Research reports

The impact of the European olive (*Olea europaea* L. subsp. *europaea*) on grey box (*Eucalyptus microcarpa* Maiden) woodland in South Australia

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Summary

The European olive (*Olea europaea* L. subsp. *europaea*) is an introduced plant that has naturalized in several areas of Australia. Originating from the Mediterranean region, *O. europaea* was imported to Australia and cultivated for its oil content. Escapees from cultivated olive groves have invaded native vegetation communities, including grey box (*Eucalyptus microcarpa* Maiden) woodland of conservation significance in South Australia. This paper examines the history of and processes leading to *O. europaea* invasion, the impact of *O. europaea* invasion on *E. microcarpa* woodland, and the relevance of olives as an environmental weed. Results of this study show that *O. europaea* invasion leads to a reduction in native species richness and abundance in *E. microcarpa* woodland. Invasion also alters the canopy structure of the woodland. Species most at risk are *E. microcarpa*, golden wattle (*Acacia pycnantha* Benth.) and sticky hop-bush (*Dodonaea viscosa* Jacq.). It is argued that the reduction in light infiltration caused by the dense canopy of *O. europaea* prevents regeneration of native species. *Olea europaea* should be considered a serious threat to the integrity of vegetation communities it invades.

Introduction

Introduced plants that invade native vegetation outside their normal ecological range can alter ecosystem-level functions, thereby adversely affecting habitat conditions or resource availability (Adair and Groves 1998). These plants, termed environmental weeds, threaten nearly all biological communities in Australia (Adair and Groves 1998) and have been described by Carr *et al*. (1986, p.150) as 'the greatest conservation problem in Australia'.

Quantitative studies have shown that environmental weeds impact on the biological diversity of the indigenous vegetation communities they invade by reducing species richness, abundance, and/or canopy cover of native species (Gleadow and Ashton 1981, Waterhouse 1988, Braithwaite *et al.* 1989, Fensham *et al.* 1994, Mullett and Simmons 1995, Rose and Fairweather 1997, Csurhes and Edwards 1998). Mullett and Simmons (1995), for example, report that increased cover and abundance of the environmental weed sweet pittosporum (*Pittosporum undulatum* Vent.) was associated with a reduction in local native species cover and diversity of 90 and 50%, respectively, in southern Victoria. In severe cases of *P. undulatum* invasion, native species may become extinct (Cronk and Fuller 1995). The conservation of species diversity is foremost in the conservation objectives for natural ecosystems and is the basis on which decisions concerning management and resource allocation are made (Adair and Groves 1998). Despite this, the impact of environmental weeds on biological diversity has not been thoroughly studied in Australia (Adair and Groves 1998).

In South Australia, the threat posed by environmental weeds on the biological diversity of remnant box grassy woodlands in temperate regions of the State has become a major conservation concern (Robertson 1993, 1996; Davies 1995, 1997). Box grassy woodlands, dominated by the rough-barked eucalypts, are some of the most poorly represented vegetation communities in South Australian National Parks and Wildlife managed reserves (Davies 1997). One such community, grey box (*Eucalyptus microcarpa* Maiden) woodland, is considered to be in urgent need of conservation (Neagle 1995). The open woodlands have been favoured by sheep graziers for the grasses and small herbaceous shrubs that thrive between the open canopy of the dominant eucalypts (Moore 1959, Gillison 1994). Hard-hooved grazing stock have altered the ecological processes operating in many of the box woodlands, paving the way for invasion by environmental weeds. These weeds endanger rare and threatened plant species that have been conserved in open box woodland reserves where grazing has now ceased (Davies 1997).

The European olive (*Olea europaea* L. subsp. *europaea*) is an invasive plant that represents a threat to the integrity and biological diversity of a substantial portion of the remaining patches of *E. microcarpa* woodland in South Australia. *O. europaea* has a detrimental impact on invaded native vegetation communities in South Australia (Specht and Cleland 1961, National Parks and Wildlife Service 1989, 1990a; Humphries *et al.* 1991, Parsons and Cuthbertson 1992, Davies 1995, 1997), Victoria (Carr *et al.* 1992), New South Wales (Dellow *et al.* 1987, Parsons and Cuthbertson 1992), Western Australia (Hussey *et al.* 1997) and south-east Queensland (Csurhes and Edwards 1998). However, the specific nature of the impact is unclear. Parsons and Cuthbertson (1992, p. 523) suggest that *O. europaea* 'alters the composition of native vegetation it invades thereby increasing the fire hazard and reducing the recreational value of parklands'. Davies (1995) lists *O. europaea* as a threat to six populations of the nationally endangered orchid (*Pterostylis cucullata* R.Br.), because of its ability to out-compete native species for ecological resources. Other authors describe *O. europaea* as a threat to native vegetation communities and argue that control is required but do not provide any evidence of its impact. This paper seeks to detail the impact of *O. europaea* in *E. microcarpa* woodland, thereby addressing the current lack of specific knowledge.

Origin and introduction of O. europaea

It is likely that *O. europaea* has been cultivated in the Mediterranean Basin since the early days of civilization, with evidence indicating that the ancient Egyptians used fruits, oil and wood of the olive tree in pre-Dynastic times (Levinson and Levinson 1984). *O. europaea* is endemic to the Mediterranean region but because it has been cultivated there for so long it cannot be safely stated that it is indigenous to the region (Parsons and Cuthbertson 1992). *O. europaea* is recorded as one of the earliest plant introductions into Australia. Reichelt and Burr (1997) suggest that *O. europaea* was first introduced into Sydney in 1800. The pioneering pastoralist, John Macarthur, is known to have planted *O. europaea* at Elizabeth Farm, Parramatta in 1805 (Dellow *et al*. 1987).

The popularity of the oil and fruit of *O. europaea* was the reason for its introduction into South Australia (South Australian Animal and Plant Control Commission (SAAPCC) 1990, Reichelt and Burr 1997). South Australia's first Governor, John Hindmarsh, brought with him an *O. europaea* tree upon arrival in 1836 (Reichelt and Burr 1997) and in 1844 five more were imported (SAAPCC 1999). In the 1870s olive production in South Australia was considered to be a more viable enterprise than wine production. This led to the establishment of a three-acre grove at Magill in 1874, which was expanded to over 100 acres by 1882, and other groves at Beaumont and Glen Osmond (Reichelt and Burr 1997). By the early 1900s, however, the olive industry in South Australia was in decline. In an attempt to revive the olive industry the South Australian Department of Agriculture established the Blackwood experimental orchard in 1908, planting 27 different varieties of *O. europaea* (Reichelt and Burr 1997). In the 1920s the decline in the industry worsened, forcing the Department of Agriculture to offer an annual bonus of ten shillings per acre over a ten year period to anyone who cultivated *O. europaea*. Interest in olives increased during the late-1940s and 1950s coinciding with a large number of immigrants from Mediterranean countries (Taylor and Dick 1999). By 1958/59, 3000 ha were under cultivation in South Australia and Victoria (Taylor and Dick 1999).

Current distribution of O. europaea in South Australia

Olea europaea is naturalized across a wide range of habitats in South Australia (Figure 1). Occurrences are predominantly within the 400 to 600 mm median annual rainfall range. The highest concentrations occur on the western foothills of the southern and central Mount Lofty Ranges. These populations are likely to be the direct descendants of *O. europaea* cultivated in the Magill, Glen Osmond, Beaumont and Blackwood regions. The naturalization of *O. europaea* in the southern and central Mount Lofty Ranges is not a recent event. As early as the 1920s Adamson and Osborn (1924) noticed the abundance of *O. europaea* on the dry and rockier slopes of the western foothills. Specht and Perry (1948) and Specht and Cleland (1961) made similar observations. Other large concentrations can be found in the Barossa and northern Mount Lofty Ranges. The occurrence of a small number of *O. europaea* on Yorke Peninsula and in the far east of the State suggests that the species can survive in regions that receive less than 500 mm average annual rainfall.

Olea europaea ecology and biology

Olea europaea is a member of the Oleaceae family. Both *O. europaea* and the African olive (*Olea. europaea* L. subsp. *cuspidata* (Wall. ex G.Don) Cif.) are recognized as invasive plants in Australia (Dellow *et al*. 1987, Csurhes and Edwards 1998). Subspecies *cuspidata* is not as widespread in Australia but is still a problem in New South Wales (Muyt 2001). *O. europaea* is an evergreen plant with a stocky trunk and grey bark as well as an extensive root system. Specimens may reach a height of >10 m (Levinson and Levinson 1984, Parsons and Cuthbertson 1992, Hussey *et al.* 1997), with many branches forming into a dense, rounded crown. Its opposite leaves are narrow, lanceolate to oblong (2–8 cm long, 1–3 cm wide), subdued green above and whitish grey beneath. Leaf renewal occurs at regular intervals of two to three years (Levinson and Levinson 1984).

The small, white, hermaphrodite flowers (4–6 mm in diameter, carried in upright clusters) are pollinated during the flowering period (Table 1) by wind as well as insects, and are followed by green ovoid fruits (2–2.5 cm long, 1–2 cm wide) which ripen through purple to black. Approximately three per cent of the available flowers develop into fruit (Levinson and Levinson 1984). Flowers develop in axillary racemes on the previous year's wood, with fruit ripening from April to July. In many cases, fruits from naturalized plants are smaller than those from cultivated specimens (Mladovan 1998). Fruits contain a single seed dispersal unit (embryo plus endosperm) wrapped in a hard endocarp. Ripe fruit will often remain on the tree well into spring (Crossman personal observation). Fruit production of wild *O. europaea* in Spain follows a supra-annual cycle of two or three years, depending on climatic conditions (Rey and Alcantara 2000). Rey and Alcantara (2000) report a seed dormancy of up to 20 months, with epigeal germination occurring in the second winter after dispersal. Approximately one year after germination, the seedling acquires adult vegetative traits (woody stems and wax-covered leaves) and becomes a sapling (Rey and Alcantara 2000). Reproductive age is achieved after more than eight years (Rey and Alcantara 2000). Parsons and Cuthbertson (1992) report that *O. europaea* can also reproduce vegetatively from cuttings.

Olea europaea prefers regions with a Mediterranean climate with cool winters and warm summers. Although the tree is hardy and possesses drought resistant

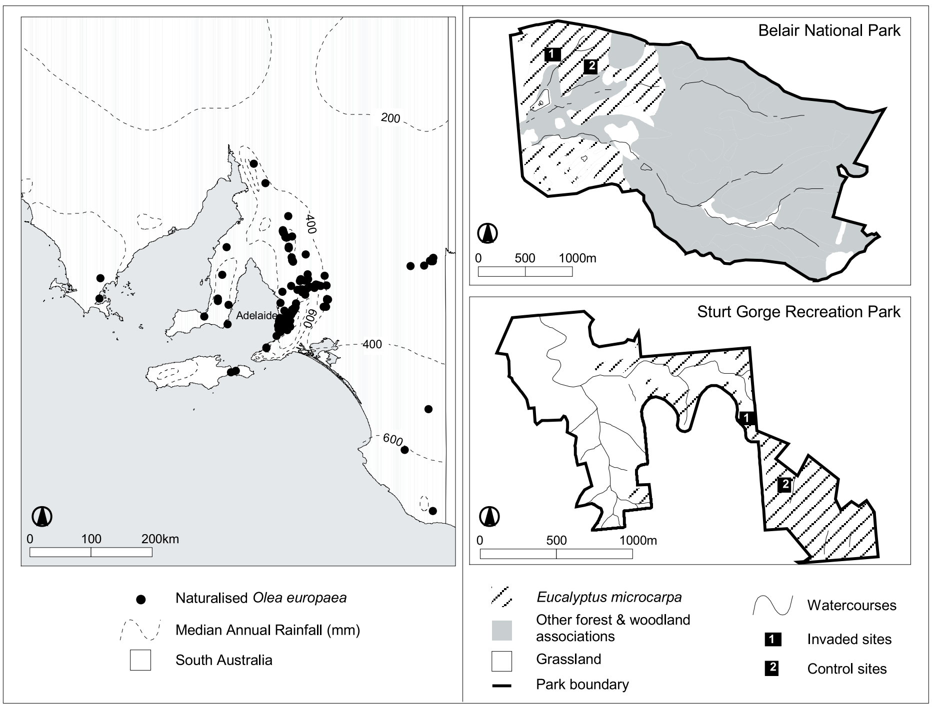

Figure 1. Current distribution of naturalized *Olea europaea* in South Australia (left) and study areas (right).

Table 1. Phenological timetable, by month, for naturalized *Olea europaea* in South Australia (solid bars represent peak times). Source: Costermans 1983, Crossman personal observation, Parsons and Cuthbertson 1992.

qualities, flowers and fruit will not form without sufficient winter and spring rainfalls (Taylor and Dick 1999). Judging by the naturalized distribution of *O. europaea* (Figure 1), average annual rainfall of approximately 500 mm is a minimum requirement for successful reproduction. *O. europaea* can grow in a wide range of soils but favours well drained, neutral to alkaline (pH of 6.5 to 8.5) sandy loams, loams and clay loams (Olives Australia 1999, SAAPCC 1999, Taylor and Dick 1999).

Dispersal of *O. europaea* seed is almost wholly dependent on birds. Jupp *et al*. (1999) found that 17 species of birds included olives in their diet, although only seven dispersed *O. europaea* seed. The Common Starling (*Sturnus vulgaris* L.) is the most prevalent disperser: a common sized flock of 100 could disperse thousands of seeds in a day (Mladovan 1998). Parsons and Cuthbertson (1992) suggest that avian frugivores defecate *O. europaea* seeds at locations away from the source. There is evidence to suggest that Emus (*Dromaius novaehollandiae*) disperse seeds in this manner (Jupp *et al*. 1999). However, Paton *et al*. (1988) and Jupp *et al*. (1999) believe that it is more common for a bird to depit the olive while consuming it at a location distant to the source rather than defecate the seed. The European fox (*Vulpus vulpus*) consumes *O. europaea* seed from naturalized plants in the southern Mount Lofty Ranges with seeds frequently found in *V. vulpus* scats during the winter months (Paton *et al*. 1988). The nursery trade and widespread cultivated orchards of *O. europaea* are responsible for long distance dispersal. The recent boom in the Australian olive oil industry accounts for the large olive orchards that have recently been planted in south-east Queensland, Larenta (100 km north of Perth) and Coonalpyn and Pinaroo (180 km south-east and 250 km east of Adelaide respectively) (Crabb 1999; Haran 1999; McIlwraith 1999). These areas have, until now, been free of *O. europaea*. An estimated 6.6 million olive trees have been sold or ordered in Australia between 1999 and 2001 (Australian Broadcasting Corporation, 1999).

Methods

Study area

Study sites were located in stands of *E. microcarpa* woodland within Sturt Gorge Recreation Park and Belair National Park (Figure 1). Two slightly different *E. microcarpa* woodland communities, as described by Williams and Goodwins (1988); Crossman (1999), were represented within the study area. *E. microcarpa* woodland at Belair has a higher level of native plant species richness and abundance, particularly in the ground stratum which is dominated by grasses and herbs. In both communities, the overstorey is dominated by *E. microcarpa*, with the presence of drooping sheoak (*Allocasuarina verticillata* (Lam.) L.Johnson) at Belair. The understorey in both communities supports shrubs such as golden wattle (*Acacia pycnantha* Benth.), kangaroo thorn (*Acacia paradoxa* DC.) and twiggy daisy-bush (*Olearia ramulosa* (Labill.) Benth.). Common grasses and herbs include native cranberry (*Astroloma humifusum* (Cav.) R.Br.), pointed mat-rush (*Lomandra densiflora* J.Black) and kangaroo grass (*Themeda triandra* Forsskal). Common exotics include *O. europaea*, boneseed (*Chrysanthemoides monilifera* (L.) Norlindh ssp. *monilifera*), bridal creeper (*Asparagus asparagoides* (L.) Wight), and quaking-grass (*Briza maxima* L.).

Annual rainfall varies from 600 mm at the drier and warmer Sturt Gorge sites to 750 mm at the higher altitude Belair sites. Average January temperature maxima range from 26.6 to 27.8°C and the average July minima are between 7.0 and 7.7°C (Bureau of Meteorology 1999). Soils at the Belair sites are predominantly acid to neutral, grey loamy sands with clayey subsoils formed on unconsolidated sediments (NPWS 1989). At the Sturt Gorge sites, soils are predominantly shallow to moderately deep gravelly and stony loams to clay loam forming on weathering Tapley's Hill Shale (NPWS 1990a,b).

Site selection

At both Sturt Gorge and Belair, two study sites were selected: one control site where *O. europaea* presence was limited to isolated individuals, and one heavily invaded site. Aspect at all four sites ranged from northeast through to east. Slope was less than 15 degrees. Sites with similar aspect and slope were selected to minimize edaphic variability. The control sites had relatively low levels of disturbance. Both control sites are located in *E. microcarpa* woodland that has high conservation value (Neagle 1995; Davies 1997).

Data collection

At each site, 6 × 50 m transects were aligned parallel along the contour. Along each transect, canopy intercepts (Brower *et al*. 1990) of herbs, shrubs and trees were recorded for three strata: upper-stratum >8 m; mid-stratum 2–8 m; and lower-stratum <2 m. An estimate of the opaqueness of *E. microcarpa* crowns (Walker and Hopkins 1990) was made under 15 randomly selected trees at control sites. Similarly, an estimate of *O. europaea* opaqueness was made under 15 randomly selected trees at the invaded sites. Species richness and abundance were recorded using 25 × 1 m^2 quadrats equally spaced along each 50m transect. The species and number of individuals were recorded for all native and introduced herbaceous and woody perennials. Nomenclature follows Dashorst and Jessop (1990) and Jessop (1993). Ten 1 m^2 quadrats were placed along each sample transect at 5 m intervals. The percentage of grass cover (native and introduced) and bareground/leaf litter was estimated using a modified Braun-Blanquet scale: 0, 1 (1–10%), 2 (11–25%), 3 (26–50%), 4 (51–75%), 5 (76–100%).

Data analysis

To assess the impact of *O. europaea* invasion on native species diversity, *E. microcarpa* woodland diversity levels (H') at the control and invaded sites were compared using the Shannon-Wiener diversity index (Zar 1999). Tests of significance for H' was determined using a *t*-test (Zar 1999). To determine the impact of *O. europaea* invasion on canopy cover of native species, linear and relative coverage indices (Brower *et al*. 1990) were calculated on the canopy cover data to establish differences between control and invaded sites. Linear coverage index (LC) is the proportion of the total transect length intercepted by a species' canopy. Relative coverage index (RC) is the proportion of the total canopy in a stratum that belongs to a species. The possible range of values for both linear and relative coverage indices is from zero to one. Where LC = 1, 100% of the total transect length is intercepted by a species' canopy. Where RC = 1, that species is the only species recorded in that stratum. Tests of significance for LC were determined using a Mann-Whitney U-test. Foliage cover, the percentage of the site occupied by the vertical projection of foliage and branches, was calculated by multiplying the linear coverage index value with the corresponding mean opaqueness for *E. microcarpa* or *O. europaea* (adapted from Walker and Hopkins 1990).

Results

Olea europaea population characteristics

At Sturt Gorge, 66% (n = 150) of 1 m^2 quadrats contained *O. europaea*. Proportions at Belair were considerably lower at 23% (n = 150). This frequency pattern is reflected in the density of *O. europaea* across the invaded sites. At Sturt Gorge, seedlings accounted for 98% (n = 928) of the total at an average density of 6.0 m^{-2}. At Belair, seedlings accounted for 73% (n = 113) of all plants at a density of 0.5 m^{-2}. The densities of mature *O. europaea* at all invaded sites were <0.4 m^{-2}.

Canopy cover of *O. europaea* was most prevalent in the mid-stratum (Table 2). Over one third of the total transect length of 300 m at Sturt Gorge was covered by *O. europaea* canopy stemming from individuals 2–8 m in height. This result is highlighted by a RC of 0.76, indicating that few other species grow in the mid-stratum. At Belair, the total *O. europaea* canopy in the mid-stratum was similar with a RC of 0.81. *O. europaea* rarely exceeds 8 m in height accounting for low LC and RC values in the upper-stratum. At Sturt Gorge, the dominance of *O. europaea* in the lower-stratum is explained by the abundance of *O. europaea* seedlings.

The foliage cover of *O. europaea* in the mid-stratum of the invaded sites varied according to the LC value. The mean opaqueness of *O. europaea* was very high. For Sturt Gorge and Belair the mean opaqueness of *O. europaea* was 82 and 77% respectively. Thus, foliage cover (LC* opaqueness) of *O. europaea* at Sturt Gorge and Belair was 28 and 15% respectively. Even though foliage cover appears low, the high opaqueness values suggest that the canopy of *O. europaea* produces intense shading.

Native species richness and abundance

At Sturt Gorge, differences in native species abundance and richness between the control and invaded sites were not as large as at Belair (Table 3). A higher frequency of quadrats at the control site contained natives. The total abundance of natives was also higher. *A. pycnantha* was the most common species in the control site. Native species richness was identical with a total of eight different species at each site. Species such as *A. pycnantha*, native lilac (*Hardenbergia violacea* (Schneev.) Stearn) and the dwarf green-hood orchid (*Pterostylis nana* R.Br.) were better represented at the control site. The rock fern (*Cheilanthes austrotenuifolia* H. Quirk and T.C. Chambers) was the only abundant native species at the invaded site. Grass cover was considerably lower at the control site (Table 3). Bareground/leaf litter was similar at both sites. Overall, the diversity of *E. microcarpa* woodland at the Sturt Gorge control site (H′ = 0.605) was significantly greater (t = 1.72, d.f. = 219, P <0.05) than at the corresponding invaded site (H′ = 0.498).

At Belair, a considerable difference existed between the richness and abundance of native species at the control and invaded sites. Four in every five quadrats at the control site contained at least one native species compared to only one in every five at the invaded site. Over twice as many native species were recorded at the Belair control site indicating that there was greater species richness than at the invaded site (Table 3). Similarly native species abundance at the control site was noticeably higher. Over 2.5 times as many native individuals were recorded in the quadrats at the control site compared to the invaded site. Species such as sticky hop-bush (*Dodonaea viscosa* Jacq.), *A. pycnantha*, *Hibbertia* spp., *A. verticillata*, brush heath (*Brachyloma ericoides* (Schldl.) Sonder), *O. ramulosa* and *P. nana* were counted on more than ten occasions at the control site. These species did not appear at all (apart from one *A. pycnantha*) at the invaded site. However, there was a considerable increase in the abundance of some native species at the invaded site. Several populations of the herb sheeps burr (*Acaena agnipila* Gand.) were recorded. Peach heath (*Lissanthe strigosa* (Smith) R. Br.) and *L. densiflora* were also better represented at the invaded site. Values for grass cover were higher at the control site whilst values for bareground/leaf litter were higher at the invaded site (Table 3). The diversity of *E. microcarpa* woodland at the Belair control site (H′ = 0.987) was significantly greater (t = 11.41, d.f. = 359, P <0.0005) than at the corresponding invaded site (H′ = 0.594).

Canopy cover

Results from only the Sturt Gorge sites are presented in this paper as they are representative of *O. europaea* impacts upon canopy cover recorded at both study sites. The values of LC and RC and the frequency of canopy cover for *E. microcarpa* and *A. pycnantha* in the upper- and mid-strata respectively are shown in Table 4. *E. microcarpa* was the only species recorded in the upper-stratum at the control site (RC = 1.00). The high LC value of 0.87 reflects this dominance. That is, 87% of the total transect length was covered with *E. microcarpa* in the upper-stratum. In sharp contrast, the total intercept length of *E. microcarpa* canopy along the combined transect samples at the invaded site was one fifth of that at the control site. This was a significant reduction (U = 36, p <0.0025). *E. microcarpa* was still the dominant species (RC = 0.77), having to share the upper-stratum with *O. europaea* (RC = 0.22, Table 2). The results for *A. pycnantha* in the mid-stratum were similar. At the control site, *A. pycnantha* was clearly the dominant species (RC = 0.84) in the mid-stratum, with 19% of the total transect intersected by its canopy. At the invaded site, the total length of *A. pycnantha* canopy cover was 75% lower than at the control site. This reduction was also significant (U = 29, p = 0.05). *O. europaea* (RC = 0.76, Table 2) was the dominant mid-stratum species.

The dominance of *E. microcarpa* in the upper-stratum along a typical transect sample at the Sturt Gorge control site is clearly indicated in Figure 2. The mean opaqueness of *E. microcarpa* canopy was 49%, giving an upper-stratum foliage cover of 43% (0.87 of 49%). This value of foliage cover is high for a woodland, although enough light passes through the canopy to allow strong *A. pycnantha* growth. *A. pycnantha* grows successfully along the first 25 and last 15 metres of the transect (Figure 2). Note the presence of a few *O. europaea* in the mid- and lower-strata along this particular transect, indicating an early stage of invasion. In complete contrast, Figure 3 shows the dominance of *O. europaea* in the mid-stratum along a transect sample that represents a late stage of invasion. Notice the almost complete absence

Table 2. Linear and relative coverage indices for *Olea europaea* canopy within invaded sites (LC = proportion of the total transect length intercepted by a species, RC = proportion of the total canopy in a stratum that belongs to a species).

Site	Stratum	Linear coverage index (LC)	Relative coverage index (RC)
Sturt Gorge	Upper	0.05	0.22
	Mid	0.34	0.76
	Lower	0.07	0.86
Belair	Upper	0.01	0.01
	Mid	0.19	0.81
	Lower	0.00	0.10

Table 3. Summary vegetation data recorded across all study sites.

	Sturt Gorge		Belair	
	Control	Invaded	Control	Invaded
% of quadrats (n = 150) with native species	51	25	80	18
Total species	11	11	20	15
Native species	8	8	18	8
Native species abundance	140	108	437	165
Introduced species	3	3	2	7
Introduced species abundance	28	945	20	207
% introduced species	27	27	10	47
mode grass cover[A]	1	5	5	2
mode bareground/leaf litter[A]	5	5	1	5

[A] Modified Braun-Blanquet scale: 0, 1 (1–10%), 2 (11–25%), 3 (26–50%), 4 (51–75%), 5 (76–100%).

of *E. microcarpa* in the upper-stratum and *A. pycnantha* in the mid- and lower-strata. The presence of *O. europaea* some distance from *E. microcarpa* is also worth noting. The dynamics of bird dispersal of *O. europaea* suggests that plants should be found beneath or near the edge of *E. microcarpa* (Jupp *et al*. 1999). The structure of canopy in Figure 3 suggests that tree clearance or other vectors of dispersal, for example *V. vulpus* or gravity, may have contributed to the distribution patterns of *O. europaea*. It is unlikely that these *O. europaea* have been planted. Note also the greater levels of grass cover, compared to Figure 2.

The sample transects presented in Figures 2 and 3 were chosen as the best representatives of the complete transect length of 300 m surveyed at the Sturt Gorge sites.

Discussion

The results show a change in floristic composition within two communities of *Eucalyptus microcarpa* woodland invaded by *Olea europaea*. The comparatively low native species diversity of *E. microcarpa* woodland at Sturt Gorge meant that reductions in species richness and abundance following *O. europaea* invasion were not as significant. However, at Belair, native species richness and abundance were reduced by more than 50% in the biologically diverse *E. microcarpa* woodland. This level of reduction is comparable to Mullett and Simmons' (1995) study on *P. undulatum* invasion in Victoria and Waterhouse's (1988) investigation of Scotch broom (*Cytisus scoparius* L. Link) invasion in New South Wales.

In many cases, native species at the control sites were completely absent from the invaded sites. Species most at risk appear to be *A. pycnantha*, *A. paradoxa*, *A. verticillata*, *B. ericoides*, *D. viscosa*, *H. violacea*, *Hibbertia* spp., *O. ramulosa*, and *P. nana*. The orchid *P. nana*, while not listed as a threatened species under the Commonwealth Environment Protection and Biodiversity Conservation Act 1999 or the South Australian National Parks and Wildlife Act 1972, should still be considered as a plant which is vulnerable. Its low-growing habit means that it is especially threatened by environmental weeds. It also belongs to the same genus as two other orchids that are directly threatened by invasive plants, including *O. europaea*, in the Mount Lofty Ranges (Bates 1989, Briggs and Leigh 1995, Davies 1995, Sorensen and Jusaitis 1995). One of those orchids, *P. cucullata*, is located in Belair National Park.

The results also show a reduction and structural alteration of native species canopy cover in *E. microcarpa* woodland following *O. europaea* invasion. *O. europaea* invasion creates gaps in the upper-stratum canopy of *E. microcarpa* in a community

Table 4. Canopy cover variables for native species at Sturt Gorge (LC = proportion of the total transect length intercepted by a species; RC = proportion of the total canopy in a stratum that belongs to a species).

Species	Site	Linear coverage index (LC)	Relative coverage index (RC)
Upper-stratum			
E. microcarpa	Control	0.87	1.00
	Invaded	0.17	0.77
Mid-stratum			
A. pycnantha	Control	0.16	0.84
	Invaded	0.05	0.12

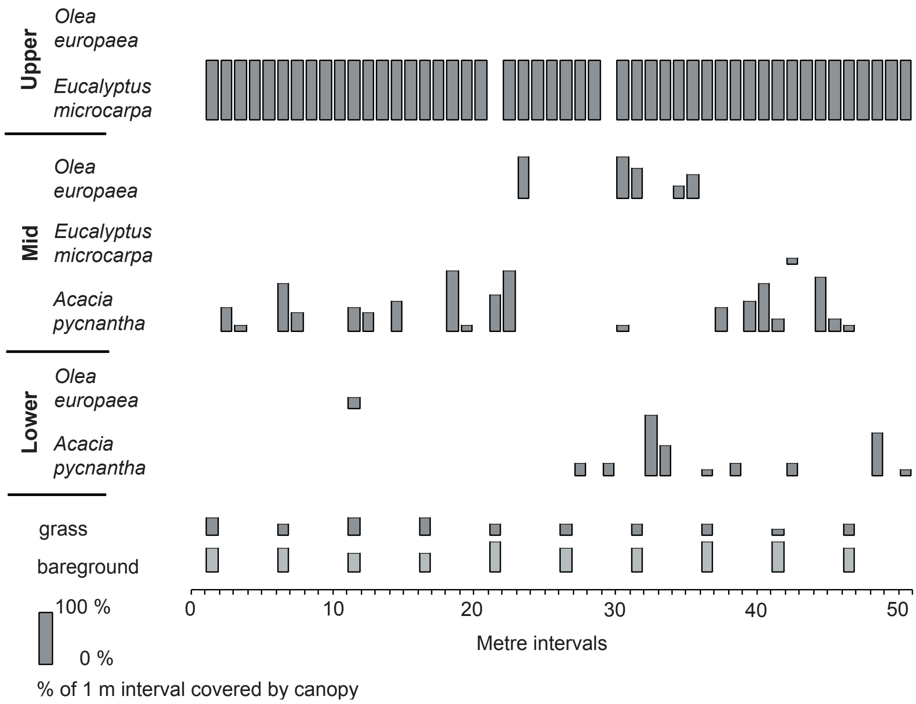

Figure 2. Representative diagram of canopy structure along transect sample 10 at the Sturt Gorge control site.

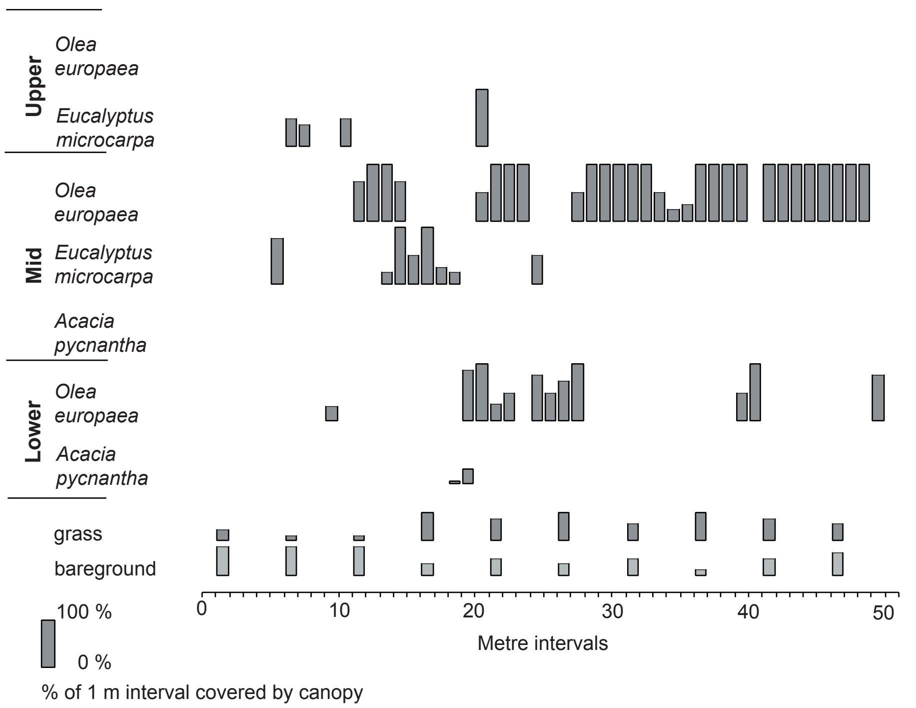

Figure 3. Representative diagram of canopy structure along transect sample 1 at the Sturt Gorge invaded site.

where almost no gaps existed prior to invasion. The canopy length of *E. microcarpa* in the upper-stratum and *A. pycnantha* in the mid-stratum were reduced by 80 and 70%, respectively, following *O. europaea* invasion at Sturt Gorge. At Belair, remnant *E. microcarpa* canopy was not as extensive, and, therefore, the reductions following invasion were not as large. With a lower percentage foliage cover, the more open *E. microcarpa* canopy allows for higher species diversity in the lower- and mid-stratums. Following *O. europaea* invasion, there is a greater reduction of native species.

Reduction of light infiltration from a highly opaque *O. europaea* canopy may be responsible for preventing native shrub and tree regeneration. The invasive plants *P. undulatum* (Gleadow and Ashton 1981, Mullett and Simmons 1995), thunbergia (*Thunbergia grandiflora* Roxb.) (Csurhes and Edwards 1998) and mimosa (*Mimosa pigra* L.) (Braithwaite *et al*. 1989) all produce deep shade under their canopies. It is possible that the canopy of *O. europaea*, whilst preventing regeneration of larger native species, may promote grass (native and introduced) and herb growth amongst the sparse ground stratum of *E. microcarpa* woodland found at Sturt Gorge. At these sites it was found that grass cover increased in *E. microcarpa* woodland invaded by *O. europaea*. Moisture may be retained longer in soil adjacent to and directly beneath *O. europaea* canopy as a result of shading.

Birds are the most common seed dispersal vector of *O. europaea*, leading to clumps of plants beneath *E. microcarpa* (Jupp *et al*. 1999). At the Sturt Gorge invaded site, the presence of *O. europaea* at a considerable distance from *E. microcarpa* canopy suggests other vectors of dispersal (e.g., humans, gravity, and/or *V. vulpus*) or a prevention of *E. microcarpa* regeneration. The eventual death of a eucalypt would traditionally be followed by con-specific replacement. The results suggest that the canopy of *O. europaea* has prevented this replacement, leading to extensive lengths of *O. europaea* canopy in the mid-stratum with no *E. microcarpa* above.

The floristic simplification and modification of *E. microcarpa* woodland following *O. europaea* invasion threatens its ecological integrity, thereby reducing the conservation value of the community. The typically open nature of *E. microcarpa* woodland in conjunction with the pressures of grazing and fragmentation has increased its vulnerability to *O. europaea* invasion. This increased vulnerability validates the decision to place the *E. microcarpa* woodland community on the list of vegetation associations that require an urgent upgrade in levels of protection. Evidence presented in this paper is consistent with research which has found other woodlands within regions of Australia that have a Mediterranean type climate are also at great risk from weed invasion. The suitability of climate, combined with the vulnerability of Eucalypt woodland to weed invasion and the rapid dispersal of *O. europaea* via birds and the booming olive oil industry, ensures that *O. europaea* will remain a threat across large parts of Australia. The findings confirm that, if left unchecked, extensive tracts of native vegetation will suffer substantial losses of biological diversity.

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