# Bridal creeper – squeezing the juice out of the citrus industry

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## Abstract
Bridal creeper, *Asparagus asparagoides* (L.) Druce, is largely regarded as an environmental weed, but to citrus growers, it is one of the worst weeds they have ever faced.

Citrus is an important horticultural industry and export earner for Australia. In 2000–01, the estimated gross value of production was $426 million, with exports valued at around $191 million. A total of 83% of Australia's citrus production occurs in southern Australia, mainly in the Riverland, Riverina and Sunraysia irrigation regions of South Australia, New South Wales and Victoria. Alarmingly, this entire region is under threat from bridal creeper invasion, which could pose a significant concern to Australia's citrus industry.

Bridal creeper has been rapidly invading citrus orchards causing a decline in tree health and fruit quality, interfering with harvesting and tree maintenance operations and increasing production costs. The continual invasion of the weed from infested shelterbelts, roadsides and bushland inflicts ongoing financial pressure on growers, which is estimated to cost growers an extra $2000 per hectare per year.

Following the initiation of a biological control program against bridal creeper in the late 1990s, growers were keen to use bridal creeper leafhoppers and rust within their orchards. Consequently, a pilot project funded by the Murray Valley Citrus Marketing Board and the Victorian Department of Primary Industries commenced in 2000 to determine if the biological control agents could persist within an intensively managed system.

The project demonstrated that the agents could establish within orchards and reach very damaging populations within a few years of release. A grower survey also revealed that pesticides and fungicides were not used as frequently as initially thought, and when they are used, they are mostly applied outside the periods of activity of the leafhopper and rust.

More recently, 'spore water' (a suspension of bridal creeper rust in water – see Overton and Overton, 2006) has been applied to citrus orchards using orchard sprayers, and an aerial application by plane has been trialled to achieve broader-scale delivery of the rust. Research into improving the spore water technique is still in progress.

More information about the biological control of bridal creeper in citrus orchards project can be found in Kwong and Clift (2004).

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# An eight year removal experiment measuring the impact of bridal creeper (*Asparagus asparagoides* (L.) Druce) and the potential benefit from its control

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## Summary
Bridal creeper is a weed with climbing annual shoot growth and extensive, underground storage tubers, and is capable of dominating native vegetation. **While its impacts appear obvious, this has been measured in few quantitative studies. In 1996, forty 3 × 3 m plots were established in a mallee remnant north of Adelaide, South Australia, to investigate this issue. Using glyphosate, bridal creeper was removed from half the plots in 1997, with follow-up treatment for the same plots in 1999.**

**In 2005 there was still no significant difference in the number of native plant species between plots with or without bridal creeper. There was also no significant difference in abundance of individual native species, except for the saltbush *Enchylaena tomentosa* ($P < 0.01$). However, there were consistent increases in cover of the chenopod and native grass understorey in the bridal creeper removed plots, even if not significant for some species. The common chenopods *E. tomentosa* and a combined dataset for *Rhagodia parabolica* and *R. candolleana* had greater shoot biomass where bridal creeper had been controlled ($P < 0.01$ and $P < 0.05$ respectively). An exotic plant, *Oxalis pes-caprae* also had higher cover in plots without bridal creeper compared to untreated plots ($P < 0.01$).**

**This study has shown that it may take many years for recovery following weed control and additional restoration work may be necessary. Dead tubers were still intact below the surface in the removal plots and their presence may be affecting seedling establishment. Recovery may also have been hindered by higher *O. pes-caprae* density. A third possibility is a lack of suitable environmental conditions in the eight year period for germination and establishment of indigenous species.**

**Keywords**: *Asparagus asparagoides*, environmental weeds, succession, weed substitution.

## Introduction
Plant invasions into natural ecosystems are a threat to indigenous biodiversity (Adair and Groves 1998). Exotic plant species that invade and impact on natural ecosystems are commonly referred to as environmental weeds (Humphries *et al*. 1991, Richardson 2001, Richardson *et al*. 2000). Managing environmental weeds requires knowledge of the impacts the weed has on indigenous communities and then

determining if indigenous communities can be restored following weed control (Gratton and Denno 2005). Predicting the influence of weed control on community dynamics is also needed when developing control strategies (Sheley and Krueger-Mangold 2003) and thought must be given to what is likely to occur after removal has been achieved (Hobbs and Mooney 1993). In Australia, few studies have measured the impact of environmental weeds (Adair and Groves 1998, Grice *et al.* 2004), with the emphasis being more on weed control, often with little consideration even to their ecological consequences (Williams and West 2000).

Of the studies that have investigated weed impacts, many have reported a reduction in indigenous plant species diversity following invasion (see Grice *et al.* 2004, Levine *et al.* 2003). Yet, the majority of these have been comparisons of community structure between invaded and weed free areas or across a gradient of weed density. For example, Mullet and Simmons (1995) reported a reduction in the abundance of indigenous species with increasing density of *Pittosporum undulatum* Vent. Sites dominated by the introduced grass, *Cenchrus ciliaris* L. (buffel grass) had fewer plant species when compared to sites without buffel grass (Jackson 2005) and in the US, Kedzie-Webb *et al.* (2001) measured plant species along a gradient of *Centaurea maculosa* Lam. (spotted knapweed). Indigenous perennial grass cover and species richness were inversely related to the spotted knapweed's cover. Although this method of documenting weed impacts allows rapid data collection, they are correlative studies and therefore lack the power of manipulative studies (Adair and Groves 1998, Grice *et al.* 2004). Woods (1997) suggested that there is a clear need for a distinction between correlation and the causes of reduced biodiversity. Although time consuming, weed removal experiments can give stronger evidence of impact, and they can also provide some indication of the long term changes that could occur after large-scale control (Adair and Groves 1998).

*Asparagus asparagoides* (L.) Druce (bridal creeper) mainly occurs as an understorey species in its native range in southern Africa and is usually found scrambling up other plants (Kleinjan and Edwards 1999). Within Australia, bridal creeper has the potential to dominate indigenous vegetation both above and belowground with its shoot biomass representing around 13% of the total biomass and extensive storage tubers found belowground (Raymond 1996). Most populations of the small endangered shrub, *Pimelea spicata* R.Br., in south-eastern New South Wales are threatened by bridal creeper (Willis *et al.* 2003). A vulnerable listed ground orchid in South Australia, *Pterostylis arenicola* M.A.Clem. & J.Stewart, is also under threat by bridal creeper (Sorensen and Jusaitis 1995) as both the weed and orchid emerge from underground organs in autumn (Vranjic *et al.* 2000). In Western Australia, it has also been reported that bridal creeper can germinate faster than the indigenous species *Clematis microphylla* DC. and therefore bridal creeper may also have an impact on this species (Fox 1984).

Even with the above studies on bridal creeper, most Australian research on bridal creeper has focused on the chemical and biological control of this weed and it now appears that the biological control program will go a long way in controlling bridal creeper (Batchelor and Woodburn 2002, Morin *et al.* 2006). But, fundamental to declaring successful management of bridal creeper is the need to investigate the impacts of bridal creeper on the whole plant community structure and determine if weed control alone is enough to enable restoration of these communities. Therefore, using the weed removal method, the aim of this study was to measure the impact of bridal creeper on perennial plant composition and abundance and hence to predict the benefit from the control of bridal creeper.

Grice *et al.* (2004) reviewed Australian studies that quantified weed impacts in native ecosystems. Of the 24 studies reviewed, none reported their impacts in terms of succession. When a weed has been establishment for some time, weed removal experiments do not measure their weed impacts *per se* but the response of the native community after weed removal (Adair and Groves 1998). Therefore experiments like the one reported here can predict the likely successionary pathways that could occur after weed control.

## Materials and methods

The study area was in a mallee vegetation community, within a small council reserve of remnant vegetation approximately 3 km north-west of Owen, South Australia 34°14′6″S, 138°31′8″E. The reserve was adjacent to a former school site and has had minimal amenity use in the past 30 years. Indigenous vegetation within the reserve is dominated by an overstorey of mature *Eucalyptus socialis* F.Muell. ex Miq., with mainly chenopod understorey. Soil is an alkaline clay loam. Rabbits were controlled during the period of the study by the Lower North Animal and Plant Control Board of South Australia.

Initially forty 1 × 1 m plots were chosen that had a dense coverage of bridal creeper and which contained an individual of both the most common shrub and groundcover species. The common shrub was either *Rhagodia parabolica* R.Br. or *Rhagodia candolleana* Moq. and the groundcover was *Enchylaena tomentosa* R.Br. Plots were then extended to 3 × 3 m, giving nine 1 $m^2$ subplots, with the two indigenous species within the central subplot. Bridal creeper was removed from half the plots on 1–3 October 1997 (with follow-up in spring 1999) using 33% Roundup® (360 g $L^{-1}$ glyphosate) with 2% Pulse Penetrant® (1020 g $L^{-1}$ polyether modified polysiloxane), sponge applied by hand to minimize off-target contact with native plants.

Between May 1996 and March 2005, vegetation was periodically sampled within the plots. From 1996, the number of individuals (or shoots) of each perennial plant species was recorded and from 1998 the percentage areal shoot cover was also recorded. The areal cover of the native trees, being *Pittosporum phylliraeoides* DC., *Santalum acuminatum* (R.Br.) A.DC., *Senna artemisioides* (Gaudich. ex DC.) Randell and *Eucalyptus socialis*, was estimated only below the height of 0.5 m. Percent areal cover was also recorded in September 2004 for the annual shoot growth of the exotic geophyte *Oxalis pes-caprae* L. (soursob), which was abundant within the study area. In addition to percentage covers, shrub shoot biomass was estimated using the 'Adelaide' hand-held unit technique (Andrew *et al.* 1979) for all portions of foliage directly above each plot, including that of plants that were rooted outside the plots. Statistical analysis was carried out using GenStat, Release 7.2 (GenStat 2003).

From the centre of each plot, soil cores below the tuber mats were taken in March 2005. Cores 5 cm deep and 5 cm in diameter were taken 5 cm below the litter layer. Cores were bulked for each treatment and any large organic matter was removed, such as tubers and other roots. Eight subsamples from each bulked sample were forwarded in airtight containers to CSBP laboratories for chemical analysis. Nitrogen ($NO_3^-$ and $NH_4^+$), extractable phosphorus, organic carbon and pH were measured.

## Results

In September 1999, October 2002 and September 2004 both the number of bridal creeper shoots and cover of bridal creeper were measured across all plots and these two measurements were found to be significantly correlated (Figure 1). Before treatments were applied, cover of bridal creeper within all plots in 1997 was 45.4% ± 2.9 (mean ± S.E.). Over the following years bridal creeper cover within untreated plots ranged from 35.5% to 44.9% whilst in the removal plots bridal creeper cover ranged from 0.1% to 2.4% (Figure 2). Before treatments were applied in 1997, mean abundance of indigenous plant species across all plots was similar (Figure 3) and the year following the initial application of herbicide the percentage cover of indigenous species was still comparable between treatments (Figure 4).

In March 2005 there was no significant difference in the number of perennial indigenous plant species between plots (Table 1). There was also no significant difference in abundance of indigenous species, except for the saltbush *E. tomentosa* (P = 0.01) (Table 1 and Figure 3). There were consistent increases in cover of the chenopod and indigenous grass understorey, even if not significant for some species (Table 2 and Figure 4). *Enchylaena tomentosa* cover was significantly lower in untreated plots (P <0.001). In September 2004, the exotic plant, *O. pes-caprae* had significantly higher cover of 42.6% in plots without bridal creeper, compared to 22.5% in untreated plots (P = 0.004).

Whilst the combined chenopod cover of *R. parabolica* and *R. candolleana* was not significantly different between treated and untreated plots (Table 2), there was approximately 45% less foliage biomass measured in March 2005 for combined *R. parabolica* and *R. candolleana* (P <0.05) (Table 3). *Enchylaena tomentosa* in untreated plots had on average only 15% of the foliage biomass compared to where bridal creeper had been controlled (P <0.01).

There was no significant difference between plots in any soil parameters that were measured in March 2005 (Table 4).

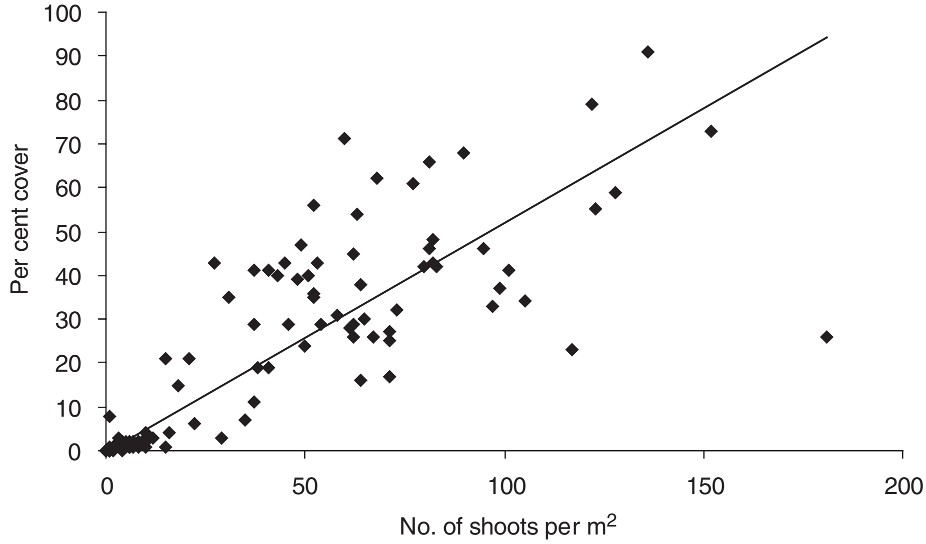


**Figure 1. Relationship between number of bridal creeper shoots and plant cover (all plots and dates, September 1999, October 2002 and September 2004 combined). (Square root transformation, $R^2$ = 83.3; d.f. 1,115; F=579.34; P <0.001).**

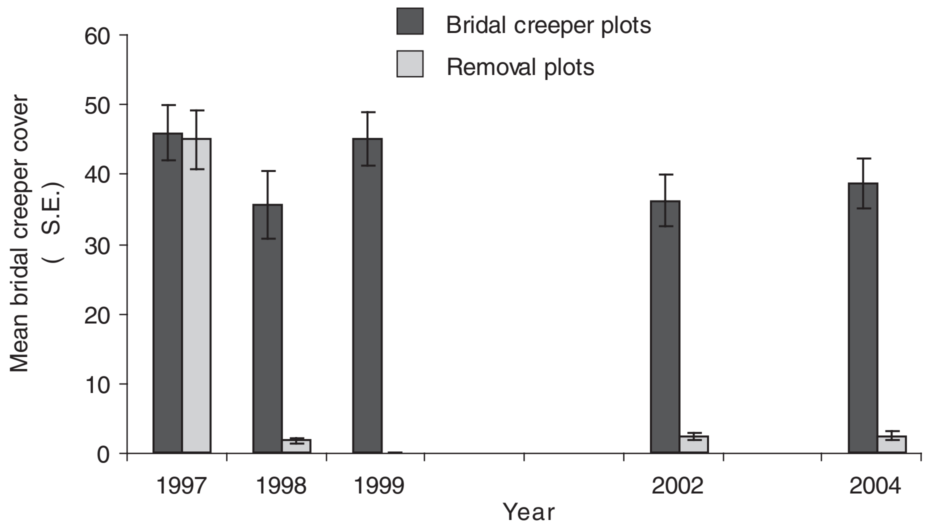


**Figure 2. Bridal creeper cover across the period of the study. The removal plots were treated with glyphosate in 1997 with a follow-up in 1999. Bridal creeper cover was not recorded in the years not shown.**

## Discussion

Any off-target effects from the glyphosate on indigenous plants was minimal given that the cover of these plants between treated and untreated plots was similar in 1998, one year after the initial application (Figure 4). Hence, differences detected between the control and herbicide treatments are largely due to differences in bridal creeper density. The experiment has shown that bridal creeper can reduce native plant biodiversity through a reduction in biomass (*E. tomentosa* and combined *R. parabolica* and *R. candolleana*) and in plant number (*E. tomentosa* and perennial grasses). However, within the eight year timeframe of this study there was no significant change detected in the number of plant species due to bridal creeper.

Adair and Groves (1998) suggested that a combination of approaches are needed when investigating weed impacts and a multi-site approach of comparing weed free areas to invaded areas, coupled with this weed removal study would increase the power of this study. This multi-site approach has been undertaken in S.W. Australia, where it was established that bridal creeper invaded sites contained fewer native plant species when compared to weed free areas (Turner unpublished). In addition, Leah (2001) while comparing infested sites of a closely related weed species, *Asparagus declinatus* L. (bridal veil) to un-infested sites, reported that the establishment of bridal veil in South Australia resulted in a reduction of species richness and diversity.

Byers *et al.* (2002) posed the question that if an environmental weed is controlled, will this removal actually benefit indigenous species. In this study, even eight years after bridal creeper control there was only a limited increase in the abundance of indigenous plants. However, this study revealed consistent increases in the cover of perennial plant species, even if not significant for some species. In a similar study, McCarthy (1997) used the weed removal method at one site to measure the response of a forest understorey community in the US. After a three year period, following removal of *Alliaria petiolata* (M.Bieb.) Cavara & Grande (garlic mustard), it was shown that garlic mustard had a negative effect on the composition and structure of the understorey community. Removal resulted in the increase in annuals, vines and tree seedlings, but the effects on slow growing perennial plant species were less clear. Response of the slow growing perennials may have become evident if the study period was increased.

Weed removal experiments are labour intensive and long-term (Adair and Groves 1998, Marrs *et al.* 2004), both with monitoring and the repeated removal of the weedy species. Due to this, replication across sites was missing from this study. It must be stressed then, that the results be viewed with caution when trying to extrapolate this study across regions invaded by bridal creeper. In moorlands in the U.K., Marrs *et al.* (2004) had replication

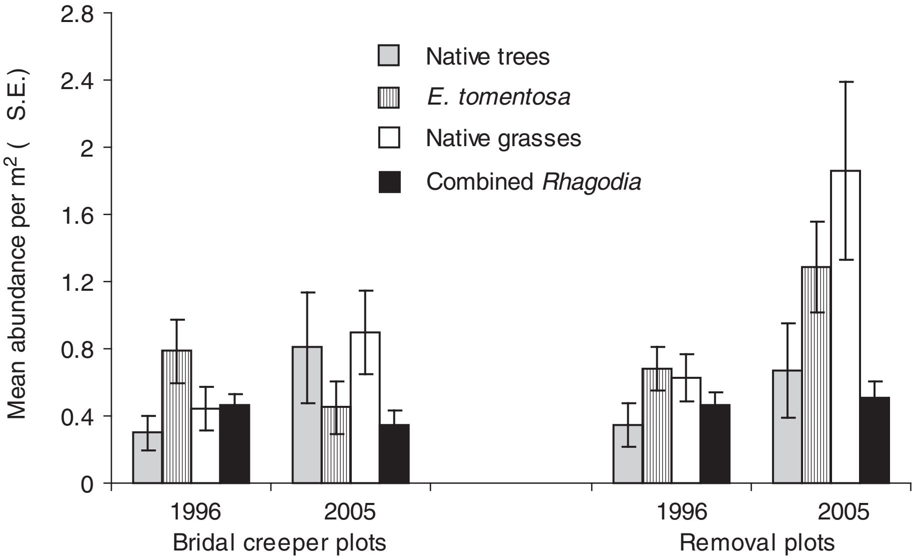


**Figure 3. Abundance of indigenous species in 1996, before treatments were applied, and in 2005 at the completion of the experiment. Native trees are a combination of *Pittosporum phylliraeoides*, *Santalum acuminatum*, *Senna artemisioides* and *Eucalyptus socialis*. Native grasses are a combination of native grasses in the genera *Austrostipa* and *Austrodanthonia*. Combined *Rhagodia* is a combined abundance for two species, *Rhagodia parabolica* and *Rhagodia candolleana*.**

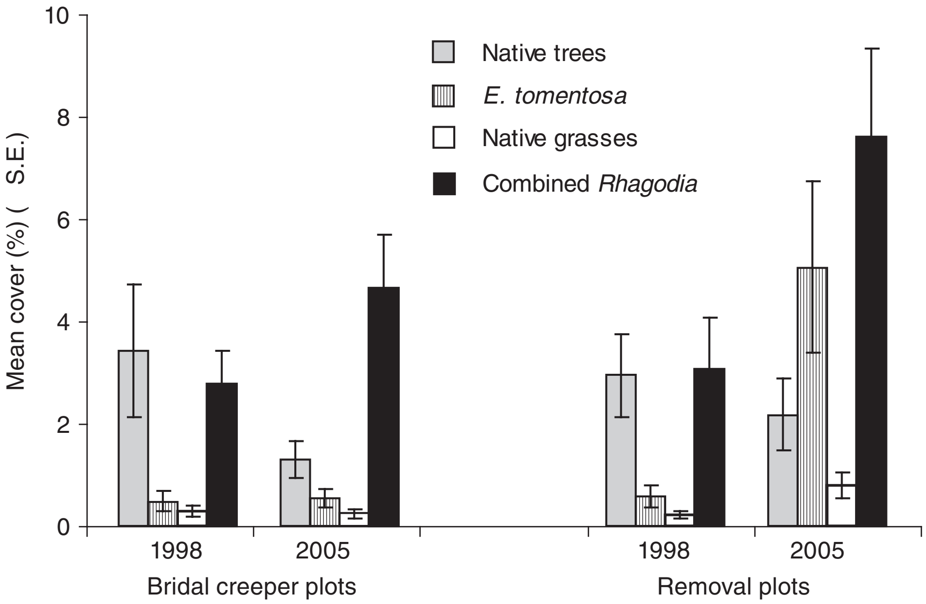


**Figure 4. Percentage cover of indigenous species in 1998, one year after the initial treatment was applied, and in 2005 at the completion of the experiment. (See Figure 3 for details on vegetation classes monitored).**

across sites and removed *Molinia caerulea* (L.) Moench (purple moor grass) with burning, grazing and glyphosate applications, however the herbicide treatment was applied once at the start of the five year study and purple moor grass had recovered in some areas by the end of the experiment. Even so, Marrs *et al*. (2004) established that different sites responded differently following weed removal and that multiple outcomes following large-scale control could be possible.

In invaded heathlands in the U.K., Marrs and Lowday (1992) controlled *Pteridium aquilinum* (L.) Kuhn (bracken) by cutting or with the use of herbicide coupled with artificial seeding of native species. After ten years it was established that the final vegetation varied greatly from their goal of a heath dominated area, through to grass heaths, being mainly a mixture of two species, to other areas where two different weeds became dominant. In these areas, there was a replacement of one weed problem with another (Marrs and Lowday 1992). In the same way, soursob, another weed from southern Africa, may have replaced bridal creeper following its control. Soursob had a higher density in the removal plots (Table 2) and recovery of indigenous species may have been hindered by this higher density. Mason *et al.* (2004) used the correlative method of comparing native sites to weed infested sites to measure the effects of *Chrysanthemoides monilifera* (L.) Norl. ssp. *rotundata* (DC.) Norl. (bitou bush) on fore and hind-dune communities. Bitou bush adversely affected native species richness in the fore-dune while in the hind-dune species richness was not affected, however in both communities control activities increased weed species richness.

It has been reported that repeated applications of glyphosate can give good control of bridal creeper (Dixon 1996, Pritchard 2002) and again this herbicide worked well in this experiment. However, it was observed at the study site that dead tubers still remained in the removal plots even eight years after the weed had been killed. This may have affected seedling establishment. Bridal creeper's underground plant parts consist of a cylindrical branching rhizome, bearing numerous tubers, which become entwined together and form thick mats (Raymond 1996). Biological control may provide better outcomes than chemical control from a conservation view point if agents were found to reduce bridal creeper's substantial root system. Intensive searches have been undertaken in South Africa to identify biological control agents that directly damages the root system, but no candidates were located (Kleinjan and Edwards 2006). However, glasshouse experiments performed on the biological control agent *Puccinia myrsiphylli*, which has been released in Australia, showed that this rust could significantly reduce vegetative growth. This in turn translated into a decrease in tuber biomass (Morin *et al.* 2002, Turner *et al.* 2004), which may allow greater seedling establish-ment.

A third possibility for the limited increase in indigenous species is a lack of suitable environmental conditions for germination and establishment over the previous eight year period. Marrs *et al.* (2004) suggested that when contemplating this kind of research a knowledge of the initial floristic composition is needed. A target for the density of indigenous species following weed removal is also needed. A target could be an historic condition, say at a level before bridal creeper invasion, however Chapman and Underwood (2000) argued that this would be illogical in ecological terms, as ecological systems do not stay constant. Instead, reference sites are recommended as well as control sites. Control sites are sites similar to the site being restored, but not subjected to the weed control. Reference sites are natural areas that represent the target for restoration (Chapman 1999). This study would have been strengthened by incorporating reference sites, areas free of bridal creeper and free of other disturbances. Comparisons between indigenous germination and survival could then have been compared to areas where the weed was never present. However, this was not possible given the limited natural areas in the agricultural region around the study area and due to the fact that bridal creeper is widespread throughout South Australia.

Adair and Groves (1998) suggested that weed removal experiments were not suited to old invasions where the invader has caused irrecoverable damage. This has not been the case with this study. The chenopod and native grass understorey has started to show trends of increased cover. However, this study has also shown that it may take many years for recovery and additional restoration work may be necessary following the control of bridal creeper to speed up the recovery process and to ensure that weed substitution does not occur. An additional site in South Australia has also been established following a wild fire. This site continues to be monitored and may provide outcomes different to those reported above, given that fire can stimulate the regeneration of native plant species.

**Table 1. Differences in mean (± S.E.) perennial plant abundance per square metre between treatments in March 2005[A].**

| Plant variable | Bridal creeper plots | Removal plots | F | P |
|---|---|---|---|---|
| No. of indigenous species | 4.10 ± 0.38 | 4.55 ± 0.24 | 1.09 | 0.304 |
| *Enchylaena tomentosa* | 0.45 ± 0.16 | 1.29 ± 0.27 | 7.25 | 0.011[E] |
| Combined *Rhagodia*[B] | 0.35 ± 0.08 | 0.51 ± 0.10 | 1.50 | 0.228 |
| Perennial native grasses[C] | 0.90 ± 0.25 | 1.86 ± 0.53 | 2.70 | 0.108 |
| Native trees[D] | 0.90 ± 0.33 | 0.83 ± 0.30 | 0.02 | 0.890 |

[A] The analysis of variance models with treatment (n=20) as the only factor. A log(10) transformation was applied to the variable native trees before the analysis. [B] Combined *Rhagodia* is a combined abundance for two species, *Rhagodia parabolica* and *Rhagodia candolleana*. [C] Perennial native grasses are a combination of native grasses in the genera *Austrostipa* and *Austrodanthonia*. [D] Native trees are a combination of *Pittosporum phylliraeoides*, *Santalum acuminatum*, *Senna artemisioides* and *Eucalyptus socialis*. [E] Statistically significant ($P < 0.05$).

**Table 2. Differences in mean (± S.E.) plant cover between treatments in March 2005 except for *Oxalis pes-caprae* cover which was measured in September 2004[A].**

| Plant variable | Bridal creeper plots | Removal plots | F | P |
|---|---|---|---|---|
| *Enchylaena tomentosa* | 0.56 ± 0.19 | 5.20 ± 1.72 | 14.32 | <0.001[B] |
| Combined *Rhagodia* | 4.77 ± 1.08 | 7.81 ± 1.78 | 1.10 | 0.300 |
| Perennial native grasses | 0.25 ± 0.09 | 0.82 ± 0.25 | 4.47 | 0.041[B] |
| Native trees | 1.34 ± 0.38 | 2.24 ± 0.73 | 0.65 | 0.426 |
| *Oxalis pes-caprae* | 22.47 ± 3.56 | 42.55 ± 5.51 | 9.17 | 0.004[B] |

[A] The analysis of variance models with treatment (n = 20) as the only factor. A log(10) transformation was applied to all variables, except Perennial native grasses and *Oxalis pes-caprae*, before the analysis. [B] Statistically significant ($P < 0.05$).

**Table 3. Differences in mean (± S.E.) foliage biomass (dry weight g $m^{-2}$) between treatments in March 2005[A].**

| Plant variable | Bridal creeper plots | Removal plots | F | P |
|---|---|---|---|---|
| Enchylaena tomentosa | 2.60 ± 0.87 | 17.57 ± 4.78 | 14.15 | <0.001[B] |
| Combined *Rhagodia* | 23.76 ± 4.33 | 43.01 ± 8.30 | 4.23 | 0.047[B] |
| Native trees | 26.07 ± 7.87 | 45.42 ± 11.70 | 0.90 | 0.350 |

[A] The analysis of variance models with treatment (n=20) as the only factor. A log(10) transformation was applied to all variables, except Combined *Rhagodia*, before the analysis. [B] Statistically significant ($P < 0.05$).

**Table 4. Differences in mean (± S.E.) soil variables between treatments in March 2005[A].**

| Soil variable | Bridal creeper plots | Removal plots | F | P |
|---|---|---|---|---|
| Nitrate (mg $kg^{-1}$) | 7.50 ± 1.45 | 9.75 ± 1.73 | 0.99 | 0.336 |
| Ammonium (mg $kg^{-1}$) | 2.75 ± 0.65 | 2.00 ± 0.33 | 0.84 | 0.374 |
| Phosphorus (mg $kg^{-1}$) | 10.75 ± 1.44 | 13.75 ± 1.56 | 2.25 | 0.156 |
| Organic carbon (%) | 2.84 ± 0.23 | 2.57 ± 0.16 | 0.96 | 0.344 |
| pH | 8.45 ± 0.03 | 8.45 ± 0.04 | 0.00 | 0.997 |

[A] The analysis of variance models with treatment (n=8) as the only factor. A square root transformation was applied to the variables ammonium, phosphorus and pH before the analysis.

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