Marsupial response to matrix conversion: results of a large-scale long-term 'natural experiment' in Australia

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Abstract

We quantified changes in forest-dependent mammal populations when the habitat in which they live remains intact but the surrounding matrix is converted from open grazed land to closed pine plantation forest. This situation is increasingly common as plantations are often established on formerly cultivated or grazed land.

We conducted a large-scale (30 Km^2), long-term (14 years) fully controlled and replicated (111 sites) 'natural experiment' in south-eastern Australia. The study focused on the effects of changes occurring in the matrix on mammals which inhabit patches of native *Eucalyptus* woodland.

We found that none of the five target species in our study (two macropods, two possums and a glider) responded negatively to pine plantation establishment. For three species (the sugar glider *Petaurus breviceps*, the red necked wallaby *Macropus rufogriseus* and the swamp wallaby *Wallabia bicolor*) the response to plantation establishment was positive (i.e. increase in colonization/patch use in sites surrounded by pine plantations) whereas the two possums (the common ringtail possum *Pseudocheirus peregrinus* and the common brushtail possum *Trichosurus vulpecula*) were positively affected by the amount of native tree cover surrounding sites, rather than pine plantation establishment.

We foresee two strong implications of our work for the conservation of mammal species in agricultural areas subject to multiple land-use changes: 1) Our results suggest that converting agricultural land to pine plantations will not affect our target mammalian species negatively; rather, it may facilitate colonization of remnant patches of native vegetation by some species. 2) Our findings underscore the critical importance of preserving remnant native vegetation within plantations, as it may decrease the risk of local extinction for some species or facilitate the colonization of new sites for others. Thus, retention of patches of remnant native vegetation should be part of the design of future plantations.

Key words: land-use change; mammals; pine plantations; habitat loss; colonization; extinction.

1. Introduction

The conversion of forested and agricultural land to tree plantations is a major driver of global change (Foley et al., 2005). Humans have converted natural forests to forest plantations for thousands of years (e.g. Romans used to convert oak forests to pine plantations for timber production; Ginanni, 1774). However, in the last few decades, there has been an unprecedented global increase in planted forests as the Food and Agriculture Organization recorded a worldwide increase of 5 million hectares/year in the period 2000-2010. This is due to an increased demand for wood and for carbon storage (Jackson et al., 2005; Paquette and Messier, 2010).

A key distinctive feature of tree plantations compared to other types of crops is that they are often extremely large (i.e. covering areas thousands of hectares in size)and need to be managed over long time periods (crop rotation length is usually 7-50 years; Pawson et al., 2013). Consequently, the conversion of agricultural lands to tree plantations can have broad scale and long-lasting impacts on landscapes. How does biodiversity respond to these large-scale and long-term changes? The majority of existing studies have focused on birds, where response to plantations has ranged from positive to negative (Luck and Korodaj, 2008; Mortelliti et al., 2014, 2015; Tomasevic and Estades, 2008; Villard and Haché, 2012). Similarly, the few studies focused on mammals have found complex responses to plantation establishment (Lindenmayer et al., 2008, 1999a; Youngentob et al., 2013), with some studies showing an increase in dispersal and connectivity due to plantation establishment (e.g. Banks et al., 2005; Taylor et al., 2007) and others suggesting an opposite effect (e.g. Lancaster et al., 2011). Previous studies, however, have focused mainly on comparing remnant patches surrounded by forest plantations with contiguous non-fragmented areas (i.e. have contrasted forest plantations vs native forested habitat, Stephens and Wagner, 2007; Youngentob et al., 2013). Other studies have compared different matrix types (e.g. pine plantations vs. others) but have typically focused on single species or have employed an observational approach (e.g. Anderson et al., 2007; Umetsu and Pardini, 2006). Therefore, clear patterns of mammalian responses to the conversion of the agricultural matrix to forest plantations have not yet emerged in the existing literature.

Afforestation of cultivated or grazed land is increasingly common worldwide (Cyranoski, 2007; Sedjo, 1999) and thus empirical evidence to support decision-making is urgently needed. To contribute to filling this knowledge gap we focused our study on the following research question: ***What happens to mammal populations when the habitat in which they live remains intact but the surrounding matrix is converted from open grazed land to closed plantation forest*?**

Vagility is a key driver of animal responses to land-use changes (Kennedy et al. 2010; Watson et al., 2014), indeed many species of non-volant mammals are limited in their movements and are therefore very sensitive to matrix modifications (Anderson et al., 2007; Gascon et al., 1999; Sozio et al., 2013). The conversion of open agricultural land to a closed plantation environment may thus facilitate the movement of forest-dependent species. Tree plantations may therefore represent an opportunity for keeping landscapes productive for commodities like wood and paper whilst also increasing connectivity between patches of remnant vegetation (Brockerhoff et al., 2008; Tomasevic and Estades, 2008).

To evaluate the effects of matrix conversion on mammals, we conducted a unique large-scale (30 Km^2) long -term (14 years) fully controlled and replicated (111 sites) landscape scale ‘natural experiment’ in south-eastern Australia. The goal of the experiment was to compare mammal populations living within 55 *Eucalyptus* patches surrounded by maturing pine (*Pinus radiata*) plantations (*treatment sites*), with mammal populations living within 56 *Eucalyptus* patches where the surrounding matrix remained unchanged (*control sites*, where the matrix remained pasture). We targeted five forest dependent marsupials varying in body size and with contrasting mobility, ranging from arboreal marsupials (the common ringtail possum *Pseudocheirus peregrinus* and the common brushtail possum *Trichosurus vulpecula*) and gliders (the sugar glider *Petaurus breviceps*), to wide-ranging macropods (the red-necked wallaby *Macropus rufogriseus* and the swamp wallaby *Wallabia bicolor*). We predicted that the target mammalian species would display an overall positive response (i.e. increase in patch use or patch colonization) to the conversion of agricultural land to forest plantations, as the new plantation matrix surrounding the native woodland was expected to provide more sheltered cover (compared to grazed land) for ground-dwelling species and facilitate movement for arboreal species.

2. Methods

2.1 Study area

Our study was conducted in the Nanangroe area (New South Wales, Australia; Fig. 1). The Nanangroe area lies approximately 500 km northeast of Melbourne (co-ordinates 34°54' - 35°4' and 148°32’ - 148°18’ E, altitudinal range: 250-750 m a.s.l.), covers approximately 30 km² and the region is characterised by hot summers and cool winters (temperate climate). The native vegetation is characterised by open woodlands dominated by white box (*Eucalyptus albens*), red box (*E. polyanthemos*), yellow box (*E. melliodora*), red stringybark (*E. macrorhyncha*) and Blakely’s red gum (*E. blakleyi*). More than 80% of the original temperate *Eucalyptus* woodland has been cleared for grazing.

2.2 Experimental design

The Nanangroe area hosts a large-scale landscape transformation experiment known as ‘the Nanangroe experiment’ (Lindenmayer et al., 2001). In 1998 and in 2000 Forest New South Wales established a series of large scale Radiata pine (*Pinus radiata*) plantations in formerly grazed areas for the purpose of a) timber and pulp production and b) carbon storage. During the establishment of the plantations native *Eucalyptus* patches remained uncleared and thus were surrounded by a changing matrix (i.e. native pastures being converted to maturing pine plantations).

The experimental design of the study is composed of 55 treatment *Eucalyptus* patches surrounded by maturing pine plantations and 56 control *Eucalyptus* patches where the surrounding matrix sites has remained composed of pastures. The selection of sites followed a replicated, random stratified procedure. Treatment sites were selected based on the following factors: a) patch size b) age of the pine plantations and c) number of boundaries between the patch and the plantations. Patch size was selected according to the following size classes: 0.5-0.9 ha (15 replicates), 1.0-2.4 ha (20 replicates), 2.5-4.9 ha (15 replicates), 5.0-10 ha (4 replicates), >10-15 ha (2 replicates). *Eucalyptus* patches were surrounded by pines belonging to two age-cohorts (cohort 1= pines planted in 1998; cohort 2= pines planted in 2000). Eleven *Eucalyptus* patches had 1-2 open boundaries with grazed land, whereas the remaining patches had 1 boundary with grazed land or were completely surrounded by pines. Each treatment site was matched with a control site of similar size. A summary of the experimental design is provided in Table S1.

2.3 Mammal surveys

Sites were surveyed for the presence of target mammal species by means of spotlighting transects (length of the transect =200m; 1 transect per site). Each *Eucalyptus* patch was surveyed by an observer holding a 50W spotlight and walking at an average speed of 3 km/h. Sites were surveyed during spring of the following years: 1999, 2001, 2004, 2006, 2009, 2011, 2012 and 2013. Surveys were conducted starting one hour after dusk and terminated 3 hours later to reduce observers fatigue and consequent bias in detectability. Additional surveys conducted within pastures and within plantations showed that mammals were virtually absent from these areas, confirming that these areas are ‘matrix’ used for occasional movements rather than habitats for the target species (Lindenmayer et al., 2008). For this reason, these data were not included in our analyses.

2.4 Data analysis

False absences are a major source of bias in wildlife distribution studies (MacKenzie, 2005), particularly in studies focusing on mammals (Mortelliti and Boitani, 2007). To control for potential bias caused by false absences, multiple season occupancy models (MacKenzie et al., 2003) were fitted on detection history data gathered through the spotlighting surveys conducted in the period 1999-2013. Each of our 111 *Eucalyptus* patches was defined as a *site* (MacKenzie et al., 2003). The transect was divided into two segments (0-100 m and 100-200m) and each segment was considered as a *visit* to a site (see discussion below on spatial replication in spite of temporal replication). Populations were assumed to be closed between visits occurring in the same year, but open to colonization/extinction between years. We emphasise that while the term ‘colonization’ and ‘extinction’ of local populations may be appropriate for our smaller target species these should be interpreted as ‘change in patch use’ for the larger macropods (MacKenzie, 2005) as these species may include several patches within their home range (Tyndale-Biscoe, 2005; Van Dyck and Strahan, 2008).

The substitution of spatial replicates for temporally repeated surveys has been adopted elsewhere (e.g. François et al., 2008; Sadoti et al., 2013; Sirami et al., 2008). Following Hines et al. (2010), a 'multiple season for correlated detection' model was preliminary fitted through software Presence (version 7.1) to check for evidence of potential spatial dependence between visits. As little support for spatial dependence was found (i.e. correlated-detection models ranked lower than the null model), analyses were conducted using ordinary multiple season models through the 'unmarked' package (Fiske and Chandler, 2011) in R.

2.4.1 Predictor variables.

Four variables were selected to be included as predictors in the occupancy models: a) treatment (e.g. pine vs grazing context), b) time elapsed since the beginning of the study c) vegetation type (based on dominant tree species, categories are listed in Table S2) and d) *Eucalyptus* tree cover surrounding the site. *Eucalyptus* tree cover was measured in a circle (250m radius) centred on each site. Tree cover included habitat patches and isolated trees and was measured by using digitised aerial photography in ArcGIS 10.1. We chose to use the tree cover in the circle rather than patch size because of the critical role played by scattered trees in determining the occurrence of Australian native fauna (Fischer et al., 2010; Manning et al., 2006). Following Fahrig (2013), tree cover in the circular area (including the *Eucalyptus* patch and scattered trees) was considered to be a proxy for habitat amount and isolation. The 250 m radius was selected following preliminary analyses (i.e. best model fit compared to other distances).

We adopted a three step approach to fitting models:

a) *Detection probability (p)* was modeled as either constant or as function of year (both as categorical and continuous covariate). The continuous covariate implied a trend in detectability whereas the categorical covariate implied a year-specific probability of detection which could be caused by several factors such as variability in weather and fluctuations in resource availability. The variable(s) included in the top ranking models (as measured through the Akaike Information Criteria) were retained in the following steps (b-c).
b) Factors influencing Ψ_1 (probability of a site being occupied in the first study year) were then modelled. Predictor variables included: vegetation type and tree cover in the 250 m circle (Table S2). Variable(s) included in the top ranking model were retained the in the following step.
c) Finally, we modelled factors affecting the *probability of colonisation (Υ)* and the *probability of extinction (ε)*. Predictor variables included: treatment (a site surrounded by pines vs control), year since the beginning of the study (continuous variable) and tree cover in the 250 m circle. Two-way interactions for these variables (e.g. treatment* year) also were included as well as models with different effects for colonization and extinctions: e.g. Υ(treatment*time), ε(tree cover). Although our sample size is relatively large for landscape ecological studies, we opted to keep the ratio of the number of parameters to number of sites relatively small, so did not include three way interactions. Furthermore, as our sample design was specifically focused on the evaluation of the treatment effect we chose not to include treatment*tree cover interactions, as the number of replicates for larger patches was limited (which is a typical feature of many large scale fragmentation studies; Gibson et al., 2013; Laurance et al., 2011).

Inference was based on model averaged estimates (including models within 2 ΔAIC; Burnham and Anderson, 2002). Goodness of fit of each model was measured using Nagelkerke's pseudo R^2(range of values:0-1).

The variable "year" should be interpreted as 'time since the beginning of the study', which in the case of treatment sites, also acted as a proxy for the time since plantation establishment. We were not able to identify a method to include a time covariate for treatments only as a value of 'zero' in the control sites would not be meaningful. Similarly, it was not possible to distinguish between the two cohorts of plantation establishment and simultaneously modelling control sites. We opted to use the variable ‘time since start of the study’ to avoid conducting separate analyses for treatments and controls. We therefore acknowledge that this variable was only a proxy for time since the establishment of the plantation. Occupancy models were fitted using the *unmarked* package for R (Fiske and Chandler, 2011).

Our experiment was designed to reduce the amount of spatial dependence between sites. Key features of the design included: 1) The inclusion of four independent plantations as ‘treatments’ (Fig. 1). 2) Woodland control sites were distributed among six different farms. 3) Average nearest neighbour distance was 507 m (range 114-1158 m) for treatment sites and 507 m (range 188-3195 m) for control sites. Nevertheless, we checked for spatial autocorrelation in the residuals of the most parameterised model by using a spline correlogram (Zuur et al., 2009). We calculated correlograms for each of the eight time periods and could not find evidence of spatial autocorrelation for any of the target species.

3. Results

Throughout the duration of the study, arboreal marsupials were detected more often than the macropods (total number of detections for arboreal marsupials: common brushtail possum=316, common ringtail possum=199, sugar glider=36; total number of detections for macropods: swamp wallaby=35 detections, red-necked wallaby=29 detections). All five species were detected in both treatment and control sites.

The detectability of the swamp wallaby and the sugar-glider increased through the years, the detectability of the common ringtail possum varied on a year-by-year basis, whereas the detectability of the common brushtail possum and the red-necked wallaby was constant throughout years (Table 1).

The variable ‘treatment’ was included as predictor in the top ranked models of the 2 macropods and the sugar glider. For all the three species the establishment of pine plantations increased the probability of a site being “colonized” during the 16-year study period (Fig. 2 and Fig. 3, Table 1). Little support was found for a treatment effect on the common brushtail and common ringtail possum (i.e. variable treatment not included as predictor for Υ and ε in top-ranking models and with large standard errors for parameter estimates,Table 1). The colonization probability of the common ringtail possum was influenced mainly by the amount of native tree cover surrounding the site: higher tree cover led to higher chances of a site being colonized (Fig. 3). The extinction probability of the common brushtail possum was affected mainly by the amount of native tree cover surrounding the site: the probability of local extinction was lower in sites surrounded by higher amounts of tree cover (Fig. 3). The amount of tree cover surrounding the site was the most important predictor of the probability of a site being occupied during the first survey (Ψ_1) for all the species apart from the swamp wallaby. Vegetation type and two-way interactions had little support in the top ranked models.

4. Discussion

4.1 Biological interpretation of the models

The positive response of the two macropods to plantation establishment is in line with existing knowledge on these species (Van Dyck and Strahan, 2008). Both species are strongly associated to forest cover (Van Dyck and Strahan, 2008) and therefore it is likely that the establishment and maturation of the plantations facilitated movement between *Eucalyptus* woodland patches.

The sugar glider is the most widespread glider in Australia; however, its distribution in fragmented landscapes is usually limited (Lindenmayer, 2002; Suckling, 1982; Tyndale-Biscoe, 2005). Although its gliding capabilities are relatively high (up to 50m), its ability to move across large open areas is inevitably limited and therefore afforestation may facilitate the dispersal and movement of this species. Our results suggest that pine plantations may have favoured the colonization of sites embedded within the changing matrix (i.e. the probability of a treatment site being colonized was almost twice the probability of a control site being colonised; Fig. 2).

Both the common brushtail possum and the common ringtail possum were relatively common in treatment and control sites, which explains the lack of a strong ‘treatment’ effect in both species. In accordance with previous studies (Lindenmayer et al., 1999b; Youngentob et al., 2013), we found that the amount of tree cover surrounding a site had a strong effect on the long term colonization/extinction dynamics of these species. In the case of the common ringtail possum, tree cover increased the chances of a site being colonized. Conversely, in the case of the common brushtail possum, tree cover surrounding a site decreased the likelihood of a local extinction. The chances of a local extinction were close to zero within woodland patches surrounded by areas with a moderate amount of tree cover (Fig. 3). These results underscore the critical role played by remnant native vegetation in highly fragmented and plantation-dominated landscapes (Lindenmayer et al., 1999b; Youngentob et al., 2013). Furthermore, our results highlight that the conversion of the agricultural matrix to pine plantation does not interfere with the dynamics of these two species.

In line with previous theoretical and empirical studies on mammals in modified landscapes we show that the matrix matters (Driscoll et al., 2013; Prevedello and Vieira, 2009) and may be actively used by individuals thus increasing connectivity (Anderson et al., 2007; Pita et al., 2007; Sozio et al., 2013). Compared to previous studies focusing on mammals in plantation landscapes, our work provides more consistent results. We show that the response of our target forest-dependent mammal species was either positive or neutral, suggesting that matrix conversion through afforestation may not have negative effects. The relatively clear patterns that we have observed are likely to be a consequence of the experimental approach that we have followed, the strong contrast between matrices and the extensive duration of the study. The patterns we have uncovered were not identified in a previous study conducted in this area (Lindenmayer *et al.*, 2008) possibly because the present study was substantially longer (14 years vs 7 years). Most importantly, our study focused on different ecological variables (population turnover rather than occupancy).

Our study was focused on two key parameters (local colonization and extinction). However, we acknowledge that future studies focusing on more detailed dynamics of the populations, such as fluctuations in abundance and assessment of connectivity through landscape genetics techniques, will provide further understanding on the mechanisms involved in determining population turnover (Mortelliti et al., 2014).

4.2 Implications for conservation

How generalizable are our results? Although our study is one of the largest empirical assessments of the impact of matrix conversion on mammals ever conducted, further studies are required before we can generalize the patterns we have observed to other areas and species. Nevertheless, we emphasize two key features of our study suggesting that the patterns we have observed may observed in other systems: 1) Our study targeted the impact of pine plantation establishment. Pines are the most commonly used species in forest plantations worldwide (i.e. they occur in 20% of plantations; Brockerhoff et al., 2008). Thus the type of plantation that we have studied is highly representative of plantation establishment globally. 2) We focused on five species with contrasting mobility and encompassing a wide variation in body size (weight ranging from approximately 130g in sugar gliders to up to 20Kg in wallabies) thus we were able to include a relatively representative variety of life-history traits for a mammalian study. Furthermore, although all our species are marsupials, these have 'ecological equivalents' in Eutherian mammals (e.g. gliders are equivalent to flying squirrels etc). We acknowledge, however, that replicating our study in different continents and with different mammalian orders will surely help disclose general patterns on mammalian response to plantation establishment.

Finally, we emphasise that the contrast between the internal structure of the native vegetation and the forest plantation will also have a strong impact on the response observed. In our case the pine plantations have a relatively dense structure whereas *Eucalyptus* patches are relatively open. Replication of our study in different environments (i.e. rainforest with dense structure) would surely help to understand how generalizable are our results.

Landscape managers all over the world are increasingly faced with the issue of whether to convert agricultural areas to forest plantations (Cyranoski, 2007) - this study was designed to support such decision making. The results of our long-term large-scale ‘natural experiment’ have two strong implications for the conservation of mammal species in agricultural areas subject to multiple land-use changes (Watson et al., 2014):

1) For the first time we provide field-based empirical evidence that the conversion of the agricultural matrix to pine plantations does not have negative effects on the forest-dependent mammals. Our results therefore suggest that plantations may be a ‘lesser evil ‘compared to other types of matrix and that a sequence of land-cover changes from open-areas to tree plantations may favour habitat specialists such as the mammals we have studied.

2) Our results underscore the critical importance of preserving patches of native vegetation within plantations therefore retention of patches of remnant native vegetation should be part of the design of future plantations.

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Figure legends.

Fig. 1. Map of the study area. Black triangles are control sites (*Eucalyptus* patches surrounded by grazed areas); black circles are treatment sites (*Eucalyptus* patches surrounded by pine *Pinus radiata* plantations). Barred areas are pine plantations.

Fig. 2. Model predictions. Model predictions (including 95% confidence intervals) based on model averaged estimates of top ranking models (ΔAIC<2) for the red-necked wallaby *Macropus rufogriseus*, the swamp wallaby *Wallabia bicolor* and the sugar glider *Petaurus breviceps*. The three species responded positively to pine plantations (increase in “colonisation probability” in sites surrounded by pine plantations).

Fig.3. Model predictions. Model predictions (including 95% confidence intervals) based on model averaged estimates of top ranking models (ΔAIC<2) for the ringtail possum *Pseudocheirus peregrinus* and the brushtail possum *Trichosurus vulpecula*. Both species were mainly affected by the amount of tree cover surrounding the sites (measured in a 250 m radius around each site).

Table 1. Model ranking according to ΔAIC (delta Akaike Information Criterion); only models <2 ΔAIC are shown. Ψ = probability of a site being occupied during the first survey, ϒ = probability of colonization; ε = probability of extinction, p = detection probability; T= Treatment; Vegtype = vegetation type (categories listed in table S2); Y = year (categorical covariate); Yn = year (numeric covariate); H = tree cover within a 250 m radius circle; nPars = number of estimated parameters; R^2 = Nagelkerke's coefficient of determination; (.) = constant model (no covariate); W= cumulative model weight (i.e. sum of the Akaike weight of the given model and higher ranked models).

Species	**model**	**nPars**	**ΔAIC**	R^2	***W***
Red-necked wallaby *(Macropus rufogriseus)*	Ψ(.)ϒ(T)ε(Yn)p(.)	6	0.00	0.18	0.47
Swamp wallaby (*Wallabia bicolor)*	Ψ(H)ϒ(T)ε(H)p(Yn)	8	0.00	0.44	0.42
	Ψ(H)ϒ(T)ε(.)p(Yn)	7	1.62	0.41	0.61
Sugar glider *(Petaurus breviceps)*	Ψ(H)ϒ(T)ε(.)p(Yn)	7	0.00	0.10	0.14
	Ψ(H)ϒ(.)ε(T*Yn)p(Yn)	9	0.45	0.13	0.26
	Ψ(H)ϒ(.)ε(T)p(Yn)	7	1.91	0.08	0.33
	Ψ(H)ϒ(T)ε(H)p(Yn)	8	1.94	0.10	0.39
	Ψ(H)ϒ(T)ε(Yn)p(Yn)	8	2.00	0.10	0.45
	Ψ(H)ϒ(T)ε(T)p(Yn)	8	2.00	0.10	0.50
Common ringtail possum (*Pseudocheirus peregrinus*)	Ψ(H)ϒ(H)ε(.)p(Y)	13	0.00	0.34	0.16
	Ψ(H)ϒ(Yn)ε(.)p(Y)	13	1.14	0.33	0.26
	Ψ(H)ϒ(.)ε(.)p(Y)	12	1.71	0.31	0.33
	Ψ(H)ϒ(H)ε(H)p(Y)	14	1.87	0.34	0.39
	Ψ(H)ϒ(H)ε(T)p(Y)	14	1.90	0.34	0.46
	Ψ(H)ϒ(H)ε(YN)p(Y)	14	1.99	0.34	0.52
Common brushtail possum (*Trichosurus vulpecula*)					
	Ψ(H)ϒ(.)ε(H)p(.)	6	0.00	0.21	0.28
	Ψ(H)ϒ(H)ε(H)p(.)	7	0.12	0.22	0.55
	Ψ(H)ϒ(Yn)ε(H)p(.)	7	1.33	0.21	0.70
	Ψ(H)ϒ(T)ε(H)p(.)	7	1.79	0.21	0.81

Table S1. Factorial study design of the Nanangroe study (Table modified from Lindenmayer *et al.* 2008).

Site	**Context**	**Cohort**	**No. edges**	**No. of replicates**
Woodland	*Pinus radiata* plantation	1998	1-2	3
Woodland	*Pinus radiata* plantation	1998	3-4	16
Woodland	*Pinus radiata* plantation	2000	1-2	8
Woodland	*Pinus radiata* plantation	2000	3-4	29
Woodland	Grazing land			55

Table S2. Main characteristics of 111 sampled sites. Context: treatment = *Eucalypt* patches surrounded by pine plantation; control = *Eucalypt* patches surrounded by grazing land. Tree cover (hectares)measured in a 250 m radius circle surrounding the site. Vegetation type: 1 = red box and red stringybark (codominant) with apple box (*E. bridgesiana*), long-leaf box (*E. goniocalyx*), and broad-leaved peppermint (*E. dives*); (2) mountain swamp gum (*E. camphora*) and other kinds of vegetation (e.g., river oak *Allocausarina cunninghamiana*); (3) yellow box, white box, red stringybark (codominant), and Blakely's red gum.

Sites	**Context**	**Tree cover**	**Vegetation type**
AWA-1	Control	4.17	3
AWA-10	Control	8.81	3
AWA-11	Control	6.74	3
AWA-2	Control	4.18	3
AWA-3	Control	2.26	3
AWA-4	Control	7.05	3
AWA-5	Control	4.90	3
AWA-6	Control	4.60	3
AWA-7	Control	4.32	3
AWA-8	Control	3.36	3
AWA-9	Control	6.78	3
GRE-1	Control	0.79	3
GRE-2	Control	0.91	3
GRE-3	Control	7.58	3
GRE-4	Control	6.78	3
JWA-1	Control	5.87	3
JWA-10	Control	2.19	3
JWA-11	Control	4.52	3
JWA-2	Control	4.87	3
JWA-3	Control	5.01	3
JWA-4	Control	6.74	3
JWA-5	Control	10.55	3
JWA-6	Control	8.32	3
JWA-7	Control	4.31	3
JWA-8	Control	0.94	3
JWA-9	Control	6.92	3

KEA-1	Control	4.67	3
KEA-2	Control	8.07	2
KEA-3	Control	3.45	3
KEA-4	Control	3.40	3
KEA-5	Control	4.63	3
KEA-6	Control	5.04	3
LUF-1	Control	3.45	3
LUF-10	Control	6.72	3
LUF-11	Control	5.87	3
LUF-12	Control	11.64	3
LUF-13	Control	13.00	3
LUF-14	Control	5.00	3
LUF-2	Control	3.31	3
LUF-3	Control	6.85	3
LUF-5	Control	9.83	3
LUF-6	Control	3.89	2
LUF-7	Control	3.13	3
LUF-8	Control	6.11	3
LUF-9	Control	7.93	3
SKI-1	Control	6.26	3
SKI-10	Control	4.25	3
SKI-2	Control	5.47	3
SKI-3	Control	8.42	3
SKI-4	Control	4.74	3
SKI-5	Control	3.56	3
SKI-6	Control	2.96	3
SKI-7	Control	2.64	3
SKI-8	Control	2.15	1
SKI-9	Control	3.27	3
BUN-1	Treatment	7.68	3
BUN-2	Treatment	6.42	3
COT-1	Treatment	8.21	1
COT-10	Treatment	4.07	3
COT-2	Treatment	2.74	1

COT-4	Treatment	0.96	1
COT-5	Treatment	2.01	1
COT-6	Treatment	2.90	3
COT-7	Treatment	1.82	3
COT-8	Treatment	3.74	3
COT-9	Treatment	9.49	3
EAB-1	Treatment	3.32	2
EAB-2	Treatment	9.06	2
EAB-3	Treatment	8.37	2
EAB-4	Treatment	3.82	2
EAB-5	Treatment	4.28	2
EAB-6	Treatment	7.03	1
EAB-7	Treatment	9.82	2
EAB-8	Treatment	10.61	2
NAN-1	Treatment	1.09	3
NAN-10	Treatment	3.13	3
NAN-11	Treatment	3.10	3
NAN-12	Treatment	5.84	3
NAN-14	Treatment	6.18	3
NAN-15	Treatment	1.44	2
NAN-16	Treatment	2.23	3
NAN-18	Treatment	9.12	3
NAN-19	Treatment	1.93	3
NAN-20	Treatment	6.12	3
NAN-23	Treatment	3.30	1
NAN-24	Treatment	3.24	1
NAN-25	Treatment	8.35	3
NAN-26	Treatment	4.25	3
NAN-27	Treatment	7.43	3
NAN-28	Treatment	5.01	3
NAN-29	Treatment	4.37	2
NAN-30	Treatment	5.72	2
NAN-31	Treatment	1.58	1
NAN-34	Treatment	5.61	3

NAN-35	Treatment	1.26	3
NAN-36	Treatment	5.63	3
NAN-37	Treatment	5.71	3
NAN-38	Treatment	7.03	2
NAN-39	Treatment	5.10	3
NAN-4	Treatment	1.39	3
NAN-40	Treatment	9.23	3
NAN-41	Treatment	14.67	3
NAN-42	Treatment	4.22	3
NAN-43	Treatment	2.22	3
NAN-44	Treatment	4.86	3
NAN-45	Treatment	1.02	3
NAN-46	Treatment	4.49	3
NAN-5	Treatment	2.83	3
NAN-6	Treatment	2.64	3
NAN-7	Treatment	8.77	3
NAN-8	Treatment	3.50	3