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Changes to tuart woodland in Yalgorup National Park over four decades

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ABSTRACT: The condition of the tuart tree (*Eucalyptus gomphocephala*), a coastal southwestern Australian woodland species, has declined dramatically within parts of its distribution over the last decade, particularly within Yalgorup National Park. Prior to the park being gazetted in 1968, some of the woodlands were used for cattle grazing. Frequent, light, understorey burns were carried out to encourage grass fodder growth. Earlier, Aboriginal use is believed to have involved a similar regime to facilitate hunting and access. Since gazettal, the majority of the park has either been excluded from fire, or burnt infrequently by wildfire and prescribed fire. Consequently, from 1968 to the present, most fires are thought to have been more intense due to increased fuel loads. Alterations in disturbance patterns (particularly fire) elsewhere, have been linked with vegetation changes (composition and structure) and in some instances, declining tree health. For tuart woodland, it has been proposed that increased abundance and vigour of the lowerstorey peppermint tree (*Agonis flexuosa*) and a decline in the health of tuart trees are consequences of reduced fire frequency. Sample plot data from the mid – late 1970s and photographs from 1957 are contrasted with the 2003/2004 situation to describe changes in tuart woodland. Declining tuart health, changes in the health and abundance of some understorey species (for example, fewer *Banksia attenuata*) and a shift towards peppermint dominance are revealed. The contribution of changing fire regimes to these trends is explored. While a link between fire and changes to the woodland may be established, factors underlying the loss of tuart dominance remain to be determined. An integrated research project is in progress to examine the range of decline factors.

1 INTRODUCTION

It is becoming increasingly apparent that the creation of conservation reserves such as National Parks, followed by a passive approach to management, does not guarantee the preservation of an ecosystem in its existing state or condition. This is in contrast to popular public opinion which ignores the long (millennial) history of human influence on the biota through modification of fire regimes (Hallam 1975, Bowman 1998, Hassell & Dodson 2003) and the recent (decades and centuries) changes in the landscape from fragmentation of vegetation, climatic variation, introduction of exotic species and the spread of plant diseases (Soule, 1991). For most Australian ecosystems, disturbance, particularly fire, performs a significant role in recycling and renewal. For example, many plant species are markedly dependent on fire for successful reproduction and recruitment (Bell et al. 1993), nutrients are returned from litter and woody debris to the soil following combustion (Grove et al. 1986, Adams et al. 2003) and the vigour of some species increases when resprouting after fire (Wallace 1966, Bowen & Pate 2004). Therefore, maintaining or restoring appropriate fire regimes is an important tool in the preservation of biodiversity. Where the goal is to maintain biodiversity, but where complete knowledge of ecosystem functioning is lacking, maintaining a diversity of fire regimes across a landscape (rather than fire exclusion) is now advocated as a practical, precautionary approach to managing fire prone ecosystems (Bond & van Wilgen 1996, Burrows & Wardell-Johnson 2003).

The modification of natural fire regimes, usually by way of increasing frequency, decreasing intensity and bringing forward the time of burning within the season, has been a practice common to the world's aboriginal peoples living in fire prone environments (Pyne 2001). Australian Aborigines used fire as a tool to increase the abundance of resources (plants and animals), in hunting and to facilitate movement through the landscape (Hallam 1975, Bowman 1998). Historical accounts of early European settlers indicate frequent burning in the summer months (Hallam 1975, Abbott 2003) with grasstree stem analysis suggesting burning every 3 to 5 years on average in central southwestern forests and woodlands (Ward 2000, Ward et al. 2001).

From grasstree data (Ward 2000, Ward et al. 2001), dating of fire scars on jarrah tree cross sections (Burrows et al. 1995) and the compilation of historical accounts (Hallam 1975, Abbott 2003, Ward 2000), there seems little doubt that a general trend towards declining fire frequency has taken place in parts of southwestern Western Australia since European settlement in 1829. Further, it follows, and evidence from Burrows et al. (1995) supports this, that greater fuel loads accumulating after longer fire free intervals have contributed to a reduction in the return interval for intense fire events. Such events have the potential to bring about drastic and long-term changes to vegetation (Hopkins & Robinson 1981).

A succession of fires with sufficient intensity to cause substantial defoliation or bark damage, could conceivably impact on the condition of resprouting canopy dominants by eroding their regenenerative capacity (depletion of storage carbohydrates or destruction of epicormic buds). Beard (1967) proposed such a process, in combination with historical timber cutting and present day insect borer attack, to underlie the decline of tree canopies in woodland of Kings Park, Western Australia. Insect and fungal attack have been observed in other areas where wood has been exposed by fire damage (McCaw 1983, Ashton 2000). Beard (1967) inferred that the increased vigour of the understorey was both a response and reinforcing agent (through competition) in the opening of the canopy. Given the strong limitations imposed by moisture availability on Leaf Area Index (LAI) in Mediterranean climates and the decline in southwestern winter rainfall totals since the 1950s (IOCI 2002), redistribution of leaf area (from the overstorey to the understorey) following disturbance, warrants further investigation as a factor in tree decline.

The view of Beard (1967) above is one of the earlier of many recent ecological papers from Australia and North America associating reduced fire frequencies in recent centuries and decades with one or more of the following: increased density of vegetation (e.g. Lunt 1998, Gilliam & Platt 1999), declining health of overstorey species (e.g. Ellis 1980, Jurskis & Turner 2002) and the sustained lack of recruitment of overstorey species (e.g. Yates 1994, Abrams 2003).

Debate surrounds the following interrelated issues: the nature of the vegetation at the time of settlement, the extent of particular vegetation types, the role of fire in the creation of these vegetation types and patterns, and current landscape management (e.g. Ryan et al. 1995 versus Benson & Redpath 1997, Jurskis 2002 versus Keith & Henderson 2002). Much of the debate stems from a lack of data on historical circumstances and process (Fensham & Fairfax 2002, Lunt 2002). Theories of tree decline explained by indirect mechanisms and with supposed application across a wide range of environments are difficult to accept in the absence of thorough, site-specific research. A great variety of complex tree decline syndromes have been identified but not fully explained in Australia (Old 2000) and the Northern Hemisphere (Manion & Lachance 1992). Therefore, there is a need to gain actual evidence and to resist accepting theories based on inference, especially repeated inference viewed as "evidence" (Skelly 1992).

This paper outlines the initial stages of a project investigating changes in vegetation, including tree health, in one landscape (Yalgorup National Park) where fire regimes have been altered dramatically. While the knowledge of fire history for this area is relatively good compared to others, it is not complete, and there are still only minimal or imperfect data sets available in relation to the vegetation history. Nevertheless, we feel that this preliminary examination of available data is a useful step in testing the assumption that changes to vegetation structure and composition have actually occurred, providing some details on the nature of these changes and proposing further research direction. A project exploring decline of a tree species (*Eucalyptus gomphocephala*) from a number of research angles and across a wider portion of the landscape is also discussed.

2 SITE: PHYSICAL FEATURES, VEGETATION AND FIRE HISTORY

Yalgorup National Park is located on the southwestern coastal plain of Western Australia between Mandurah and Bunbury. A Mediterranean climate prevails with approximately 80 % of the mean annual rainfall (870mm) occurring between May and September (Australian Bureau of Meteorology 2004). Soils are predominantly calcareous sands underlain by limestone which occasionally outcrops near ridges (Portlock et al. 1993). Chains of lakes are a conspicuous feature of the Park as is the diversity of vegetation structural types from herbfields and shrublands to forests and tall woodlands (Portlock et al. 1993). Tuart is a dominant tree species in many of the Park's woodland formations. Tuart is restricted to the western margin of the coastal plain of southwestern Western Australia, however clearing is estimated to have reduced the area of tuart woodlands and forests by 73 % since European settlement, consequently there is now a community-wide desire to conserve remnant tuart woodlands (Government of Western Australia 2003).

Ward (2000) has documented the history of burning in the area, which extends back to well before European settlement. Insight into Aboriginal and European burning practices within the woodlands was gained from the analysis of 58 grasstrees scattered across the National Park and nearby private property, plus interviews with local identities and the examination of historical literature. An average fire return interval of 2-4 years up until the 1850s (Aboriginal burning) was then maintained at the shorter end of this range in areas where cattle grazing occurred. A substantial and sudden increase in fire return time (between 1 and 0 per decade) has occurred within the National Park area and private property since the 1960 – 1970 period. Today, grasstrees unburnt for 80 years or more indicate the presence of some very long unburnt areas within the Park.

Regular burning, two or more times per decade by Aborigines, and then by European graziers, was likely to have involved two features perpetuating one another: low intensity fire and an open understorey dominated by a non-woody, possibly grassy fuel-bed (Bunbury quoted in Ward 2000, Bradshaw 2000). This has been reported for other areas in the southwest (Scott 1994) and beyond (Anon. 1903, Booysen & Tainton 1984, Bowman 1998, Lunt & Morgan 2002) where burning on short rotations has been carried out. The condition of mature tuart trees, which like other *Eucalyptus* spp. have thick bark, foliage at considerable height and the ability to resprout, would likely have been largely unaffected directly by such fires. This contrasts with the current situation where the quantity of accumulated fuel raises the potential for significant canopy damage. Smith (1975) discusses a number of intense fires (some extensive) that took place in the Park between 1970 and 1974 and a wildfire resulting in major, long-term canopy loss in a section of tuart woodland occurred in 1996 (Bradshaw 2000, McCaw & Sneeuwjagt 2002).

Tuart woodlands in Yalgorup National Park were noted to be "… mostly in very good condition …" in 1993 (Portlock et al. 1993, p. v), a description that does not seem appropriate today. Dead branches extending from dead trunks or sparse crowns now remain in place of the original woodland canopy in many parts of the Park. The condition of tuart trees in metropolitan reserves has deteriorated since the 1960s (Beard 1967, Fox & Curry 1980). Widespread decline or death of trees from all age classes at Yalgorup has been observed since the mid-1990s and is still apparent today. The prevailing view implicates sustained borer attack in response to other as yet unidentified predisposing stress/stresses (Bradshaw 2000, Longman & Keighery 2002). Speculation on the possible agents of decline has also implicated fire, frost, drought, hydrological changes, nutritional factors, fragmentation and fungal disease (Longman & Keighery 2002).

Concerns have also been expressed about the lack of tuart recruitment in Yalgorup (Smith 1975, Backshall 1983, Bradshaw 2000) and elsewhere (Anon 1921, Beard 1967, Fox & Curry 1980) as well as the apparent increase in understorey growth and density in Yalgorup (Bradshaw 2000, Ward 2000) and elsewhere (Anon 1921, Beard 1967). This thickening of the understorey has been suspected as a source of competition, diminishing tuart seedling survival (Anon 1921, Bradshaw 2000, Ward 2000) and possibly tuart sapling and tree vigour (Beard 1967, Bradshaw 2000). The rise in prominence of WA peppermint (*Agonis flexuosa*) as a result of fire frequency reduction is thought to have been the most conspicuous and influential change in the understorey south of Mandurah (Bradshaw 2000, Ward 2000). Bradshaw (2000) outlines the main features of peppermint which enable it to flourish under a variety of conditions including when fires are absent, infrequent, or frequent following a period of some absence (presumably in the order of five or more years): The ability of peppermint recruitment to occur both following fire, between fires when competition is high (including under canopy, i.e. a tolerant species) and the tendency of seedlings to develop a substantial lignotuber with time. The latter explaining not only the resistance of the sapling/tree to intense fire but also the vigorous resprouting and increased density of stems reported following even the most intense fires. Post-fire recruitment of peppermint is possible due to the combination of some protection being afforded to seed in seed capsules (although somewhat thin-walled) and their storage at height (with greater heights following longer fire free intervals) (Authors' pers. obs.). Data are yet to be presented as to the degree or extent of these understorey changes or whether these mechanisms are relevant for the decline of tuart trees at Yalgorup. That the development of a more vigorous understorey may be a symptom rather than a cause of canopy decline, should not be overlooked in this, and other cases where fire frequencies have changed. As noted by Florence (1996) inhibition of understorey development is conceivable in the absence of disturbance for long periods, under a complete and healthy tree canopy.

Applicable evidence from the literature to provide a convincing argument that extended periods without fire may have led to widespread canopy decline in tuart forests and woodlands, is lacking. In fact, representations of some of the more intact canopies can be found in areas that have remained long unburnt (Woodman Point and Leschenault Peninsula). Given the strong reliance on fire for tuarts to regenerate by seed (Ruthrof et al. 2002), it logically follows that some period without fire (related to tuart longevity) would be sufficient to lead to stands with a greater proportion of over-mature trees more vulnerable to decline or simply senescing naturally – cohort senescence theory (Mueller-Dombois 1992). However, for Yalgorup this process can be largely dismissed due to the rapid rate of decline and the range of age classes affected. Until experimental data are forthcoming, speculation will continue on the link between fire and tuart canopy decline. A more detailed knowledge of the historical patterns of change in the vegetation will assist in directing experimental work into the causes of tuart decline.

3 METHODS

A selection of vegetation monitoring plots established by John Fox and his Ecology students from Curtin University in 1976 (Table 1) was reassessed in 2004 using the original assessment methodology. Height and diameter of trees (stems greater than 20 cm diameter) were recorded within a circular area of 500 m^2, while species and percentage cover (projected crown) data were collected for shrubs nested within a 100m^2 circular area of the larger area. Plots selected for re-measurement met all of the following criteria: located in tuart woodland, readily accessible, reliable original data were available and each original plot centre was able to be located. Thirteen plots met these criteria and were measured in the same way as in 1976/1977. The plots were located in several clusters along 4 transect lines in different areas of the National Park. Aerial photographs covering one transect of plots (36 – 40) at various times from 1957 to 2003 were also available for comparison at the time of the study.

4 RESULTS AND DISCUSSION

A change in the dominant tree species occurred in at least one of the plots within each of the four transects (Table 1). In each case *A. flexuosa* either replaced *Banksia attenuata*, *E. gomphocephala* or *Allocasuarina fraseriana* as the dominant tree species, or became the dominant tree species where none was present before (within the 500m^2 monitoring plot and employing the definition of tree used in the original survey). Given, the presence of *A. flexuosa* as a seedling or sapling in the treeless plots in 1977 (141,143 and 144) and the fire event of 1976, this change probably reflects the recovery and development of these individuals rather than a process of invasion. However, invasion or other major increases in density prior to the time of the initial surveys is a possibility. Two of the five plots in the study where *A. flexuosa* was absent in 1976 now have seedlings or saplings of this species growing within them, suggesting some expansion of the range of this species has in fact occurred between monitoring times. But, this phenomenon has not been widespread as plots 37, 38 and 52 (last burnt prior to 1965), all located within 100m of another plot where *A. flexuosa* was recorded in 1977, still did not have the species present some 27 years later. As seeds of this species tend to be produced annually on a mature individual (Bennett 1996) and are both small (0.1 x 0.03 cm) and winged (Boland et al. 1984), they could potentially be carried many tens of metres in the wind. Factors other than seed dispersal are likely to have limited the advance. A number of studies have emphasized the importance of environmental factors (particularly soil conditions) and disturbance regimes, in determining patterns of vegetation change within landscapes (e.g. - Callaway & Davis 1993, Burrows & Wardell-Johnson 2003). No relationship between fire history and changes in the abundance or cover of the species was apparent given the limited, unreplicated nature of the data.

A trend within some of the plots reflecting the wider decline of the tuart overstorey within the region, can be seen in the reduction of height of the tallest tree (*E. gomphocephala*) in plots 38, 39 and 40. In plot 40, an *E. gomphocephala* was the tallest tree in 1977 (19 m), but crown dieback led to an *A. flexuosa* becoming the tallest tree in 2004 (15 m). Observations made in 1977 for these plots also noted the poor condition of some tuart crowns and referred to evidence of fire and borer damage (D. Ward pers. comm.). A scorch height up to 20 m was reported following the fire in the vicinity of plots 141 – 144 in April 1976 (F. Maso pers. comm.). The role of those fires that occurred after the formation of the Park, in the direct decline, or the predisposition to decline of tuart crown health (given that further decline has continued in the absence of fire), is worthy of further investigation.

A dramatic difference in the number of living *B. attenuata* trees across plots 36 – 40 was recorded during the study period. Total numbers fell from 19 to just five. This may simply reflect the senescence of mature individuals combined with a lack of recruitment opportunities. The plots have not been burnt for a considerable time (> 30 years) and fire is important in the recruitment of this species (Cowling & Lamont 1987). Further work is required to determine if such a decline is widespread and what possible factors may be involved. *Banksia* spp. are particularly susceptible to *Phytophthora cinnamomi* (Shearer et al. 2004, Wills 1993) and the chance that this or a similar plant pathogen was introduced at the time of the original surveys cannot yet be discounted.

The majority of plots had different species dominating the shrub layer in 2004 compared to in 1977. The most notable trend was the rise in dominance of tall growing species: *Spyridium globulosum* (plots 36 and 38) and *Templetonia retusa* (plots 40 and 144), both of which can attain heights of 4 m (Bennett 1988), and *Acacia rostellifera* (plots 141 and 143), which in the absence of fire develops into a tree (Powell and Emberson 1981). Such a pattern is not unexpected considering the length of time since fire last occurred across the study sites.

Bradshaw (2000) speculates that *A. flexuosa* may have come to dominate when fire frequency declined in the 1950s and 1960s. A time series of aerial photographs (Figure 1) covering the area surrounding plots 36 – 40, and commencing in 1957 (prior to the formation of the National Park), also provides support for the increased height and density of the understorey. The texture of the ground cover has become more pronounced over time, and this is most noticeable in the large open

Table 1. Fire history and changes in vegetation attributes for monitoring plots between 1976/1977 (normal type) and 2004 (bold type). Dominance defined as greatest projected (crown) cover. Absence of tree species denoted by "-". Braun-Blanquet cover classes[1] in brackets (last two columns) where available.

Plot	Position[2] Northing Easting	Date of last fire	Trees			Shrubs
			Dom. species	Max. ht (m)	Saplings/ Seedlings[6]	Dom. species
36	6362581 375743	Pre-1960[3,4]	Ba **Ba**	Ba (11) **Ba (15)**	Af (1) Ba (1) **Af (+)**	Hh (1) **Sg (2)**
37	6362594 375870	Pre-1960[3,4]	Eg **Eg**	Eg (9) **Eg (9)**	- **-**	Ma (2) **Ma (2)**
38	6362613 376016	Pre-1960[3,4]	Eg **Eg**	Eg (14) **Eg (9)**	- **-**	Ma (2) Gt(2) **Sg (2)**
39	6362588 376247	4/72[3]	Eg Ba Af **Eg Ba Af**	Eg (15) **Eg (13)**	Af (+) Ba (+) **-**	Hh (1) Mr (1) **Hh (1) Mr (1)**
40	6362600 376350	4/72[3]	Ba **Af**	Eg (19) **Af (15)**	Af (1) **Af (1)**	Hh (2) Mr (2) **Tr (2)**
51	6366198 373831	Pre-1965[3]	Eg **Eg**	Eg (21) **Eg (22)**	Eg (+) **Eg (1)**	Ap (1) **Xp (2)**
52	6366128 373828	Pre-1965[3]	Eg **Af**	Eg (20) **Eg (21)**	Af (2) **Af (2) Eg (1)**	Gt (1) **Xp (2)**
141	6380773 372576	1990[5]	- **Af**	- **Af (13)**	Af (2) Ar (1) **Af (+) Ar (2)**	Hh (1) **Hh (1)**
143	6380930 372700	1990[5]	- **Af Ar**	- **Af (10)**	Ar (1) **Af (1) Ar (3)**	Ap (+) Hh (+) **Hh (1)**
144	6381006 372772	1990[5]	- **Af**	- **Af (10)**	Af (1) **Af (3)**	A (1) Gt (1) **Tr (2)**
138	6382096 372746	1990[5]	Alf **Af**	Alf (12) **Eg (13)**	Eg Afl Af **Alf Af**	Hh **Hh**
139	6382086 372852	1990[5]	Eg **Eg**	Eg (17) **Eg (20)**	- **Eg (+) Alf (1) Af (1)**	Hh **Hh**
140	6382064 372950	1990[5]	Alf **Alf**	Alf (8) **Alf (14)**	- **Af (+)**	Hh **Hh**

Notes: Af (*Agonis flexuosa*), Alf (*Allocasuarina fraseriana*), A (*Acacia spp.* unknown), Ap (*Acacia pulchella*), Ar (*Acacia rostellifera*), Ba (*Banksia attenuata*), Eg (*Eucalyptus gomphocephala*), Gt (*Grevillea thelemanniana*), Hh (*Hibbertia hypericoides*), Ma (*Melaleuca acerosa*), Mr (*Macrozamia riedlei*), Sg (*Spyridium globulosum*), Tr (*Templetonia retusa*), Xp (*Xanthorrhoea preissii*)

[1] Braun-Blanquet classes for projected cover: + = < 1%, 1 = 1-5 %, 2 = 6-25 %, 3 = 26–50 %, 4 = 51– 75 %, 5 = 76-100%; [2] UTM zone 50 S; [3] Smith (1975); [4] From reference to student observations in 1976, assumed unburnt in 1972. These plots occur near the boundary of the 1972 fire reported by Smith (1975); [5] S. Dutton (Department of Conservation and Land Management) pers. comm.; [6] Within 100m^2 zone of monitoring plot.

area between the woodland formations extending from the lakes. Additionally, light patches (assumed to represent bare ground/herbs/leaf litter) between tree crowns within the woodlands in the photos from the earlier periods are less apparent in 1991 and virtually absent in 2003. Thus, for the area covered by these photographs, it appears that much of the increased understorey growth occurred from the mid-1970s. Distinguishing the influence of grazing and fire regimes from the changes between 1957 and 1976 is not possible, but the increasing cover and absence of grazing for the latter time periods suggests an increase in shrub cover occurred as fire frequency declined. Actually identifying an association will require an examination of other comparable areas with contrasting fire histories.

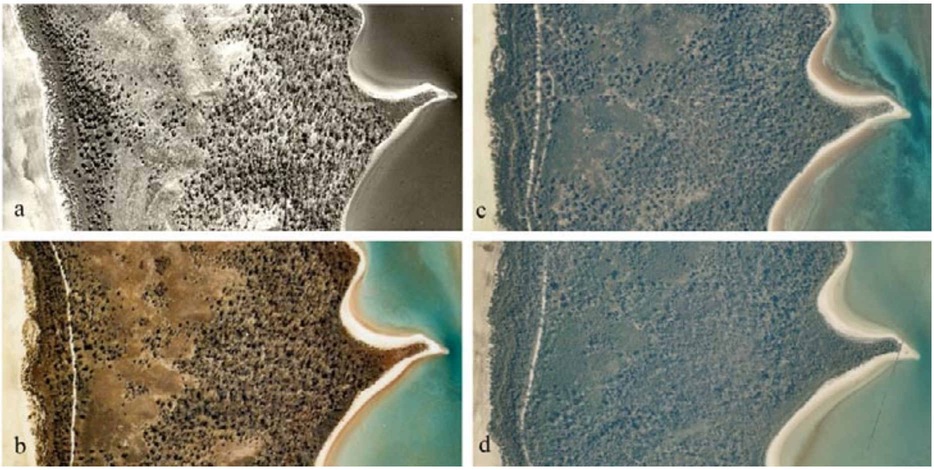

Figure 1. Aerial photographs of land between Lake Preston (left) and Lake Yalgorup (right) taken in a) 1957 b) 1976 c) 1991 and d) 2003. Plots 36 – 40 located in this area. (Reproduced by permission of the Department of Land Information, Perth, Western Australia, Copyright Licence 38/2004)

4.1 *Future Research Directions*

With the availability of aerial photographs at 1:25 000 scales or less over a range of years and reasonably good records of the fire history since the National Park was formed, there is considerable potential to test for an association between fire, vegetation structural change and tuart canopy health. Fensham et al. (2002) have demonstrated a robust method enabling the quantification of vegetation cover change with image scales up to 1:40 000. The availability of other data such as soil type, offers the further possibility to explore interactions between vegetation change, fire and other environmental factors. Data collection in this way would represent a superior approach to the standard "space for time" method in isolation. In conjunction with this, on-ground experimental work is being established to examine the response of tuart, co-occurring tree species and the understorey to fire intensity as well as competition and post-fire grazing effects on seedling regeneration.

A broader project investigating tuart tree decline in Yalgorup and elsewhere has commenced. It acknowledges that the causes of tree decline are rarely simple or single-factored. Preliminary work suggests there is no single cause of the decline/declines across the distribution of tuart (Longman & Keighery 2002, Edwards pers. comm.). Primary and secondary agents, complex interactions and time lags between cause and effect are all possible. Collaboration between research groups and the co-ordination of research across several areas: fire, insects, fungal pathogens, nutrition, hydrology and water relations, represents the best hope for gaining the knowledge necessary to undertake any remedial work or prevent any further decline.

5 CONCLUSION

This study supports observations that the understorey in parts of Yalgorup National Park has in fact increased in height and density and historical aerial photography suggests that the trend began before the onset of the major episode of tuart decline. A fuel structure with the potential to lead to further acute and possibly long term canopy damage from wildfire has developed. With current knowledge it seems that a continued commitment to deliberate, controlled burning could have contributed to a more stable vegetation structure in parts of the park by reducing the incidence of intense fires which result in tuart canopy damage and peppermint stem proliferation. Nevertheless, evidence from this and other studies indicates increasing understorey densities under a range of possible fire regime scenarios. Further work is needed to quantify the understorey changes pre-decline and post-decline, and clarify the association (and mechanism if an association is present) between understorey, fire and tuart canopy decline. Active management of fire regimes will undoubtedly be an essential component of any future initiatives to restore and maintain healthy tuart woodland in the Park.

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