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Restoration rocks: integrating abiotic and biotic habitat restoration to conserve threatened species and reduce fire fuel load

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Abstract With rapid urban expansion, biodiversity conservation and human asset protection often require different regimes for managing wildfire risk. We conducted a controlled, replicated experiment to optimise habitat restoration for the threatened Australian pink-tailed worm-lizard, *Aprasia parapulchella* while reducing fire fuel load in a rapidly developing urban area. We used dense addition of natural rock (30 % cover) and native grass revegetation (*Themeda triandra* and *Poa sieberiana*) to restore critical habitat elements. Combinations of fire and herbicide (Glyphosate) were used to reduce fuel load and invasive exotic species. Rock restoration combined with herbicide application met the widest range of restoration goals: it reduced fire fuel load, increased ant occurrence (the primary prey of *A. parapulchella*) in the short-term and increased the growth and survival of native grasses. Lizards colonised the restored habitat within a year of treatment. Our study documents an innovative way by which conflicts between biodiversity conservation and human asset protection can be overcome.

Keywords Ecological restoration · Fire management · Habitat loss · Invasive species · Urban ecology · Wildland-urban interface

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Introduction

Ecological restoration is essential to halt global declines in biodiversity (Newbold et al. 2015) but conflicts between biodiversity conservation and the demands of human communities often prevent successful restoration (Redpath et al. 2013). One prominent conflict arises when biodiversity conservation and human asset protection require different regimes for managing fire fuel loads (Driscoll et al. 2010). Broad-scale prescribed burning is frequently advocated to reduce house loss during bushfire (e.g. Boer et al. 2009) even though it might be ineffective for both asset protection (Gibbons et al. 2012) and biodiversity conservation (Kelly et al. 2015; Smith et al. 2013). Despite these conflicts, management agencies remain under strong pressure to maintain low fire fuel loads; a pressure that increases with rapid urbanisation (Radeloff et al. 2005). Management techniques that minimise fire fuel load while also restoring natural habitat are highly desirable but rarely achieved (Costanza et al. 2013; Madden and McQuinn 2014).

Rocks are a critical resource for many species, providing shelter (McGrath et al. 2015), structures for thermoregulation (Chapperon and Seuront 2011), prey hot-spots (Wong et al. 2011) and, for plants, safe-sites for seedling establishment (Pierson et al. 2013). Rock removal can cause declines in animal populations (Buisson et al. 2015), particularly when coupled with nutrient enrichment and associated changes in plant community composition (Prober and Lunt 2009). The importance of restoring rock habitat is exemplified by increases in reptile abundance following artificial rock installation (e.g. concrete pavers, corrugated steel or roof tiles, Croak et al. 2010; Michael et al. 2012). However, there are few scientific reports of habitat restoration using natural rock (but see Buisson et al. 2015) which is a problem for two reasons. First, compared with artificial habitat, natural rock might have different thermal and structural properties (Croak et al. 2010), be more feasibly applied at a landscape-scale (Menz et al. 2013) and be more likely to restore the landscape's natural aesthetic (Gobster et al. 2007). Second, when combined with biotic restoration, rock restoration could substantially reduce fire fuel loads, providing an innovative way to overcome conflicts between biodiversity conservation and human asset protection.

We conducted an experiment to develop a restoration technique that could be used to maintain low fire fuel loads while restoring habitat for the threatened Australian pink-tailed worm-lizard, *Aprasia parapulchella* Kluge (Pygopodidae). The species occupies natural temperate grasslands with lightly-embedded surface rock along the western slopes of the Great Dividing Range, south-east Australia (Wong et al. 2011). *A. parapulchella* inhabits ant burrows under rocks and eats ant eggs and larvae (Wong et al. 2011). The stronghold of the species distribution occurs around the city of Canberra, an area undergoing rapid urban development (ACT Government 2011). Federal planning laws require development projects to protect threatened species (ACT Government 2011) while the local government must maintain low fuel loads in zones between conservation land and housing (the wildland-urban interface). Within these zones, patches of high-quality *A. parapulchella* habitat remain which would be damaged by conventional fuel control methods (e.g. slashing, stock grazing and annual hazard reduction burning). Widespread surface rock removal has eliminated critical *A. parapulchella* habitat and the grassland plant community is now dominated by invasive exotic species. Land managers around the world face conflict between maintaining low fuel loads while conserving threatened species (e.g. Ryan et al. 2013) and this conflict intensifies with urbanisation (Bar-Massada et al. 2014). Thus, our restoration technique will be relevant to grassland conservation in any region of the world where rocks are a key feature of the landscape and fire management is essential.

The aim of this study was to determine the best combination of abiotic and biotic restoration methods that would increase the quality of *Aprasia parapulchella* habitat while also reducing fire fuel load. Habitat was considered high quality if it had high cover of rocks and native plant species, low cover of exotic invasive species and high abundance of ants, the primary prey of *A. parapulchella* (Wong et al. 2011). Our replicated experiment consisted of five treatments (including controls) which we compared to a reference site in nearby high-quality habitat. To reduce the cover of invasive plant species, we used treatments that could feasibly be applied broadly across degraded landscapes and that have been previously successful in grassland restoration: fire and herbicide (Prober and Thiele 2005). We planted native grasses and applied a dense (30 %) cover of natural rock to reduce fire fuel load and restore *A. parapulchella* habitat. Our specific research question was: which combination of treatments would deliver the strongest positive change in habitat quality? By combining biotic and abiotic habitat restoration, we aimed to provide a practical solution for land-managers under pressure to protect human assets and conserve grassland biodiversity.

Materials and methods

Study design

The study took place at Coppins Crossing in the Molonglo Valley, 7.5 km west of Canberra, Australian Capital Territory (ACT), Australia (Fig. 1). Native grasslands dominated by *Themeda*, *Austrostipa*, *Poa*, *Bothriochloa* and *Rhytidosperma* species have been heavily degraded by livestock grazing, introduced pastures and other weeds. Exotic species (chiefly *Avena* sp., *Hypericum perforatum* and *Rubus fruticosus*) now dominate, increasing fuel load and the risk of grassland fire (Driscoll et al. 2014). Small ($\sim$2 ha) patches of native grassland with 30–50 % rock cover persist in the area, which are assumed to have substantially lower fuel loads.

We established six replicate sites [separated by an average of 150 m (range 100–200 m)] within a linear grassland strip, approximately 100 m wide, which separates the Molonglo River from the area zoned for future urban development (Fig. 1). For each replicate, we selected sites using random points generated in ArcMap 10 (ESRI), then visited the first point to assess whether it met our criteria for restoration: dominated by exotic invasive species with a low proportion of native species. When these criteria were not met, we moved to the next point until we found a suitable site.

At each site, we established six 4 $\times$ 4 m plots, consisting of a 2 $\times$ 2 m core with a 1 m buffer: reference, control, plants + rocks, fire-only, herbicide-only and fire + herbicide (Fig. S1). Rocks and tubestock grasses (*Themeda triandra* and *Poa sieberiana*, commonly associated with *A. parapulchella* habitat) were applied to all plots except the reference and control. Experimental treatments, rocks and grasses were applied across the whole plot, including the buffer. Treatments were randomly assigned within the site and arranged so that each plot shared a boundary with at least one other plot (Fig. S1). Reference plots were placed in high quality *A. parapulchella* habitat, an average of 50 m (range 25–65 m) from the treatment plots (Fig. 1).

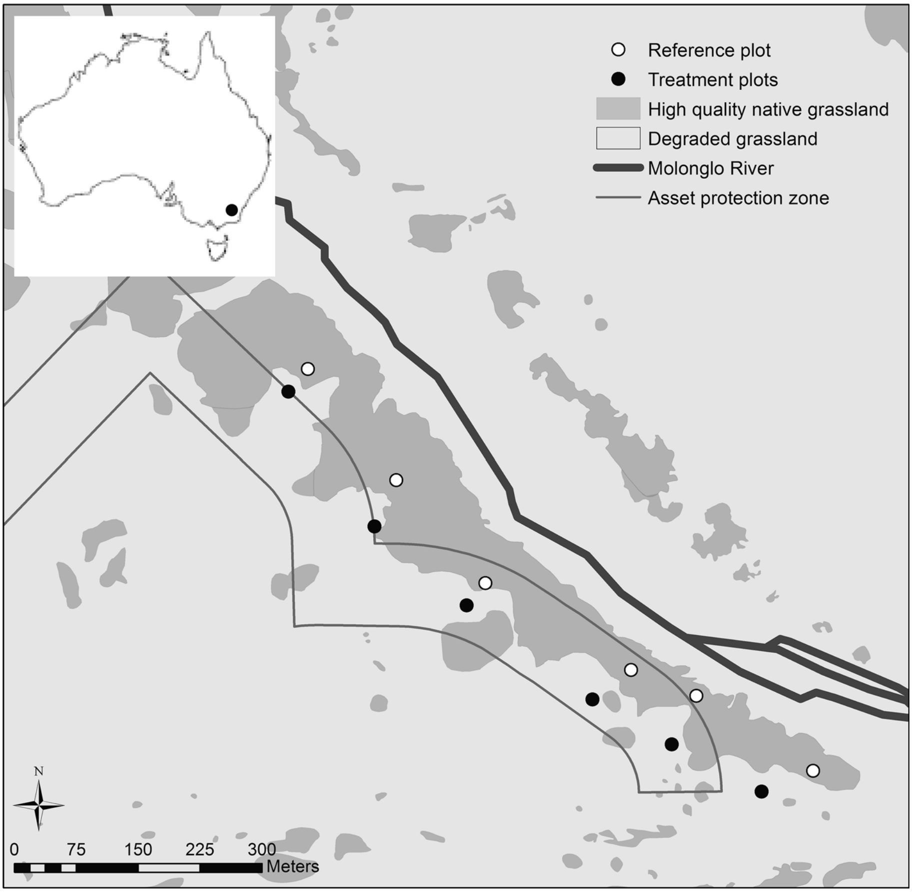

Fig. 1 Six replicate sites were located in grassland, heavily invaded by exotic species. Responses to restoration treatments were compared to nearby reference sites in high-quality native grassland which has been mapped as potential habitat for *Aprasia parapulchella*. The local government is required to maintain low fire fuel load in the asset protection zone which separates the grassland and river from a rapidly developing urban area. Layout of plots within sites is shown in Fig. S1 in supplementary material

Glyphosate (1:100 glyphosate:water) was applied to the herbicide-only and fire + herbicide plots on 09 April 2014. A blowtorch was used to burn the fire-only and fire + herbicide plots on 29 April 2014. Rocks were placed on plots 1–2 May 2014 at approximately 6.5 rocks/m^2 (approximately 30 % rock cover or 100–110 rocks per plot). Rocks were sourced locally from the urban development area and were approximately 225–625 cm^3, corresponding with the preferred rock size of *A. parapulchella* (Wong et al. 2011). To reduce the risk of introducing weeds into the restoration site (from soil attached to imported rocks), we used bedrock crushed to the identified size range. A total of 1488 *Poa* and 1488 *Themeda* was planted 5–7 May 2014, with 62 of each species in each plot. *Poa*, sourced from one location in ACT, was grown from seed collected in Jan 2014. *Themeda*, sourced from three provenances in New South Wales and ACT (Supporting Information), was grown Jan–May 2014 (provenance effects were considered in the analysis). *Themeda* was planted in rows of 5–8 individuals, with a row of 7–9 *Poa* individuals planted between each. Individuals were planted approximately 40 cm apart.

We measured habitat quality before and after restoration treatment via seven metrics: (1) plant community composition, (2) percentage cover of dominant invasive exotic species and species groups (native grasses, native forbs, exotic annuals and exotic perennials), (3) fire fuel load, (4) soil chemistry (to quantify plant nutrient variation), (5) growth and survival of planted native grasses, (6) ant abundance and (7) presence of *A. parapulchella* individuals.

Plant community and fire fuel load

Pre- (29 Nov–13 Dec 2013) and post- (27 Nov–3 Dec 2014) treatment vegetation surveys were undertaken using a linear point transect to monitor plant community composition. We placed a 2 m transect tape at 0, 50, 100, 150 and 200 cm on each plot. At every 10 cm point along each transect, all plant species touching the tape were recorded (200 points per plot). We used these values to calculate the percentage cover of each individual species and species groups (native grasses, native forbs, exotic annuals and exotic perennials). The cover of non-vegetative material (rocks, leaf litter and bare ground) was also recorded. Fire fuel load was measured using grassland fire hazard scores, calculated by multiplying the average of height of all grasses by the percent grass cover on each plot (ACT Government 2009). Fire management standards in ACT require the grassland fire hazard to be <35 when grass curing is >70 % (ACT Government 2009).

Soil analysis

Soil was sampled to quantify plant nutrients variation before (2 April 2014) and after (16 April 2015) treatment. For the primary samples, we took three cores from the 2 $\times$ 2 m area in each plot: one in the centre and two in opposite corners. For each core, we cleared organic matter from the surface and augured 20 cm into the soil profile. The three cores were combined in a labelled plastic bag. To measure bulk density, a second sample was taken in a 5 cm metal core, compressed into the soil in the centre of the plot and sealed. The primary samples were air-dried for 4 days, ground with a mortar and pestle and sieved through 0.06–2 mm mesh. A blank sample was prepared for each test to identify potential contamination. We analysed soil properties important for plant growth (Richardson et al. 2009): pH, electrical conductivity, bulk density, nitrate, ammonium, phosphate, total N and total C (Supporting Information).

Themeda and *Poa* growth and survival

The height and diameter of every tubestock plant within the 4 $\times$ 4 m plot was measured 8–15 May 2014 (time 0, immediately after planting), 24–28 July 2014 (time 1) and 28 Jan–5 Feb 2015 (time 2). Survival was calculated at time (t) 1 and 2 as N_t/N_0, where N is the number plants alive.

Ant and lizard surveys

We undertook one detailed ant community survey (29 July 2014, approximately 3 months post-treatment) and one general survey of ant colonies and eggs (5 Feb 2015, approximately 9 months post-treatment). We were unable to obtain pre-treatment ant data because there were no rocks on our restoration plots before treatment and other sampling methods (e.g. traps) would not have yielded comparable ant community data. In the 2014 survey, we turned 30 rocks on each 2 × 2 m plot (excluding the buffer) and recorded the number of burrows, number of morphospecies and estimated the total number of ants. We also quantified the number of *Paratrechina* spp. and *Iridomyrmex* spp. separately as these ant genera comprise a large component of the *A. parapulchella* diet (Wong et al. 2011). Ants were scooped in approximately 15 cm^3 of soil, placed in a plastic container and taken to the laboratory for identification to genus (Shattuck 2000). During the 2015 survey, we turned all rocks on each 4 × 4 m plot and recorded the number of ant colonies with and without eggs and the number of ants in an abundance class (1–10). At this time, we also recorded all plots on which *A. parapulchella* or their shed skins were seen. We defined the presence of lizards or skins as colonisation events (Driscoll 2007) because the species could not previously use sites without rocks (Wong et al. 2011). All rocks were returned to their original position following the count.

Analysis

We investigated treatment effects on plant community composition using nonmetric multidimensional scaling (NMDS) in the R (R Core Team 2013) package 'vegan' (Oksanen et al. 2013). We used a Bray-Curtis dissimilarity matrix and included all species with three or more occurrences. To analyse treatment effects on the other response variables, we used mixed-effects models in 'lme4' (Bates et al. 2013) in R. We used normal linear models for all response variables except tubestock survival (binomial) and ant abundance (Poisson) for which we fit generalised linear mixed models with logit and log link functions, respectively. Our global model included an interaction between treatment and time. *Themeda* models included a main fixed effect for provenance. Two random effects were fitted to all models (with exceptions outlined below): site, to account for spatial dependence among plots within sites, and plot, to account for repeated sampling of plots over time. The height and diameter models for *Poa* and *Themeda* included a third random effect for plant, to account for repeated measures of individuals. The ant models included only a site random effect as one measurement per plot was taken. For *Poa*, survival at time 1 was 100 %, so we modelled data only from time 2 and with only a site-level random effect. There was also 100 % *Poa* survival on the herbicide-only treatment so we removed that level from the analysis. We analysed individual plant species data for species that we observed to have a strong treatment effect in the field.

To determine if the importance of terms, *P* values were calculated using Wald tests (Welsh 1996). We removed the interaction between treatment and time when it was >0.1, a relaxed level to maintain the design in our model structure when trends existed. We used Fisher's Least Significant Differences (LSD) to determine if there were significant differences between each pair of levels within terms (e.g. control vs fire treatment). Significant differences were inferred when confidence intervals did not include zero.

Results

We recorded significant effects of restoration treatments on habitat quality measured by fire fuel load, dominant invasive exotic species, soil properties, growth and survival of native grasses and ant abundance (Table 1). In general, habitat quality increased with the

Table 1 The direction of response for variables with significant effects of fire-only, herbicide-only and fire + herbicide

Habitat quality metric	Specific response variable	Baseline for comparison	Fire-only (F)	Herbicide-only (H)	Fire + herbicide (FH)
Dominant invasive exotic species	*Avena* sp.	Control		↓	↓
Fire fuel load	Grassland fire hazard score	Control	↓	↓	↓
Soil properties	Soil pH	Control			↓
	Soil electrical conductivity	Control		↑	↑
Growth and survival of planted native grasses	*Themeda* height	Plants + rocks			
	Themeda diameter	Plants + rocks		↑	↑
	Themeda survival	Plants + rocks		↑	↑
	Poa height	Plants + rocks		↑	↑
	Poa diameter	Plants + rocks		↑	↑
	Poa survival	Plants + rocks		↑	↑
Ant abundance	Total number of ants, ant burrows, *Paratrechina* sp. and *Iridomyrmex* sp.	Plants + rocks	↑ (short term)	↑ (short term)	↑ (short term)

Growth and survival of native grasses and ant abundance subjected to fire and/or herbicide were compared to the plants + rocks treatment because control plots did not contain planted grasses or rocks under which ants could be detected

herbicide-only and the fire + herbicide treatments but not with the fire-only treatment (Table 1). The direction of treatment effects are summarised in Table 1 with significant effects shown in Figs. 2, 3, and 4. Weak effects or those unrelated to our primary research question (e.g. provenance effects) are presented in the Supporting Information.

Nonmetric multi-dimensional scaling (NMDS) showed that, prior to treatment, the reference plots were different to the treatment plots and there were no pre-existing differences among treatment plots (Fig. S2 in supplementary meterial). Following treatment, plant community composition became more similar to the reference plots along NMDS axis 1, while differences on axis 2 became greater (Fig. S2 in supplementary meterial). A high degree of similarity among treatments at the end of the experiment suggested minimal treatment effects on plant community composition (Fig. S2 in supplementary meterial).

Fire-only, herbicide-only and fire + herbicide significantly reduced grassland fire hazard, such that fuel load was not different from the reference sites at the end of the experiment (interaction $P < 0.001$, Fig. 2a). *Avena* sp. cover (the dominant exotic grass) significantly increased between years on the control, plants + rocks and fire-only treatments (interaction $P < 0.001$, Fig. 2b). Herbicide-only and fire + herbicide eliminated this year effect and *Avena* cover on these treatments was significantly lower than control, plants + rocks and fire-only at the end of the experiment (Fig. 2b). At the end of the experiment, soil pH was higher on the reference than all treatment plots and significantly lower on fire + herbicide compared with the control (interaction $P = 0.005$, Fig. 2c). Electrical conductivity was higher on the herbicide-only than reference and control plots

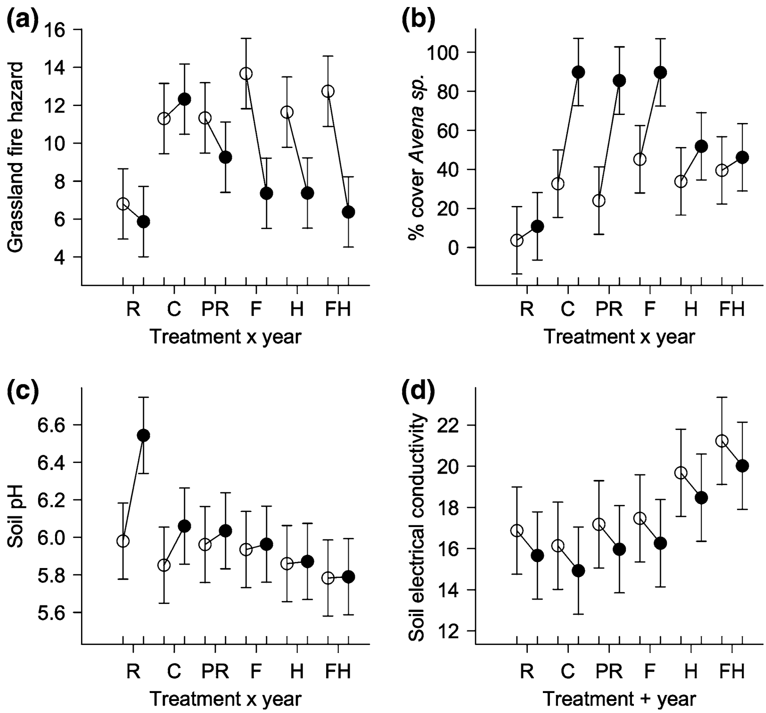

Fig. 2 Estimated effect (and 95 % confidence interval) of restoration treatment on **a** grassland fire hazard, **b** *Avena* sp. cover, a dominant exotic grass, **c** soil pH and **d** soil electrical conductivity. *Open circles* = pre-treatment, *closed circles* = post-treatment, R = reference, C = control, PR = plants + rocks, F = fire-only, H = herbicide-only, FH = fire + herbicide

and was higher on fire + herbicide than reference, control, plants + rocks and fire-only plots (no interaction, Fig. 2d). None of the soil nutrients (NH_4^+, NO_3^-, PO_4^{3-} and total N and total C) showed significant effects of treatment.

None of the treatments effectively increased native grass and forb cover or reduced exotic species cover to the level of the reference sites (Fig. S3 in supplementary material). The cover of native grasses (Fig. S3a in supplementary material) and exotic annuals (Fig. S3c in supplementary material) increased on control plots between years but not on the treatment plots suggesting a potential moderating effect of the treatments on natural annual variation. In addition to *Avena* sp. (described above), herbicide-only and/or fire + herbicide significantly increased the cover of four individual exotic species (Fig. S4 in supplementary material, these are not dominant species that contribute strongly to fire fuel loads).

Immediately after planting (time 0), there were no treatment differences in *Themeda* and *Poa* height or diameter. There were interactive effects of treatment and time on *Themeda* height ($P = 0.005$), diameter ($P = 0.004$) and survival ($P = 0.07$), and on *Poa* height ($P < 0.001$), diameter ($P < 0.001$) and survival ($P = 0.03$). By the end of the experiment (time 2), *Themeda* height was greater on plants + rocks than fire-only (Fig. 3a). *Poa* height was greater on herbicide-only and fire + herbicide than on plants + rocks and fire-only (Fig. 3b). *Themeda* diameter was greater on herbicide-only and fire + herbicide than plants + rocks (Fig. 3c). *Poa* diameter was lower on fire-only than on plants + rocks,

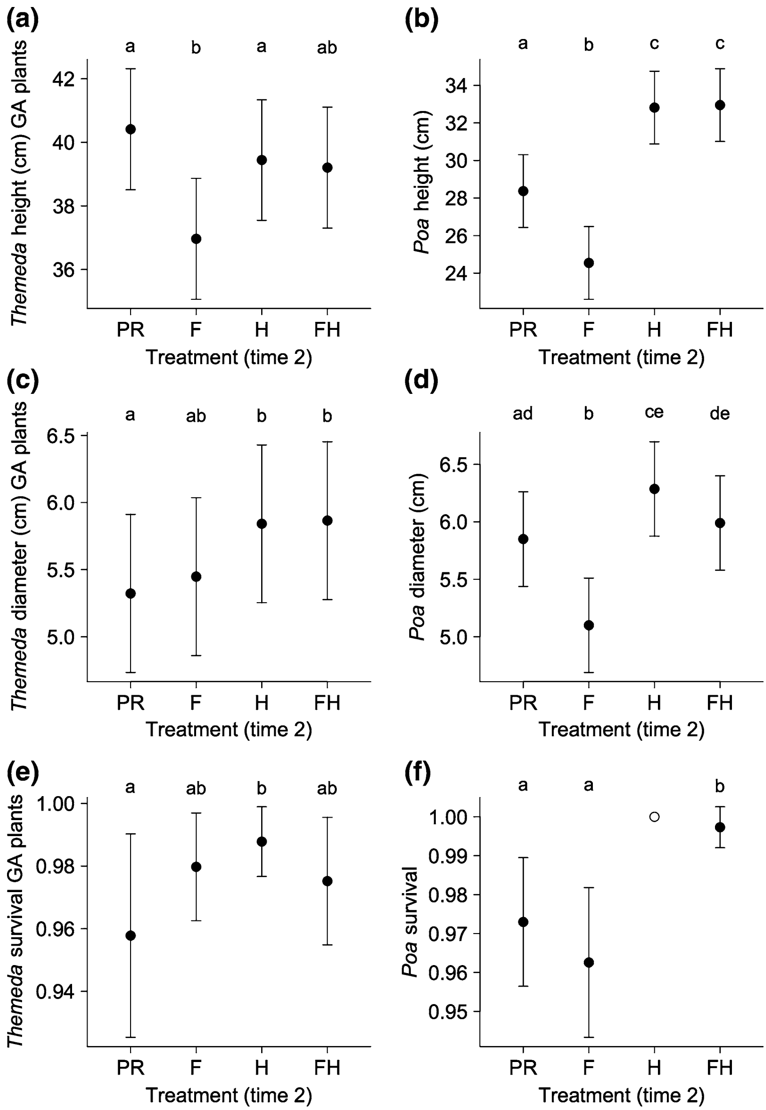

Fig. 3 Estimated effect (and 95 % confidence interval) of restoration treatment on **a**, **b** height, **c**, **d** diameter and **e**, **f** survival in *Themeda triandra* and *Poa sieberiana*. Models included an interaction between time and treatment and values are shown for the end of the experiment (time 2). The *Themeda* model included provenance (Fig. S5 in supplementary material) and values are shown here for GA plants. *Poa* survival was 100 % on herbicide-only at time 2, thus herbicide-only was not modelled but is shown as an *open circle* (**f**). PR = plants + rocks, F = fire-only, H = herbicide-only, FH = fire + herbicide. Shared letters indicate no difference

herbicide-only and fire + herbicide, while herbicide-only had greater diameter than plants + rocks (Fig. 3d). At time 2, *Themeda* survival was greater on herbicide-only compared to plants + rocks but not to the other treatments (Fig. 3e). *Poa* survival was greater on fire + herbicide than plants + rocks and fire-only at time 2 (Fig. 3f). There

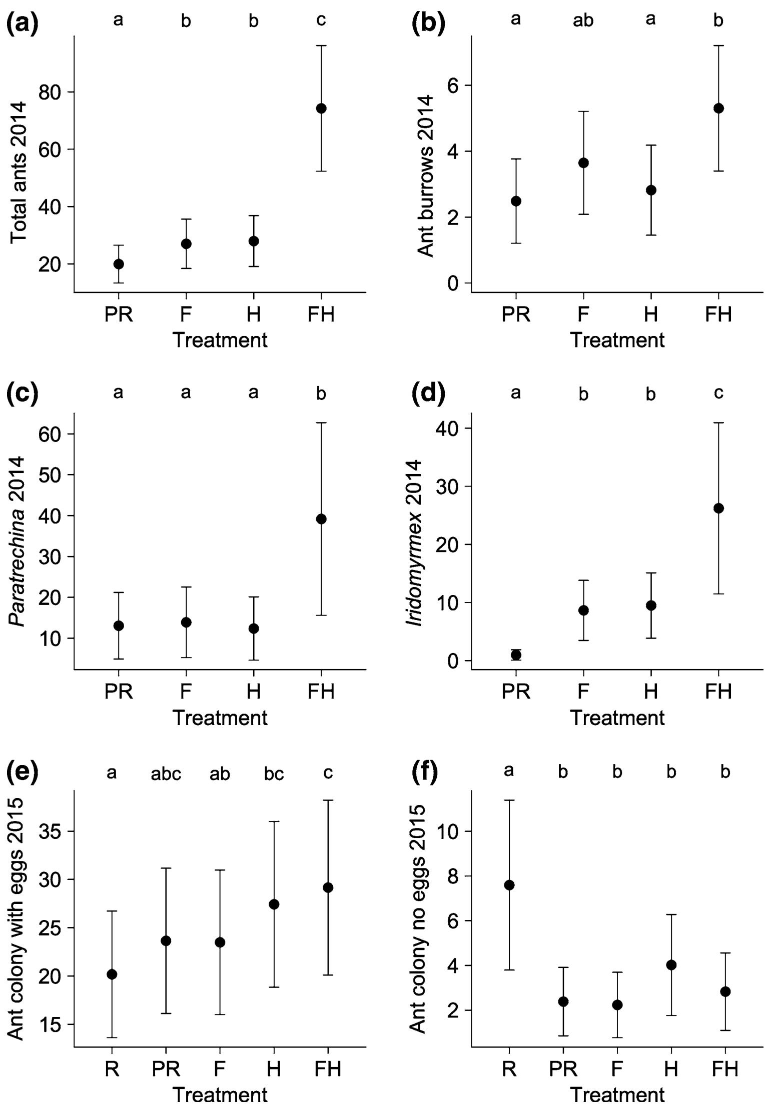

Fig. 4 Estimated effect (and 95 % confidence interval) of restoration treatment on the abundance of **a** ants, **b** ant burrows, **c** *Paratrechina* species, **d** *Iridomyrmex* species and on ant colonies **e** with and **f** without eggs. Surveys were undertaken approximately **a**–**d** three and **e**, **f** nine months after treatment. R = reference, C = control, PR = plants + rocks, F = fire-only, H = herbicide-only, FH = fire + herbicide. Shared letters indicate no difference

were significant effects of provenance on *Themeda* growth and survival ($P < 0.001$, Fig. S5 in supplementary material).

Three months after treatment, fire + herbicide significantly increased the number of ants ($P < 0.001$), ant burrows ($P = 0.05$), *Paratrechina* spp. ($P < 0.001$) and *Iridomyrmex* spp. ($P < 0.001$) compared with the control (Fig. 4). The fire-only and herbicide-only treatments had a weaker but significant effect on total ant abundance and *Iridomyrmex* abundance (Fig. 4). Nine months after treatment there were no significant treatment effects on ant abundance compared to plants + rocks (Fig. 4e, f). We recorded 14 *A. parapulchella* individuals (four each on reference, plants + rocks and fire + herbicide; two on fire-only) and four individual skins (three on reference; one on plants + rocks).

Discussion

Restoration of rocks and biotic habitat elements lowered fire fuel load, changed soil properties and increased the growth and survival of native grasses. Adding rocks enabled the threatened *A. parapulchella* to colonise restored habitat within a year of treatment, despite the treatment plots being 10–20 m from existing habitat. Our restoration technique therefore holds potential for increasing population sizes of this species in the wildland-urban interface, while also reducing fire risk. Repeated application of herbicide is likely to be necessary until native grasses can establish to a level that prevents high-biomass invasive species (such as *Avena*) from dominating. How long this will take and whether this will increase native plant species richness is a longer-term unanswered question. Results from our study are currently being used to implement larger-scale restoration in our study region (Fig. 5). This is being done in an experimental framework, so that management decisions can be adapted when new data become available (Cundill et al. 2012).

Our restoration technique could be used in any grassland region where rocks are an important resource for native species and there is a need to maintain low fire fuel loads. Globally, rock removal for agriculture, quarrying and landscaping has caused declines in the distribution and abundance of native species (Wong et al. 2011) and in biological processes such as recruitment and recolonisation (Croak et al. 2010; Pierson et al. 2013). Our study suggests that rock restoration can help reverse these negative effects. In the Mediterranean Basin, Buisson et al. (2015) found that the addition of natural rock provided a physical nurse effect on planted seedlings which increased their survival and growth. Thus, rock restoration for target species of conservation concern (as in our study) is likely to have broader positive effects on biodiversity. Rocks can be applied across the landscape using earthmoving equipment (Fig. 5) making it a feasible solution for broad-scale restoration and fuel reduction. We recommend that only sustainably-sourced rocks are used in conservation projects which can be achieved by recycling materials excavated from local development projects.

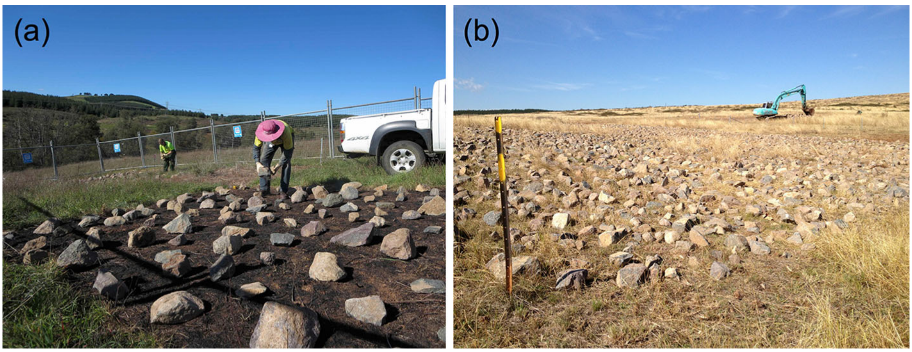

Fig. 5 Rocks are placed **a** by hand during the experiment (the fire-only treatment is shown) and **b** by an earth mover at a larger scale across the study region following the experiment. Photos: **a** RNCM **b** ALS

In rapidly expanding urban areas, applying fire at a landscape-scale is expensive and risky (Keane et al. 2014) and conservation managers often need to consider alternatives to fire. In our study, the herbicide-only and the fire + herbicide treatments both showed strong responses, in the desired direction, of restoration success, while the fire-only treatment generally did not (Table 1). The only gain that the fire + herbicide treatment had over herbicide-only was a reduction in soil pH. Although fire can have positive effects on *Themeda* establishment (Prober and Thiele 2005), we found that herbicide alone increased its growth and survival rates, probably by reducing competition. Our results suggest that biomass control through herbicide and rock restoration is an effective solution for restoring habitat for threatened species while also reducing fire hazard to meet planning regulations.

Introducing native species through revegetation is necessary to establish a plant community that can compete with exotic species and reduce the need for weed control in future. *Themeda* can reduce soil nutrients and help buffer restored grasslands to further invasion (Prober and Lunt 2009), thus this species should be a priority for primary restoration in Australia's temperate grasslands. Although our restoration treatments did not influence soil nutrients, the increase in electrical conductivity and decrease in soil pH might have enhanced the growth of tubestock following fire and herbicide treatments (Cole et al. 2005). Locally sourced seed increased establishment of *Themeda* in our study but predicting suitable provenances for future climates will be important in this study system (Bragg et al. 2015).

Our results represent short term responses and continued studies are needed to confirm the suitability of this technique for the long-term viability of *A. parapulchella* and other species. However, urban development in our study system is occurring rapidly (55,000 people will be housed over 1356 ha within 30 years) and conservation projects must keep up with this pace. Government decisions are typically made on time scales much shorter than ecological research projects. It is very important that environmental decisions are based on the best available scientific evidence. Our study represents a successful collaboration between a government agency and academic scientists, to ensure that conservation during rapid urban expansion had the best possible scientific input within the urgent timeframes. Rapid urban expansion presents a challenge for conserving biodiversity while also protecting human life and property and we documented one way to meet this challenge. This combination of biotic and abiotic habitat restoration should be relevant to grasslands in any region of the world where rocks are a key feature of the landscape and biodiversity conservation is critical.

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