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Chainsawing for conservation: Ecologically informed tree removal for habitat management

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Summary In many ecosystems, increases in vegetation density and the resulting closure of forest canopies are threatening the viability of species that depend upon open, sunlight-exposed habitats. Consequently, we need to develop management strategies that recreate open habitats while minimizing the impacts on non-target areas. Selective logging creates canopy gaps, but may result in undesirable effects in other respects. Thus, chainsaws have not been a popular tool for conservation. We conducted a landscape-scale experiment to test whether selective tree removal can restore patch-level habitat quality for Australia's most endangered snake (*Hoplocephalus bungaroides*) and its main prey (the lizard *Oedura lesueurii*). We selectively removed canopy trees surrounding 25 overgrown rock outcrops and compared the resultant habitat structure and abiotic conditions to 30 overgrown, shady outcrops and 20 open, sunny outcrops. Removing vegetation decreased canopy cover by 19% in experimental plots and increased incident radiation and thermal regimes. These changes increased the availability of suitable shelter sites for our target species by 131%. At the landscape scale, our manipulations had a trivial effect on forest habitat; by increasing the area of sun-exposed outcrops, we decreased forest cover by <0.1%. Our results show that targeted canopy removal can increase the availability of sun-exposed habitat patches for endangered species in biologically meaningful ways. Thus, selective tree felling may be an effective conservation tool for open-habitat specialists threatened by vegetation overgrowth.

Key words: *habitat quality, open habitat, rock outcrops, thermal regimes, vegetation encroachment.*

Introduction

Habitat degradation is a leading threat to biodiversity. In many countries, long-term fire suppression and/or the loss of herbivores has resulted in increases in tree density, canopy cover and shading, which produce microhabitat conditions that favour shade-tolerant species (Bond *et al.* 2005; Nowacki & Abrams 2008; Fuhlendorf *et al.* 2009; Levick *et al.* 2009; Staver *et al.* 2009). Such 'mesophication' (sensu Nowacki & Abrams 2008) can produce rapid changes in the composition of plant and animal communities in formerly open habitats (e.g. Jäggi & Baur 1999; Skelly *et al.* 1999; Anderson *et al.* 2006; Blaum *et al.* 2007; Kaphengst & Ward 2008; Pike *et al.* 2011). For example, in the northern Swiss Jura Mountains, Asp Viper *Vipera aspis* populations have disappeared from sites with higher tree densities and it has been suggested that vegetation overgrowth has contributed to local extinctions of this species (Jäggi & Baur 1999).

In many cases, habitat restoration is urgently needed to conserve habitat specialists that depend on open habitats (e.g. Jäggi & Baur 1999; Pringle *et al.* 2009; Shoemaker & Gibbs 2010). Prescribed fire is widely used to combat vegetation thickening, but can result in undesirable effects by encouraging the spread of invasive species (Keeley 2006), altering soil nutrient availability (Prosser & Williams 1998) or causing increased mortality of fauna (Lyet *et al.* 2009). In addition, fire is not appropriate or effective at providing the desired canopy thinning in all ecosystems. For example, rocky habitats often have heterogenous fuel loads, which prevent fires from carrying well and thus having a predictable impact (Hunter *et al.* 1998; Clarke 2002a,b). Thus, we need to develop and test alternative techniques to restore open habitats suffering from increased vegetation density.

Selective timber removal is one management tool often overlooked in habitat management for wildlife, but may offer a rapid way to target overgrown habitat at spatial scales relevant to management (Jäggi & Baur 1999). Large-scale selective logging is often associated with habitat degradation because of negative impacts on habitat structure and species diversity (e.g. Vitt *et al.* 1998; Waldrop *et al.* 2008, 2010; Kalies *et al.* 2010), suggesting that such techniques may be inappropriate for restoring habitats. However, in habitats where prescribed fire cannot restore habitats, targeted tree removal may be a cost-effective method for improving the habitat quality of open-habitat specialists (Webb *et al.* 2005; Fagg & Bates 2009).

We used a replicated field experiment to investigate whether tree removal can increase habitat availability for Australia's most endangered snake, the Broad-headed Snake (*Hoplocephalus bungaroides*). Broad-headed Snake has a small geographic distribution and is restricted to sandstone rock outcrops within 250 km of Sydney. This snake and its main prey (the Velvet Gecko, *Oedura lesueurii*) utilize sun-exposed rock habitats for up to 9 months of the year (Webb & Shine 1997, 1998a,b). During the cooler months, both species thermoregulate beneath sun-exposed rocks (Webb & Shine 1997, 1998b), and geckos lay their eggs in rock crevices in exposed locations (Pike *et al.* 2010a). Broad-headed Snake has declined throughout their geographic range (Shine *et al.* 1998; Newell & Goldingay 2005) because of multiple factors, including the loss of open rock habitat owing to increases in vegetation density (Pringle *et al.* 2009). Although the exact cause of vegetation thickening is not completely understood, a decline of large macropods throughout the region (Webb *et al.* 2005) and/or potential changes in fire regimes may be responsible (Pringle *et al.* 2009). Because prescribed fire is often ineffective for thinning eucalypts in rocky habitats (Hunter *et al.* 1998; Clarke 2002a,b), we need alternative techniques that are effective at reducing vegetation density (Webb *et al.* 2005).

In this study, we investigated whether targeted tree removal can restore open rock habitats for reptiles at the landscape scale. We build on a pilot study demonstrating that trimming small amounts of overhanging vegetation directly above individual rocks formerly used by snakes increased rock temperatures and facilitated recolonization by snakes (Webb *et al.* 2005). Our current project sought to determine whether we can scale-up tree removal to the outcrop level while having the same positive effects, thereby substantially increasing the availability of rocks with microclimates suitable for reptiles throughout the landscape. We assessed the effectiveness of our patch-level canopy manipulations based on the microhabitat variables most important to our target species (canopy structure, sunlight exposure and microhabitat temperatures; Pringle *et al.* 2003) and supplemented this with longer-term data on reptile community responses (Pike *et al.* 2011).

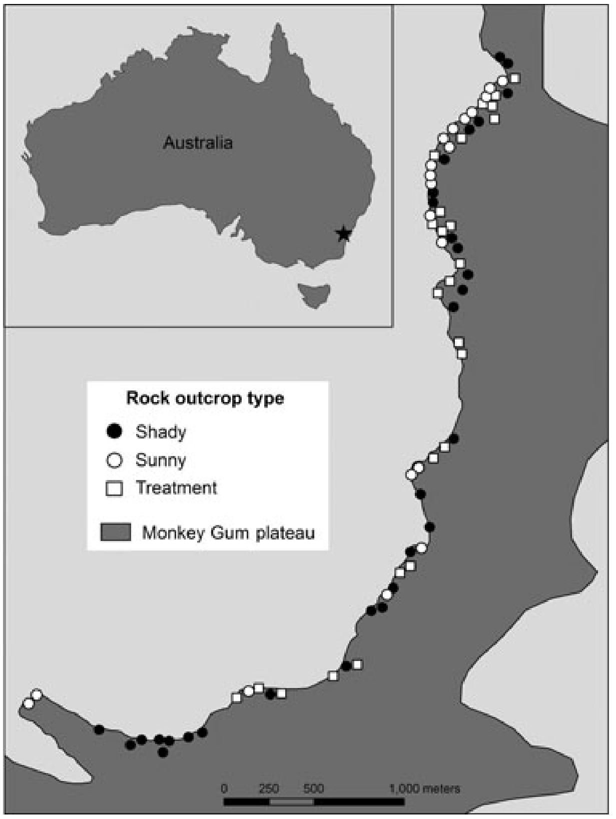

Figure 1. Location of our study outcrops along Monkey Gum plateau within south-eastern Australia. The plateau and surrounds are eucalypt forest, but the plateau is ca. 50 m higher in elevation than surrounding areas. Note that the location of sunny outcrops was limited by a scarcity of bare rock habitat.

Methods

Study area

Our study plateau (Monkey Gum) was located west of Nowra in south-eastern New South Wales, Australia (Fig. 1). The plateau and surrounding valleys are dominated by closed-canopy eucalypt forest, except for bare rock habitats located near cliff edges. At this site, bare rock habitat has declined by 24% over the past 65 years because of an increase in woody vegetation density (Pringle *et al.* 2009).

The reptile assemblage consists of 10 common species (eight lizards and two snakes) that vary in activity patterns (from diurnal to nocturnal), habitat preference (forest dwelling to rock outcrop specialists) and home range (Table 1). Most of the lizards maintain home range sizes of only a few hundred square metres (Pike *et al.*, unpubl.), while the snakes range much more widely (males have an average home range of 3.3 ha; Webb & Shine 1997). Despite the snakes having larger home ranges, individual snakes often return to use the same individual rocks, both within and among years (Webb & Shine 1997). Consequently, individual rocks represent important habitat components (Pike *et al.* 2010b).

Experimental design

Our goal was to convert overgrown, shady rock outcrops into open bare rock habitats that mimic naturally occurring sun-exposed habitat. Thus, our experimental design consisted of three replicated outcrop types: (i) shady controls; (ii) sunny controls; and (iii) treatment plots (formerly shaded sites from which surrounding trees were removed). Each outcrop consisted of a contiguous rock platform containing a minimum of five surface rocks under which reptiles could shelter. We identified potential sites by walking a 5.6 km stretch on the western side of the plateau and searching for 'sunny' or 'shady' outcrops. We later confirmed these designations using hemispherical photography and gap light analysis (see detailed Methods and Results). To minimize the effects on non-target arboreal species during tree removal, we selected treatment outcrops that contained one or two common trees that lacked tree hollows. Habitat availability dictated our sample sizes; because open bare rock is scarce along this plateau (Pringle *et al.* 2009), we used all sunny outcrops that met our criteria (n = 20).

Table 1. Summary of species responses to our canopy manipulation (modified from Pike *et al.* 2011), showing the activity times (diurnal, nocturnal), habitat preferences and the outcome of our manipulation in terms of whether our manipulation increased or decreased the relative abundance of individuals and rock usage in treatment patches (as compared to baseline shady habitat). Shown are species found in all three outcrop types. See Pike *et al.* (2011) for full details

Species	Common name	Activity	Habitat preference	Outcome	
				Relative abundance	Rock usage
Lizards					
Acritoscincus platynotum	Red-throated Skink	Diurnal	Forest	Decreased	Decreased
Amphibolurus muricatus	Jacky Dragon	Diurnal	Open forest	Increased	Increased
Cryptoblepharus pulcher	Wall Skink	Diurnal	Rock outcrops	Increased	Increased
Ctenotus taeniolatus	Copper-tailed Skink	Diurnal	Rock outcrops	Increased	Increased
Eulamprus quoyii	Eastern Water Skink	Diurnal	Rock outcrops	Increased	Increased
Lampropholis delicata	Delicate Skink	Diurnal	Forest	Decreased	Decreased
Lampropholis guichenoti	Garden Skink	Diurnal	Forest	Decreased	Decreased
Oedura lesueurii	Velvet Gecko	Nocturnal	Rock outcrops	Increased	Increased
Snakes					
Hoplocephalus bungaroides	Broad-headed Snake	Nocturnal	Rock outcrops	Increased	Increased
Rhinoplocephalus nigrescens	Small-eyed Snake	Nocturnal	Forest	Increased	Increased

A further 55 outcrops were assigned as shady controls ($n = 30$) or treatments ($n = 25$), from which canopy cover was removed (total $n = 75$). Sites were scattered along the plateau, with adjacent outcrops separated by an average distance of 80 ± 10.2 m (Fig. 1). Inter-patch distances were sufficiently large that we recorded only a few individual reptiles moving between patches (<1% of all captured reptiles; Pike *et al.*, unpubl.). Mean patch size was 107.0 ± 9.3 m^2. We did not manipulate sunny and shady outcrops, but we removed trees shading the treatment outcrops on 21 April 2007. One person used a chainsaw to fell trees and cut them into manageable lengths, and two people removed fallen trees from the rock outcrops. All trees that were shading the rock outcrop were removed; most trees were 0.5–15 m from the outcrop (mean distance = 5.5 ± 0.2 m). We applied herbicide to the stumps to prevent resprouting and trimmed additional growth in the summers of 2008 and 2009 (one person-day of effort per year).

On each outcrop, we assigned each rock large enough to shelter a lizard with a unique number (written underneath with a paint pen) and recorded its size (length × width × maximum thickness; each to 0.5 cm). We obtained a GPS position for each outcrop and calculated the straight-line distance between successive outcrops using ArcGIS 9.3 ESRI, Redlands, California, USA.

Canopy cover and solar radiation

Canopy cover affects the amount of solar radiation reaching the ground, which in turn influences microhabitat temperatures (Shine *et al.* 2002; Pringle *et al.* 2003). We used hemispherical photography and Gap Light Analyzer software (Version 2.0; Frazer *et al.* 2000) to quantify canopy cover and model sunlight penetration. Photographs were taken by placing a digital camera with a fisheye lens (to record 180° views of the canopy) directly on the top of numbered rocks in each outcrop. Prior to analysing the photographs, we obtained site-specific information on locality (latitude/longitude) and day length (i.e. sunrise/sunset times and the number of sunshine hours, obtained from a nearby weather station and averaged monthly throughout the calendar year). Gap Light Analyzer uses this information to quantify canopy cover above each rock (expressed as % openness), the total incident radiation transmitted through the canopy (mols/m^2/day) and the amount of time that each rock is exposed to direct sunlight (min/day; see Doody *et al.* 2006). We took photographs above all rocks prior to canopy manipulation and above rocks in the treatment outcrops the week following canopy manipulation (Fig. 2). Eucalypt trees maintain their foliage throughout the year (Williams & Brooker 1997), making additional photographs of control outcrops unnecessary.

We tested for differences among outcrop types using ANOVA with canopy cover and incident radiation as the dependent variables. To compare the amount of sunlight received throughout the course of the year, we used a repeated measures ANOVA with outcrop type as the factor, duration of direct sunlight as the dependent variable (averaged monthly for each outcrop) and month as the repeated measure.

Thermal regimes

We recorded hourly temperatures under rocks by placing miniature dataloggers (Thermochron iButtons; diameter 15 mm, height 6 mm, factory calibrated to 0.5°C) beneath two randomly selected rocks in each outcrop. We replaced dataloggers periodically and gathered thermal data from January 2007–November 2009. We compared thermal regimes among habitats using repeated measures ANOVAs with outcrop type as the factor and temperature as the dependent variable. Using the average of the hourly temperature readings, we ran separate analyses for the weeks before vs after canopy removal.

Magnitude of change in habitat quality

Pringle *et al.* (2009) quantified the amount of bare rock, forest and heath habitats in the northern portion of our study site. To determine the effect of our manipulations on habitat availability, we calculated the proportional increase in bare rock habitat

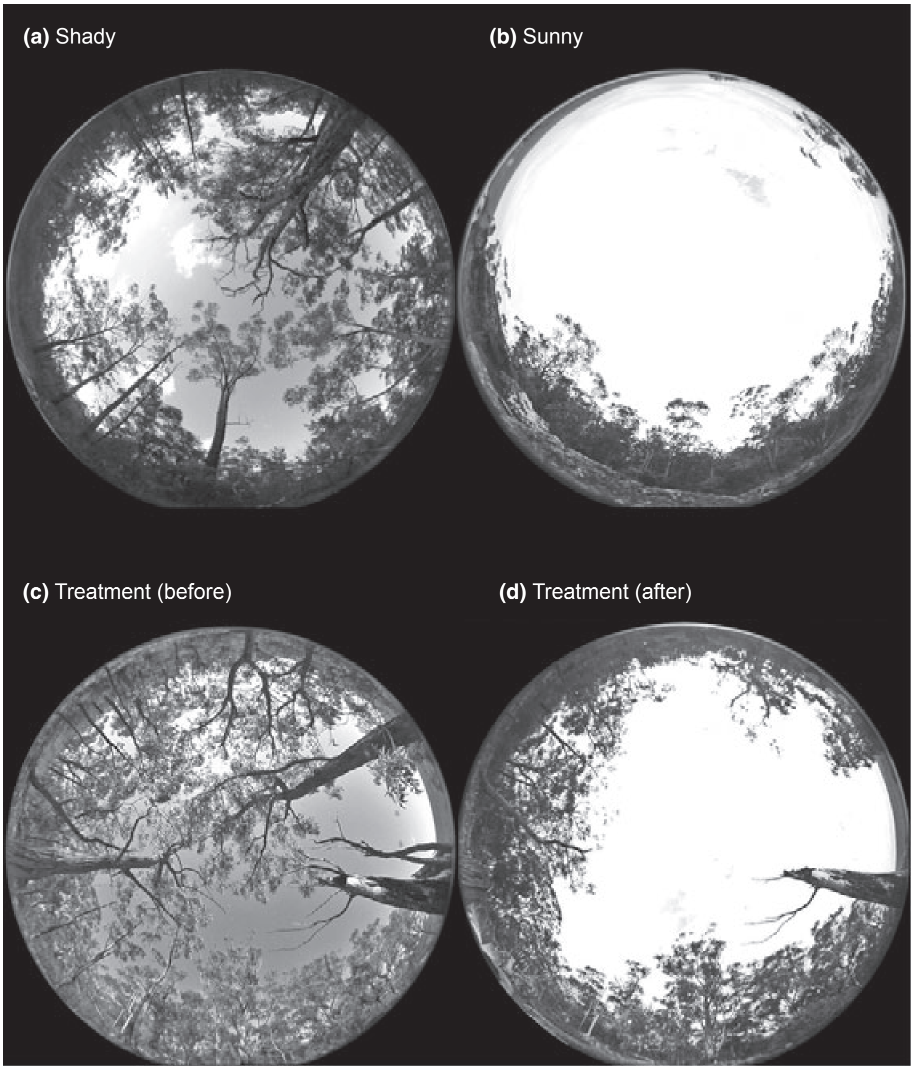

Figure 2. Gap light photographs taken above individual rocks that are representative of (a) shady outcrops, (b) sunny outcrops and treatment outcrops (c) before and (d) after canopy manipulation. Photographs (c) and (d) were taken above the same rock and show how canopy cover changed following canopy removal. Note in (c) and (d) that we did not remove the dead, hollow-bearing tree on the right-hand side of the photograph.

and the concomitant decrease in forested habitat within the subset of our treatment outcrops that overlapped the Pringle *et al.* (2009) study area.

Results

Canopy manipulation and site characteristics

We removed 199 trees on the 25 treatment outcrops (mean of 7.9 ± 0.87 trees per outcrop), primarily *Eucalyptus piperita* Smith (55.3%), *E. gummifera* Gaertner (21.1%), *Banksia* sp. Linnaeus (7.5%), *Syncarpia glomulifera* (Smith) (5.0%), *E. agglomerata* Maiden (3.0%), *E. punctata* de Candolle (2.5%), *E. luehmanniana* von Mueller (2.5%), *Acacia* sp. Linnaeus (2.1%) and *E. paniculata* Smith (1.0%). These trees ranged in size from 5.7 to 92.0 cm in diameter at breast height (mean dbh = 24.2 ± 1.05 cm) and from 1.6 to 49 m in height (mean height = 12.6 ± 0.3 m).

Canopy cover and solar radiation

We took hemispherical photographs above 5–24 rocks per outcrop. Before canopy removal, canopy openness differed among outcrop types; sunny outcrops were more open than both shady and treatment outcrops (Table 1; Figs 2,3a). Our manipulation decreased outcrop-level canopy cover by an average of 18.5 ± 2.31% (range: 2.5–45.6%), which reversed this trend. Following canopy manipulation, sunny and treatment outcrops had similar mean levels of canopy openness, while shady outcrops were substantially more closed-in (Table 2; Figs 2,3a).

Solar radiation transmitted through the forest canopy followed the same pattern. Prior to manipulation, shady and treatment outcrops were similar and received less solar radiation than did sunny outcrops (Table 2; Fig. 3b). After canopy removal, treatment outcrops received similar solar radiation as sunny outcrops, which was substantially higher than in shady outcrops. By removing canopy cover, we increased outcrop-level solar radiation by an average of 6.2 ± 0.88 mols/m^2/day (range: 1.5–15.6 mols/m^2/day).

Before canopy manipulation, sunlight exposure of rocks differed among outcrop types, such that shady and treatment outcrops were similar (Table 2, Fig. 4). Shady and treatment outcrops did not differ significantly in sunlight exposure prior to manipulation (Table 2). Following canopy removal, monthly sunlight exposure again differed by outcrop type, with treatment outcrops now receiving more direct sunlight per day than did shady outcrops (Table 2). By removing canopy cover, we increased the sun exposure of rocks in each outcrop by an average of 164.7 ± 19.3 min/day (range: 10.9–353.7 min/day).

Thermal regimes

Rock sizes (area and thickness) were similar among outcrop types (all $P > 0.20$). Equipment failure lowered sample sizes in some instances; sample sizes ranged from 126 to 137 rocks, except for the week of rainy weather following canopy manipulation, when $n = 68$ rocks (sample sizes were similar among outcrop types).

In the week before canopy manipulation, temperatures beneath rocks were affected by an interaction between time (h) and outcrop type (Table 2; Fig. 5a). Mean nocturnal temperatures were similar

Table 2. Statistical results comparing canopy cover, solar radiation, sunlight duration and thermal regimes between patch types (sunny, shady or treatment in which surrounding vegetation was removed) before vs after removal of trees surrounding treatment patches. Canopy cover, solar radiation and sunlight duration data were unavailable from one treatment patch for logistical reasons, so these comparisons use data from 74 patches

Variable and comparison	*F*	df	*P*	Site comparison *P*-values	
				Shady vs treatment	Sunny vs treatment
Canopy cover					
Before manipulation	62.02	2,71	<0.00001	>0.05	<0.0001
After manipulation	66.54	2,71	<0.00001	<0.0001	>0.05
Solar radiation					
Before manipulation	40.99	2,71	<0.00001	>0.07	<0.0001
After manipulation	98.82	2,71	<0.00001	<0.0001	>0.08
Sunlight duration					
Before manipulation	Patch type: 52.75	2,71	<0.00001	$F_{1, 52} = 2.51$; $P = 0.12$	
	Patch type*time: 0.52	22,781	0.52	$F_{11, 572} = 0.23$; $P = 0.99$	
After manipulation	Patch type: 60.84	2,71	<0.00001	$F_{1, 52} = 71.42$; $P < 0.0001$	
	Patch type*time: 1.02	22,781	0.44	$F_{11, 572} = 1.86$; $P = 0.04$	
Thermal regimes					
Before manipulation	Patch type: 20.89	2,134	<0.00001	$F_{1, 97} = 2.80$; $P = 0.10$	
	Patch type*time: 11.48	46,3082	<0.00001	$F_{23, 2231} = 1.30$; $P = 0.28$	
After manipulation	Patch type: 11.78	2,65	<0.00001	$F_{1, 47} = 18.92$; $P < 0.0001$	
	Patch type*time: 7.08	46,1495	<0.00001	$F_{23, 1081} = 14.89$; $P < 0.0001$	

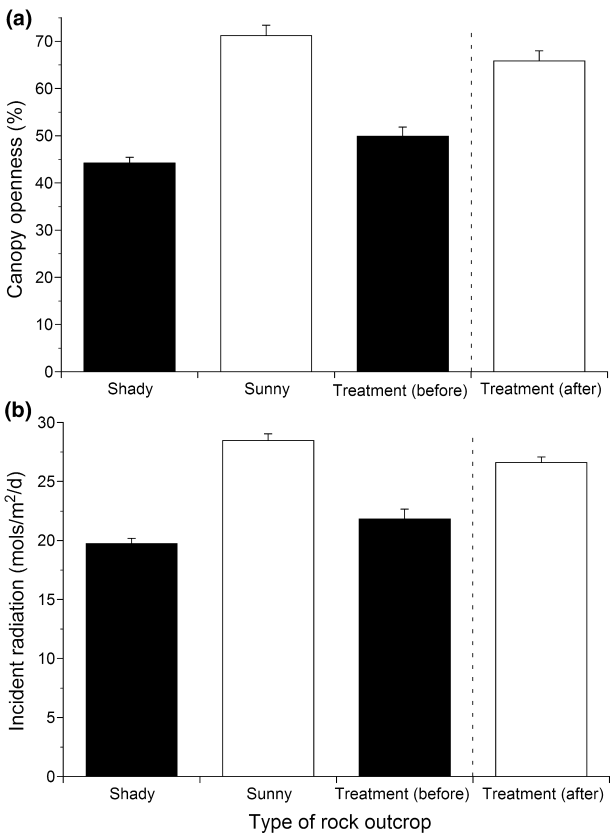

Figure 3. Mean (a) canopy openness (% + SE) above and (b) incident radiation (mols/m^2/day + SE) received by rocks in each of the three types of outcrops. Data for treatment outcrops are shown before and after canopy removal. Treatment outcrops were similar to shady outcrops before canopy removal, but similar to sunny outcrops following removal.

among outcrop types, but diurnal mean temperatures were higher in sunny outcrops than in shady and treatment outcrops (Fig. 5a). Treatment and shady outcrops exhibited similar thermal regimes (Table 2; Fig. 5a).

In the week following canopy removal, mean temperatures in sunny and treatment outcrops were more similar to each other than to those in shady outcrops (Table 2; Fig. 5b). Thermal regimes under rocks in shady vs treatment outcrops now exhibited a significant interaction with the time of day (Table 2; Fig. 5b). Nocturnal temperatures were similar in shady and treatment outcrops, but by day, treatment outcrops averaged up to 3.1°C warmer than shady outcrops (Fig. 5b). Canopy removal thus rapidly changed the daily thermal regimes beneath rocks in treatment outcrops, by increasing temperatures during the afternoon.

Weather conditions became cooler the week after manipulation compared with the week before (note mean temperatures in Fig. 5a vs 5b). Nonetheless, relative thermal regimes beneath rocks in treatment outcrops increased in temperature during the week after canopy manipulation (as compared with the week before manipulation), whereas the sunny outcrops showed a decrease in temperatures over the same period (Fig. 5c). Thus, canopy manipulation increased temperatures beneath rocks in treatment outcrops (Fig. 5c).

The effect of canopy removal on rock temperatures persisted throughout the remainder of 2007 (Fig. 4b) and during 2008 and 2009 (Fig. 6). By selectively removing trees over a 1-day period, with

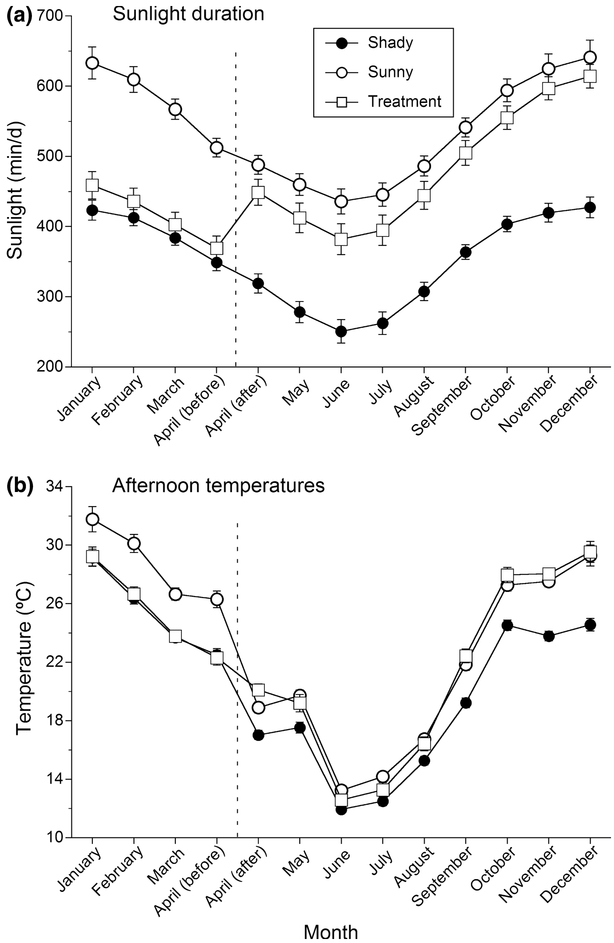

Figure 4. (a) Mean duration of direct sunlight per day (min ± SE) received by rocks within shady, sunny or treatment outcrops, averaged monthly. (b) Temperatures beneath rocks in each outcrop type during 2007, the year in which canopy removal occurred. Shown are mean temperatures (±SE) taken at 1500 hours (the warmest part of the day) and averaged over the months shown. Canopy cover was removed in April (indicated by the dashed vertical line), and data for April are shown before and after manipulation. Error bars are too small to be seen in some cases.

1-day follow ups in each of the next 2 years, we significantly increased afternoon temperatures in reptile shelter sites for the entire study period.

Magnitude of change in habitat quality

Overall, in the 18 treatment outcrops that overlapped the study area of Pringle *et al.* (2009), we created an additional 1865 m^2 of bare rock. This increased the total amount of bare rock habitat on that section of the plateau by 2.4%, but only reduced the total forest canopy by <0.06%. By removing a small proportion of the forest canopy in the region, we more than doubled the total number of sun-exposed rocks in the landscape in a single day (from 266 rocks in sunny outcrops to 614 rocks in sunny plus treatment outcrops), increasing the availability of suitable shelter sites by 131%. Thus, a minor change in vegetation cover dramatically increased habitat critical for the endangered Broad-headed Snake.

Discussion

Increases in vegetation density have occurred in many habitats worldwide, and restoration is often necessary to reverse such trends. In many regions, resource managers use prescribed fire to create open habitats (Inman *et al.* 2007; Fielder *et al.* 2010; Kalies *et al.* 2010; Kane *et al.* 2010), but this technique is inappropriate for thinning eucalypt forests in rocky habitats because the fuel loads are often heterogenous and thus do not carry fire well (Hunter *et al.* 1998; Clarke 2002a,b). We used targeted tree removal to create open habitats for a critically endangered snake. Our field experiment showed that this technique created open, sunny habitat patches with thermal environments similar to those occurring naturally. Habitat manipulation was quick (five person-days), cost-effective (requiring only a chainsaw and herbicide) and immediately improved habitat quality for our target species at both the outcrop and landscape scales. Longer-term monitoring revealed that this technique was highly successful; within 30 months, reptile species, including Broad-headed Snake and Velvet Gecko, increased in abundance at our treatment sites (Pike *et al.* 2011; individual species responses are summarized in Table 1).

Our manipulation was substantially different from traditional selective logging practices, many of which target the removal of trees that can be salvaged for economic gain (including specific species and/or size classes; Lindenmayer *et al.* 2008). Our approach was applied in a very precise manner to target the specific habitat components that reptiles rely on and which are known to be decreasing in availability (Pringle *et al.* 2009). Consequently, selective logging should not be misinterpreted as habitat creation for open-habitat specialists unless shown otherwise.

Despite the success of our experiment, we acknowledge that tree removal may be inappropriate for some systems. For example, harvesting individual trees in Amazonian rainforests creates more extreme microhabitat temperatures than those found in natural gaps, rendering such man-made canopy gaps unsuitable for many ectotherms (Vitt *et al.* 1998). In rainforests, canopy removal is unlikely to produce functionally useful changes in habitat quality for most species of native fauna. We conclude that selective tree removal is compatible with habitat restoration goals when (i) detailed habitat requirements of target taxa are understood, (ii) *a priori* conservation outcomes are clearly defined and (iii) follow-up monitoring is used to

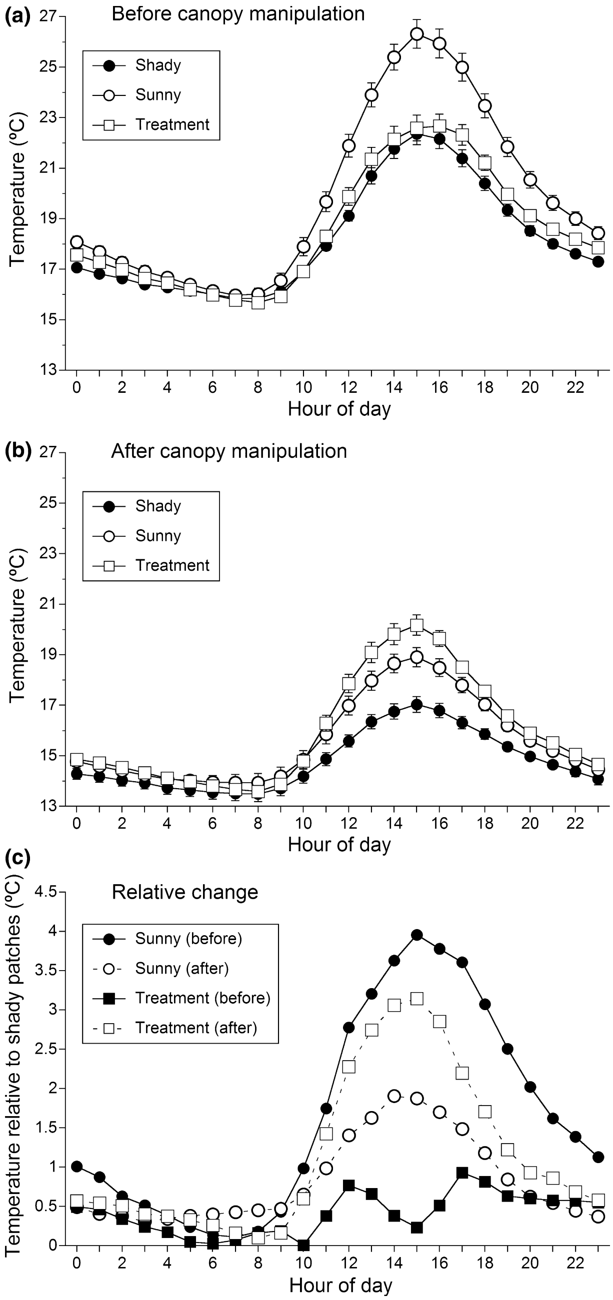

Figure 5. Daily thermal regimes beneath rocks in shady, sunny or treatment outcrops. Shown are mean temperatures (±SE) taken at hourly intervals and averaged over 24 h for: (a) the week prior to canopy manipulation (14–20 April 2007) and (b) the week following canopy manipulation (22–28 April 2007). Also shown are (c) sunny and treatment outcrop temperatures relative to shady outcrops, shown before and after canopy manipulation. Note that relative temperature beneath rocks in sunny outcrops decreased before vs after canopy manipulation (because of ambient weather), but that relative temperatures beneath rocks in treatment outcrops increased (because of canopy manipulation). By removing canopy cover in treatment outcrops, we increased temperatures beneath rocks during the day, when nocturnal reptiles use them as shelter. Error bars are too small to be seen in some cases and are not shown in (c).

evaluate changes in habitat characteristics important to the target taxa.

The most important issue in habitat restoration is not whether the process employed is 'natural' (e.g. fire, grazing, etc.), but whether it mimics the effects of natural processes that maintain components of the habitat (Ausden 2007). Targeted tree removal largely achieved this goal; we created a more open canopy structure (Figs 2,3), which increased sunlight penetration (Figs 3b,4a) and therefore increased temperatures beneath rocks (Figs 4b–6). At the landscape scale, we increased both the numbers of sun-exposed rock outcrops and the number of thermally suitable rocks. Within 30 months, manipulated habitat patches were colonized by our target species and were used by a higher diversity and abundance of reptiles than shady sites (Pike *et al.* 2011). Thus, by all measures, our restoration effort was a success. Similar approaches have been suggested for other endangered reptiles (e.g. Massasauga Rattlesnake, *Sistrurus catenatus*; Shoemaker & Gibbs 2010) and have been used to increase the availability and growth rates of native food trees for endangered parrots (Puerto Rican Parrot, *Amazona vittata*; Inman *et al.* 2007) and to increase butterfly host plants (Bergman 2001).

Although alternative techniques (e.g. prescribed fire, mechanical tree removal, application of herbicide) can be used to reduce vegetation cover, these practices may be more difficult to control at small spatial scales, can take longer to improve habitat quality and/or can negatively impact non-target taxa. Recent studies on the effectiveness of different management techniques have shown that tree thinning in conjunction with fire was most effective in reducing tree density and canopy cover (e.g. Fielder *et al.* 2010; Kane *et al.* 2010), while fire alone was least effective (Kane *et al.* 2010). However, these management techniques are more costly to implement and use of heavy machinery to thin trees could break or damage rocks that reptiles rely on. By felling individual trees, we targeted specific components of the habitat (e.g. specific tree species, of specific ages, and in specific locations) while simultaneously protecting ecologically important components of habitat structure (e.g. tree hollows, rare tree species, rocks) and maintaining fine-scale spatial heterogeneity in tree cover.

Habitat management is most effective when the causal reasons for habitat changes can be addressed (Pike *et al.* 2011). Although changes in fire regime

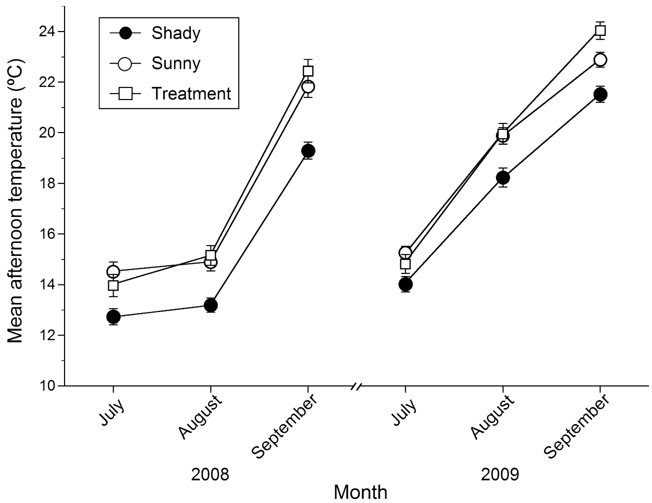

Figure 6. Temperatures beneath rocks in shady, sunny or treatment outcrops during winter and early spring (July–September), a critical period of rock use by nocturnal reptiles. Shown are mean temperatures (±SE) taken at 1500 hours (the warmest time of the day) in 2008 (1 year post-manipulation) and 2009 (2 years post-manipulation). Error bars are too small to be seen in some cases.

(frequency and/or intensity) can contribute to vegetation thickening, an equally important contributor can be the loss of herbivores, or a combination of removing these two disturbances (e.g. Fuhlendorf *et al.* 2009; Levick *et al.* 2009; Staver *et al.* 2009). If both herbivory and fire shape the structure of forested habitats (Fuhlendorf *et al.* 2009; Levick *et al.* 2009; Staver *et al.* 2009), then simply applying fire is unlikely to recreate historical patterns of habitat availability in the absence of herbivores and could even result in more closed canopies (because vegetation growth following fire is no longer suppressed by herbivores). With the loss of local herbivores (as has been suggested in our study system; Webb *et al.* 2005), active management is crucial for maintaining suitable habitat for species dependent on open habitats.

In conclusion, we used an active management approach to rapidly transformed shady overgrown habitats into sun-exposed rock outcrops. By doing so, we reversed historical declines in a habitat type critical for an endangered species and its prey and increased the abundances of those species restricted to rock outcrop habitats (Pike *et al.* 2011; Table 1). In habitats where vegetation overgrowth threatens the viability of declining species, selective tree removal provides a rapid and effective means of habitat restoration. Although this approach may not provide a long-term solution to the problem, it can help buy time for critically endangered taxa until alternative approaches (e.g. reintroduction of herbivores, reimplementation of natural disturbances) are experimentally tested and implemented (Waldrop *et al.* 2008, 2010). Given the logistical difficulties of implementing fire in wilderness areas and fragmented habitats, a range of management techniques may be necessary to conserve open-habitat specialists threatened by vegetation overgrowth.

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Box 1. Implications for Managers

- **Historical changes in disturbance regimes have decreased the habitat availability of many species that rely on habitats with open canopies.**
- **In many instances, the reasons behind these habitat changes are unknown, but traditional management practices (such as fire) may be ineffective or impractical to restore historical habitats.**
- **Selectively removing individual trees that are responsible for shading important habitat components for fauna can be extremely effective at increasing habitat availability.**
- **Careful application of this technique and follow-up monitoring are crucial to ensuring that such practices are beneficial to wildlife.**

