

Review of rural tree decline in a changing Australian climate

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Abstract

Rural tree decline (RTD) in Australia is a complex biological issue involving dieback of native trees following the removal of native vegetation, the introduction of improved pasture species, soil fertilisation and the grazing of livestock. The extent and severity of RTD has reached historically high levels in the past two to three decades. As this period has coincided with an unprecedented increase in atmospheric carbon dioxide concentration, it may be speculated that climate change, a result of the Greenhouse Effect, has played a significant role in RTD. Ecological similarities between two geographically and climatically distinct Australian case studies (the Midlands of Tasmania and the New England Tablelands of New South Wales), both characterised by severe RTD, suggest that the underlying causes of this phenomenon have a common basis.

The biological and ecological effects of rural land use are manifold. These include exposure of remnant trees to increased wind and soil drying; decreased water availability due to competition with improved pasture and decreased infiltration due to soil compaction by cloven-hooved livestock; increased salinity; loss of ectomycorrhizal fungal diversity; changed nutrient balance due to direct addition of fertiliser and/or nitrogen fixation by clover-based pastures; changed pest/predator and biodiversity balance; and lack of seedling recruitment due to browsing by livestock and/or

native animals. Superimposed on these factors are below-average rainfall and above-average temperatures over the past three decades, phenomena that are associated with climate change. Climate change has been linked to eucalypt decline in landscapes with little or no rural land use, and to tree decline across Europe. The contributions of climate change and rural land use to RTD are considered in this review and recommendations for management are discussed.

Introduction

Rural tree decline (RTD) and dieback in Australia have long been observed (Norton 1886) but in the past three decades have reached historically high levels (Williams and Nadolny 1981; Rees 2000). RTD has been particularly well documented for two agricultural regions, the New England Tablelands of New South Wales (Williams and Nadolny 1981; Nadolny 1984; Curtis 1989, 1991a, b; Jones *et al.* 1990) and the Midlands of Tasmania (Grice 1995; Neyland 1996; Kirkpatrick *et al.* 2000; Doyle 2001), where RTD is particularly severe (Photo 1). Rural tree decline has also been documented in Queensland (see Wylie *et al.* 1993 for a review; Fensham and Holman 1999) and Western Australia (Wills *et al.* 2001), although the severity of RTD in the Tablelands and the Midlands may indicate a future, more widespread phenomenon as both are at high altitude relative to the regions studied in Queensland and Western

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Photo 1. Dead and dying paddock trees with no evident eucalypt regeneration in the Midlands of Tasmania.

Australia, and the Greenhouse Effect has the largest consequences for higher altitude plant communities (Fanta 1992).

Rural tree decline is associated with serious environmental and economic costs. Trees in agricultural regions provide shelter to livestock, crops and pastures, prevent soil erosion and salinisation, conserve endemic species and provide habitat for native flora and fauna. It has been estimated that it would cost in excess of \$20 billion to replace trees (assuming successful revegetation; see Close and Davidson 2003a, b) currently at risk of RTD in agricultural regions of Australia alone (Reid and Landsberg 2000).

Rural tree decline in *Eucalyptus* species is characterised by a thinning of the crown that begins at branch ends and progresses towards the trunk. Dead branches typically protrude beyond the remaining foliage (Photo 2). New leaf development subsequently occurs from epicormic buds and may re-occur for years before eventual death. Factors affecting RTD are generally thought to affect the water relations of trees. For example, the recent decline in the

Eucalyptus gunnii alpine woodlands in central Tasmania has been attributed to below-average rainfall over the last two decades (Potts *et al.* 2001a). The decline of eucalypts in regenerating, even-aged forests of New South Wales has occurred in response to severe competition for water ('self-thinning') (Glasby *et al.* 1987) and RTD in the Western Australian wheatbelt has been attributed to salinity stress (Greenwood *et al.* 1995). The overall objective of this review is to consider climate change as a factor in RTD; research conducted in two relatively well-studied cases, the Midlands of Tasmania and the New England Tablelands of New South Wales, will be reviewed. Furthermore, considering the biological effects of rural land use, recommendations for addressing RTD are suggested.

Climate change and tree decline in natural ecosystems

The global climate has warmed by more than 0.5°C during the past 125 years, and is estimated to rise by up to 4°C by the mid twenty-first century (Gates 1990a).



Photo 2. Early stages of rural tree decline are indicated by dead branches protruding beyond the remaining foliage.

By then, rainfall in many regions will decrease, rainfall patterns are predicted to alter and the incidence of drought to increase (Gates 1990a; Fanta 1992; Körner 1992). High latitudes and altitudes will warm more than lower ones and winters more than summers (Gates 1990a). An increasing rate of climate change in the past three decades has been indicated through tree-ring chronology of the climatically sensitive Huon pine (*Lagarostrobos franklinii*) (Cook *et al.* 1991) growing on the west coast of Tasmania. Movements of forest zone boundaries at a rate of 2.5 km/yr

are predicted, which is nearly ten times the known natural rate (Gates 1990b) and will exceed the dispersal abilities of many plant species (Dyer 1995), particularly given artificial barriers such as cultivated agricultural lands and monospecific coniferous forests (Fanta 1992).

Pollen records indicate that a temperature increase of as little as 1°C over 100 to 200 years, coupled with intense agricultural land use, led to large-scale forest decline on the dry and poor sandy soils of the north-western European lowlands (Lamb 1984).

Hughes *et al.* (1996) calculated the mean annual temperature range and mean annual rainfall ratio (the ratio of the highest to lowest rainfall per annum) of 819 eucalypt species Australia-wide using the GUMNUT database that contains over 44 000 point-location records for the species. Twenty-five per cent of the *Eucalyptus* species investigated have a natural temperature range of less than 1°C, 53% less than 3°C and 70% less than 5°C in mean annual temperature (Hughes *et al.* 1996). The natural mean annual rainfall ratio is less than 1.1 for 17%, less than 2.0 for 60% and less than 3.0 for 80% of the eucalypt species investigated (Hughes *et al.* 1996). Note that these data do not indicate that species do not survive outside their exhibited natural ranges, as demonstrated in agricultural or ornamental plantings. But even considering factors additional to rainfall and temperature that confer an ecological niche, such as competition, disturbance, or soil type, the predicted change in future rainfall and temperature patterns has serious implications for *Eucalyptus*. For example, conservative estimates indicate that by 2070, populations at the southern boundary of 53% of eucalypt species are predicted to be exposed to warmer temperatures than any presently tolerated by their most northerly populations (Hughes *et al.* 1996). Furthermore, in forest communities containing eucalypt species that can persist under warmer conditions and changed total and seasonal rainfall patterns, the new conditions may confer competitive advantage to other species (Lowman and Heatwole 1992).

Widespread decline of forests across Europe in the 1980s was considered to be due to drought resulting from the Greenhouse Effect (Pearce 1990). *Fagus grandifolia* (Roberts 1989), *Pinus ponderosa*, *P. taeda* (Krupta and Kickert 1989), *Betula papyrifera*, *B. alleghaniensis*, *Acer saccharum*, *Fraxinus nigra* (Gates 1990a) and *Picea abies* (Fenger *et al.* 1993) are considered vulnerable to the effects of climate change. Fensham and Holman (1999) showed that extensive tree death occurred in *Eucalyptus* savanna in

northern Queensland, following three severe droughts this century. This was particularly severe following recent drought between 1990 and 1997 (Fensham 1998; Fensham and Holman 1999). Further, tree decline in many *Eucalyptus wandoo* woodlands in south-western Western Australia has been attributed to below average rainfall from 1997 to 2000 and the lowest rainfall on record (since 1887) for 2001 (Wills *et al.* 2001). Decline of *E. gunnii* on the high altitude central plateau of Tasmania has been attributed to below-average rainfall over the last two decades (Potts *et al.* 2001a).

Climate change and rural tree decline in Australia

Meteorological data indicate that rainfall in the Midlands of Tasmania (42°18'S, 147°22'E) has been below the long-term average for 20 of the last 25 years (Figure 1) and that the average annual rainfall over the past 25 years has been 492 mm compared to the long-term average of 552 mm. The seasonal pattern of rainfall has also changed, from a more even distribution to a pronounced dry period in late summer-autumn (Kirkpatrick *et al.* 2000). In addition, an increase of 1.5°C in the mean daily maximum temperature occurred over the period 1945–1995 (Kirkpatrick and Gibson 1999). The worst affected areas of RTD in the Midlands are those in areas of lowest rainfall (Grice 1995) (see Figure 2).

Meteorological data indicate that whilst rainfall in the New England Tablelands of New South Wales (30°31'S, 151°40'E) has only been below the long-term average for 15 of the last 25 years (Figure 3), greater deviation below the average has dropped the annual average for the past 25 years to 735 mm compared to the long-term average of 786 mm: the period between 1978 and 1987 particularly contributed to this observation (Figure 3). It has been shown that the worst affected areas of RTD around the New England Tablelands coincide with soils that have relatively low moisture-holding

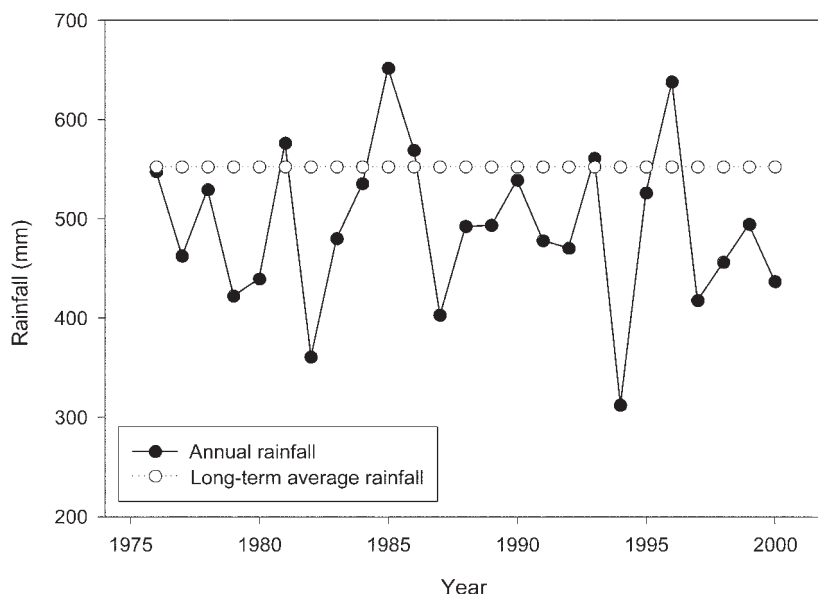


Figure 1. Average and annual rainfall from 1976 to 2000 at the Oatlands Post Office in the Midlands, Tasmania, 42°18'08"S, 147°22'12"E.

capacity (Jones *et al.* 1990). Coincident with decreasing rainfall, average temperatures have risen to 0.4°C above the long-term average for the same 25-year period.

Regional similarities in rural tree decline in New South Wales and Tasmania



Rural tree decline in the New England Tablelands of New South Wales and the Midlands of Tasmania is typified by the decline of isolated trees in pasture or of small patches of trees where the native understorey has been cleared and/or grazed. The severity of RTD is correlated with the extent of native vegetation removal and the degree of pasture improvement (Mackay 1978; Mackay *et al.* 1984; Nadolny 1984; Wylie and Johnston 1984; Landsberg 1988), which affect local water relations. Visually, the New England Tablelands and the Midlands are strikingly similar, with trees in varying stages of RTD scattered throughout improved pastures used for sheep grazing (Photo 1). There are also ecological similarities such as differences in susceptibility of eucalypt species to

RTD, large variation in soil types and precipitation, high levels of specialisation of plant species to slope and soil type, and even similarities of species particularly vulnerable to RTD; for example, *Eucalyptus viminalis*. Both regions are in relatively high altitudes and experience cold winters. Before clearing for agriculture, both regions were mosaics of grassy eucalypt forests and woodlands (Williams 1963; Curtis 1989 for New South Wales; Fensham 1989 for Tasmania). These similarities may suggest common underlying causes of RTD.

Tree decline in natural eucalypt ecosystems occurs in response to severe drought events (Fensham 1998; Fensham and Holman 1999; Potts *et al.* 2001a; Wills *et al.* 2001). Although it is difficult to quantify the relative contributions of the effects of rural land-use management and natural fluctuations in climate on tree decline, Fensham (1998) showed that after severe drought in 1992–1994, there was no effect of cattle- versus macropod-grazed sites on drought-related tree decline and death. Caveats of this finding included (1) that grazing severity was only light to

LEGEND

Tree Decline Class

- Moderate  10–40% of branches are dead within the tree's canopy
- Severe  > 40% of branches are dead within the tree's canopy – trees dead or dying

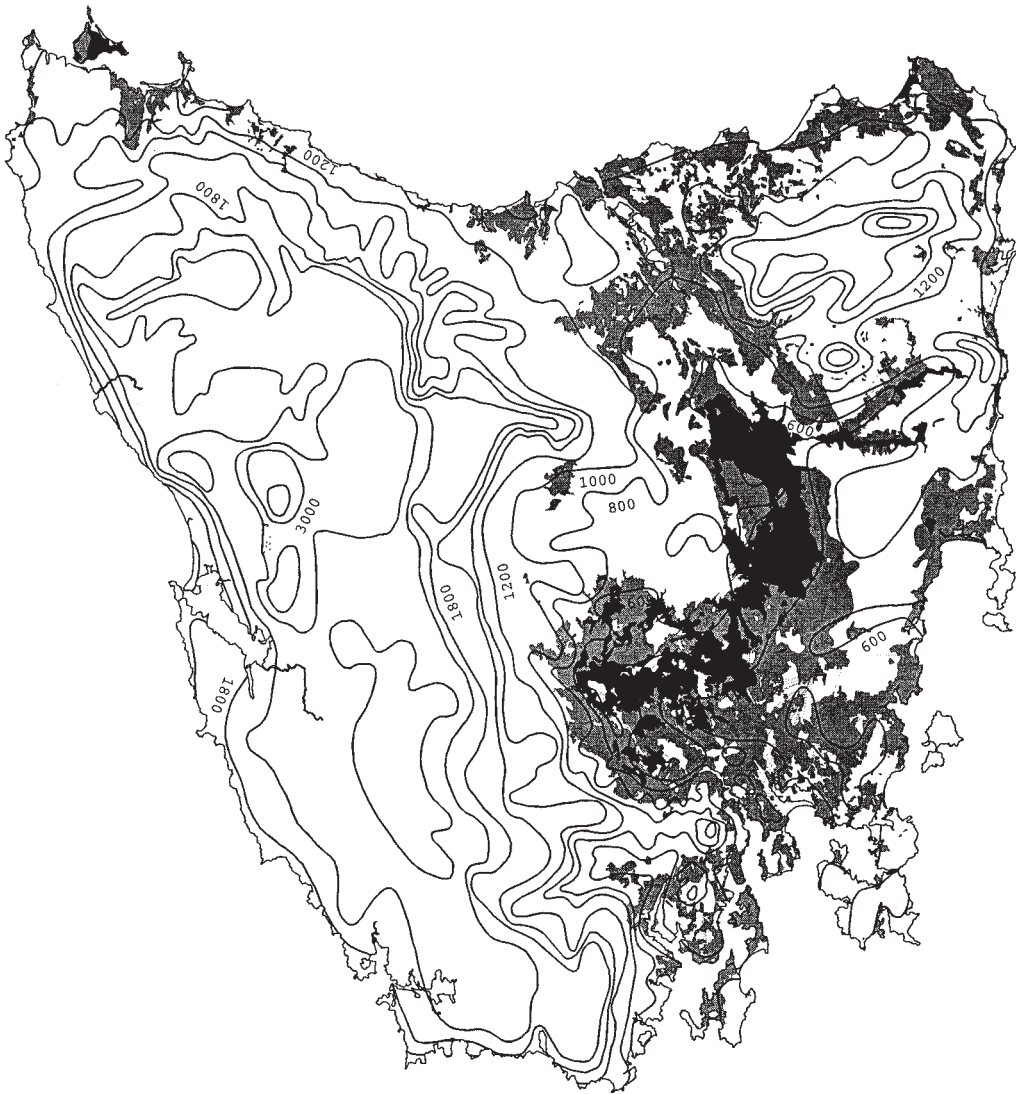


Figure 2. The extent of tree decline in Tasmania as mapped in 1992 (redrawn from Grice 1995).

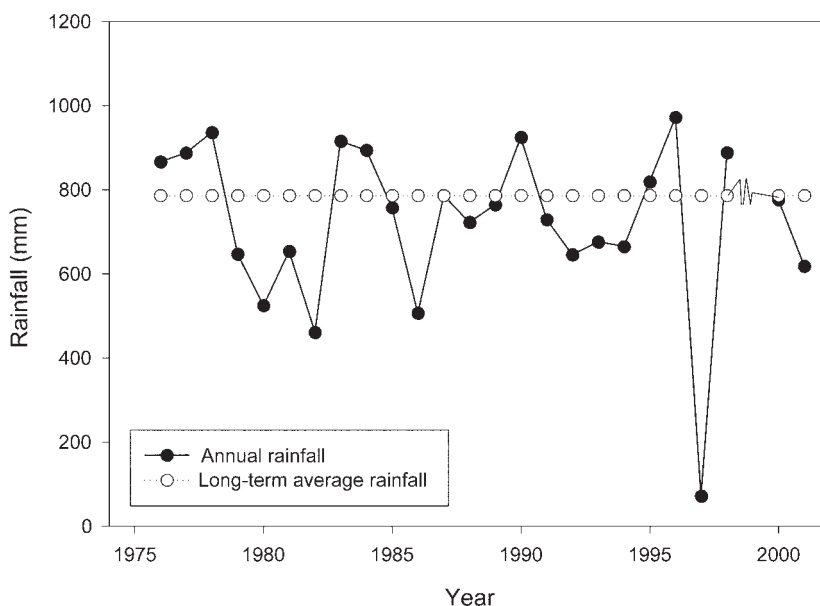


Figure 3. Average and annual rainfall from 1976 to 1996 at the Armidale radio station, New South Wales (30°31'00"S, 151°40'05"E) and from 1997 to 2001 (data not available for 1999) at the Armidale airport (30°31'38"S, 151°36'57"E).

moderate, and (2) that grazing may have a greater effect on drought-related tree decline on soils of lower moisture-holding capacity relative to the well-developed ferrosols of the experimental sites. The low intensity of land use characterised by the study sites of Fensham (1998) compared with the relatively high intensity of land use of the case-study regions of the New England Tablelands and the Midlands of Tasmania investigated in this review are clearly distinct. It might be expected that the greater intensity of land use in the case-study regions may contribute to tree decline above that caused by natural climatic fluctuation. In support of this, Kirkpatrick *et al.* (2000) showed that fenced-off (livestock-excluded) plots suffered less decline than non-fenced plots in the Midlands of Tasmania.

Effects of rural land use

Clearing the majority of native vegetation, and the sowing and fertilising of introduced pasture species induces changes in the

physical, chemical and biological environment. Application of fertilisers leads to vigorous growth of introduced pasture species that very efficiently compete for soil moisture and nutrients (Adams *et al.* 2003). In addition, the pasture-grass cover may prevent moisture reaching deeper parts of the soil profile, particularly during periods of low rainfall or in dry areas, and may form a dense mat of roots that interferes with tree root growth (Fensham and Kirkpatrick 1992). In contrast, native grasslands are composed principally of tussock-forming species, separated by bare ground, that together allow infiltration of rainfall deep into the soil profile. Soil compaction by cloven-hooved sheep impedes infiltration (Willat and Pullar 1983), particularly around trees commonly used by sheep for shelter or shade. Drought-stressed trees have low seed sets (Heatwole and Lowman 1986) and, together with competition by grass for water, and grazing by sheep, this prevents natural regeneration in landscapes affected by RTD. Tree clearing has increased the

incidence and severity of highly saline areas in the Midlands of Tasmania affected by RTD but in higher country (e.g. the Central Plateau of Tasmania) salinity is not associated with RTD.

About 95% of plants worldwide have mycorrhizal symbiotic fungal associations (Tommerup and Bougher 2000) that act to increase plant available nutrients (Grove and Le Tacon 1993) and water (Duddridge *et al.* 1980; Grove and Le Tacon 1993; Garbaye and Churin 1997). Soil conditions that discourage ectomycorrhizal fungi (e.g. loss of topsoil, litter and organic matter: Tommerup and Bougher 2000) and decreased levels of inoculum have been described in areas of tree decline outside Australia (Amaranthus and Trappe 1993). Growth of eucalypt seedlings was inhibited in soils without ectomycorrhizal inoculum and such soils occur beneath trees growing on established pastures (Ellis and Pennington 1992). Consistent with these findings, trees in paddocks have poor ectomycorrhizal fungal development whereas, in the absence of cultivation, stands as small as 0.1–0.3 ha can have significant ectomycorrhizal fungal development (Tommerup and Bougher 2000).

Nutrient input

Increased nutrient input, either through direct fertilisation, N-fixation in clover-based pastures or from stock excrement deposited whilst sheltering under trees, has detrimental effects on ectomycorrhizal fungi (Tommerup and Bougher 2000) as well as increasing tree nutrient uptake and consequently foliar nutrient levels. Average foliar nitrogen and phosphorus content of young foliage of *Eucalyptus blakelyi* was, respectively, 1.88% and 0.21% of leaf dry weight (DW) on pasture sites but 1.44% and 0.16% of DW in healthy forest (Landsberg *et al.* 1990). Granger *et al.* (1994) reported that RTD of *E. ovata* and *E. camphora* at Yellingbo, Victoria, was caused by 'nitrogen-saturation' of the remnant stands which

were surrounded by, and downslope from, vegetable, poultry and pig farms. This has the additional effect of increasing the palatability of foliage to insect and vertebrate pests (such as possums) as well as speeding up the lifecycles of insects, minimising the mortality of pupae and supporting larger populations of insects (Landsberg *et al.* 1990).

Rural regions with and without tree decline

Whilst severe RTD occurs in the New England Tablelands, in the landscape around Tamworth in New South Wales (31°05'S, 150°56'E) where similar rural land use occurs, RTD is absent (Beckman and Davidson 1990). A comparison of their climate data (Table 1: Bureau of Meteorology 2001 website) indicates that, although average rainfall is less around Tamworth than Armidale, average wind speed is lower, average temperatures and relative humidity are higher and there are more clear and fewer cloudy days at Tamworth. All these factors are more severe in the Midlands of Tasmania than the New England Tablelands and Tamworth (note relative humidity data are not available for the Midlands) and rainfall is far lower (Table 1). Differences between these climatic parameters are greater if data for spring only (September–November) are compared (data not shown). In combination, this information indicates that the growing season, in terms of duration and favourability, ranks Tamworth above the New England Tablelands, with the Midlands of Tasmania ranked lowest. The spatial distribution of RTD in Tasmania correlates closely with a region that is in the rain shadow of westerly and easterly weather patterns and is therefore the driest in Tasmania (Figure 2). It can be concluded that the capacity of trees to recover from stress is limited in areas where RTD is observed. It has been observed that dieback is less severe on western slopes in the New England Tablelands of New South

Table 1. Climate averages for Tamworth airport (31°05'20"S, 150°56'80"E, 404 m a.s.l.; recorded since 1876), Armidale Post Office (30°31'00"S, 151°40'05"E, 980 m a.s.l.; recorded since 1857) and Oatlands Post Office (42°18'08"S, 147°22'12"E, 406 m a.s.l.; recorded since 1882).

	Tamworth NSW	Armidale NSW	Oatlands Tasmania
Mean annual rainfall (mm)	670	790	552
Average 9 am temperature (°C)	16.4	13.3	9.7
Average 3 pm temperature (°C)	22.5	18.9	13.9
Average minimum temperature (°C)	10.2	7.1	5.0
Clear days	134	106	33
Cloudy days	90	133	129
Relative humidity (%)	65	47	-
Windspeed 9 am (km/h)	6.9	8.6	10.6
Windspeed 3 pm (km/h)	11.5	11.3	14.2

Wales (Curtis 1991a). During the afternoon, western slopes receive sun that may effectively increase the growing/recovery period of stressed trees.

The causes of rural tree decline

Evidence of tree decline in response to below-average rainfall over recent decades in landscapes where little or no rural land use occurs (Pearce 1990; Grice 1995; Neyland 1996; Fensham and Holman 1999; Potts *et al.* 2001a; Wills *et al.* 2001) implicates climate change as a factor in the increased phenomenon of RTD over the last two to three decades. In addition to the effects of climate, practices associated with rural land use have been associated with RTD. In the early 1960s in the New England Tablelands of New South Wales (Williams and Nadolny 1981), dieback coincided with pasture improvement by aerial application of superphosphate (Heatwole and Lowman 1986). The severe drought of 1964–65 and the moist summers of the early to mid 1970s that supported an explosion of insect populations (Carne *et al.* 1974) catalysed large-scale RTD (Williams and Nadolny 1981). Of particular interest were environmental and tree conditions

that were conducive to recovery. Trees recovering from RTD in the 1979–81 period had greater than 25% foliage remaining at the beginning of the recovery period whereas trees with less than 25% continued to decline despite favourable growing conditions (Nadolny 1984). A correlation between dieback and pre-drought basal area (indicative of site occupancy) suggested that the lack of competitive ability of trees may be a partial cause of dieback (Fensham and Holman 1999). Furthermore, White (1986) concluded that changed rainfall pattern induced water stress in rural trees on the New England Tablelands that increased their susceptibility to the insect attack observed between 1950 and 1980. Likewise, Landsberg and Wylie (1983) and Pook and Forrester (1984) emphasise the role of water stress in initiating the process of RTD. These investigations support the contention that increased RTD over the past two to three decades may be related to climate change. While changes in species distribution have occurred (Potts and Reid 1988), the rapid increase in atmospheric carbon dioxide concentration and the apparent change in climatic conditions over the past two to three decades (Kump *et al.* 2000) may be at rates too rapid for species boundaries to adjust.

Recommendations for addressing rural tree decline

The following recommendations are aimed at minimising, halting, reversing and preventing RTD. These recommendations are for agrosystems where there are isolated remnant trees or remnant patches of bushland, and that combine livestock grazing with regimes of cropping and/or improved pasture in regions which naturally supported grassy woodlands. Such recommendations are particularly relevant to regions such as the New England Tablelands or the Midlands of Tasmania that have experienced climate change over the past two to three decades and are forecast to experience climate change to a greater degree than lower altitude agricultural regions.

1. Retain remnant patches

If vegetation is retained in paddocks for stock shelter, it is imperative to retain remnant patches of woodland rather than isolated trees. Large remnant patches are more resistant to RTD than small patches because there is better retention of understorey and health of soil ecosystems (Tommerup and Bougher 2000). Loss of understorey and tree cover is typical in the New England Tablelands and the Midlands of Tasmania where RTD is severe. Such loss of understorey and tree cover, together with a shift towards homogeneity of the soil ecosystem due to grazing, cultivation and fertiliser application, decreases the diversity of mycorrhizal fungal communities (Tommerup and Bougher 2000). The growth of eucalypts is inhibited in soils without ectomycorrhizal inoculum and such soils occur beneath trees growing in isolation on established grasslands (Ellis and Pennington 1992). Roots growing in soil under such grassland did not develop mycorrhizal associations and this led to the conclusion that the decline of isolated trees in established grasslands was principally caused by lack of ectomycorrhizal inoculum (Ellis and Pennington 1992). It has been hypothesised that this condition causes

the vegetation boundary between forest and grassy woodlands, and grasslands in Tasmania (Fensham and Kirkpatrick 1992). Consistent with these findings, trees in paddocks have poor ectomycorrhizal fungal development whereas stands of trees as small as 0.1–0.3 ha can have ectomycorrhizal fungal development similar to that of native forest (Tommerup and Bougher 2000).

2. Fencing to exclude livestock and browsers

Fencing to exclude livestock is probably the single most important management tool for improving the health of remnant bushland. There are instances across the New England Tablelands and Midlands regions where trees in pastures suffered severe RTD, but trees in adjacent blocks that were fenced to exclude livestock remained healthy (Landsberg and Wylie 1983; Nadolny 1984; Beckman and Davidson 1990; Kirkpatrick *et al.* 2000; Doyle 2001). These observations do not negate a link between RTD and climate change but rather are indicative of the complicated nature of RTD. Fencing allows seedling recruitment of both tree and shrub species, successful coppicing from existing trees in decline, and protects seedlings during regeneration. The establishment and/or maintenance of an understorey in a multi-aged eucalypt grassy woodland increases the diversity of flora and fauna. Successful recruitment of tree seedlings is critical as trees that are old or in decline have relatively low seed set (Landsberg 1988; Lowman and Heatwole 1992). Since the landscape was cleared for farming in the 1800s, sheep grazing has prevented seedling recruitment and thus many rural trees may be old (Beckman and Davidson 1990). A formal investigation of stand structure change with time since settlement may well be warranted, although clearly many trees suffering RTD are old in comparison to younger, smaller bole-diameter trees on roadside reserves and in remnant bush patches.

Fencing will prevent soil compaction by stock (Willat and Pullar 1983; Kirkpatrick *et al.*



Photo 3. Roadside trees often appear to suffer less from the effects of tree decline than those in the adjacent paddocks.

2000) and the nutrient enrichment of soils by stock excrement (Beckman and Davidson 1990). Many studies have shown the benefits to seedling growth of reducing compaction commonly measured as decreased soil bulk density (e.g. Corns 1988; Minore and Weatherly 1990; Kube 1993; Conlin and van den Driesche 1996) particularly where soils have high clay content or hardpans. Compacted soils also have lower hydraulic conductivity that increases run-off on slopes during precipitation and waterlogging on flats (Douglas and Crawford 1998). Decreased infiltration, resulting in lower soil moisture content, may have consequences for plants and ectomycorrhizal fungi (Tommerup and Bougher 2000). If complete stock exclusion is not economically viable, remnant areas should be excluded from livestock for as long as possible, minimising the adverse affects (Landsberg *et al.* 1990). Rotational grazing improves regeneration as it allows for long periods of recovery between short periods of grazing (Morgan and Creighton 1991).

Bushland restoration in regions suffering RTD can be facilitated by erecting fences around one or more isolated trees in pasture. Ideally, fences should be placed at a distance from the tree equivalent to 1.5 times the tree height on the leeward side of prevailing winds, as this is where the majority of seed falls (Curtis 1991a). Seed from isolated trees or from trees within remnant stands has, on average, reduced vigour (Borrhalho and Potts 1996) due to low outcrossing rates (Hardner *et al.* 1996). However, outcrossing can be above that expected on the basis of their isolation because the trees may provide resting places or habitat 'islands' for pollinators travelling across the landscape (B.M. Potts, pers. comm.). Thus, a proportion of the seed gathered from trees in remnant stands or in isolation is likely to be vigorous and provide a valuable genetic resource if seed material of local provenance from dense stands is not available. Chemical control of weeds and vigorous pasture species within the fenced area will maximise the potential for tree and shrub



Photo 4. Repeated browsing by possums results in sparse foliage and torn leaves.

seedling recruitment (Beckman and Davidson 1990). The removal or 'scalping' of the top few centimetres of soil has been recommended to decrease weed competition because it removes the soil layer containing the majority of weed seed (Loney 1990) and with the highest nutrient content (Hobbs 1991). This practice must be used with caution, however, as this layer also contains the majority of organic matter and therefore has a high moisture-holding capacity (Lichter 1998). An alternative method for removing excess nutrients is repeated harvesting of weed biomass (Marrs 1985).

Possums are a major source of stress to rural trees in the Midlands of Tasmania (Photo 4). Possums reach unnaturally high populations in these landscapes where nutritious pasture, vegetable and fodder crops are grown and where many nesting sites are available in the skeletons of dead eucalypts (Neyland 1996). Browsing by insects and possums can be factors accelerating RTD (Curtis 1991b). However, a study on possum guards has indicated that they had no effect on the likelihood of tree death from RTD (Rees 2000), although the health of trees prior to placement of the possum guards was not assessed.

Kirkpatrick *et al.* (2000) demonstrated that the exclusion of possums resulted in improved tree health at two of three experimental sites in the Tasmanian Midlands. Based on this investigation, fencing around rural trees to exclude browsing possums is recommended where feasible.

If it is not possible to fence isolated trees in pasture, a number of management practices can be employed to minimise the potential for RTD. These include avoiding application of fertiliser and the sowing of introduced pasture species in an area 1.5 times the diameter of the canopy around the base of a tree (Reid and Landsberg 2000), the introduction or retention of native grass species (Reid and Landsberg 2000), and irrigation during very dry periods (Doyle 2001) that may be feasible if crops are being irrigated nearby.

3. *Mimic ecosystems*

Fast growing, pioneer species planted near the perimeters of remnant areas and scattered within fenced areas can act as nurse crops for trees recovering from decline and facilitate regeneration of native understorey species. This sets in motion a natural

succession and has recently been termed the 'ecosystem mimic concept' (Pate and Bell 1999). This approach uses plant species in similar density and distribution to those of the vegetation originally inhabiting the given site requiring restoration. The construction of a complex, self-sustaining, natural ecosystem improves the robustness of remnants in regions susceptible to RTD (Curtis 1991a; Reid and Landsberg 2000).

Nurse crops confer many micro-environmental benefits. A nurse crop decreases wind speed that in turn reduces the direct drying of soils (Ball *et al.* 1995; Doyle 2001), reduces evapotranspirational stress and physical damage to plants (termed thigmomorphogenesis; Jaffe 1980), and increases humidity near the ground (Dulohery *et al.* 2000). A nurse crop can also decrease diurnal temperature variation (Marrs *et al.* 1982; Ray and Brown 1995; Langvall and Örlander 2001) and thereby moderate extremes of temperature (Curtis 1991a; Ray and Brown 1995; Snowcroft and Jeffrey 1999).

The shade provided by a nurse crop can decrease photoinhibition and photodamage. Photoinhibition occurs when plants absorb more light than they can utilise when the efficiency of photosynthesis is depressed by drought (e.g. Björkman and Powles 1984), cold (e.g. Close *et al.* 2001a, b) or nutrient deficiency (e.g. Close *et al.* 2000, 2001a, b). Photodamage occurs when photoinhibition is severe. Leaf tissues become bleached and plants can suffer mortality (Wise and Naylor 1987). Some nurse-crop species can increase nitrogen availability through atmospheric nitrogen fixation by symbiotic bacterial associations (DeBell *et al.* 1997; Khanna 1997; Bauhus *et al.* 2000). Conspicuously flowering nurse-crop species can attract pollinating insects and birds (Beckman and Davidson 1990) that facilitate pollen dispersal and outcrossing. Care should be taken not to use exotic nurse-crop species near remnant stands of closely related natural species where there is the possibility of genetic pollution through hybridisation

(Potts *et al.* 2001b). Although hybrids do not usually survive in undisturbed natural communities, this is not the case for hybrids introduced as exotics (Stace 1975) and also may not be the case in rural landscapes affected by climate change.

4. *Link remnants*

Corridors of vegetation between remnants allow for movement of wildlife (Wilson and Lindenmayer 1996) that is essential for viable native animal populations (Nadolny 1984; Saunders *et al.* 1991; Estrada *et al.* 2000). Soil disturbance by animals is required for the distribution of ectomycorrhizal fungi (Tommerup and Bougher 2000).

5. *Maximise biodiversity*

Alterations to the understorey, partial tree removal and the growing of introduced pastures change the balance between pests and predators. Nectar from native species is an essential food source for many parasitic wasps and flies (Davidson 1982) as well as an attractant to predatory birds. These vectors are essential for cross pollination and seed dispersal of trees. Larvae and adults of some predatory flies and beetles require shelter provided by leaf and bark ground litter. Birds in a healthy woodland on the New England Tablelands consume as much as 60% of the eucalypt-pest insects (Ford and Bell 1982; Ford *et al.* 1985). Factors associated with foliar morphology and chemistry of trees in decline, relative to healthy trees, confer greater susceptibility to pest insect damage (Landsberg 1988). Thus the balance between natural pests and their predators is disrupted when RTD occurs.

Steps to ameliorate biodiversity imbalance can include the inclusion of species of varied form (i.e. grasses, forbs, shrubs, small and large trees) in plantings or augmentation of these species in degraded remnants or existing plantings and exotic weed removal, that maximises patchiness and habitat types available and further provides an

understorey necessary for the completion of lifecycles of many insects. Species with promiscuous, nectar-producing flowers help attract predatory insects and birds that consume insect pests.

6. Management for climate change

If management of land-use-associated factors, such as that outlined in the preceding recommendations, does not reverse the rising incidence of RTD, and the overriding effects of long-term climate change continue to contribute to RTD, management options are limited. Irrigation may be possible in some instances to manage the effects of lowered rainfall. However, rising air temperatures will likely cause widespread, unmanageable RTD as species are trapped within an unsuitable bioclimatic envelope (Hughes *et al.* 1996). This scenario will necessitate revegetation if trees are to remain in these landscapes (see Close and Davidson 2003c). Particular microsites, such as break of slopes, natural springs, drainage lines or southern and/or eastern aspects may provide the best opportunities for ensuring the survival of planted local indigenous species and provenances. Furthermore, judicious selection of exotic eucalypt species, or provenances of local indigenous species, may be required for successful revegetation of sites maximally exposed to the effects of

climate change (e.g. sites of north-facing aspects) under the drier, hotter conditions forecast for the near future.

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While demonstrated impacts of climate change on tropical forests include reductions in plant diversity (14), changes in plant species composition (15), and increases in tree growth, mortality, and biomass (16), little is known about the impact of climate warming on rainforest arthropods (17, 18).^Â If the declines in arthropod resources negatively impact avian abundance, then species with primarily granivorous or frugivorous diets should suffer smaller declines than those consuming a greater proportion of insects. Australian climate trends. The Bureau of Meteorology and other science agencies employ a range of atmospheric, terrestrial and marine sensors to track climatic trends. For example, the Australian Climate Observations Reference Network ^{â€} Surface Air Temperature dataset is based on a network of over 100 stations, with data for more than half starting in 1910. The 2016 State of the Climate report states that Australia^{â€™}s climate has warmed since national records began in 1910, especially since 1950. Mean surface air temperature has increased by around 1.0^Â°C since 1910. Australia^{â€™}s top five warmest