadam, R. (1990). Principles, problems and priorities for restoring degraded rangelands, *Australian Rangeland Journal* **12**: 30–3.

Luke, R. H. and McArthur, A. (1978). *Bushfires in Australia*, Australian Government Publishing Service.

Lunt, I. (1997). Effects if long-term vegetation management on remnant grassy forests and anthropogenic native grasslands in south-eastern Australia, *Biological conservation* **81**: 287–297.

Lunt, I. D. (1990). Species-area curves and growth-form spectra for some herb-rich woodlands in western Victora, Australia, *Australian Journal of Ecology* **15**: 155–161.

Lunt, I. and Morgan, J. (2002). The role of fire regimes in temperate lowland grasslands of southeastern australia., *in* R. Bradstock, J. williams and M. Gill (eds), *Flammable Australia*, Cambridge University Press.

Major, R. E., Christie, F. J., Gowing, G., Cassis, G. and Reid, C. A. (2003). The effect of habitat configuration on arboreal insects in fragmented woodlands of south-eastern Australia, *Biological conservation* **113**: 35–48.

Major, R. E., Smith, D., Cassis, G., Gray, M. and Colgan, D. J. (1999). Are roadside strips important reservoirs of invertebrate diversity? A comparison of the ant and beetle faunas of roadside strips and large remnant woodlands, *Australian journal of zoology* **47**: 611–624.

Mark, A. F. (1994). Effects of burning and grazing on sustainable utilisation of upland snow tussock (chionochloa spp) rangelands for pastoralism in South island New Zealand., *Australian Journal of Botany* **42**: 149–161.

Marston, F., Young, W. and Davis, R. (1995). Catchment management support system nutrient generation rates data book, *Technical report*, CSIRO Land and Water.

McCarthy, G. and Tolhurst, K. (2001). Fire management: effectiveness of broadscale fuel reduction burning in assisting with wildfire control in parks and forests in Victoria, *Research report 51*, Forest Science Centre, Orbost.

McKergow, L. A., Prosser, I. P., Hughes, A. O. and Brodie, J. (2005). Sources of sediment to the Great Barrier Reef World Heritage Area, *Marine Pollution Bulletin* **51**: 200–211.

McKergow, L. A., Weaver, D. M., Prosser, I. P., Grayson, R. B. and Reed, A. E. (2003). Before and after riparian management: sediment and nutrient exports from a small agricultural catchment, Western Australia, *Journal of Hydrology* **270**: 253–270.

Moliere, D., Saynor, M. and Evans, K. (2005). Suspended sediment concentration-turbidity relationships for Ngarradj – a seasonal stream in the wet-dry tropics., *Australian journal of water resources* **9**: 37–47.

Morgan, J. W. and Lunt, I. D. (1999). Eects of time-since-Rre on the tussock dynamics of a dominant grass (*Themeda triandra*) in a temperate Australian grassland, *Biological Conservation* **88**: 379–386.

Motha, J. A., Wallbrink, P., Hairsine, P. and Grayson, R. (2003). Harvested areas and unsealed roads as sources of suspended sediment in a forested water supply catchment in south-eastern Australia, *Water Resources Research* **39**: 1056, doi:10.1029/2001WR000794.

National Health and Medical Research Council (2004). Australian drinking water guidelines, *National water quality management strategy*.

Nichols, O. G. and Nichols, F. M. (2003). Long-term trends in faunal recolonization after bauxite mining in the jarrah forest of Southwestern Australia, *Restoration Ecology* **11**: 261–272.

Nisbett, T. (2001). The role of forest management in controlling diffuse pollution in UK forestry, *Forest Ecology and Management* **143**: 215–226.

Olley, J. and Scott, A. (2002). Sediment supply and transport in the Murrumbidgee and Namoi Rivers since European settlement, *Technical Report 9/02*, CSIRO Land and Water.

Olley, J. and Wasson, R. (2003). Changes in the flux of sediment in the Upper Murrumbidgee catchment, southeastern Australia, since European settlement, *Hydrological Processes* **17**: 3307–3320.

Palis, R., Okwach, G., Rose, C. and Saffigna, P. (1990). Soil erosion processes and nutrient loss. 1. The interpretation of enrichment ratio and nitrogen loss in runoff sediment, *Australian Journal of Soil Research* **28**: 623–639.

Parris, K. and Lindenmayer, D. (2004). Evidence that creation of a *Pinus radiata* plantation in south-eastern Australia has reduced habitat for frogs, *Acta Oecologica* **25**: 93–101.

Pettersson, K. (1998). Mechanisms for internal loading of phosphorus in lakes, *Hydrobiologia* **373-374**: 21–25.

Pierson, F., Robichaud, P. and Spaeth, K. (2001). Spatial and temporal effects of wildfire on the hydrology of a steep rangeland watershed, *Hydrological Processes* **15**: 2905–2916.

Pittock, A. B. (2005). *Climate change: turning up the heat*, CSIRO Publishing.

Prosser, I. P. (1991). A comparison of past and present episodes of gully erosion at Wangrah Creek, southern tablelands, New South Wales, *Australian Geographical Studies* **29**: 139–154.

Prosser, I. P. (1996). Thresholds of channel initiation in historical and Holocene times, southeastern Australia, *in* M. G. Anderson and S. M. Brooks (eds), *Advances in hillslope processes*, John Wiley and Sons, New York, chapter 31, pp. 687–708.

Prosser, I. P., Chappell, J. and Gillespie, R. (1994). Holocene valley aggradation and gully erosion in headwater catchments, south eastern highlands of Australia, *Earth Surface Processs and Landforms* **19**: 465–480.

Prosser, I. P. and Williams, L. (1998). The effect of wildfire on runoff and erosion in native Eucalyptus forest, *Hydrological Processes* **12**: 251–265.

Prosser, I., Rustomji, P., Young, W., Moran, C. and Hughes, A. (2001). Constructing River Basin Sediment Budgets for the National Land and Water Resources Audit, *Technical Report 15/01*, CSIRO Land and Water.

Purdie, R. W. and Slatyer, R. O. (1976). Vegetation succession after fire in sclerophyll woodland communities in south-eastern Australia, *Australian Journal of Ecology* **1**: 223–236.

Quinn, J. M., Boothroyd, I. K. and Smith, B. J. (2004). Riparian buffers mitigate effects of pine plantation logging on New Zealand streams 2. Invertebrate communities, *Forest Ecology and Management* **191**: 129–146.

Radford, J. Q., Bennett, A. F. and Cheers, G. J. (2005). Landscape-level thresholds of habitat cover for woodland-dependent birds, *Biological conservation* **124**: 317–337.

Raison, R. J., Woods, P. V. and Khanna, P. (1986). Decomposition and accumulation of litter after fire in sub-alpine eucalypt forests, *Australian Journal of Ecology* **11**: 9–19.

Reid, I. M. and Dunne, T. (1984). Sediment production from road surfaces., *Water Resources Research* **20**: 1753–1761.

Reyes, O. and Casal, M. (2002). Effect of high temperatures on cone opening and on the release and viability of *Pinus pinaster* and *P. radiata* seeds in NW Spain, *Annales des Sciences Forestieres* **59**: 327–334.

Roberts, S., Vertessy, R. and Grayson, R. (2001). Transpiration from Eucalyptus sieberi (L. Johnson) forests of different age, *Forest Ecology and Management* **143**: 153–161.

Rossiter, N. A., Setterfield, S. A., Douglas, M. M. and Hutley, L. B. (2003). Testing the grass-fire cycle: alien grass invasion in the tropical savannas of northern Australia, *Diversity and Distributions* **9**: 169–176.

Schnell, M. R., Pik, A. J. and Dangerfield, J. M. (2003). Ant community succession within eucalypt plantations on used pasture and implications for taxonomic sufficiency in biomonitoring, *Austral Ecology* **28**: 553–565.

Scott, A. (2001). Water erosion in the Murray-Darling Basin: Learning from the past, *Technical Report 43/01*, CSIRO Land and Water.

Scott, D. and Van Wyk, D. (1990). The effects of wildfire on soil wetability and hydrological behaviour of an afforested catchment., *Journal of Hydrology* **121**: 239–256.

Select committee into the recent Australian bushfires (2003). *A nation charred: report on the inquiry into bushfires*, The Parliament of the Commonwealth of Australia.

Sinclair, J. and New, T. (2004). Pine plantations in south eastern Australia support highly impoverished ant assemblages (Hymenoptera: Formicidae), *Journal of Insect Conservation* **8**: 277–286.

Sinclair Knight Mertz (2004). Bushfire Recovery Program. The Impact of the 2003 Alpine Bushfires on Streamflow. Task 1 - Broadscale assessment, *Consulting report to Department of Sustainability and Environment, Victoria.*

Singh, G., Kershaw, A. P. and Clark, R. (1981). Quaternary vegetation and fire history in australia., *in* A. M. Gill, R. Groves and I. Noble (eds), *Fire and the Australian biota*, Australian Academy of Science.

Smith, A. and Lindenmayer, D. (1988). Tree hollow requirements of Leadbeater's possum and other possums and gliders in timber production ash forests of the Victorian Central Highlands, *Australian Wildlife Research* **15**: 347–362.

Smyth, A., MacNally, R. and Lamb, D. (2002). Comparative Influence of Forest Management and Habitat Structural Factors on the Abundances of Hollow-Nesting Bird Species in Subtropical Australian Eucalypt Forest, *Environmental Management* **30**: 547–559.

Specht, R. (1970). Vegetation, *in* G. Leeper (ed.), *Australian Environment*, CSIRO and Melbourne University Press.

Starr, B. (2004). A landscape stability trend analysis of the upper Cotter River catchment, *Report prepared for ActewAGL*, Barry Starr Pty Ltd.

Stevens, D. P., Cox, J. W. and Chittleborough, D. J. (1999). Pathways of phosphorus, nitrogen, and carbon movement over and through texturally dierentiated soils, South Australia, *Australian Journal of Soil Research* **37**: 679–693.

Takken, I. and Croke, J. (2004). Road to stream connectivity in forestry catchments, *in* J. Croke, S. Mockler and I. Takken (eds), *Forestry management workshop proceedings, Canberra 23-25 March 2004*, Cooperative research centre for catchment hydrology.

Talsma, T. (1983). Soils of the Cotter catchment area, A.C.T.: Distribution, chemical and physical properties, *Australian Journal of Soil Research* **21**: 241–255.

Thrall, P. H., Millsom, D. A., Jeavons, A. C., Waayers, M., Harvey, G. R., Bagnall, D. J. and Brockwell, J. (2005). Seed inoculation with effective root-nodule bacteria enhances revegetation success, *Journal of applied ecology* **42**: 740–751.

Tongway, D. (1990). Soil and landscape processes in the restoration of rangelands, *Australian Rangeland Journal* **12**: 54–7.

van Dijk, A., Hairsine, P. and Dowling, T. (in review). Forestry, water resources and salinity reduction in the Murray Darling Basin: a review of knowledge and spatial planning methods., *Forest Ecology and Management* .

van Loon, A. P. (1977). Bushland fuel quantities in the Blue Mountains: Litter and understorey, *Research note 33*, Forestry Commission of New South Wales.

Vertessy, R. (2003). Vegetation re-growth in the upper catchment post the 2003 bushfires., *in* R. Hardie (ed.), *Preliminary review of selected factors that may change future flow patterns in the River Murray System*, Earth Tech.

Vertessy, R., Hatton, T., Benyon, R. and Dawes, W. (1996). Long term growth and water balance predictions for a mountain ash (*E. regnans*) forest subject to clearfelling regeneration, *Tree Physiology* **16**: 221–232.

Vertessy, R., Watson, F. and O'Sullivan, S. (2001). Factors determining relations be-

tween stand age and catchment water balance in mountain ash forests, *Forest Ecology and Management* **143**: 13–26.

Vertessy, R., Zhang, L. and Dawes, W. (2003). Plantations, river flows and river salinity, *Australian Forestry* **66**: 55–61.

Walker, J. (1981). Fuel dynamics and australian vegetation., *in* A. M. Gill, R. Groves and I. Noble (eds), *Fire and the Australian biota*, Australian Academy of Science.

Walker, J., Raison, R. and Khanne, P. (1986). Fire, *in* J. Russell and R. Isbell (eds), *Australian soils: the human impact*, University of Queensland Press.

Wallbrink, P. J. and Croke, J. (2002). A combined rainfall simulator and tracer approach to assess the role of Best Management Practices in minimising sediment redistribution and loss in forests after harvesting, *Forest ecology and management* **170**: 217–232.

Wallbrink, P. J., Olley, J., Murray, A. and Olive, L. (1996). The contribution of subsoil to sediment yield in the Murrumbidgee River basin, New South Wales, Australia, *Erosion and sediment yield: global and regional perspectives*, IAHS Publication no. 236, pp. 347–355.

Wallbrink, P., Martin, C. and Wilson, C. (2003). Quantifying the contributions of sediment, sediment-P and fertiliser-P from forested, cultivated and pasture area at the landuse and catchment scale using fallout radionuclides and geochemistry, *Soil and tillage research* **69**: 53–68.

Wallbrink, P., Roddy, B. and Olley, J. (2002). A tracer budget quantifying soil redistribution on hillslopes after forest harvesting, *Catena* **47**: 179–201.

Ward, S. (2000). Soil development on rehabilitated bauxite mines in south-west Australia, *Australian Journal of Soil Research* **38**: 453–464.

Wasson, R., Mazari, R., Starr, B. and Clifton, G. (1998). The recent history of erosion and sedimentation on the Southern Tablelands of southeastern Australia: sediment flux dominated by channel incision, *Geomorphology* **24**: 291–308.

Wasson, R., Worthy, M., Olley, J. and Mueller, N. (2004). Sources of turbidity in Bendora Reservoir, *Report prepared for ActewAGL*, Centre for Resource and Environmental Studies.

Watson, D., Freudenberger, D. and Paull, D. (2001). An assessment of the focal-species approach for conserving birds in variegated landscapes in southeastern Australia, *Conservation Biology* **15**: 1364–1373.

Watson, F., Vertessy, R. and Grayson, R. (1999). Large scale modelling of forest hydrological processes and their long term effect on streamflow, *Hydrological Processes* **13**: 689–700.

Webster, I. T., Ford, P. W. and Hancock, G. (2001). Phosphorus dynamics in Australian lowland rivers, *Marine and Freshwater Research* **52**: 127–37.

Whetton, P. H., Katzfey, J. J., Hennessy, K. J., Wu, X., McGregor, J. L. and Nguyen, K. (2001). Developing scenarios of climate change for Southeastern Australia: an example using regional climate model output, *Climate Research* **16**: 181–201.

White, I. (2005). Water quality in the cotter catchment, Canberra Times.

White, I., Wade, A., Worthy, M., Mueller, N., Daniell, T. and Wasson, R. (submitted). The vulnerability of water supply catchments to bushfires: impacts of the January 2003 wildfires on the Australian Capital Territory, *Australian Journal of Water Resources* .

Williams, R., Griffiths, A. and Allen, G. (2002). Fire regimes and biodiversity in the savannas of northern australia., *in* R. Bradstock, J. williams and M. Gill (eds), *Flammable Australia*, Cambridge University Press.

Williamson, T., Newman, B., Graham, R., P. and Shouse (2004). Regolith water in zero-order chaparral and perennial grass watersheds four decades after vegetation conversion, *Vadose Zone Journal* **3**: 1007–1016.

Wilson, C. J. (1999). Effects of logging and fire on runoff and erosion on highly erodible granitic soils in Tasmania, *Water Resources Research* **35**: 3531–3546.

Windsor, D. and Clemments, A. (2001). A germination and establishment field trial of *Themeda australis* for mine site rehabilitation in the Central Tablelands of New South Wales, *Restoration Ecology* **9**: 104–110.

Yates, C. and Hobbs, R. (1997). Temperate Eucalypt Woodlands: a Review of Their Status, Processes Threatening Their Persistence and Techniques for Restoration, *Australian Journal of Botany* **45**: 949–973.

Young, W., Marston, F. and Davis, J. R. (1996). Nutrient exports and land use in Australian catchments, *Journal of environmental management* **47**: 165–183.

Zhang, L., Dawes, W. and Walker, G. (2001). The Response of mean annual evapotranspiration to vegetation changes at catchment scale, *Water Resources Research* **37**: 701–708.

Zhang, L., Hickel, K., Dawes, W., Chiew, F., Western, A. and Briggs, P. (2004). A rational function approach for estimating mean annual evapotranspiration, *Water Resources Research* **40**: W02502.

Ziegler, A. D., Sutherland, R. A. and Giambelluca, T. W. (2001). Runoff generation and sediment production on unpaved roads, footpaths and agricultural land surfaces in northern Thailand, *Earth Surface Processes and Landforms* **25**: 519–534.

Zierholz, C. and Hairsine, P. (1995). Runoff and soil erosion in the bushland following the Sydney bushfires, *Australian journal of soil and water conservation* **8**: 28–37.

Zierholz, C., Prosser, I., Fogarty, P. and Rustomji, P. (2001). In-stream wetlands and their significance for channel filling and the catchment sediment budget, Jugiong Creek, New South Wales, *Geomorphology* **38**: 221–235.

A Project Brief

TERMS OF REFERENCE FOR PROJECT BRIEF (30 AUGUST 2005)

Scientific Research Paper

Revegetation of Water Supply Catchments

1. Collate and synthesise local, national and international data and literature relating to the revegetation of water supply catchments relevant to ACT catchments after fire with an emphasis on:

    - natural vegetation of similar or different genetic material;
    - radiata pine or similar species
    - revegetation from pine or exotic species back to native vegetation
    - natural re-establishment versus debris clearing, then site preparation and then replanting; and
    - grassland (native and/or exotic)

2. Identify literature supporting contentions of the effectiveness of the various revegetation types for addressing the following natural resource and land management factors:

    - water yield
    - water quality
    - controlling sediment yield to streams
    - biodiversity
    - bushfire management implications
    - weed control
    - road network requirements and whether they vary for different vegetation types
    - ongoing maintenance
    - fire sensitivity and regenerative capacity after fire

- The study should particularly focus on studies conducted in a landscape that is similar to the Lower Cotter Catchment.

3. Prepare a report that includes:

    - an analysis of the information and a summary of key issues addressed in scientific literature
    - an assessment of the relativities of different vegetation types for the range of factors identified above (in reference 2)
    - identifies sensitivities and treatments for slope and soil types in relation to erosion control; and
    - an assessment of the interactions between all of the factors identified above (reference 2)

4. The consultant will be required to:

    - consult with appropriate experts including from ANU, UC, ACT and Commonwealth Departments and Agencies and ACTEW to obtain data and information;
    - provide a copy of the report for peer review; and
    - finalise the report by the end of October 2005