

Mistletoes increasing in 'undisturbed' forest: a symptom of forest decline caused by unnatural exclusion of fire?

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Summary

Increases in populations of mistletoes were observed in undisturbed eucalypt forest near Eden, New South Wales. Repeated counts in an ecological research area showed that there were large increases in mistletoes over 13 years. Populations of mistletoes quadrupled in areas that had few or no prescribed burns. Mistletoes doubled in areas that were patchily burnt by fires of generally low intensity six times in 13 years, but this increase was not statistically significant. Hypotheses advanced to explain the perceived proliferation of mistletoes in rural lands cannot account for the increases in undisturbed forests. However, the results of this study are consistent with a general hypothesis of tree decline in rural lands and forests caused by chronic abiotic stress. Unnatural exclusion of low intensity fire may impair the health of eucalypt forests and cause outbreaks of pests, pathogens and parasites. Populations of mistletoes or other parasites could readily be monitored as indicators of ecological imbalance.

Keywords: mistletoe; eucalypt; undisturbed forest; forest decline; fire regimes

Introduction

Increasing populations of mistletoes in rural areas of Australia have been reported by many observers since the late nineteenth century but no data have been published to confirm these reports (Reid and Yan 2000; Ward 2005b). Reid and Yan (2000) suggested that the perceived increases were due to better habitat provided by more vigorous trees responding to increased water and nutrients as a result of partial clearing and development. They suggested that mistletoes were not generally regarded as a problem in public forests and conservation reserves. However, mistletoes have long been perceived as a problem in the box–ironbark forests of north-central Victoria where extensive areas of forest were treated, from the 1920s to the 1950s, to remove dodder-laurel vines (*Cassytha melantha*) and mistletoes (Kellas 1991). Jacobs (1955) stated that 'the mistletoe problem has attracted considerable attention and work is being undertaken by CSIRO, the Forestry and Timber Bureau and the Forest Departments to counter the pest' by sprays, injections and lopping. In the conservation reserves of the Mount Lofty Ranges, South Australia, 29% of pink gums, *Eucalyptus fasciculosa*, are currently infected by mistletoe, and infected trees have 58%

canopy dieback (Ward 2005a). High densities of mistletoes are now apparent in some stands of trees that were not normally susceptible to infection, such as blackbutt, *E. pilularis*, in Peachester State Forest in south-eastern Queensland (*pers. obs.* VJ).

Mistletoe infestation is especially apparent on isolated trees in paddocks (Reid and Yan 2000), suggesting that increased apparency could be a result of relatively stable densities of mistletoes compared to reduced densities of host trees. Alternatively, increasing populations of mistletoes in Australian and North American forests have been attributed to changed fire regimes (e.g. Koonce and Roth 1980; Liddy 1982; Reid and Yan 2000; Shaw *et al.* 2004). There is a widely held view that fire controls mistletoes by killing them (e.g. Reid and Yan 2000; Shaw *et al.* 2004; Ward 2005b). Jurskis (in press) has proposed that high populations of parasites including mistletoes, native cherries (Santalaceae) and dodder-laurel vines (*Cassytha* spp.), like high populations of folivorous animals or fungal pathogens, are symptoms of declining health of forests caused by unnatural fire regimes. Exclusion of fire may change the soil environment around the roots of trees and affect their physiology, improving the nutritional value of their roots, sap and leaves for pests and parasites (Jurskis and Turner 2002; Jurskis in press).

High densities of mistletoes and declining trees are apparent in many 'undisturbed' forests in south-eastern Australia, Europe and North America (Koonce and Roth 1980; Jaggars 2004; Jurskis 2004a,b; Shaw *et al.* 2004; Tsopelas *et al.* 2004; Ward 2005a,b). Ecologists are advocating the reintroduction of more natural fire regimes to improve forest health, and more information is needed on the interaction between fire regimes and mistletoe populations (Shaw *et al.* 2004).

Over about 15 years, populations of mistletoes increased very noticeably in a large, undisturbed Flora Reserve on the Wallagaraugh River near Eden, along with a general deterioration in the health of the trees (Jurskis 2004a). Managers have largely excluded fire from this area since it was reserved as State forest about 30 years ago. This present study examines changes in populations of mistletoes over 13 years in the nearby 'Eden Burning Study Area'. Possible relationships between fire regimes and the population dynamics of mistletoes are discussed.

Study area

The study area is dry sclerophyll eucalypt forest that had never been used for grazing, timber production or other post-European human enterprises prior to the establishment of the Eden Burning Study in 1987 (Binns and Bridges 2003). It is dominated by yertchuk *E. consideriana*, white stringybark *E. globoidea*, silvertop ash *E. sieberi*, and blue-leaved stringybark *E. agglomerata* (Binns and Bridges 2003). The understorey is mostly scattered small shrubs, but there are many thickets of large shrubs and small trees such as sheoak *Allocasuarina littoralis*. There is a discontinuous groundlayer of grasses and graminoids with some bracken (*Pteridium esculentum*) (Keith and Bedward 1999; Binns and Bridges 2003). The coarse soils are derived from Devonian adamellite. They are mostly poorly structured and infertile. The climate is mild to warm and the mean annual rainfall is approximately 900 mm.

The study area adjoins a strip of reserved forest, at least 500 m wide, straddling the Wallagaraugh River, and is surrounded by an uninterrupted tract of about 40 000 ha of native forest and woodland. Most of the study area had not been burnt since a wildfire in summer 1972/1973, which burnt at low intensity.

In 1987/1988, logging treatments were applied to about half of the ecological study area in a pattern of alternating logged and unlogged coupes; this study of mistletoes was confined to the unlogged treatments. Within the logged treatments approximately 10% of the gross area was excluded from logging to protect wildlife and water quality (Bridges in press), and within the net logged area 51% of trees comprising 39% of the basal area were retained for various reasons (Binns and Bridges 2003). A similar alternate coupe pattern was applied to much of the surrounding landscape between 1980 and 1994 (Anon. 1994). Thus the mistletoe study was conducted in units of unlogged forest within a permanent native forest landscape that has not been affected by agricultural or urban development.

The burning study has a randomised block design, consisting of three blocks of six treatments (Binns and Bridges 2003). There are three replicates of three unlogged burning treatments: Unburnt; Routinely Burnt (burnt three times in 13 years); Frequently Burnt (burnt six times in 13 years). The burns were generally low-intensity burns conducted according to routine operational prescriptions during autumn and winter. Details of weather conditions and field comments for each burn, except the sixth frequent burn, were provided by Binns and Bridges (2003, pp. 54–62). The burns were patchy and the effects within any sampled patch varied from none to severe.

Methods

Mistletoes were sampled in nine treatment units with an average area of 26 ha. Ten sample plots of 50 m radius were used in each treatment unit. These were at reference points that had been established to repeatedly sample birds, small mammals and reptiles (Binns and Bridges 2003). The plots were randomly located at six overstorey reference points and four understorey reference points. Three of the 10 plots were located in an upper-slope stratum and three in a lower-slope stratum. The other four plots in each unit were selected so that all points were at least

100 m apart, all points were at least 50 m from the edge of a treatment unit, and points sampled the range of topography within each unit. Reference points were located only in areas that would be subject to logging or burning if routine forestry operations were carried out; i.e. areas which would be 'filter strips', 'wildlife corridors', etc., were not sampled (Bridges in press). All trees in the plots were searched by one observer from the ground, and the number of mistletoes in the trees was counted. Both living and dead standing trees and mistletoes were counted. The species of each tree with mistletoes and the species of each mistletoe were recorded. Mistletoes were counted in October and November of 1990 and April 1991 (Turner 1991), and for the second time in February 2004 by the same observer.

The numbers of living mistletoes and the numbers of infected living trees in the sample plots were compared between the two counts using the ANOVA procedure (Zar 1984; SAS Institute Inc. 1990). The number of mistletoes at each count were compared amongst the three treatments using the log likelihood ratio method (Zar 1984). This method was also used to compare changes in mistletoe populations amongst the treatments and numbers of mistletoes infecting the different eucalypt species in 2004.

The operational scale and nature of the burning treatments applied in a heterogenous environment resulted in very variable outcomes, complicating the analysis of treatment effects (Binns and Bridges 2003). Under these circumstances it was considered reasonable to treat the sample plots as replicates in assessing the effects of the burning treatments on numbers of mistletoes.

The average tree species composition of the study area was obtained from 54 plots, each of 0.2 ha, comprising the 'overstorey reference points' (Binns and Bridges 2003).

Results

Most of the mistletoes (1028 or 96%) were *Amyema pendulum*. The remainder (46 or 4%) were *Muellerina eucalyptoides*. No other species of mistletoe was encountered in the study area. The mistletoes occurred almost exclusively in eucalypts, the exceptions being one plant of *Amyema pendulum* in each of two wattles (*Acacia mearnsii*). It was often difficult to distinguish plants of *Amyema pendulum* from clumps of epicormic foliage on the branches of the trees, especially yertchuk. For this reason and because there were more mistletoes, it took longer to sample most plots in 2004 than in 1990/1991.

In 1991 the mistletoe density was greater ($P < 0.001$), at 6 ha⁻¹, in the units that were to be frequently burnt than in the other units where there were 4 ha⁻¹ (Table 1). There were substantial increases in mistletoes between 1991 and 2004. The number of infected trees across all treatments increased ($P < 0.01$) from 195 to 545, and the number of living mistletoes increased ($P < 0.05$) from 349 to 1072 (Table 1). Mistletoe populations doubled in the frequent burning treatment whilst they quadrupled with less frequent burning or exclusion of fire (Table 1). The increases in both mistletoes and infected trees were significant ($P < 0.001$) in the unburnt and the routinely burnt treatments. The increase in mistletoes in the frequently burnt treatment was not statistically significant, but the increase in numbers of infected trees was significant ($P < 0.05$). The differences in growth of

Table 1. Counts and mean densities (no. ha⁻¹) of infected trees and mistletoes by sampling year and treatment

Year	Frequent burning				Routine burning				Exclusion of fire				Total	
	Trees	Density	Mistle-toes	Density	Trees	Density	Mistle-toes	Density	Trees	Density	Mistle-toes	Density	Trees	Mistle-toes
1990/1991	69	3	150	6	60	3	99	4	66	3	100	4	195	349
2004	148	6	313	13	199	8	397	17	198	8	362	15	545	1072

mistletoe populations according to treatments were statistically significant ($P < 0.001$). Four mistletoes ha⁻¹ and less than one infected tree ha⁻¹ died between the two counts, but many more mistletoes established.

In 2004 the density of mistletoes was smaller in the frequently burnt units than in the other units ($P < 0.01$). This was despite the fact that the highest densities of mistletoes in individual sample plots (62 ha⁻¹ in 1990, and 89 ha⁻¹ in 2004) occurred in two plots within a frequently burnt unit. There were 100 trees per hectare with diameter at breast height over 30 cm within these plots; half of them were yertchuk, and 39 of these were infected by mistletoes. Many mistletoes (18 ha⁻¹) and a few infected trees (1 ha⁻¹) in these plots died during the study, but recruitment of mistletoes far outweighed mortality.

Yertchuk was more heavily infested with mistletoe ($P < 0.001$) than other tree species (Table 2). Only 38% of trees in the study area were yertchuk, but they supported 73% of the mistletoes. No other species supported more than 7% of the mistletoe population. Disproportionately low numbers of mistletoes occurred in silvertop ash, and the stringybarks.

Discussion

This study supports the view that mistletoes are increasing in undisturbed forests and that they are increasing right across the landscape. Increased visibility as a result of reduced density of host trees is clearly not involved in this case because the density of host trees has not been reduced.

Environmental and operational heterogeneity made it difficult to assess the effect of frequent burning on mistletoes. Neither mistletoes nor yertchuk are evenly distributed across the landscape. They mostly occur on flat to gently undulating low

Table 2. Tree species composition of the study area and occurrence of mistletoes in eucalypt species

	Tree density		Mistletoe	
	No. ha ⁻¹	%	No. of	%
<i>E. considiana</i>	78	38	785	73
<i>E. globoidea</i>	30	15	54	5
<i>E. sieberi</i>	27	13	73	7
<i>E. agglomerata</i>	26	13	51	5
<i>E. muelleriana</i>	17	8	38	4
<i>E. radiata</i>	12	6	49	5
<i>E. cypellocarpa</i>	8	4	8	1
<i>E. obliqua</i>	3	2	0	0
<i>E. angophoroides</i>	1	<1	14	1

ridges and slopes with poor drainage. (e.g. Boland *et al.* 1984; Keith and Bedward 1999). Burns within treatment units were patchy (Binns and Bridges 2003).

Nevertheless, the study suggests that frequent burning has moderated increases in mistletoe populations. There were higher populations of mistletoes in frequent burning units at the outset, and higher populations in the other treatments 13 years later. The relationship between fire regimes and the dynamics of mistletoe populations warrants further examination.

Reid and Yan (2000) listed five hypotheses (they were not mutually exclusive) that have been proposed to explain increases in mistletoes in temperate Australia. These hypotheses mostly relate to agricultural or urban development and therefore may have limited relevance to the increases observed in relatively undisturbed forests. They are briefly considered hereunder.

- (i) *There has been a reduction in predators of mistletoes:* Populations of food plants are not regulated by their natural predators, rather populations of their predators are limited by the quantity of high quality food plants in their environment (White 1993, 2001, 2004). Given the increases in populations of mistletoes, it seems likely that any specialist predators of mistletoes might also increase.
- (ii) *There has been increased dispersal of seed by mistletoe-birds (Dicaeum hirundinaceum):* It is likely that numbers of mistletoe-birds are limited by numbers of mistletoes because the birds feed almost exclusively on their fruits (Liddy 1983; White 1993). The birds might increase in response to increasing populations of mistletoes (e.g. White 1993, 2001, 2004), but increases might be difficult to detect because mistletoe-birds are nomadic (e.g. Liddy 1982). Periodic bird counts in the study area have yielded insufficient numbers of this bird to reveal any trends.

The establishment rate of mistletoe seeds is low by comparison with their rate of dispersal to natural hosts and their rate of germination. Establishment rates of around 6% have been reported for *Amyema* (Lamont 1983). Although these rates are high compared to many terrestrial plants (e.g. Jacobs 1955; Lamont 1983), it is apparent that seedling establishment rather than seed dispersal is the factor that mostly limits recruitment of mistletoes as well as other native plants. It is more likely that the increasing mistletoe populations in the study area are a result of increased establishment of seedlings than of increased dispersal of seed.

- (iii) *Better habitat is provided by more persistent branches on open grown trees, and (iv) Better habitat is provided by trees that are more vigorous as a result of partial clearing, watering and/or fertilising:* These hypotheses cannot explain the observed increases in relatively undisturbed forests.

Furthermore, trees in improved pastures with enhanced nutrient and water supply are typically declining rather than increasing in vigour as suggested by hypothesis (iv).

- (v) *Mortality has reduced as a result of fire suppression*: It is commonly suggested that frequent burning controlled populations of mistletoes by killing them (e.g. Reid and Yan 2000; Shaw *et al.* 2004; Ward 2005b). This hypothesis appears to be based on a misconception of pre-European fire regimes and the way that fire regimes have changed since European settlement (e.g. Jurskis *et al.* 2003). Dendrochronology typically reveals increased fire scarring of eucalypt trees since European settlement. For example, when European settlers reduced the frequency of low intensity fires in jarrah forests, wildfires became more intense and there was a large increase in the frequency of fire-scars on jarrah trees (Burrows *et al.* 1995; Ward *et al.* 2001). Mistletoes have been increasing with reduced occurrence of low intensity fires and increased occurrence of high intensity fires. Reduced mortality as a consequence of fire suppression is an unlikely explanation for increased populations of mistletoes.

Complete crown scorch may be necessary to kill mistletoes (e.g. Reid and Yan 2000, p. 368). Mistletoes in this study mainly occurred in stands of mature yertchuk that were about 20 m high (Binns and Bridges 2003) and were growing on low ridges and slopes. In the frequent burning treatment, most of these stands and trees were not affected by intense fires or crown scorch during the study (Binns and Bridges 2003, pp. 54–62). Crown scorch from burning treatments was noted in only one treatment unit on one occasion, and this unit contained relatively few yertchuk and mistletoes.

The rate of increase in mistletoes was lowest in the frequent burning treatment, despite the fact that the highest density of mistletoes occurred in this treatment in two plots that sampled an extensive stand of yertchuk on a lower slope near a creek. Four of the six prescribed burns did not reach this area, but most of the area burnt at a relatively high intensity in 1994 (Forests NSW unpublished data). Although mortality of mistletoes was relatively high in these two plots, there was no evidence that it was caused by fire scorch, and recruitment far outweighed mortality. The study did not support the hypothesis that frequent fires control population growth by killing mistletoes.

Liddy (1982) attributed high densities (>200 ha⁻¹) of *Amyema cambagei* in open forest in south-east Queensland to cessation of annual burning 15 years previously. This type of open forest had a natural regime of frequent, low intensity fires before European settlement (Reid and Yan 2000; Jurskis *et al.* 2003). Box-ironbark forests in Victoria that have been 'protected' from fire are heavily infested with mistletoes (Kellas 1991; Jurskis 2004b). In western North America, there were moderately low levels of infection of ponderosa pine (*Pinus ponderosa*) by dwarf mistletoe (*Arceuthobium campylopodum*) prior to the development of effective fire suppression systems (Koonce and Roth 1980). Dwarf mistletoes have now become the most serious 'disease' of trees in much of this area, causing very large losses to timber production (Hawksworth 1983). Increased outbreaks of other pests and diseases in the region also appear to be related to fire suppression (Dwire and Kauffman 2003).

Hessburg and Agee (2003) argued that, over millenia, fire was the most important ecological 'disturbance' process in Inland Northwest North America. They suggested that fire suppression since European settlement and subsequent industrial development has been associated with changes in forest structure and composition, increasing spread of pest insects and pathogens across the landscape, declining forest health and more extensive outbreaks of intense fires, pests and diseases. Reduced use of low-intensity fires in temperate Australian forests during recent decades has also coincided with declines in forest health, increases in pest and disease populations, and more extensive high intensity fires (Jurskis and Turner 2002; Jurskis *et al.* 2003; Jurskis in press). Most mature yertchuk in the study area had clumps of epicormic foliage distributed along their woody branches indicating that the trees were stressed (e.g. Jacobs 1955; Jurskis 2004a).

An alternative hypothesis to explain increasing mistletoes

It is proposed here that outbreaks of pests, pathogens and parasites including mistletoes, cherries and dodder-laurel vines (*Cassytha* spp.) are symptomatic of forests that are declining as a result of chronic abiotic stress (Jurskis in press). Jacobs (1955) suggested that native cherries and mistletoes have similar interactions with their hosts. Mistletoes, dodder-laurel vines and native cherries have proliferated in forests that have been 'protected' from fire (e.g. Kellas 1991; Henderson and Keith 2002; Jurskis 2002).

Decline of both rural and forest trees can be explained by the eutrophication of topsoil. Unnatural accumulation of nitrogen, moisture and 'mulch' in topsoils stresses eucalypts and changes their physiology, so that pests, diseases and parasites are favoured (Jurskis and Turner 2002; Jurskis in press). Eutrophication can occur in 'undisturbed' forests from which fire has been unnaturally excluded and in rural lands as a result of agricultural development (Landsberg *et al.* 1990; Reid and Yan 2000; Jurskis 2003; Jurskis *et al.* 2003; McCulley *et al.* 2004). Eutrophication favours microbes and fungi that attack tree roots (e.g. Ellis and Pennington 1992; Balchi and Halmschlager 2003), insects that attack leaves (e.g. Landsberg *et al.* 1990) and may also favour recruitment of mistletoes (Turner 1991; Reid and Yan 2000), native cherries (e.g. Jacobs 1955) and dodder-laurel vines that obtain water and nutrients from trees.

Mistletoes typically have low water potential relative to their hosts (Fisher 1983; Reid and Yan 2000). Unhealthy eucalypts can have higher water potentials than healthy trees (e.g. Landsberg 1985; Marsh and Adams 1995; Zubrinich *et al.* 2000) and this could increase the ability of mistletoes to obtain water and nutrients from them by increasing the water potential gradient between the host and parasite. The recruitment and survival of mistletoes might be enhanced in the same way that high levels of nutrients and water in the leaves of unhealthy trees (Landsberg 1990) enhance the recruitment and survival of folivorous insects (Landsberg *et al.* 1990).

Eutrophication will particularly affect trees growing on depositional soils on concave topography in lower slope positions (e.g. Jurskis 2004a,b), and symptoms of eucalypt decline, including infestations of mistletoe (e.g. Jurskis 2004a), typically appear first on lower slopes. This is also where low intensity burning first becomes difficult in forests after a period of fire

exclusion (Jurskis *et al.* 2003) and where intensive pasture improvement occurs in rural lands.

Implications for nature conservation of increases in mistletoes

It is often assumed or implied that increasing populations of native species, and particularly 'charismatic' species such as birds and koalas are indicative of healthy ecosystems (e.g. Watson 2002; Kavanagh 2003; Kavanagh and Stanton 2005). On the other hand, high numbers of some native animals such as caterpillars and beetles, and even koalas and possums, have been recognised as signs of ecological imbalance (e.g. Martin 1985; Neyland 1996). High numbers of mistletoes are regarded as a problem by some (e.g. Kellas 1991) but are regarded favourably by others (e.g. Turner 1991; Watson 2001). Watson (2002) suggested that mistletoe was a 'keystone resource' because high densities of mistletoes (92 ha⁻¹) had a positive effect on the number of species of birds and the number of birds of most species in two 'woodland remnants'. However he did not consider whether the high numbers of species and individual birds in these stands of trees were likely to be sustainable in the long term.

Increasing populations of native plants and animals can be symptomatic of a breakdown of natural processes of nutrient cycling, competition, mortality and recruitment (e.g. Granger *et al.* 1994; Marsh and Adams 1995; Lunt 1998; Jurskis and Turner 2002; Jurskis *in press*). Frequent low-intensity fires were a natural feature of open eucalypt forests and woodlands before European settlement (Jurskis *et al.* 2003). Fire regimes have just as large an influence on the distribution of vertebrate fauna in north-eastern New South Wales as major environmental variables including rainfall and temperature, and a much larger influence than logging (Kavanagh and Stanton 2005, Fig. 4).

Mistletoes or other parasitic plants could be monitored more efficiently than either the animals that have been proposed as 'direct indicators of major environmental change' (Kavanagh and Stanton 2005) or recognised pest insects, because they are sedentary, and are relatively easy to see and to count. Increasing populations of mistletoes appear to be a symptom of ecological imbalance (e.g. Watson 2001) associated with unnatural exclusion of low intensity fires from eucalypt forests and a general decline in the health of trees.

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