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Does thinning regrowth restore habitat for biodiversity?

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More information

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EXECUTIVE SUMMARY

Globally, forest regrowth represents > 50 % the world's forest cover and management of such forests is a key issue for biodiversity, particularly for vegetation communities where regrowth occurs as high densities of small-sized stems, with high intra-stand competition and little to no self-thinning. *Callitris glaucophylla* (white cypress pine) is a native conifer that is geographically widespread in Australia and often occurs in mixed woodland communities. However, fire exclusion and selective removal of associated eucalypts and other co-occurring tree species has resulted in *C. glaucophylla* dominating stands in many parts of its distribution, often at high densities, with little evidence of self-thinning for up to 200 years in low rainfall environments. Thinning is a common silvicultural practice that has been applied to *C. glaucophylla* regrowth in the Pilliga forests of north-west New South Wales (NSW) since the 1930s to reduce stand density and accelerate tree growth. Thinning may also be applied for specific ecological outcomes, such as restoring stand complexity of regrowth to benchmark levels and restoring habitat for biodiversity. Currently, the science is lacking to identify benefits of thinning for biodiversity and this hinders the provision of forest management guidelines but also more broadly, the impacts of uncontrolled regrowth on land degradation within agricultural landscapes.

In this study, we examined the short- and longer-term effects of thinning on vegetation structure and biodiversity by sampling across a chronosequence of time since thinning and establishing a before-after-control-impact thinning experiment. We also assessed the responses of focal taxa (*Nyctophilus* spp., Long-eared Bats) considered to be sensitive to thinning and contrasted these with those of another species (*Vespadelus vulturnus*, Little Forest Bat) that is thought to be less sensitive. The structure and composition of woody vegetation as well as the volume of hollow-bearing coarse woody debris were strongly influenced by time since thinning. Thinning immediately reduced the density of small live (<10 cm dbhob) and dead trees and shrubs which was maintained over the long-term (21–40 years) and resulted in live stem densities comparable to long (>80 years) undisturbed forest. The effect of thinning on structural complexity was mixed and varied with time since thinning. Thinning also increased the volume of CWD which was maintained for at least 21–40 years post-thinning, at which time CWD produced larger hollows. For many other

structural attributes (ground cover, large tree density, hollow density), the effect of thinning was neutral. Overall responses by biodiversity to thinning at both short and longer-term intervals were positive or neutral. However, the different taxa appeared to be poor surrogates for each other, emphasizing the importance of developing management actions that consider multiple taxa. Positive responses to thinning were recorded for the diversity of bats, reptiles and birds. Neutral responses were recorded for the diversity of non-volant mammals and plants. Thinning was associated with a relatively more even distribution of insect biomass among size classes >8 years post-thinning and a greater representation of beetles to overall insect biomass compared to a dominance of smaller moths in unthinned stands.

Day-roosting and nocturnal activity of multiple bat species revealed a complex pattern of habitat use across the thinned-unthinned mosaic that was only partially predicted by ecomorphology. Two closed-space (*N. corbeni* and *N. gouldi*) species considered sensitive to thinning avoided roosting in relatively open, thinned areas as predicted, whereas the few roosts located for *N. geoffroyi* were in thinned patches or in more open areas, suggesting that this species may be less sensitive to thinning. An edge-space species (*V. vulturnus*) roosted in unthinned and thinned patches, though sample sizes were low for this species. In contrast, flight activity of *Nyctophilus* spp. was evenly spread across the forest mosaic, suggesting they actively foraged in both cluttered (unthinned) and uncluttered (thinned) forest. Systematic trapping revealed *N. gouldi* was captured more frequently in unthinned forest, while no difference was found for *N. geoffroyi* and *N. corbeni*. The activity, but not capture rate, of *V. vulturnus* (edge-space) was higher in thinned than unthinned forest as predicted by ecomorphology. Such varied patterns of habitat use have implications for managing the forest landscape and emphasise the value of heterogeneous landscapes for biodiversity conservation. Our findings suggest that it is important to retain some unthinned regrowth forest to provide a mosaic forest structure suitable for a diverse suite of flora and fauna. We recommend that broad-scale application of thinning should be avoided to ensure that heterogeneity of forest structure is maintained across the landscape. Instead, we suggest a patchwork or mosaic of different vegetation densities would be of most value to biodiversity.

CONTENTS

1. GENERAL INTRODUCTION

There is conflicting evidence for the biodiversity benefits of thinning forest regrowth in NSW. Natural regeneration of cypress pine (*Callitris glaucophylla*) can increase densities to the point described as 'locked-up' and associated with a loss of species richness (Clayton-Greene and Aston 1990). Although thinning is currently employed as a routine forest management activity to increase yields of wood products (McHenry et al. 2006), the legitimacy of this practice to have co-benefits in terms of biodiversity has been challenged by recent studies that show no association of *Callitris* spp. density and loss in species richness (Thompson and Eldridge 2005b; Hunter 2013). Further studies have shown impacts to be contingent on logging and thinning histories (Eyre *et al.* 2015). Currently, the science is lacking to identify benefits of thinning for biodiversity and this hinders the provision of forest management guidelines but also more broadly, the impacts of uncontrolled regrowth on land degradation within agricultural landscapes.

The Pilliga forests of north-west NSW support extensive stands of cypress, which is one of the priority species proposed to be restored by thinning. Historical thinning activities in this area have been recorded by State Forests (now Forestry Corp NSW - FCNSW) providing a unique opportunity to examine the impact of past thinning activities on biodiversity over the long (>20 years) and short-terms (<8 years). In addition, thinning manipulation treatments can be imposed to examine immediate (before and after) impacts of thinning on state forest land compared to other tenures. Although ecological and silvicultural thinning are not identical, alleviating competition for resources is the mechanism by which both seek to modify the growth of retained trees.

In this report, we describe the responses to thinning of vegetation structure and biodiversity in dense cypress regrowth forests. First we report on the short- and longer-term effects of thinning on vegetation structure and biodiversity (Chapter 2) as examined by a chronosequence assessment. These components of the broader study comprise two manuscripts, one which has been published (waters et al. 2018) and the other which is in review (Gonsalves et al. submitted – *Forest Ecology and Mangement*).

Next we report on the immediate (<1 year) responses of vegetation structure and key biodiversity taxa to two stages of regeneration (early thinning and second thinning) in a before-after-control-impact experiment (Chapter 3). This component of the broader study will be written up as a manuscript for publication following resampling of sites at a second time point post-thinning.

We then describe the responses of focal taxa (Long-eared Bats) considered to be sensitive to thinning and contrast these with those of other taxa (Little Forest Bats) that are thought to be less sensitive (Chapter 4). This component of the broader study has been submitted (Law et al. submitted – *Diversity*) for publication and is in review. Finally, we draw on key findings from each component of the study to make an assessment of the value of thinning to habitat values and biodiversity in dense cypress regrowth forest (Chapter 5). Management recommendations are also presented. Because Chapters 2-4 have been written to form stand-alone peer reviewed papers, some repetition of content among the introduction and methods sections of these chapters could not be avoided.

2. THE SHORT- AND LONGER-TERM EFFECTS OF THINNING ON VEGETATION STRUCTURAL ATTRIBUTES AND BIODIVERSITY IN DENSE CYPRESS REGROWTH FOREST

2.1. INTRODUCTION

Globally, forest regrowth represents > 50 % the world's forest cover (FAO 2010) and management of such forests is a key issue for biodiversity. This is particularly the case for vegetation communities where regrowth occurs as high densities of small-sized stems, with high intra-stand competition and little to no self-thinning. Thinning is a common silvicultural practice used to reduce stand density to accelerate tree growth (Tappeiner et al. 1997; Finkral and Evans 2008; Kariuki 2008; Horner et al. 2010), with thinnings either retained on site (non-commercial) or recovered as a product (commercial), such as sawlog or firewood. Thinning can also be applied for specific ecological outcomes, such as restoring stand complexity of regrowth to benchmark levels (see Gorrod et al. 2017). Nevertheless, all thinning changes the physical structure and habitat value of forest regrowth. For example, thinned stands of regrowth produce larger stems and a greater number of hollow-bearing trees than unthinned stands with higher stem densities (Horner et al. 2010).

In the short-term, thinning can reduce tree and shrub canopy density by up to 50% (Harrod et al., 2009) and increase vertical canopy gaps which has implications for both fauna and flora. In the longer-term, thinning has been shown to increase the rate of hollow formation in retained trees (Horner et al., 2010). The responses of North American biodiversity to thinning have generally been positive or neutral across taxa for both diversity and abundance, although thinning intensity, time since thinning and the type of thinning applied partially drives the magnitude of response (Kalies et al. 2010; Verschuyl et al. 2011). In Australia, Eyre et al (2015) identified mixed responses to thinning from functional groups of reptiles when controlling for logging effects. Larger skinks and dragons were 7-times more abundant in thinned compared to unthinned white cypress pine *Callitris glaucophylla*, while arboreal geckos and snakes and fossorial skinks and snakes were twice as abundant, respectively (Eyre et al. 2015). Small skinks, however, were twice as abundant in unthinned than thinned stands (Eyre et al. 2015). For functional groups of birds, responses to thinning when controlling for logging effects were neutral (Eyre et al. 2015). Blakey et al. (2016) found that bat activity was 60% lower in unthinned river red gum *Eucalyptus camaldulensis* regrowth compared to thinned and reference stands, with responses of individual bat taxa to thinning mostly neutral. The responses of insect taxa to thinning were also variable and tended to mostly be neutral (Blakey et al. 2016). These studies demonstrate that the responses of fauna to thinning are complex and taxa-specific (e.g., Kalies et al. 2010; Eyre et al. 2015; Blakey et al. 2016).

Callitris glaucophylla is a native conifer that is geographically widespread in Australia (Thompson and Eldridge 2005a; Lunt et al. 2006; Whipp et al. 2012). The species often occurs in mixed woodland communities, including various eucalypts and other tree species such as *Allocasuarina luehmannii* (buloke) (Boland et al. 1984). Fire exclusion and selective removal of associated eucalypts and other co-occurring tree species (Lindsay 1967) has resulted in *C. glaucophylla* dominating stands in many parts of its distribution, with >80 % of basal area represented by the species (Thompson and Eldridge 2005a). The species is slow growing and long-lived, tolerating intense intra-specific competition at high densities (Lacey 1972; Lacey 1973; FCNSW 1988), with little evidence of self-thinning (Thompson and Eldridge 2005b) for up to 200 years in low rainfall environments (Read 1995). Recruitment of the species occurs episodically under suitable conditions (Horne 1990a; Horne 1990b) and is influenced by climate, existing stand densities and land use (Lacey 1973). In many state forests estimated densities of regenerating cypress vary from 163,000 trees ha^{-1} to 620,000 trees ha^{-1} (Horne 1990b).

Thinning of *C. glaucophylla* regrowth has been undertaken in the Pilliga forests in the north-west of New South Wales (NSW) since the 1930s, with non-commercial and commercial treatments applied to dense regrowth (Knott 1995). Time since thinning is a fundamental consideration when assessing responses to thinning, particularly for biodiversity as vegetation structure may change with time, altering habitat. The extensive history of thinning provided an opportunity to assess the short- and longer-term effects of thinning on vegetation structural attributes and biodiversity using a chronosequence of time since thinning. Where possible, our assessment focused on non-commercial thinning treatments, which left residue on the forest floor at the end of the operation and only included sites that had not been logged after thinning. Specifically, we predicted that key forest structural attributes would respond positively in thinned relative to unthinned treatments. For example, thinning treatments would have greater shrub and ground cover due to increased light penetration and greater levels of coarse woody debris (CWD) than the unthinned treatment and potentially greater numbers of hollow trees in the old thinning treatment. For biodiversity, we compared the diversity of different taxa among unthinned, thinned and long undisturbed forest treatments using the geometric mean of species abundances (Buckland et al. 2011). We also developed a composite biodiversity index (di Stefano et al. 2013) and considered species composition. For mobile taxa, we predicted that bat diversity would increase rapidly with thinning (Blakey et al. 2016; Gonsalves et al. in press), while volant insect biomass or composition would not be affected (Blakey et al. 2016; Gonsalves et al. in press). Bird diversity was expected to be positively associated with an increase in the density of understorey shrubs (*Acacia* spp.) associated with thinning (Brown et al. 1991), though initial effects of thinning can be neutral (Kalies et al. 2010). For less mobile taxa, we predicted that the diversity and abundance of reptiles would respond positively over time to thinning (Craig et al. 2009) as reduced stem density and sub-canopy cover in thinned sites may increase the penetration of sunlight (Wetzel & Burgess 2001) for basking (Vitt et al. 1997, 1998). For non-volant mammals, we predicted no general response to thinning (Craig et al. 2009), but rather responses would be species-specific (Converse et al. 2006). We also explored associations between individual species and habitat characteristics using ordination techniques.

2.2. METHODS

2.2.1. Study area and design

The study was undertaken in the Pilliga forests in north-west NSW. Approximately half of these forests are managed by Forestry Corporation NSW for timber production, while the remaining areas are managed for conservation by the National Parks and Wildlife Service, with much of this area previously managed for timber production. In all, the managed lands of the Pilliga cover 535 000 ha and constitute the single largest block of inland plains forest and woodland in Australia. Dominated by white cypress pine *Callitris glaucophylla* and narrow-leaved ironbark *Eucalyptus crebra*, the area has been important for timber production since the 1800s. Much of the Pilliga area was thought to be originally open woodland with a similar plant species composition to today (Rolls 1981; van Kempen 1997). However, increasing tree density, particularly *Callitris* appears to have resulted from a combination of altered burning regimes, introduction of rabbits and interactions with drought and flood years. The vegetation of the Pilliga is today dominated by dense stands of *C. glaucophylla*, black cypress pine *C. endlicheri*, buloke *Allocasuarina luehmannii*, with *Acacia* spp. and *E. crebra* scattered throughout the forest (Thompson and Eldridge 2005a; Law et al. 2016b; Waters et al. 2018). Silvicultural treatment to maximise tree growth for timber production (Forestry Commission of NSW 1986) has developed a forest with a range of management histories, including logging and thinning of various ages.

Thirty sampling sites within the Pilliga were selected to encompass the variability that exists in forest productivity across the entire area and have been previously described in Waters et al. (2018). Sampling took place in six site clusters across the Pilliga in eight State Forests (Baradine, Cumbil, Euligal, Jacks Creek, Pilliga East, Pilliga West, Wittenbra and Yearinan), one Flora Reserve (Wittenbra) and two State Conservation Areas (Pilliga West and Yearinan) (Fig. 2.2.1).

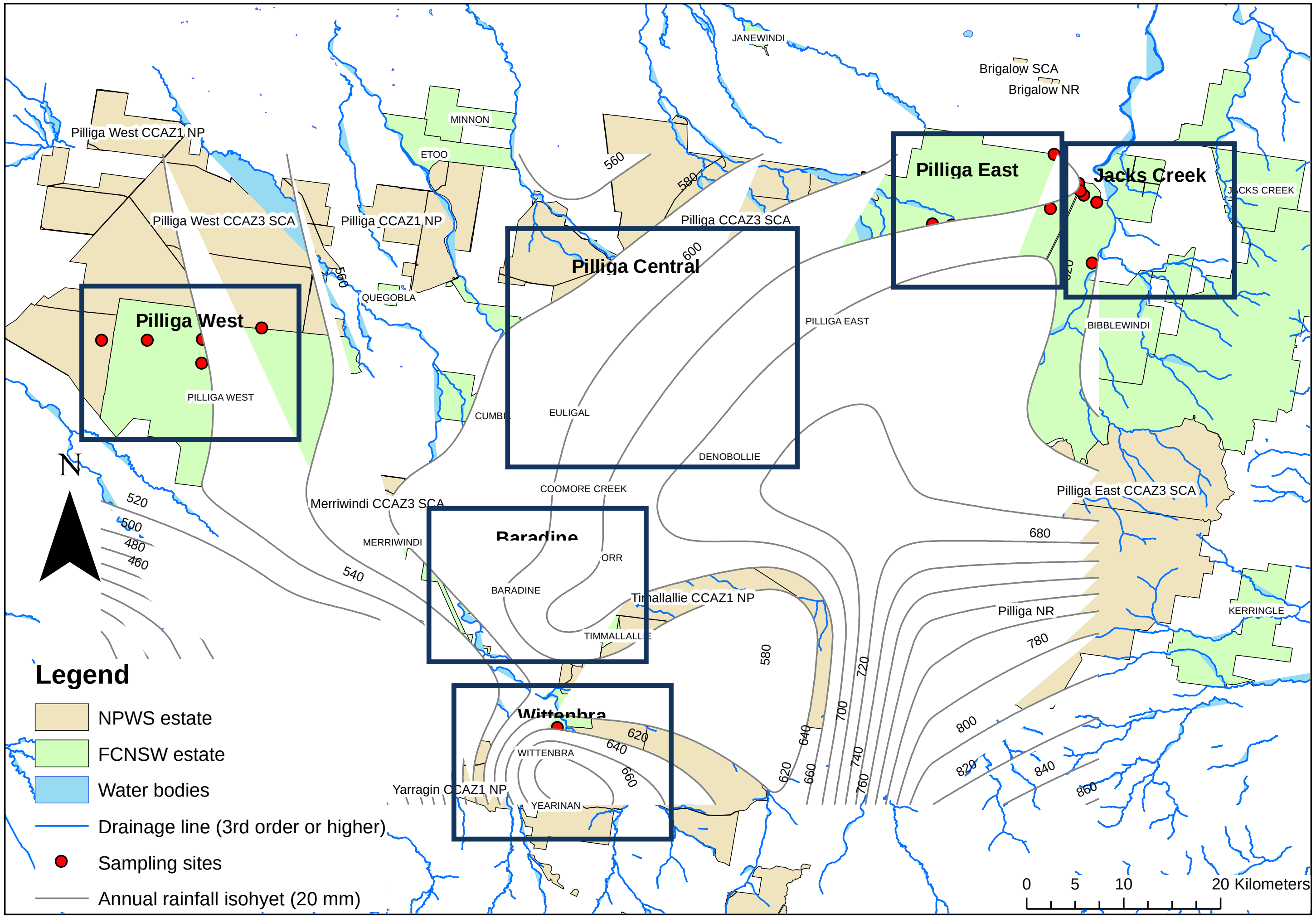


Fig. 2.2.1. Map of study area indicating location of sampling sites representing five forest management treatments within each of six site clusters (after Waters et al. 2018).

To control for variability across the large study area, each site cluster contained five different forest treatments (unthinned, recent thinning, intermediate thinning, old thinning and long undisturbed reference). Unthinned sites supported a high density of small (<10 cm diameter at breast height over bark - dbhob) stems that were suitable for thinning. Recent thinning sites had been thinned <8 years ago using mechanical (chopper-roller) and manual (brushcutting) thinning techniques, with thinnings left on site. Intermediate thinning sites were thinned 8-20 years ago. However, thinning operations at these sites also targeted larger stems for sawlogs which were removed from the site, resulting in a relatively more patchy thinning treatment than our recent thinning treatment. Old thinning sites had been thinned > 20 years ago, with thinnings left on site. Thigh-height small diameter stumps were evident in this treatment, which had persisted over time due to termite resistance of white cypress. All thinning treatments were typically 20-30 ha in size. Long undisturbed reference sites were those that represented the greatest period without disturbance within a 5 km radius of the recent thinning site for a particular cluster. They were characterised by few stumps and large diameter trees (cypress and ironbark), and often contained patches of dense *C. glaucophylla*.

At each site, three 10 × 20 m plots located at approximately 20, 70 and 120 m along each transect were used to measure habitat features (Table 2.2.1), while a 200 m transect for sampling biodiversity was established through areas representative of each forest treatment. To examine whether the response of biodiversity to thinning varies with time, we surveyed a range of fauna groups (bats, birds, invertebrates, non-volant mammals and reptiles) and plant floristics. Surveys were undertaken in spring 2015 to optimise detections for most taxa, although supplementary sampling in the following autumn was also undertaken for reptiles.

Table 2.2.1. Key measured habitat variables associated with vegetation structure and composition and the structure, and the abundance of downed coarse woody debris (DCWD) and landscape features describing site locations

Component	Variable	Description	Units of measurement
Vegetation structure and composition	Density of trees and shrubs[1.]	Number of standing trees and shrubs for each of 4 size categories (0-10; 10-30; 30-50; 50+ cm) at breast height diameter (dbhob) per plot	Number of trees and shrubs ha-^{1}
	Density of dead trees	Number of standing dead trees per plot	Number of dead trees ha-^{1}
	Density of cut tree stumps[1.]	Number of stumps allocated to 2 size categories, small ($\leq$ 5cm) and large (> 5cm) per plot	Number of small stumps ha^{-1} Number of large stumps ha^{-1}
	Density of standing hollows	Number of hollow bearing trees per plot	Number of standing hollow ha^{-1}
	Density of hollows	Total number of hollows in trees, stumps and DCWD per plot	Total number of hollows ha^{-1}
	Density of standing fissures	Number of fissures in trees per plot	Number of standing fissures ha^{-1}
	Density of fissures	Total number of fissures per plot	Total number of fissures ha^{-1}
	Ground cover	Proportion of bare, litter, CWD, cryptogam along a 20 m transect within each plot	Proportion of bare, litter, CWD and cryptogam cover
	Vegetation cover[1.]	Proportion of ground (< 2 m), mid (2-6 m), sub-canopy (6-14 m) and canopy (>14 m) along a 20 m transect within each plot	Proportion of shrub, mid, sub-canopy and canopy cover
	Thinning intensity	Ratio of the basal area of stumps:total area (live trees and stumps)	Value (ratio) for each site

DCWD structure and abundance	Density of fissures in logs	Number of fissures in logs per plot	Number of DCWD with fissures ha^{-1}
	Density of logs	Number of logs per plot	Number of logs ha^{-1}
	Density of hollow bearing logs	Number of logs with hollows per plot	Number of DCWD with hollows ha^{-1}
	Size of log hollows	Length and diameter of log hollows per plot	Mean log length and diameter ha^{-1}
	Log decomposition state	Three categories (little, minor and extensive) allocated to each log within a plot	Category little, minor or extensive
	Log source	Classification of each log as either a trunk or branch	Category trunk or branch
	Large end log diameter	Large end diameter (cm) per plot	Mean log diameter ha^{-1}
	Small end log diameter	Small end diameter (cm) per plot	Mean log diameter ha^{-1}
	Log volume	Log length x π x (average log diameter/2)^2	DCWD volume m^3 ha^{-1}
Landscape variables	Elevation	Elevation (m) for site	Mean elevation per site (m)
	Distance to drainage line	Distance to nearest > 3rd order drainage line for site	Mean distance per site (m)
	Slope	Distance to nearest 20m contour for site	Mean distance per site (m)
	Rainfall	Mean annual rainfall at each site (0.01 degree grid) for site	Mean rainfall per site (mm)
	Grazing	Presence of high, medium and low grazing pressure based on observed dung within a site	Category low, medium and high per site
	Soil type	Australian soil classification	Category for each site

2.2.2. Sampling vegetation structure and composition

At each site, the diameter at breast height over bark (dbhob) (1.3 m, bark, where present was included in the measurement) of all live trees and shrubs within each plot was measured and allocated to one of four categories (≤10 cm; >10 cm to ≤30 cm; >30 to ≤50 cm; and >50 cm). The number of dead trees and shrubs was also recorded within each plot but not allocated to size classes. The number and diameter (top of each stump) of all stumps was also measured in each plot and categorised as either small (≤5 cm) or large (>5 cm); the height of each stump was not measured. All trees, shrubs and stumps were identified to genus. A count of the number of hollows and fissures in trees and stumps was also measured within each plot. The thinning intensity for each plot was calculated as the ratio of the cross-section area of cut stumps:total cross-section area of trees and stumps.

A point intercept method was used to quantify ground and vegetation cover using a 20 m transect running through the centre of each 10 × 20 m plot. A pole with a laser pointer mounted at a height of 1.5 m was used to record four ground cover categories: bare, cryptogam, litter and fine woody debris (FWD, all woody material with ≤10 cm diameter, of any length not rooted in the soil)) and five vegetation structure categories: shrub (<0.5 m); understorey (0.5–2 m); mid-storey (2.1–6 m); sub-canopy (6.1–14 m) and canopy (>14 m) at 1 m intervals along the 20 m transect. The height of shrub and understorey plants was measured using the laser pointer at the point where the laser beam intersected with a plant. Using a densitometer positioned at the top of the same pole used for ground cover/shrub cover/understorey plant height, the remaining vegetation categories were measured following the methods described in AusPlots (2012). All vascular plants along transects were identified to genus or the lowest taxonomic classification possible.

A visual estimate of grazing intensity based on the presence of fresh and old dung was recorded for each plot. The amount of dung was categorized as low (rare or infrequent sighting); medium (some present) and high (frequently sighted) for each of three herbivore types (macropod, mostly kangaroos; goats/sheep and horses). Identification of dung followed the procedure outlined in Landsberg et al. (1994).

2.2.3. Sampling CWD structure and abundance

CWD was defined as woody material ≥10 cm diameter at the mid-point and >1 m in length and not rooted in the soil. For each site, CWD structure and abundance was measured along a 20 m transect running through the centre of each plot used for sampling vegetation structure and composition. The number of logs, log length, large and small end diameter and categories for log source (trunk or branch) and state of log decay (Little = solid log, bark and twigs present; Minor = decomposition commencing, bark largely absent, exposed soft sapwood, twigs absent; Extensive = intermediate to late stage decomposition, bark absent, exposed soft sapwood, partial heartwood exposure, log breaking up or fragmented) was recorded for each plot. The presence and diameter of hollows (minimum entrance diameter 2 cm) and the presence of fissures were recorded for all CWD along each transect.

2.2.4. Biodiversity surveys

2.2.4.1. Floristics surveys

Within each of the plots used to measure vegetation structure and compositionn, five smaller quadrats (1 m^2) were sampled and each understorey vascular plant species (<2 m) within a quadrat was identified and counted. Plant species were identified to species where possible in the field. Since some plants were not flowering at time of collection, it was not possible to ascertain the species-level identification for all specimens. For these, a morphospecies approach was adopted and plants of the same genus were assigned to a unique species number (e.g., *Austrostipa* sp. 1).

2.2.4.2. Reptiles

Reptile diversity and abundance was assessed using pitfall trapping and nocturnal spotlighting. Pitfall trapping was undertaken at each site using two pitfall traps that were installed ~50 m and ~150 m along each site transect. At each site, pitfall traps were either a pipe (0.15 m diameter and depth of 0.5 m) or bucket (0.30 m diameter and depth of 0.3 m), and centred on three drift fences (3m length, damp course) in a ‘T-formation’. At the end of one randomly selected drift fence of each pitfall trap, a funnel trap was also installed to survey reptiles that may be able to escape from pitfall traps (e.g., snakes). Pitfall traps and funnel traps were opened for eight days in October and November 2015 and for four days in March 2016. All captured animals were identified to species and then released at the point of capture immediately. Funnel traps were not employed in March since capture rates in November were low (4 captures) and represented those reptiles that were commonly recorded in pitfall traps.

Nocturnal spotlighting was carried out once at each site, with a sampling effort that equated to 20 min by two observers along the 200 m transect. Observers surveyed within 10 m either side of the transect and all observed reptiles were identified to species and recorded. In addition to pitfall trapping and spotlighting, incidental observations of reptiles were also recorded. However, only data from pitfall trapping and spotlighting were combined and used in analyses.

2.2.4.3. Birds

Birds were recorded at each site using aural and visual surveys at two stationary points in each site between dawn and midday. The two stationary points were situated 50 m and 150 m along each site transect. At each stationary point, birds were surveyed over a 20 minute period. All species observed within 50 m of the stationary point were identified to species and their abundance recorded. Bird surveys were repeated at each site by a second observer in reverse order to ensure that sites that were surveyed later in the morning by observer 1 were surveyed earlier by observer 2. Each bird species was later assigned to a bird abundance class (0=absent, 1=1-2 individuals, 2=3-4 individuals, 3>4 individuals) for analysis.

2.2.4.4. Non-volant mammals

Camera traps were used to survey non-volant mammals at each site. A single wildlife camera (Reconyx model Hyperfire HC600) was deployed ~100 m along the transect at each site along with a lure to attract mammals from within the immediate area of the site. The lure used was a peanut butter, honey and oats bait plus sesame oil on a cotton wool pad which was placed inside a PVC tube and secured in the ground using a tent peg. Cameras were set at a height of 1 m on a tree, while the lure was placed at a distance of 1.5 m from the base of the tree. Cameras were programmed to have a high sensitivity, a RapidFire picture interval with 5 pictures per trigger, a 1 minute delay (quiet period) between triggers and a balanced night mode. The sampling period for all sites was 14 days. For each set of 5 consecutive images ('an event'), a species name was assigned if that species was tagged in any of the images that comprised an event. The total number of events assigned was collated to provide an 'activity index'. Mammal taxa were assigned to three categories: 'native mammal', 'introduced herbivore' and 'introduced predator'.

2.2.4.5. Bats

Bats were surveyed at each site for 2-3 nights using a single acoustic Anabat detector (Titley Scientific – Brendale QLD) which was deployed 100 m along the site transect. Detector microphones were set at a height of 1 m and oriented at a 45° angle to the ground to face gaps within vegetation. This ensured detectors at each site were sampling a similar volume of airspace and avoided confounding by differences in vegetation density among treatments. All files were analysed using automated software, AnaScheme (Adams et al. 2010), in association with a key for bats of the Pilliga (unpublished – B. Law). Bat calls with fewer than three valid pulses (i.e. minimum of six data points and model quality of 0.8) were not analysed by AnaScheme. Because multiple bat species may call simultaneously, calls were assigned to a species only if >50% of pulses within the sequence were attributed to that species and only passes with a minimum of three pulses classified to the same species were identified. All calls that could not be assigned to a bat taxon were included in counts of total bat activity but were labelled as 'unidentified'. Since it is not possible to distinguish reliably between calls of certain species (e.g., *Nyctophilus corbeni*, *N. geoffroyi* and *N. gouldi,* or *Mormopterus planiceps* and *M. petersi*, or *Scotorepens greyii* and *S.* sp.), these calls were assigned to a species group (e.g., *Nyctophilus* spp., or *Mormopterus planiceps/petersi*, or *Scotorepens greyii*/sp.) by AnaScheme. For each detector and each night, the number of bat passes for each species and total bat activity (all identified and unidentified bat calls) was tabulated.

2.2.4.6. Invertebrates

Volant invertebrates were surveyed at a sub-set of sites (15) representing each forest management treatment (i.e., three replicates of each treatment) using blacklight traps. Blacklight traps were deployed at each of these sites and sampled invertebrates overnight for two nights. To ensure that blacklight traps did not influence bat surveys, sampling of invertebrates and bats was not undertaken on the same night. All invertebrates sampled by blacklight traps were killed in 70 % ethanol. A random sub-sample of invertebrates trapped on each night and at each site were identified to the taxonomic level of Order and assigned to one of three three size classes: small (0-4 mm), medium (4.1-10 mm) and large (>10 mm). The abundance and dry mass of insect orders in each size class was recorded. The entire sample for each site was then dried and weighed to provide a measure of biomass of invertebrates at a given site. The relative composition of invertebrate orders and size classes represented in each sub-sample were subsequently applied to the entire sample at a given site to provide a measure of biomass for each invertebrate order and size class. The biomass of all insects was then rescaled as a score between 0 and 10 for each insect taxa and also each size class. Rescaled data were plotted to illustrate insect biomass distribution among insect taxa, size classes and also forest management treatments. Rescaling for each taxa or size class involved regression analyses to fit a linear relationship between insect biomass and the quartile midpoints of the biomass distribution among forest management treatments. Quartile midpoints corresponded to the 12.5, 37.5, 62.5 and 87.5 percentiles of insect biomass distribution and were rescaled from 0 to 10 to prevent extreme values from distorting the scaling of insect biomass.

2.2.5. Data analyses

Patterns in landscape and vegetation structural attributes (Table 2.2.1) were explored using principal components analyses (PCA). Two separate PCAs were used to examine patterns between (i) clusters and landscape characteristics and (ii) forest management treatments and habitat variables (vegetation structure and composition and CWD volume and composition). Variables with small or negligible weightings on all of the first three components were excluded and the PCA was repeated. The results were displayed as biplots and mean principal component scores were calculated for key vegetation structural attributes (Table

2.2.1). Generalised linear models (GLMs) were also used to compare each vegetation structural attribute with landscape variables. Generalised linear mixed models (GLMMs) were used to compare all vegetation structural attributes among forest management treatments, with cluster and treatment:cluster fitted as random terms. The number of trees/shrubs in each of four dbhob size classes (≤10 cm; >10 cm to ≤30 cm; >30 to ≤50 cm; and >50 cm) and the number of stumps, small (≤5cm) and large (>5cm) were dependent variables, with treatment as a fixed effect.

Each tree and shrub and all stumps were also allocated to a genus. For trees, these categories were *Acacia*, *Allocasuarina, Callitris*, *Eucalyptus* and other. For stumps, categories were *Allocasuarina, Callitris* and *Eucalyptus*. Then the number of trees or stumps belonging to each genus within each of the size classes was calculated. These data were then analysed using GLMMs with cluster, cluster:treatment and cluster:size fitted as random terms and the log count of trees, shrubs or stumps were dependent variables, with treatment as a fixed effect.

Cover measurements obtained using the point intercept method were assigned to four ground cover categories (bare, litter, FWD and cryptogam) and to five vegetation strata according to height. The five strata were ground (<0.5 m); understorey (0.5–1.5 m); mid-storey (≥1.5–6 m); sub-canopy (>6–14 m) and canopy (≥14 m). Cover was defined as the number of encounters recorded in a ground cover or vegetation stratum within a plot divided by the number of recordings made in each plot (20). GLMMs were used to compare the cover of each stratum among forest management treatments with cluster, cluster: treatment and plot included as random effects.

The counts of (i) logs with/without fissures or hollows and logs that were sourced from either (ii) trunks or (iii) branches were summed within plots whereas decomposition scores were calculated as a mean score per plot and all variables were compared among forest management treatments using GLMMs with cluster and treatment:cluster fitted as random terms.

Diversity indices were calculated for each site and taxa (floristics, bats, birds, native non-volant mammals and reptiles) as well as a composite biodiversity index (i.e., diversity calculated using data from all components of biodiversity). Geometric mean was used as the index of diversity as this reflects changes in the abundance of species even if their relative abundances remain constant, unlike other indices such as Shannon's and Simpson's (Buckland et al. 2011). Due to a low number of reptile captures, reptile richness and diversity was also compared among forest management treatments by extrapolating from the sampled data following Chao et al. (2014) and using iNEXT Online (Chao et al. 2016). This method is designed to control the dependence of empirical species diversity estimates on sampling effort and sample completeness (Chao and Chiu 2016).

For all taxa and composite biodiversity, GLMMs were used to evaluate the effect of forest treatments. In each GLMM, forest treatment was used as a main effect, while site cluster was initially included as a random effect in all models given the variation in annual rainfall across the sampling area (Fig. 1). Sampling night was also included as a random effect in models for nightly bat activity. Model selection procedures were used to determine whether to retain random effects in each of the models (Burnham and Anderson 1998). Akaike's Information Criterion scores corrected for small sample sizes (AICc) were used to identify the most supported models (<δ2AIC points) for each taxa and composite biodiversity. Response variables modelled were bat activity (($\log_{10} x$+1) transformed), bat diversity, reptile abundance ($\log_{10} x$+1), reptile diversity, bird diversity, plant diversity, mammal activity ($\log_{10} x$+1), mammal diversity and invertebrate biomass. Various distributions and link functions were fitted to data in all models and assessed using histograms and AICc.

For each taxa, differences in assemblages among forest management treatments were tested using the package, 'mvabund' (Wang et al. 2012). We used the negative binomial distribution for all response variables and checked assumptions of each model by plotting residuals and observing whether a linear or curvilinear relationship was present. Significance of the effect of forest management treatment on assemblages of each biodiversity component was tested using the likelihood ratio test (LRT) statistic. P-values were estimated using 999 resampling iterations and adjusted for multiple comparisons.

To explore associations between fauna species and vegetation structural attributes and productivity (net primary productivity; NPP), we used Canonical Correspondence Analysis (CCA) carried out separately for bird and non-bird taxa. All species detected at ≥3 sites were included in analyses. Biplots with points representing species and vectors representing habitat structural variables were visually inspected to identify associations. All habitat structural vectors were initially included in CCAs before correlated vectors and/or short vectors (indicating low variable importance) were removed iteratively. Both species activity/abundance and vegetation structural attributes were $\log_{10}(x+1)$ transformed prior to analysis Palmer (1993).

SPSS version 23 or the statistical package R Core Team (2016) and the ASReml package (Gilmour et al., 2004) implemented in R (Butler et al., 2009) was used to fit all GLMMs while multivariate analyses were carried out in R only (R Development Core Team 2016). PAleontological STatistics (PAST) version 3.15 was used to perform all CCAs.

2.3. RESULTS

2.3.1. Vegetation structural attributes

Individual weightings on the first four principal components (PC) accounted for 98% of the variation between clusters whereas 69% of the variation between treatments was accounted for by landscape and vegetation structural attributes, respectively. Site clusters varied in relation to five landscape variables (Drainage, Grazing, Elevation, Rainfall and Slope). Macropod (largely kangaroo) dung was found across almost all sites, but goat dung occurred most frequently within the Pilliga West cluster and horse dung in the Baradine and Wittenbra clusters. In addition, the highest number of small stems (≤10 cm dbhob) was found in southern clusters Baradine (4271 ± 346 stems ha^{-1}) and Wittenbra (2535 ± 346 stems ha^{-1}) and the lowest number of small stems in the Pilliga West cluster (1569.5 ± 346.42 stems ha^{-1}). For forest management treatments, the density of small stems (≤10 cm dbhob), the proportion of cypress trees, density of cypress stumps and dead trees were influential variables in separating the unthinned treatment from the recent thinning treatment. The old thinning treatment was similar to the recent thinning treatment, but with more *Acacia* cover and CWD hollows, while the long undisturbed treatment showed inverse conditions. The intermediate thinning treatment was characterised by less litter cover and more hollows and fissures than other forest management treatments (Fig. 2.3.1).

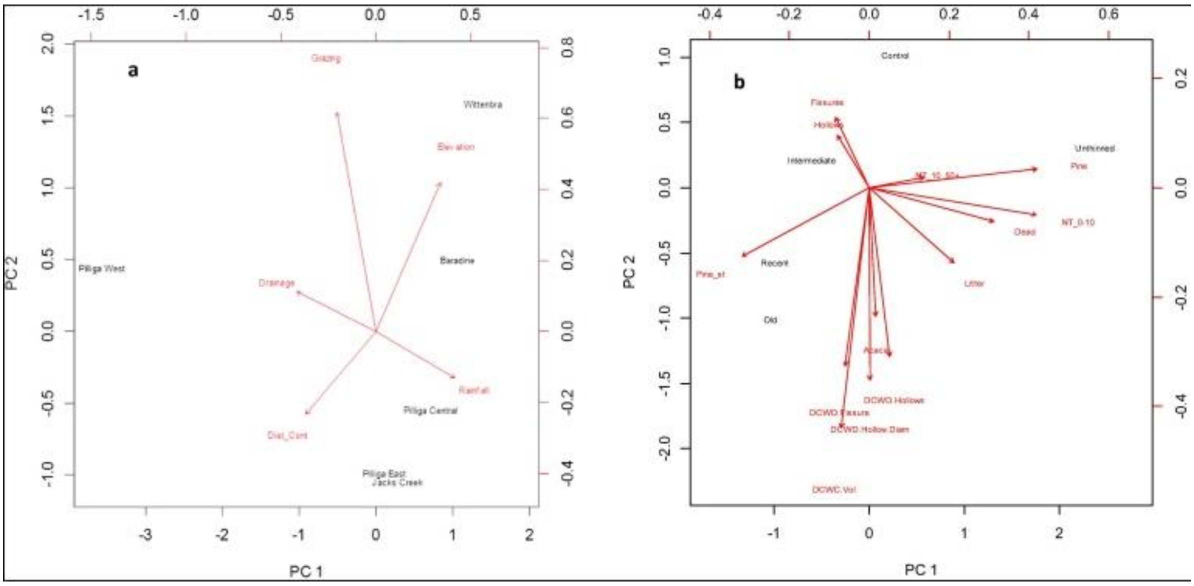


Fig. 2.3.1. Biplots of influential variables for the first two principal components (a) Six site clusters (Baradine; Jacks Creek; Pilliga Central; Pilliga East; Pilliga West; Wittenbra) and (b) habitat features for five different thinning treatments (Unthinned (>80 years); Recent = recent thinning (7–8 years); Intermediate = Intermediate thinning (9–20 years); Old = thinning (21–40 years); Control = Long undisturbed (>80 years). [a: Grazing = grazing intensity; Elevation = Distance to nearest >3rd order drainage line for site; Rainfall = Mean annual rainfall at each site; Dist_Cont = slope or distance to nearest 20 m contour for site; Drainage = Distance to nearest >3rd order drainage line for site; b: Hollows = Total number of hollows in trees, stumps and CWD per plot; Fissures = Total number of fissures per plot; NT_0_10/NT_10_50+ = Number of standing trees and shrubs in size categories 0–10 and 10–50+ cm dbhob per plot respectively; Pine = Number of cypress trees per plot; Dead = Number of standing dead trees per plot; litter = Proportion of litter along a 20 m transect per plot; Acacia = Number of Acacia spp. per plot; CWD.Hollows = Number of logs with hollows per plot; CWD.Hollow.diam = Diameter of log hollows per plot; CWD.Vol = Log volume per ha; CWD.Fissure = Number of fissures in logs per plot; Pine_st = Number of Cypress pp. stumps per plot].

2.3.2. Vegetation structure and composition

Significant differences in stem density of woody vegetation were found among forest management treatments ($P < 0.001$). Density of small trees/shrubs (≤10 cm dbhob) in the unthinned treatment was ~4 times greater (6030.5 ± 796 stems ha^{-1}) than all other forest management treatments ($P < .001$, Fig. 2.3.2a). There were no significant differences in stem density among treatments for all other dbhob size categories (>10 to ≤30, $P = .07$; >30 to ≤50 cm, $P = .79$; >50 cm, $P = .10$). There were also no significant differences ($P = .15$) in the number of large (>50 cm dbhob) trees among treatments. The number of dead trees and shrubs was significantly ($P = .003$) greater in the unthinned treatment (388 ± 79 trees ha^{-1}) compared to the lowest number in the old thinning treatment (111 ± 35 trees ha^{-1}) (Fig. 2.3.2b).

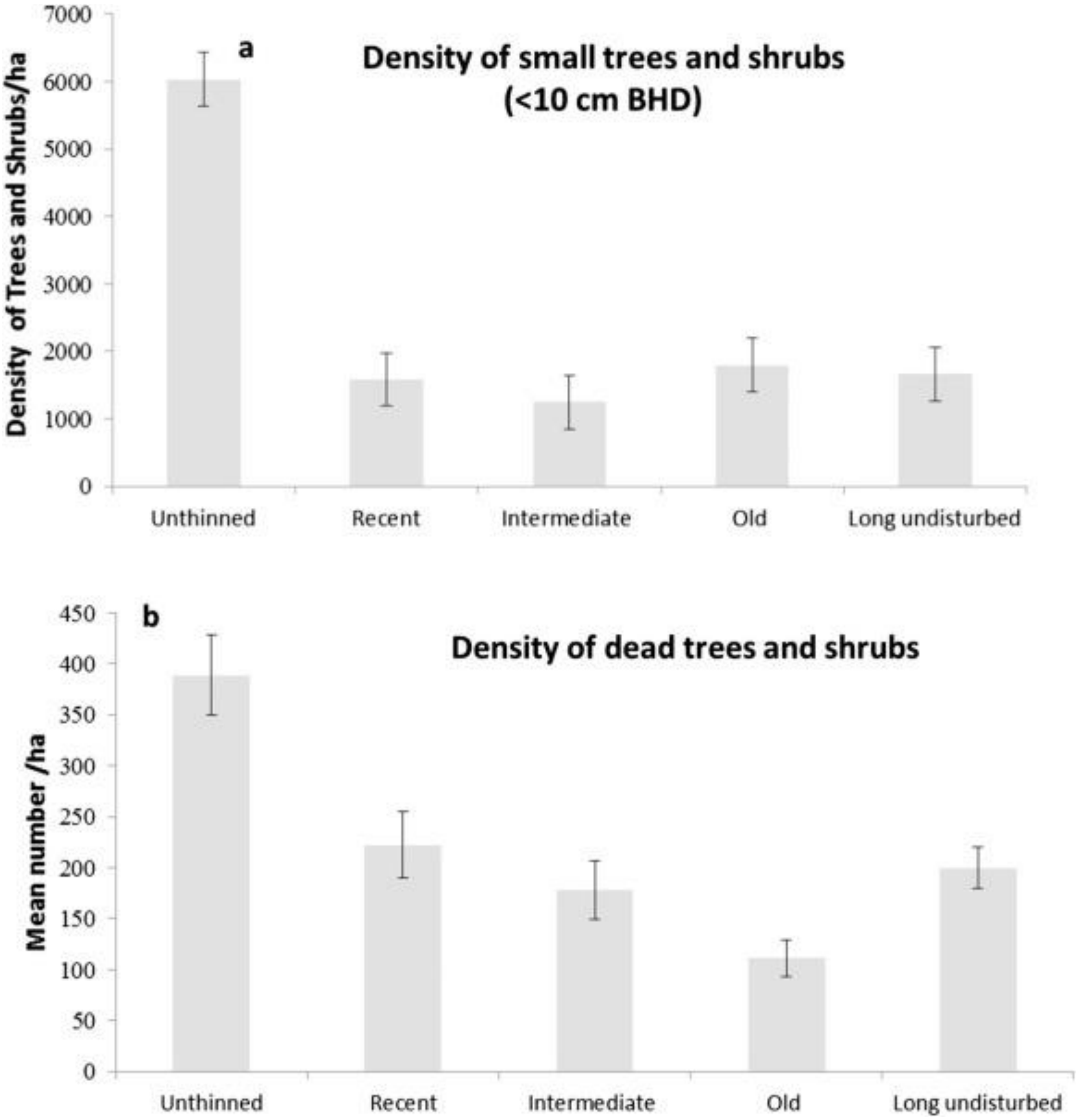


Fig. 2.3.2. The predicted mean (se) density of (a) small trees (≤10 cm dbhob) (b) the number of dead trees and shrubs for five forest management treatments.

While there were no significant differences among treatments for stem density of *Acacia* spp. (P = 0.92), Bulloak (*Allocasuarina luehmannii*) (P = 0.18) and *Eucalyptus* spp. (P = 0.47), there were differences among size classes for some of these taxa. For example, *Acacia* spp. almost exclusively fell into small stem size classes (≤10 cm dbhob) (Fig. 2.3.3a), and the recent thinning treatment had 2–3 times more *Acacia* spp. stems (456 ± 159 stems ha^{-1}) than other forest management treatments ($P < 0.001$; Fig. 2.3.3a); the stem density of small (<10 cm dbhob) Bulloak (*Allocasuarina luehmannii*) was significantly ($P < 0.05$) higher for the unthinned treatment (1185 ± 552), with 2–12 times as many stems as other forest management treatments (Fig. 2.3.3b). Significant differences among treatments (P<0 .01) and size categories (P<0.01) for cypress (*Callitris* spp.) stem density were also found, with ~6 times as many stems in the unthinned treatment compared to other treatments (Fig. 2.3.3c). The density of eucalypts <40 cm dbhob was greater in thinned treatments relative to unthinned (Fig. 2.3.3.d).

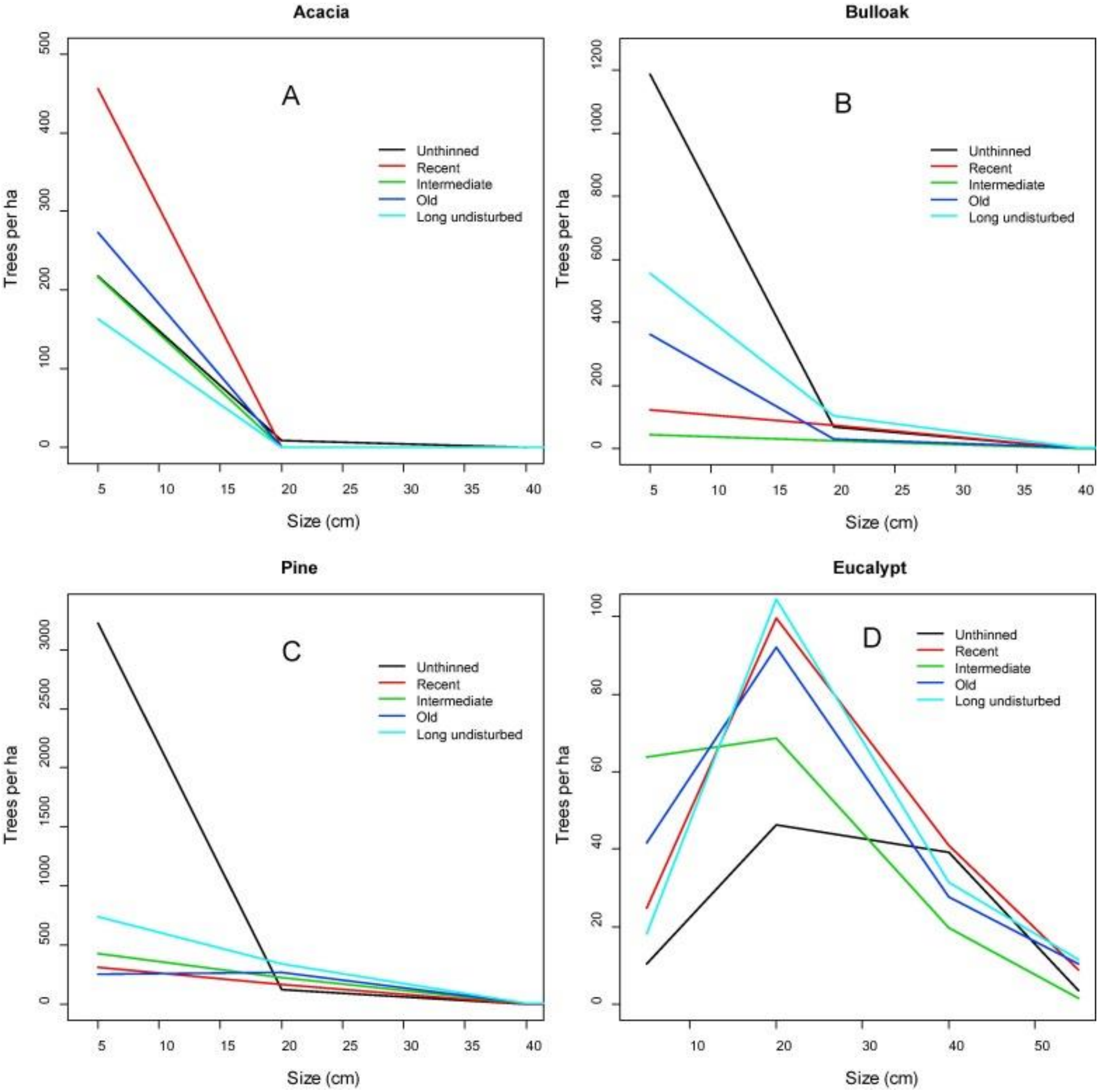


Fig. 2.3.3. Predicted mean number of (a) Acacia (*Acacia* spp.), (b) Bulloak (*Allocasuarina luehmannii*.), (c) Pine (*Callitris* spp.) and (d) Eucalypt (*Eucalyptus* spp.) in different diameter breast height (over bark), size categories for five different forest management treatments.

Thinning intensity differed significantly (P < .001) among treatments with thinning intensity in the intermediate thinning treatment (ratio of 10.4 ± 1.1) 10 times greater than the long undisturbed treatment and ~2 times greater than the recent and old thinning treatments. The number of small (≤5 cm) diameter stumps in recent and old thinning treatments was ~4 times greater than other treatments (Fig. 2.3.4a), while the density of large stumps in all thinning treatments was ~10 times that of the unthinned and long undisturbed treatments (Fig. 2.3.4b).

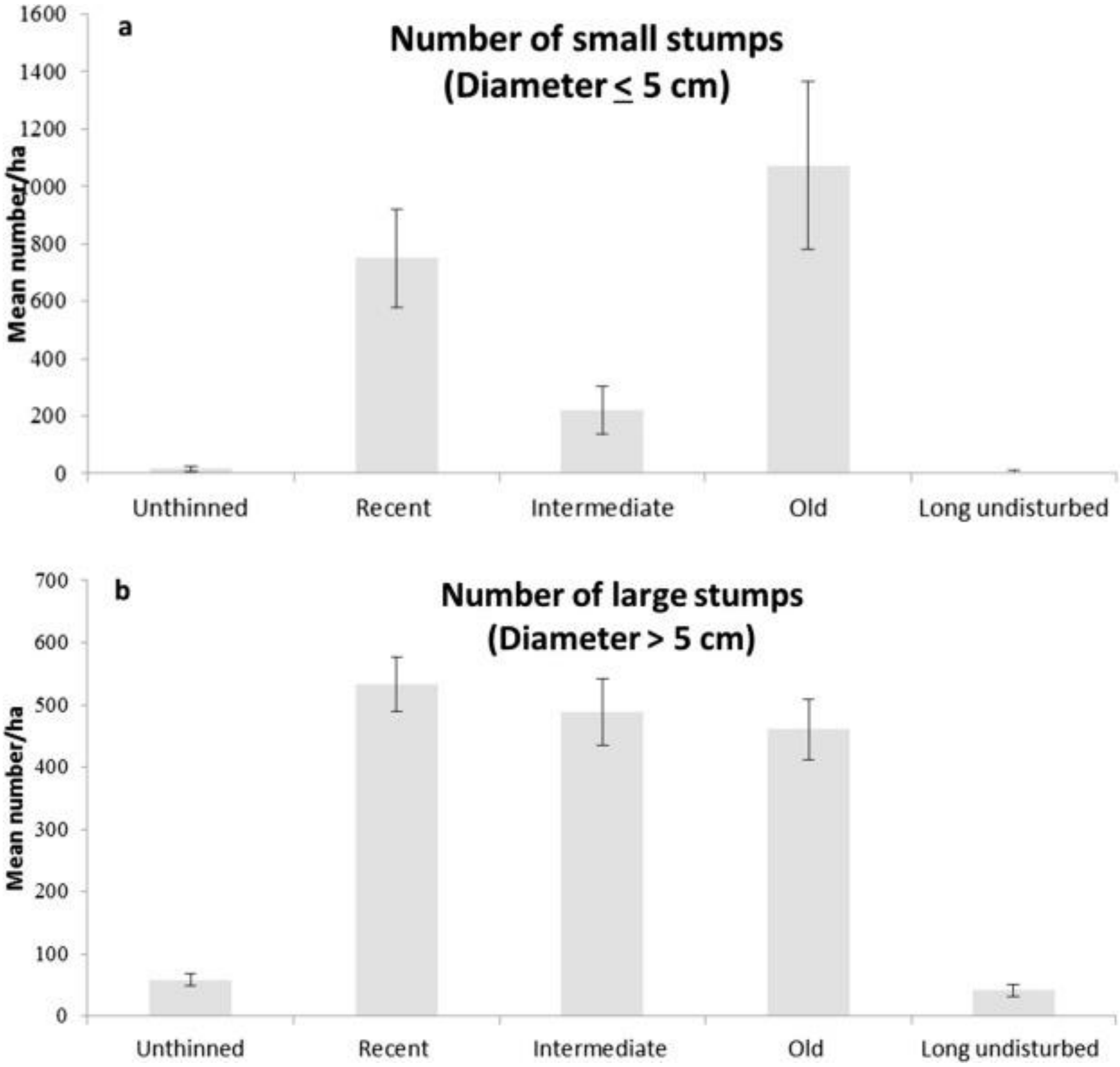


Fig. 2.3.4. Predicted mean (se) (a) number of small (≤5 cm diameter) and (b) large (>5 cm diameter) stumps for five forest management treatments.

Significant ($P<0.001$) differences among forest management treatments were found for sub-canopy cover, with the highest proportion of sub-canopy cover found in the unthinned and long undisturbed treatments (Fig. 2.3.5). Near significant ($P=0.06$) differences among treatments were found for mid-storey cover, with the recent thinning treatment having the lowest proportion of cover (0.04 ± 0.033) and the unthinned treatment the highest (0.17 ± 0.033). The proportion of shrub ($P=0.23$), understorey (0.5–2 m) ($P=0.97$) and canopy (>14 m) ($P=0.38$) cover did not differ among thinning treatments. For ground cover categories, bare ($P=0.40$), litter ($P=0.23$), FWD ($P=0.49$), cryptogam ($P=0.19$) did not differ among forest management treatments. There were also no significant differences among forest management treatments for the number of hollows ($P=0.32$) and fissures ($P=0.61$) in trees. The long undisturbed treatment had the highest (95 ± 16.8 ha^{-1}) whereas recent/unthinned treatments had the lowest (61 ± 16.8 ha^{-1}) density of standing hollows.

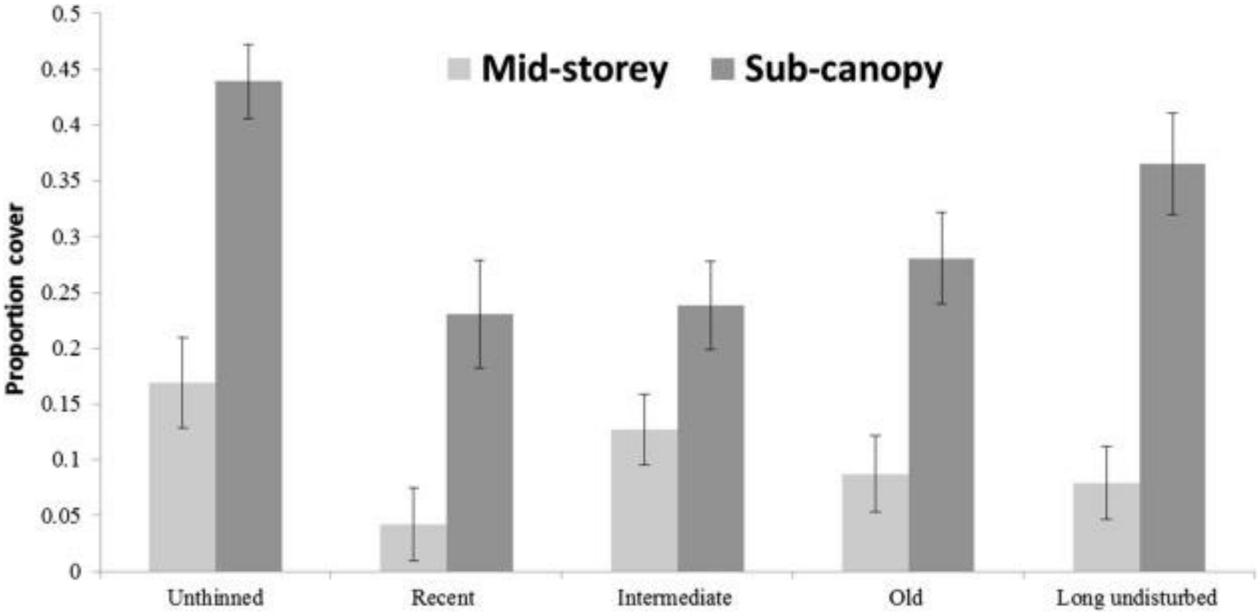


Fig. 2.3.5. Proportional cover of mid-storey and sub-canopy vegetation for five different thinning treatments.

2.3.3. CWD structure and abundance

Significantly ($P<0.05$) different volumes of CWD were found among forest management treatments (Fig. 2.3.6a). The long undisturbed treatment had the lowest volume (10.9 ± 19.62 m^3ha^{-1}) of CWD whereas the highest volumes were recorded at the old thinning treatment

(85.7 ± 19.62 m^3ha^{-1}). There were no significant differences among forest management treatments for the number of fissures ($P = 0.06$) and hollows ($P = 0.43$) in CWD. However, the old thinning treatment had the highest number of CWD with hollows (Fig. 2.3.6b). The old thinning treatment had significantly ($P < 0.05$) larger diameter hollows (6.2 ± 1.29 cm) than all other forest management treatments (Fig. 2.3.6c). The volume of CWD originating from tree trunks in the recent thinning treatment (32.4 ± 4.54 m^3ha^{-1}) was significantly ($P < 0.001$) higher than the unthinned treatment (9.3 ± 5.54 m^3ha^{-1}). There were no significant differences among forest management treatments for the amount of CWD originating from tree branches. The long undisturbed treatment had significantly higher ($P < 0.01$) decomposition scores than other forest management treatments.

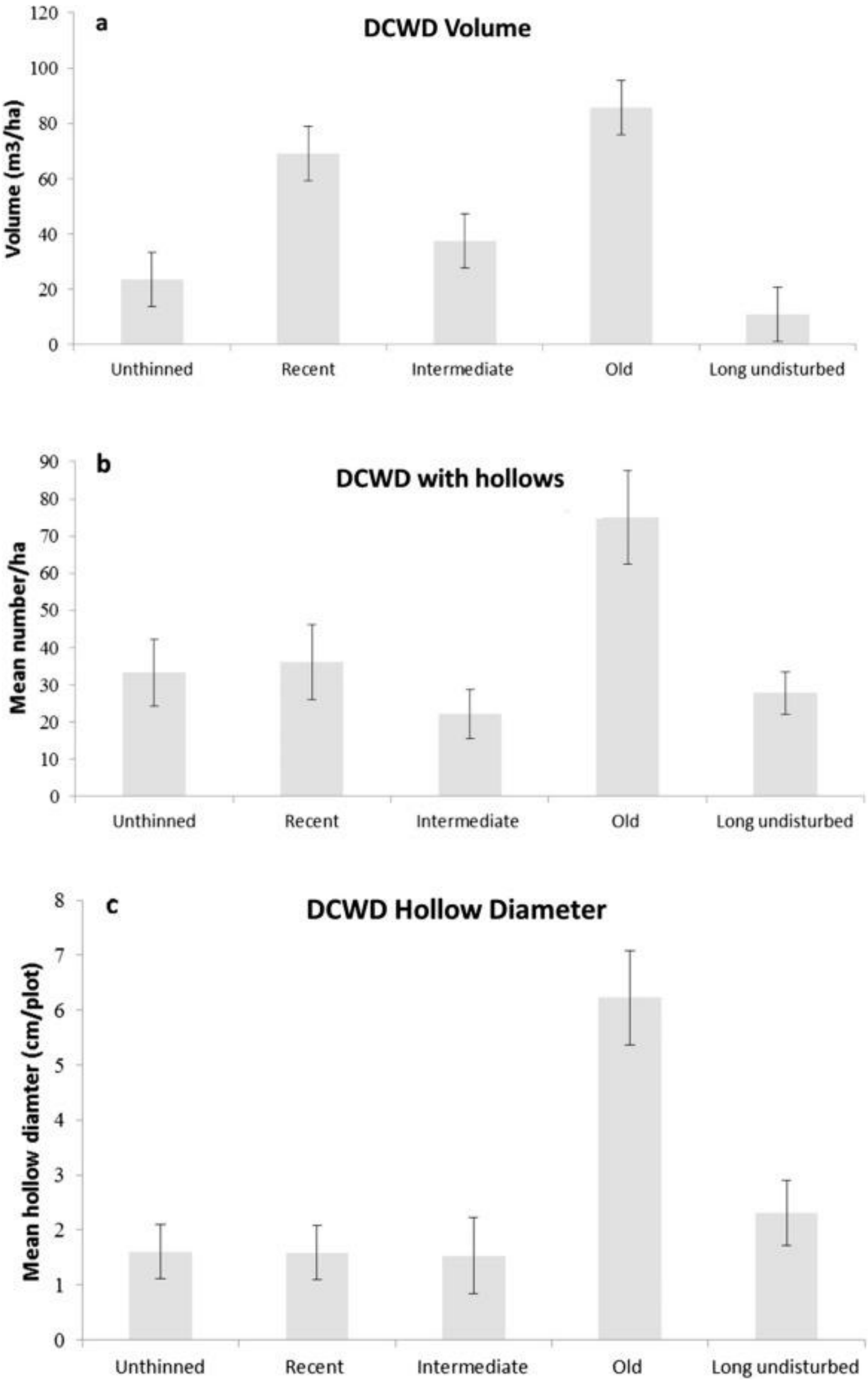


Fig. 2.3.6. Predicted mean (se) (a) Coarse woody debris (CWD) volume (b) number of hollow bearing logs and (c) CWD hollow diameter for five thinning treatments.

2.3.4. Understorey plants

In all, 7859 plants, representing 123 species were recorded across all sites (Table S2.3.1). Plant diversity at the species level ($F_{25,4}$=1.581, P=0.210; Table S2.3.2; Fig. 2.3.7a) and composition ($LRT_{25,4}$=257.6, P=0.253) did not differ significantly among forest management treatments.

2.3.5. Birds

In all, 1447 bird visits representing 69 species were observed across all sites, with rufous whistler *Pachycephala rufiventris*, yellow thornbill *Acanthiza nana* and weebill *Smicrornis brevirostris* the most commonly recorded species (Table S2.3.3). Bird diversity was significantly affected by forest management treatment. ($F_{25,4}$=2.994, P=0.038; Table S2.3.2; Fig. 2.3.7b). Bird diversity in the unthinned treatment was ~18 % lower than recent thinning (t_{25}=-2.773, P=0.010) and old thinning (t_{25}=-2.871, P=0.008) treatments, but did not differ from the intermediate thinning (t_{25}=-2.043, P=0.052) and long undisturbed (t_{25}=-1.306, P=0.203) treatments. All other treatments did not differ from one another (P>0.05). Bird assemblages did not differ among forest management treatments ($LRT_{25,4}$=251.4, P=0.085).

2.3.6 Bats

In all, 9,657 bat calls were recorded across all sites. Of these 7,034 were identified to 10 species and three species groups (Table S2.3.4). Nightly bat activity (no. passes night^{-1}) differed significantly among forest management treatments ($F_{64,4}$=3.969, P=0.006; Table S2.3.2; Fig. 2.3.7c), with the unthinned treatment 19-23 % lower than recent thinning (t_{64}=-3.410, P=0.001), old thinning (t_{64}=--3.302, P=0.002) and long undisturbed (t_{64}=--2.219, P=0.030) treatments, but not different from the intermediate thinning (t_{64}=-1.471, P=0.146) treatment. Nightly activity in the recent thinning treatment was ~16 % higher than the intermediate thinning (t_{64}=-2.066, P=0.043) treatment, while all other treatments did not differ from one another (P>0.05).

Nightly bat species diversity differed significantly among forest management treatments ($F_{64,4}$=2.952, P=0.027; Table S2.3.2; Fig. 2. 3.7c). Bat diversity in the unthinned treatment was ~29-50 % lower than recent thinning (t_{64}=-3.153, P=0.002) and intermediate thinning (t_{64}=-2.097, P=0.040) treatments, but did not differ from old thinning (t_{64}=-1.186, P=0.240) and long undisturbed (t_{64}=-1.478, P=0.144) treatments. Bat diversity was similar among thinning treatments and long undisturbed sites (P>0.05). Bat species composition did not differ among forest management treatments ($LRT_{25,4}$=69.45, P=0.364).

2.3.7. Insects

Total nightly insect biomass was not affected by forest management treatment ($F_{10,4}$=0.697, P=0.611, Table S2.3.2, unthinned: 6.6±2.0 g; recent thinning: 7.4±0.3 g; intermediate thinning: 7.6±0.5 g; old thinning: 6.8±0.5 g; long undisturbed reference: 5.5±0.2 g). The distribution of insect biomass varied among taxa and forest management treatments (Fig. 2.3.8a). Among forest management treatments, Lepidopteran (moth) biomass was greatest in unthinned and intermediate thinning treatments followed by old thinning, recent thinning and the long undisturbed reference treatment. Coleopteran (beetle) biomass was greatest in the recent thinning treatment and lowest in the unthinned treatment, while old thinning, long undisturbed reference and intermediate thinning treatments had moderate levels of beetle biomass. The biomass of dipterans (flies) was greatest in the unthinned treatment and lowest in the recent and old thinning treatments, with intermediate thinning and long undisturbed reference treatments having moderate levels of biomass. The biomass of other insects (e.g., leaf hoppers) was greatest in the unthinned treatment, with moderate to low levels in all other treatments.

The distribution of insect biomass varied among size classes and forest management treatments (Fig. 2.3.8b). Among forest management treatments, the biomass of small (<4 mm) insects was greatest in the unthinned treatment and lowest in the recent thinning treatment. The biomass of medium (4-10 mm) sized insects was also greatest in the unthinned treatment and lowest in the recent thinning and long undisturbed reference treatments. The biomass of larger (>10 mm) insects was greatest in the recent thinning treatment and lowest in the unthinned treatment.

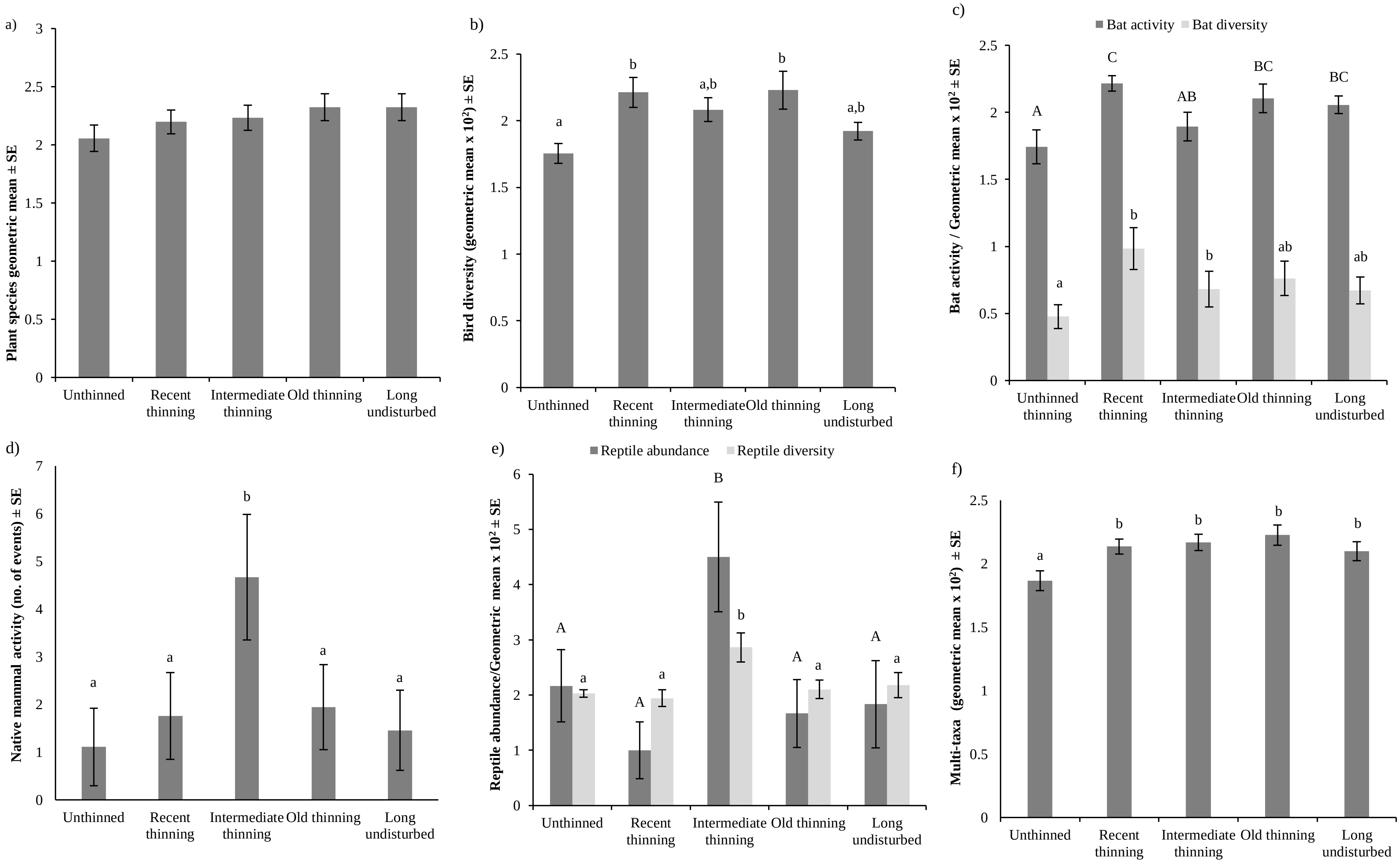
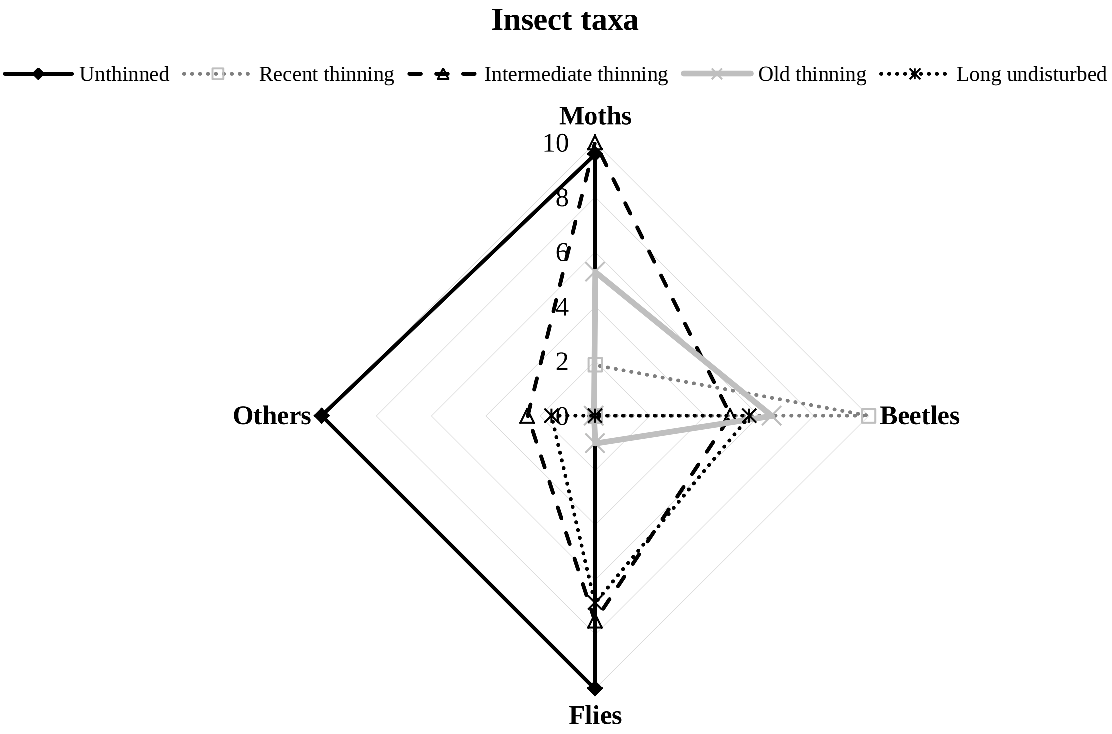


Fig. 2.3.7. Column graph illustrating: a) plant species diversity, b) bird diversity, c) nightly bat activity (log$_{10}$ *x*+1-transformed) and diversity, d) native mammal activity (no. of events per 2-week sampling period), e) reptile abundance and diversity and f) composite biodiversity recorded in all forest

a)



b)

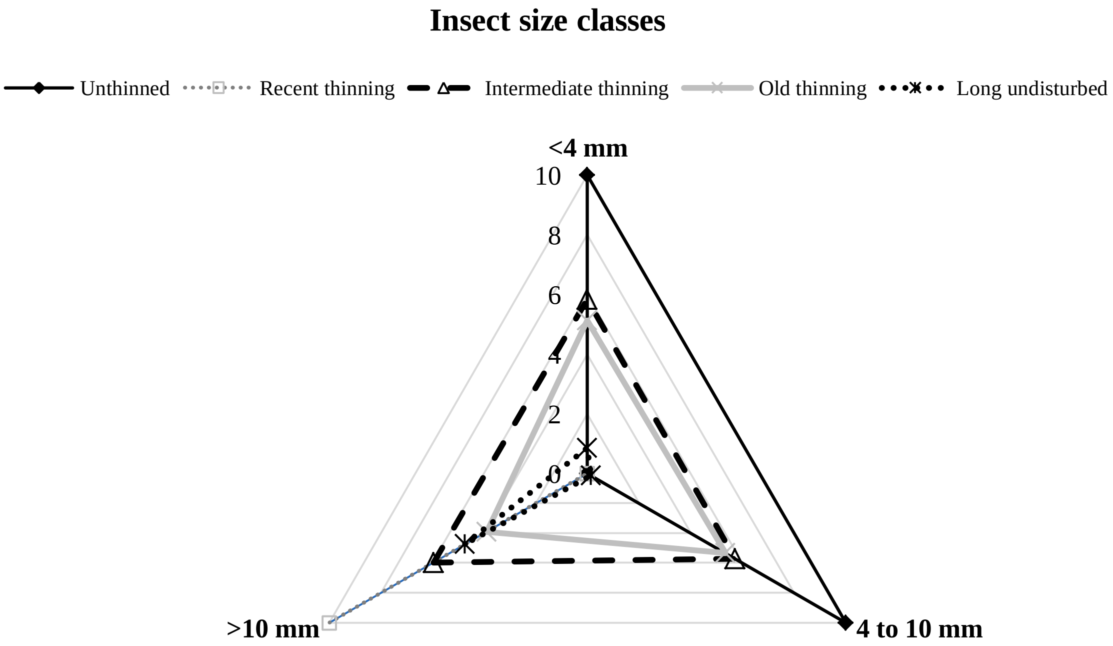


Fig. 2.3.8. Spider plots illustrating distribution of biomass of: a) insect taxa and b) insect size classes by forest management treatment. Each axis represents scaled values for insect biomass in each respective taxa or size class.

2.3.8. Non-volant mammals

In all, 2,533 images were captured across 29 sites (theft of camera at BR resulted in loss of data for this site), representing 94 mammal events and eight mammal taxa (Table S2.3.5). Eastern grey kangaroo *Macropus giganteus* was the most common native mammal recorded with 46 events per two-weeks of monitoring. Native mammal diversity ($F_{24,4}$=1.340, P=0.284, Table S2.3.2) and species composition ($LRT_{24,4}$=28.26, P=0.171) were not affected by forest management treatment. However, native mammal activity differed significantly among forest management treatments, with activity in the intermediate thinning treatment ~3-4 times greater than all other forest management treatments ($F_{24,4}$=3.211, P=0.030, Table S2.3.2, Fig. 2.3.7d). Introduced herbivore ($F_{24,4}$=1.334, P=0.286, Table S2.3.2) and predator activity ($F_{24,4}$=0.718, P=0.588, Table S2.3.5) did not differ among forest management treatments, respectively. The red fox *Vulpes vulpes* was the most common introduced predator, with six events recorded per two-weeks of monitoring, while goats *Capra hircus* were the most common introduced herbivore (11 events per two-weeks of monitoring).

2.3.9. Reptiles

In all, 85 reptiles were recorded across all sites, representing 21 species (Table S2.3.6). Of these, 68 were either captured in pitfall traps or observed during nocturnal spotlighting surveys, with eastern stone gecko *Diplodactylus vittatus* the most commonly recorded species. Reptile abundance differed significantly among forest management treatments ($F_{25,4}$=4.147, P=0.010; Table S2.3.2; Fig. 2.3.7e), being ~2-4.5 times greater in the intermediate thinning treatment than unthinned (t_{25}=-2.518, P=0.019), recent thinning (t_{25}=3.994, P=0.001), old thinning (t_{25}=-2.998, P=0.006) and long undisturbed (t_{25}=-2.572, P=0.016) treatments. Reptile abundance was similar among all other treatments (P>0.05).

Reptile diversity differed significantly among forest management treatments ($F_{25,4}$=4.137, P=0.010; Table S2.3.2; Fig. 2.3.7e), being ~1.4-1.5 times greater in the intermediate thinning treatment than unthinned (t_{25}=3.392, P=0.002), recent thinning (t_{25}=3.337, P=0.003), old thinning (t_{25}=2.688, P=0.013) and long undisturbed (t_{25}=2.161, P=0.040) treatments. Reptile diversity was similar among all other treatments (P>0.05). Extrapolation of sample data for reptiles indicated that reptile diversity (Shannon-Weiner) in intermediate thinning, long undisturbed reference and old thinning was greater than the unthinned treatment which did not differ from recent thinning (Fig. S2.3.1).

Reptile species composition also differed among forest management treatments ($LRT_{25,4}$=57.69, P=0.022). There was a trend for greater abundance of brown-blazed wedgesnout ctenotus *Ctenotus allotropis* in the intermediate thinning and long undisturbed treatments compared to the unthinned treatment, while the abundance of eastern stone gecko *Diplodactylus vittatus* was lower in recent, long undisturbed, and to a lesser extent intermediate thinning treatments, when compared with the unthinned treatment (Fig. 2.10). The abundance of tree dtella *Gehyra variegata* was lower in the recent thinning treatment, but marginally higher in the intermediate thinning treatment when compared with the unthinned treatment, whereas Bynoe's gecko *Heteronotia binoei* abundance was greater in intermediate and old thinning treatments when compared with the unthinned treatment (Fig. 2.3.9). For all other reptile taxa, abundance did not vary among forest management treatments.

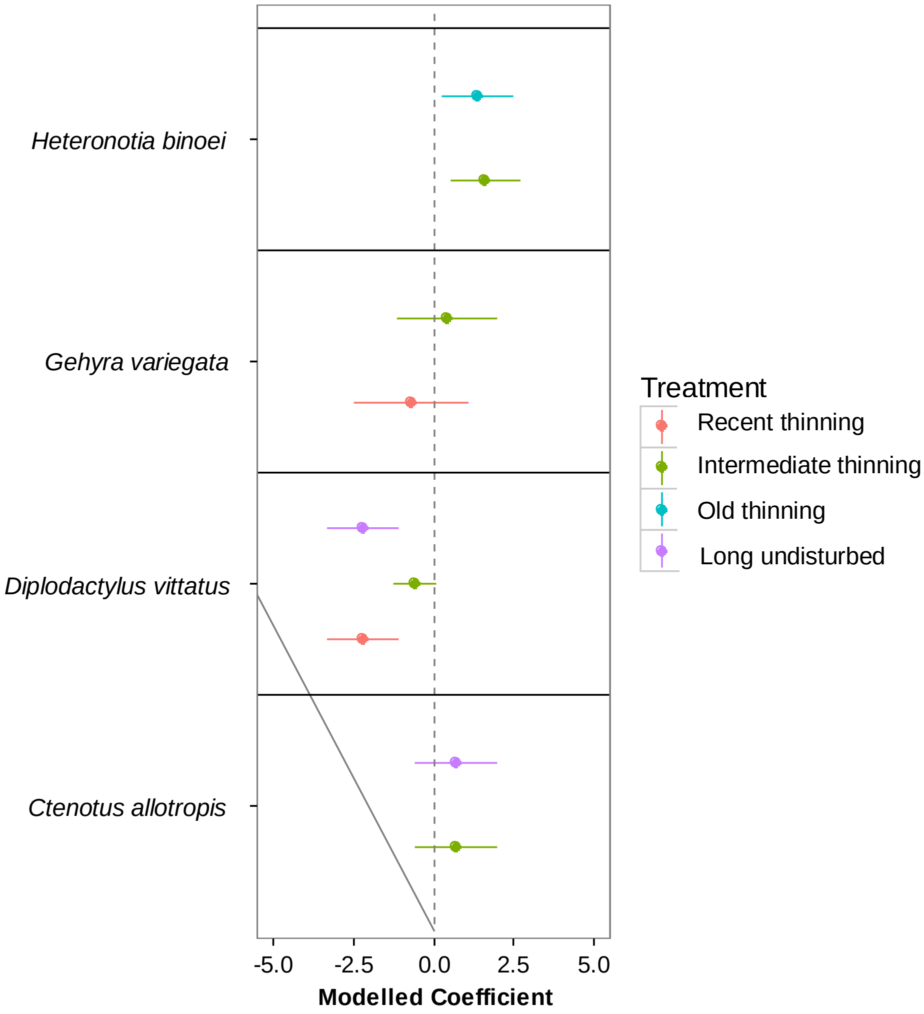


Fig. 2.3.9. Mean coefficient estimates (±SE) for generalized linear models relating positive or negative responses of reptile abundance to thinning.

2.3.10 Composite biodiversity (multi-taxa) index

The composite index was significantly affected by forest management treatment ($F_{25,4}$=3.481, P=0.022; Table S2.3.2; Fig. 2.3.7f), with the unthinned treatment ~16-22 % lower than recent thinning (t_{25}=-3.023, P=0.006), intermediate thinning (t_{25}=-3.226, P=0.003), old thinning (t_{25}=-3.465, P=0.002) and long undisturbed (t_{25}=-2.326, P=0.028) treatments. All other treatments did not differ from one another (P>0.05).

2.3.11 Relationships between fauna and habitat structure

A CCA revealed that some bird taxa were more strongly associated with particular habitat structural elements than others (Fig. 2.3.10). For example, noisy miner *Manorina melanocephala*, galah *Eolophus roseicapilla*, white-plumed honeyeater *Lichenostomus penicillatus*, spiny-cheeked honeyeater *Acanthagenys rufogularis*, common bronzewing *Phaps chalcoptera*, Horsefield's cuckoo *Chrysococcyx basalis* and sulphur-crested cockatoo *Cacatua galerita* were associated negatively with cypress stem density and positively with stump density and CWD, especially in recently thinned sites. Conversely, spotted pardalote *Pardalotus punctatus*, western gerygone *Gerygone fusca*, striated thornbill *Acanthiza lineata* and superb fairy-wren *Malurus cyaneus* were associated with a high density of *C. glaucophylla* stems. Buff-rumped thornbill *Acanthiza reguloides*, sacred kingfisher *Todiramphus sanctus*, Australian magpie *Cracticus tibicen*, red-capped robin *Petroica goodenovii*, and speckled warbler *Pyrrholaemus sagittatus* were positively associated with the density of eucalypts at intermediate thinning sites, but negatively associated with the density of *A. luehmannii* and vegetation cover <2 m. Most other bird taxa were distributed towards the middle of the CCA, indicating no strong associations with measured habitat structural elements. Among species of conservation concern, grey-crowned babbler *Pomatostomus temporalis* and varied sittella *Daphoenositta chrysoptera* were associated with large trees (negative with eucalypt density) often in old thinned sites, while brown treecreeper *Climacteris picumnus* was recorded on just two sites (both thinned) and so were not included in the analysis.

Most bat taxa showed a weak positive association with the density of stumps (Fig. 2.3.11). The exception to this trend was white-striped freetail bat *Austronomus australis*, yellow-bellied sheathtail bat *Saccolaimus flaviventris* and chocolate wattled bat *Chalinolobus morio*, with these species having a weak positive association with *C. glaucophylla* and *A. luehmannii* density. Among reptiles, *H. binoei*, litter skink *Lygiasaurus foliorum* and Boulenger's skink *Morethia boulengeri* were negatively associated with the density of *C. glaucophylla* stems, but positively associated with the density of eucalypts, plant cover and NPP (Fig. 2.3.11). Eastern spiny-tailed gecko *Strophurus williamsi*, nobbi dragon *Diporiphora nobbi* and *C. allotropis* were associated with a greater density of large trees (>60 cm dbhob). *Gehyra variegata* showed a weak, positive association with the density of *C. glaucophylla*, while *D. vittatus* was negatively associated with the density of eucalypts and positively associated with the density of *C. glaucophylla*. Among non-volant mammal taxa, goats *Capra hircus* were positively associated with Acacia cover, while European rabbit *Oryctolagus cuniculus* and eastern grey kangaroo *Macropus giganteus* were negatively associated with the density of *C. glaucophylla* and positively associated with the density of eucalypts (Fig. 2.12). The red fox *Vulpes vulpes* and short-beaked echidna *Tachyglossus aculeatus* were positively associated with density of *C. glaucophylla* and *A. luehmannii*, respectively.

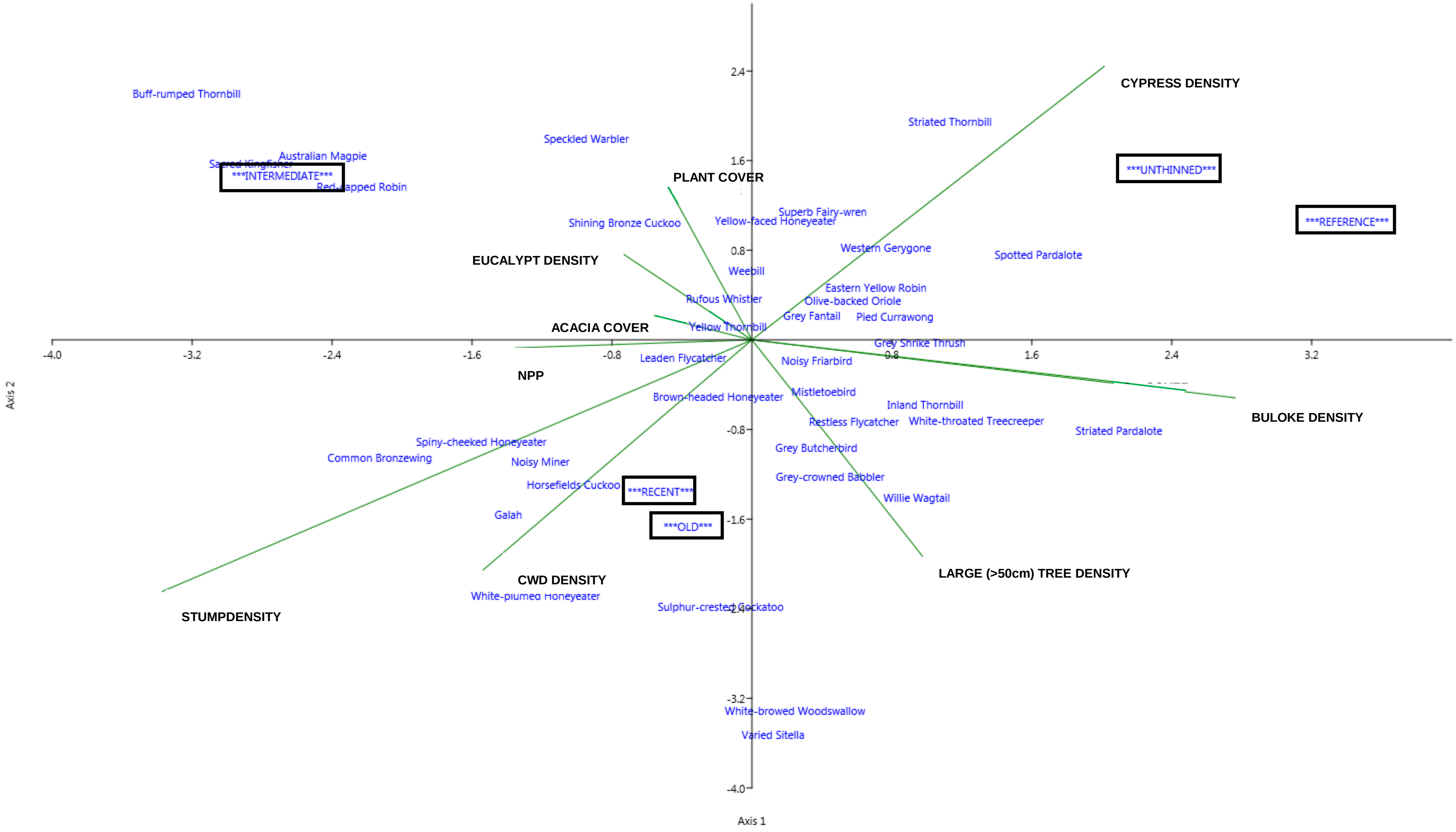


Fig. 2.3.10. Canonical correspondence analysis relating bird abundance to vegetation structure data (Waters et al. 2018). Centroids for forest management treatments are also plotted.

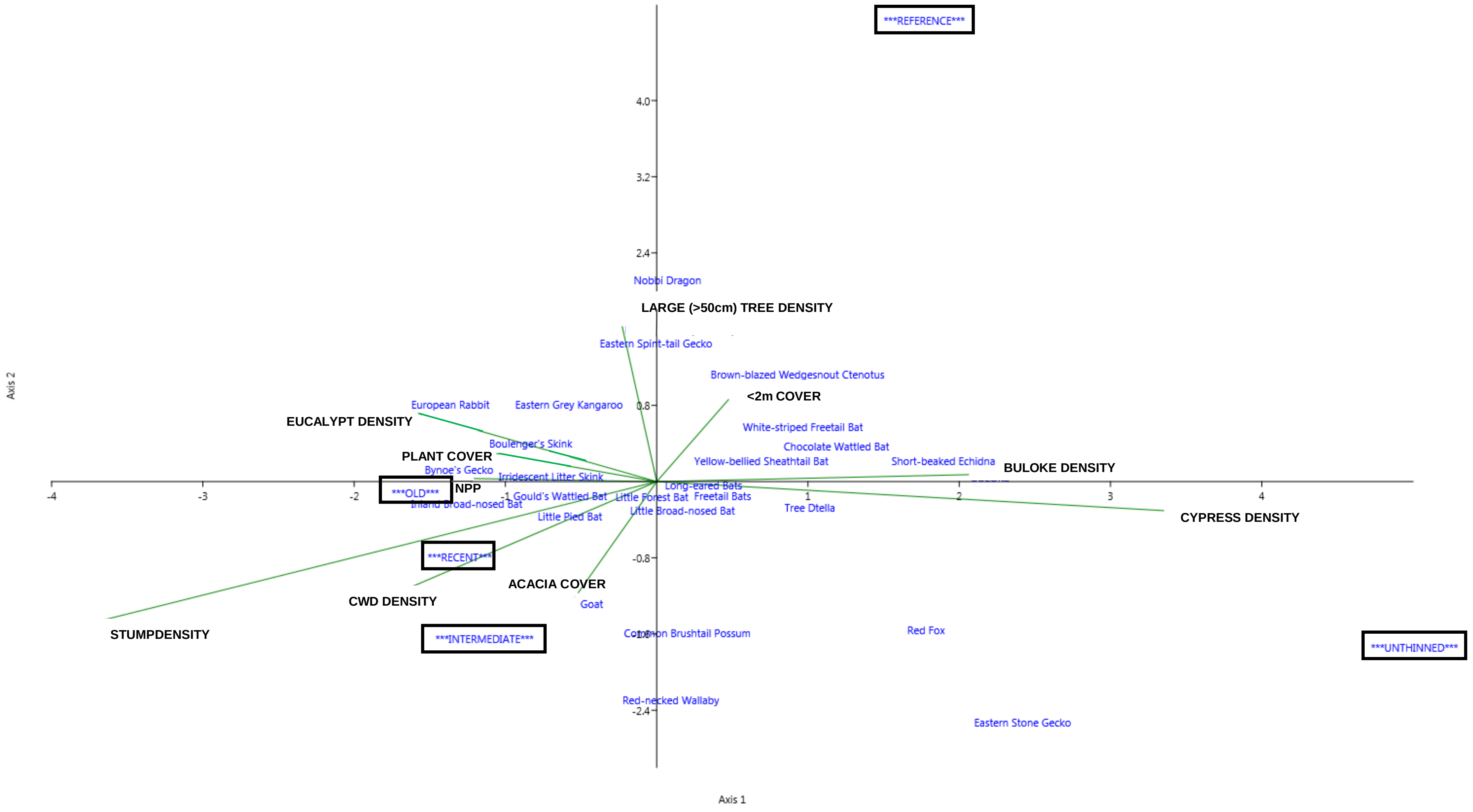


Fig. 2.3.11. Canonical correspondence analysis relating bat (activity), reptile (abundance) and mammal (activity) to vegetation structure data (Waters et al. 2018). Centroids for forest management treatments are also plotted.

2.4. DISCUSSION

In the low rainfall environment of this study, the structure and composition of woody vegetation as well as the volume and size of hollows in CWD were strongly influenced by time since thinning. Modelling of growth, survival, recruitment and competition processes in *C. glaucophylla* woodlands indicated responses to thinning may be due to differences in site productivity (drainage, elevation and rainfall) (Ross et al., 2008). While we found variations in drainage, elevation and rainfall across our extensive study area in the Pilliga forest, our sampling design accounted for these variations. Thinning immediately reduced the density of small live (<10 cm dbhob) and dead trees and shrubs which was maintained over the long-term (21–40 years) and resulted in live stem densities comparable to the long (>80 years) undisturbed treatment. The effect on structural complexity was mixed and occurred at different time frames with the cover of mid-storey vegetation (2–6 m) reduced immediately after thinning, but recovered in the long-term (21–40 years) and was comparable to the long undisturbed treatment. Thinning also increased the volume of CWD which was maintained for, again, at least 21–40 years post-thinning, at which time CWD produced larger hollows. For many other structural attributes (ground cover, large tree density, hollow density), the effect of thinning was neutral.

Overall responses by biodiversity to thinning of dense *C. glaucophylla* regrowth at both short and longer-term intervals were positive or neutral. However, the different taxa appeared to be poor surrogates for each other, emphasizing the importance of developing management actions by considering multiple taxa (Clarke 2008; di Stefano et al. 2013). At the taxa level, responses were generally positive or neutral. Positive responses were recorded for diversity of bats (all thinned treatments), reptiles (intermediate) and birds (recent and old thinning). Neutral responses were recorded for the diversity of non-volant mammals and plants. Thinning was associated with a relatively more even distribution of insect biomass among size classes >8 years post-thinning and a greater representation of beetles to overall insect biomass compared to a dominance of smaller moths in unthinned stands.

2.4.1. Vegetation responses to thinning

We identified distinct differences in vegetation structure among treatments. Unthinned sites were consistently dominated by a high density of small (≤10 cm dbhob) stems and contained higher cover in the mid-storey (2–6 m) and sub-canopy (6–14 m) than all other forest management treatments. While reduction in these layers after thinning could be interpreted as simplifying structural complexity, the mid-storey consisted primarily of *Callitris* spp., *A. luehmannii* and *Acacia* spp., but the dominance of each varied temporarily following thinning. For example, immediately post-thinning (recent thinning; <8 years) the native shrub *Acacia* spp. dominated the mid-storey, which is consistent with *Acacia* described as an early successional species (Weigand et al., 2000; Ross et al., 2008). A similar result was found for shrub cover in subtropical cypress forests, with thinned sites associated with greater shrub cover relative to unthinned sites (Eyre et al., 2015). While densities of regrowth have been reported to exceed 125,000 ha^{-1} (Lacey, 1973), the high density of small stems (~6000 ha^{-1}) we found in unthinned *Callitris* spp. dominated forests has not previously been reported largely because most other studies have not accounted for small stem diameter trees and shrubs (<10 cm dbhob) (Whipp et al., 2012; Hunter, 2013; Eyre et al., 2015). In our study, thinning reduced small (<10 cm dbhob) stem densities to levels similar to the long undisturbed treatment ~ 1500 stems ha^{-1}. However, the range of densities of medium sized trees (>10 to <30 cm dbhob) among all forest management treatments was 372–516 stems ha^{-1} and is comparable to pre-European estimates of 200–400 stems ha^{-1} for this tree size category (Whipp et al., 2009; Gibbons et al., 2010). *Callitris* spp. tend to form single-aged dense stands (Horne, 1990a, 1990b; Thompson and Eldridge, 2005a), and there is little evidence of self-thinning (Thompson and Eldridge, 2005a). Instead *Callitris* spp. tolerate intense intra-specific competition at high densities which prevents growth for long periods, perhaps up to 200 years in low rainfall environments (Read, 1995).

The higher densities of *Allocasuarina luehmannii*, as well as *Callitris* spp., in unthinned regrowth suggests that the two species may behave similarly in competitively excluding *Eucalyptus* spp. In this study, we found the highest density of medium sized *Eucalyptus* spp. (20 cm dbhob; ~60–100 trees ha^{-1}) when some thinning activity had occurred (recent, intermediate and old thinning treatments) or where densities of *Allocasuarina* and *Callitris* regrowth were reduced (long undisturbed treatment). However, while the high density of the medium size class of *Eucalyptus* spp. in the recent thinning treatment suggests the occurrence of this species may reflect historic, past thinning, this is a question for future research. However, we observed no effect of thinning on the density of large trees (>50 cm dbhob). Thinning did not produce more hollow-bearing trees, which contrasts with what has been found in river red gum *Eucalyptus camaldulensis* forests (Horner et al., 2010). Of the thinning treatments, intermediate thinning had the lowest density of large (20–30 cm, dbhob) *Eucalyptus* spp. and this may be associated with the type of thinning activity.

In our study, the intermediate thinning treatment represented commercial thinning, removing merchantable material (e.g. saw logs) and thus contributing less residue *in*-situ than non-commercial thinning activities. Although the intermediate thinning treatment supported ~twice the volume of CWD as the long undisturbed treatment, the removal of logs from the intermediate treatment sites has been shown to reduce the number of favourable microsites for *Eucalyptus* seedlings compared to recent and old thinning treatments in which non-commercial thinning retained logs on site (Bailey et al., 2012). While we did not directly measure the number of *Eucalyptus* spp. seedlings, lower number of small (<10 cm dbhob) stem *Eucalyptus* spp. were found in unthinned and long undisturbed treatments, which were also associated with lower levels of CWD. Retaining logs during thinning operations can play a role in allowing greater recruitment of *Eucalyptus* spp. and perhaps a subsequent increase in hollow bearing trees or CWD hollow formation. Dense stands of *Allocasuarina luehmannii* that include senescent individuals provide hollows used by insectivorous bats (Law et al., 2016b). The dominance of small size categories of both dead *Allocasuarina* and *Callitris* (e.g. mean = 8.5 cm dbhob), suggests little potential for hollow formation in these stems and thinning would not result in a long-term net loss of hollows; noting that we also found no differences in the number of hollows and fissures in trees among forest management treatments. One reason for a lack of effect of thinning on hollow formation may be that the chronosequence did not span a sufficient time frame for hollow formation in termite resistant cypress or that thinning activities are retaining a sufficient number of larger trees with hollows, including dead trees.

The dominance of small stemmed *Acacia* spp. immediately post-thinning (recent thinning) was evident (>400 shrubs ha^{-1}), but over time the density of this genus almost halved. This suggests that while some disturbance through thinning is necessary to counteract competitive dominance of *Allocasuarina luehmannii*, and *Callitris* regrowth, any dominance from *Acacia* regrowth after thinning will be relatively short-lived. Thinning typically reduces canopy cover (Harrod et al., 2009), and in our study, sub-canopy cover took between 21–40 years post-thinning to reach levels that were comparable to the long undisturbed treatment, indicating the effect of thinning on vegetation structure persists for a moderate period of time. This slow time to increase sub-canopy cover contrasts with the modelling by Ross et al. (2008), where thinning would be followed by rapid and dense recruitment in cypress forests, if staged thinning treatments are not ongoing. The longer time frame indicated from our study may reflect a grazing influence, as grazing was widespread across the study location and could reduce seedling recruitment.

2.4.2. Coarse woody debris responses to thinning

In our study, an immediate 'pulse' effect of thinning on CWD volume was evident when thinnings remained *in-situ*, evidenced by the highest volumes of CWD recorded in the recent and old thinning treatments. Our CWD volume of 11 $m^3 ha^{-1}$ in the long undisturbed treatment is very similar to the pre-European benchmark estimated for *Callitris* forest (9 $m^3 ha^{-1}$; Gibbons et al., 2008), which was ~4–8 times lower than values recorded in thinning treatments, including our intermediate thinning treatment where commercial thinning removed logs from the site (see also Eyre et al., 2015). Such benchmarks have been questioned because they do not account for the effects of pre-European fire management by aboriginal people, which are thought to have reduced CWD levels (Jurskis, 2011).

2.4.3 Influence of thinning on biodiversity

Bird diversity was associated positively with thinning (in recent thinning and old thinning treatments), which is consistent with studies in other forest types (Luck and Korodaj 2008; Barr et al. 2011), but not wet sclerophyll forests where lower bird diversity was found after recent (<7 years) thinning (Brown et al. 1991; Kutt 1996). Thinning did not alter species composition, indicating many species were widely distributed in the cypress-ironbark forests of the Pilliga (Date et al. 2002). Bird diversity is influenced by vegetation structure and is higher in structurally complex forests compared to more simple habitats (Seddon et al. 2003; Date et al. 2002; Hurlbert 2004). Date et al. (2002) suggested that thinning of homogeneous dense tree stands to promote grassy and mixed grass/shrub understorey could benefit many ground-foraging and ground-nesting species and in our study thinning increased CWD volumes, acacia cover and ground cover (plants and cryptogams), but not litter cover (Waters et al. 2018). In sub-tropical cypress woodland, thinning of cypress and eucalypts did not affect bird species richness (Eyre et al. 2015), despite greater diversity of small birds in sites with tall shrubs (Eyre et al. 2015; see also Craig et al. 2015). The positive association between bird diversity and tall shrubs may be confounded by the presence of the aggressive noisy miner in sites with fewer tall shrubs. Noisy miners can depress bird diversity (Maron et al. 2013), even when they occur in relatively low abundances (Eyre et al. 2015). In our study, although noisy miners were negatively associated with cypress density, they were recorded at just four sites in total (all thinned, but with low acacia cover), and likely suppressed diversity of small birds at these sites (3.3±1.4 species vs. 6.7±0.4 at all other sites). This is consistent with the observation that cypress forests, where eucalypt density is low, provide a refuge for small birds against the noisy miner (Eyre et al. 2015). More extensive areas of thinning may encourage noisy miners and reduce bird diversity.

Thinning had a significant positive effect on the diversity and activity of bats. Overall, nightly bat activity and diversity was greater (24-30 % and 60-100 %) in recent and old thinning treatments than the unthinned treatment, while there was a marginal trend for greater activity in the intermediate relative to the unthinned treatment. Thinning significantly reduced the density of stems (~6500 stems ha^{-1}) in all thinning treatments to ~1600 stems ha^{-1} (Waters et al. 2018), with a more patchy thinning operation applied in the intermediate thinning treatment (commercial thinning that targeted small and larger stems to provide a timber product). The response of bats is consistent with other studies that have found positive associations with a reduction in stem density (Humes et al. 1999; Blakey et al. 2016, 2017). Thinning opens up the forest structure, reducing clutter, which facilitates foraging by echolocation, particularly for those species that are fast-flying with low maneuverability (Humes et al. 1999; Fenton 1990; Gonsalves et al. 2013a), even though there may be similar levels of insect (bat prey) biomass in thinned compared to unthinned stands (Gonsalves et al. in press). While responses of bats to thinning may vary by species (Patriquin and Barclay

2003), we found no effect on species composition, suggesting that thinning did not preclude activity of particular species, including clutter-adapted bats that may be more active in unthinned stands (Blakey et al. 2016). Given that the threatened south-eastern long eared bat *Nyctophilus corbeni* selectively roosts in dead trees within dense stands (Law et al. 2016b) and dead trees can be collateral damage of thinning, specific efforts should be made to retain these when hollows or large fissures are present. Retention of dead trees in clumps would help to ensure maintenance of spatial connectivity and heterogeneity of stem density and other habitat components.

The biomass of nocturnal insects was similar among forest management treatments, but lowest in long undisturbed sites, and was approximately four times greater than levels recorded in coastal and floodplain wetlands (Gonsalves et al. 2013a & 2013b; Blakey et al. 2016; Gonsalves et al. in press), but markedly lower than linear remnants in an agricultural landscape after widespread rain (Lentini et al. 2012). Nevertheless, the distribution of insect biomass varied among taxa, size classes and among forest management treatments. In our study, thinning was also associated with a greater representation of beetles to overall insect biomass, with this trend strongest in the recent thinning treatment. The effects of thinning on insects are not well understood (Duguay et al. 2000; Schowalter et al. 2003; Verschuyl et al. 2011) and complicated by responses often being specific to insect functional groups. For instance, increases in the abundance of herbivorous insects in thinned stands may be associated with increased availability of canopy- and lower level foliage. While we did not distinguish between predatory and herbivorous beetle taxa in our study, recent thinning treatments contained 2-3 times more Acacia stems than the unthinned treatment (Waters et al. 2018). In *E. camaldulensis* vegetation, the response of invertebrates to thinning was neutral (Gonsalves et al. in press; Blakey et al. 2016). Elsewhere, invertebrate communities have shown positive, negative and neutral responses to thinning, though a summary of available research identified an overall positive response of functional groups (e.g., herbivores, predators and detritivores) to forest thinning (see Verschuyl et al. 2011). However, in our study, the unthinned treatment was associated with a relatively high biomass of moths, flies and other insects, highlighting the value of unthinned stands within the landscape of the study area.

Thinning also had a significant positive effect on the diversity and abundance of reptiles, though this was only the case for the intermediate thinning treatment. The abundance of reptiles in this treatment was more than double the abundance recorded in all other treatments, while diversity was ~ 40 % greater than all other treatments. Comparisons using extrapolated sample data support this pattern. Several studies have reported greater reptile species richness in forested habitats that are relatively open (Hall et al. 1997; Craig et al. 2009), with high ambient light levels suitable for basking (Brown & Nelson 1993; Vitt et al. 1997, 1998; Wetzel & Burgess 2001). But this may be offset by greater acacia cover in the recent thinning treatment (Waters et al. 2018). Acacia appears to be important for sap-licking geckoes (McNab 2014), and we often observed a number of gecko species on Acacia branches, though most commonly in intermediate thinning. Eyre et al. (2015) found larger skinks were associated with thinned cypress forests, but smaller skinks were less abundant in thinned stands where litter cover was greater, while no strong response to CWD was recorded. In the Pilliga, higher abundances and diversity of reptiles have been recorded on large and small logs compared to other habitat types (Date and Paull 2000), and a positive association with CWD volumes has been reported elsewhere (Craig et al. 2009). The trend for lower abundance of reptiles in our recent thinning treatment is unlikely to be related to litter cover which was similar among all treatments, nor a shortage of CWD volume which was greatest in this treatment (Waters et al. 2018). Instead, this low abundance may represent a short-term impact of thinning disturbance, with reptile abundance recovering over a longer period of time. It is unclear why a positive association between reptile abundance and thinning was only recorded for the intermediate thinning treatment. This treatment was representative of commercial thinning that was patchy in nature with moderate CWD volumes and higher ground cover (Waters et al. 2018), which may provide a local mosaic of habitat types for reptiles. Alternatively, the patches left unthinned may provide small refuges from the thinning disturbance facilitating faster recolonisation of thinned patches.

Thinning had no significant effect on the diversity and composition of native non-volant mammals. However, activity of native non-volant mammals was significantly greater in the intermediate thinning treatment than all other treatments, respectively, a result that was similar for reptiles. Again the greater cover of intermediate thinning and its patchiness leaves small refuges potentially important for ground mammals. Slashing of shrubs is known to increase the activity of *M. giganteus* and *O. cuniculus* rabbits (Davis et al. 2016) and these species were negatively associated with cypress stem density in our study. Thinning in ponderosa pine *Pinus ponderosa* revealed positive responses by small mammals that were species-specific and associated with the availability of CWD and/or increased understorey vegetation cover and food resources (Converse et al. 2006). Initial positive responses of non-volant mammals to thinning have been attributed to increases in invertebrate food sources (Ahlgren 1966), herbaceous food and cover (Wilson and Carey 2000) and CWD cover (Carey and Johnson 1995, Suzuki and Hayes 2003). Small non-volant native mammals were very rare in our study and their lack of recovery (see also Craig et al. 2009) may be associated with the low level of ground cover, which despite greater cover in intermediate and old thinning treatments (Waters et al. 2018), was considerably less than the benchmark reported for C. glaucophylla forests (Gibbons et al. 2008). This in turn could be related to the widespread occurrence of introduced herbivores, especially *C. hircus*, and native herbivores, such as *M.* giganteus. Low ground cover is likely to be detrimental for small mammals and reptiles, although feral predators (*V. vulpes*) were not frequently recorded in this system (six events across four sites per two-weeks of monitoring).

Thinning had no effect on the diversity and composition of shrub and understorey plants, a result previously found in a number of studies (Thompson and Eldridge 2005b; McHenry et al. 2006). Instead, differences in plant diversity were attributed to other site-based factors (McHenry et al. 2006). It should be acknowledged that in our study, tree diversity was not included the characterisation of plant diversity and thinning may influence competitive interactions between some tree species (Waters et al. 2017). In North America, the response of plant species to thinning has mostly been positive (Verschuyl et al. 2011). For example, plant species richness increased with thinning of *P. ponderosa* forests in the south-west of the United States. These positive responses of plant diversity to thinning were also recorded 3-years post-thinning (Phillips and Waldrop 2008). These patterns contrast with Hunter (2013) who suggested above ground biomass of *C. glaucophylla* had a positive effect on native species density (a decrease in trees and shrubs, but an increase in herbaceous species), though only stems > 10 cm dbhob were measured. A neutral response of plant diversity to thinning in our study may reflect the low annual rainfall in our study area, with diversity of understorey plants positively associated with mean annual rainfall in *C. glaucophylla* forests

(Thomspon and Eldridge 2005b). While thinning had no detectable effect on plant diversity, structural attributes like acacia cover were greater after thinning (Waters et al. 2018).

2.4.4. Management Implications

The short- and longer-term responses of biodiversity to thinning of C. glaucophylla regrowth were mostly positive or neutral. Such results fit with the concept of thinning as a restoration tool for 'renewal' of simple stands of regrowth (Bowman et al. 2017), even when the focus is not on ecological outcomes as was the case in our study (noting that in most of our sites thinning was a non-commercial activity that left residues in the forest). Yet individual species responded idiosyncratically across the chronosequence landscape and unthinned forest represented habitat of similar value to thinned forest for some taxa. This highlights the importance of retaining some unthinned regrowth forest to provide a mosaic forest structure suitable for a diverse suite of flora and fauna and avoiding broad-scale practices that homogenise the landscape (Date et al. 2002; Hunter 2013; Kay et al. 2016; Law et al. 2016a; Eyre et al. 2015). To more fully elucidate the ecological outcomes of product removal after a commercial thinning operation, we have established a replicated thinning experiment incorporating product removal, the results of which will be presented elsewhere.

3. EXPERIMENTAL THINNING TRIAL FOR CYPRESS REGROWTH IN PILLIGA FORESTS

3.1. INTRODUCTION

Variable environments and past land-use in the Pilliga forests can make ecological interpretation of subtle management effects difficult. For example, most areas have experienced variable patterns and intensities of logging and thinning in the past and the interaction of these two treatments can potentially obscure individual treatment effects. Establishment of experimental field trials that manipulate one aspect of the environment provides a rigorous approach to testing management effects. The need for trials that manipulate tree density have emerged from studies that have simulated cypress growth data (Ross et al. 2008) and highlighted a level of uncertainty and variability in relation to the outcomes of thinning treatments, especially regarding tree growth. In particular, they suggested that gaps and disturbance created by thinning could promote recruitment and rapid refilling perpetuating a dense stand cycle. Experimental thinning using a Before-After-Control-Impact design and direct measurement of responses in vegetation structure and biodiversity would help to resolve such uncertainties. While thinning trials have been undertaken in the Pilliga in the past (Knott 1995), these are plot based to allow measurement of tree growth and their small scale therefore makes them unsuitable for measuring the response of biodiversity.

3.2. METHODS

3.2.1. Experimental design

A Before-After-Control-Impact (BACI) design thinning trial was used to assess the effect of recent thinning on vegetation structure and habitat values (living-, dead- and hollow-tree density, vegetation cover and coarse woody debris), as well as biodiversity (bats, birds, reptiles, ground mammals). In all, 20 experimental sites (12 ha each) representing thinning undertaken at two different stages of cypress regeneration: 1) Early regeneration that is 'locked up' ('early thin'); and 2) Mature cypress ('second thin'). Each thinning treatment had paired controls as well as two long undisturbed references ('early thin reference' and 'second thin reference') (Fig. 3.2.1).

Site selection for thinning was undertaken by Forestry Corporation of NSW with the aid of information on management history to identify areas of dense vegetation that would be suitable for thinning. Early thinning sites were located in Compartment 82 of Pilliga West State Forest, with each treatment and paired control separated by at least 200 m. Second thinning sites were split between compartment 333 in Baradine State Forest and compartment 95 in Pilliga West State Forest. Two long undisturbed sites were located in Pilliga West State Forest and two in Cumbil State Forest.

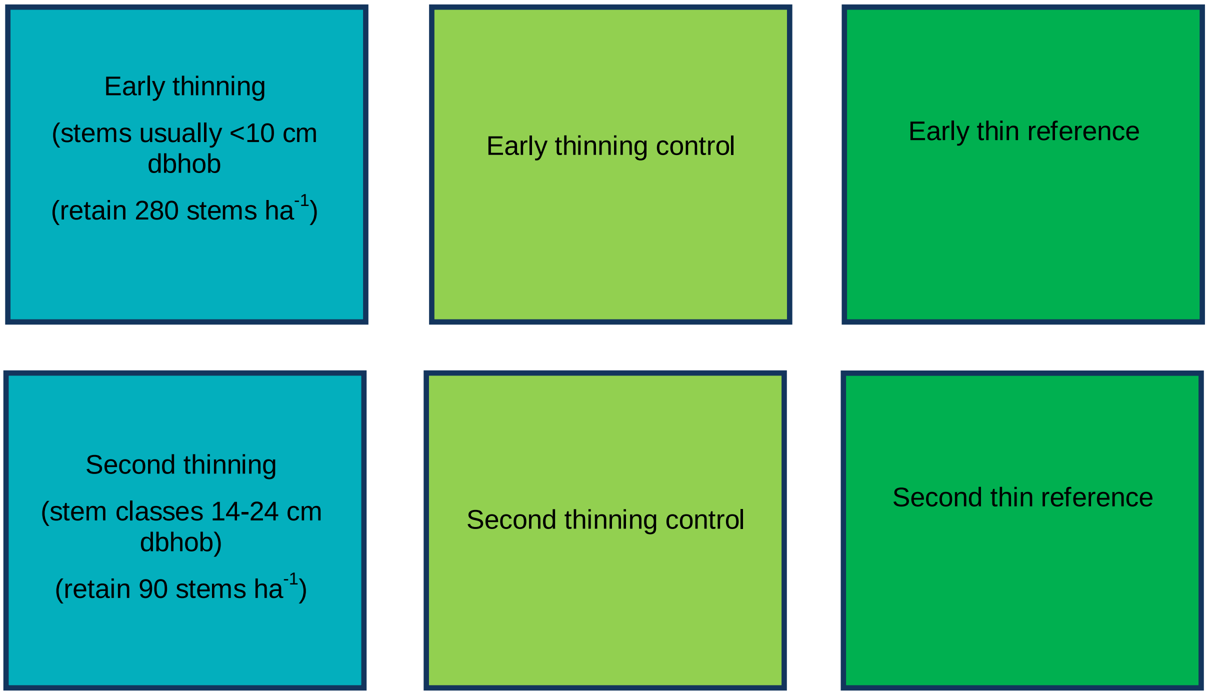


Fig. 3.2.1. Experimental design used for thinning trial. Area of treatment for thinned sites was 12 ha. Early thinning treatment targeted small stem size classes and aimed to retain ~280 stems ha^{-1} while second thinning treatment targeted larger size classes (14-24 cm) and aimed to retain ~90 stems ha^{-1}.

The early thin treatment predominantly targets suppressed and competing sub-dominant and co-dominant small sized cypress stems (<10 cm dbhob), although some larger stems may also be thinned to meet the nominal density (280 stems ha^{-1} of retained cypress stems. Second thinning targets larger cypress stems (14-24 cm dbhob) than early thinning and retains fewer stems (90 stems ha^{-1}). These nominal stem densities were selected to reflect a gradient in treatment effects rather than any fixed, 'desirable' state. Furthermore, actual stocking densities achieved in the experimental thinning treatments were influenced by the market for thinnings and so thinning may not have reached the nominal densities described. Long undisturbed reference sites ('early thin reference and 'second thin reference') were characterised by a lack of stumps and, the presence large cypress and/or ironbark trees. There were 4 replicate sites for each respective thinning treatment and paired controls, and two replicates for each long undisturbed reference treatment.

Thinnning was achieved using a small mechanized harvester that created alleys (outrows) about every 10 m to allow the machine access to stems either side (Fig. 3.2.2a). The same alleys were used to remove cut material from each site using a forwarder (Fig.3.2.2b). Despite removing timber products from the forest, considerable residue was left in situ, especially smaller stems and tree heads but also some larger stems. 1326 and 1404 tonnes of timber were removed from early thinning and second thinning sites, respectively (FCNSW unpubl. data). Thinning operations were carried out between May and October 2016.

Fig. 3.2.2. A small mechanized harvester used in an early thin site (a) and a forwarder used in a second thin site to remove the stock pile of stems (in background) from the forest (b).

At each site a 200 m transect for sampling biodiversity was established across the central core of the site to avoid edges and allow sampling of areas representative of the respective treatment. To examine the responses of vegetation structure, habitat values and biodiversity to thinning, we surveyed vegetation structure and floristics, and a range of fauna groups (bats, birds, non-volant mammals and reptiles) before (March 2016) and after thinning (March 2017).

3.2.2. Vegetation structure and composition

Surveys were undertaken in March 2016 at each site along the site transect. Three 20 x 10 m plots located at approximately 50, 100 and 150 m along each transect were used to measure habitat features (Table 3.2.1). Both the number of and diameter at breast height over bark (dbhob, 1.30 m), of all trees and shrubs within each plot was measured and allocated to one of four categories (< 10 cm; >10 cm to < 30 cm; >30 to < 50 cm; and >50 cm). The number of dead trees was also recorded within each plot but not allocated to size classes. The number and diameter (top of each stump) of all stumps was also measured in each plot and categorised as either small (< 5cm) or large diameter (> 5cm). All trees, shrubs and stumps were identified to genus. The number of hollow-bearing trees was also recorded within each plot.

A line intercept method was used to quantify cover of vegetation using a 20 m transect running through the centre of each 20 x 10 m plot. A pole with a laser pointer mounted at a height of 1.5 m was used to record six ground cover categories (bare, cryptogam, litter, fine woody debris (FWD), plant and dung). The height of vascular plants was measured using a densitometer positioned at the top of the pole used for ground cover categories following the methods described in AusPlots (2012). All vascular plants along transects were identified to genus. Five vegetation structure categories; ground (<0.5m); understorey (0.5 to 2 m); mid-storey (2.1 to 6 m); sub-canopy (6.1 to 14 m) and canopy (>14 m) were recorded at 1 m intervals along the 20 m transect.

A visual estimate of grazing intensity based on the presence of fresh and old dung was recorded for each plot. The amount of dung was categorized as low (rare or infrequent sighting); medium (some present) and high (frequently sighted) for each of three herbivore types (kangaroos, goats/sheep and horses). Identification of dung followed that outlined in (Landsberg et al. 1994).

3.2.3. Sampling CWD structure and abundance

Coarse Woody Debris was considered to be woody material > 10 cm diameter and >1 m in length and not rooted in the soil. For each site, CWD structure and abundance was measured along a 20 m transect running through the centre of each plot. The number of logs, log length, large and small end diameter and categories for log source (trunk or branch) and state of log decay (Little= solid log, bark and twigs present; Minor=decomposition commencing, bark largely absent, exposed soft sapwood, twigs absent; Extensive= intermediate to late stage decomposition, bark absent, exposed soft sapwood, partial heartwood exposure, log breaking up or fragmented) was recorded for each plot. The presence and diameter of hollows (minimum entrance diameter >2 cm) and the presence of fissures was recorded for all CWD along each transect.

Table 3.2.1. Key measured habitat variables associated with vegetation structure and composition and the volume of coarse woody debris.

Component	Variable	Description	Units of measurement
Vegetation structure and composition	Density of trees and shrubs	Number of standing trees and shrubs for each of 7 size categories (<5; 5-10->10-20; >20-30; >30-40; >40-50; >50 cm diameter at breast height over bark (dbhob)) and major tree taxa per plot	Number of trees and shrubs ha^{-1}
	Density of dead trees	Number of standing dead trees per plot	Number of dead trees ha^{-1}
	Density of cut tree stumps	Number of stumps allocated to each of 7 size categories (<5; 5-10->10-20; >20-30; >30-40; >40-50; >50 cm diameter at top of stump) per plot	Number of stumps ha^{-1}
	Density of hollow-bearing trees	Number of hollow bearing trees per plot	Number of hollow-bearing trees ha^{-1}
	Ground cover	Proportion of bare, litter, CWD, cryptogam along a 20 m transect within each plot	Proportion of bare, litter, CWD and cryptogam cover
	Vegetation cover	Proportion of ground (< 2 m), mid (2-6 m), sub-canopy (6-14 m) and canopy (>14 m) along a 20 m transect within each plot	Proportion of shrub, mid, sub-canopy and canopy cover
DCWD structure and abundance	Volume of logs with a fissure	Volume of CWD with at least one fissure per plot	Volume m^{3}ha^{-1}
	Volume of hollow-bearing logs	Volume of logs with at least one hollow per plot	Volume m^{3}ha^{-1}
	Large end log diameter	Large end diameter (cm) per plot	Mean log diameter ha^{-1}
	Small end log diameter	Small end diameter (cm) per plot	Mean log diameter ha^{-1}
	Log volume	Log length x π x (average log diameter/2)^2	Volume m^3 ha^{-1}

3.2.4. Reptiles

Reptile diversity and abundance was measured using pitfall trapping, diurnal transect searches and nocturnal spotlighting. Pitfall trapping was undertaken at each site using two pitfall traps that were installed ~50 m and ~150 m along each site transect. One of the two pitfall traps at each site consisted of a pipe (0.15 m diameter and depth of 0.5 m) with three drift fences (3m length, damp course) set as arms, while the other consisted of a 20 L bucket (0.30 m diameter and depth of 0.3 m) also with three drift fences (3 m length, damp course). Pitfall traps were opened for up to eight days in March 2016 (pre-thinning) and March 2017 (post-thinning). All captured animals were identified to species and then immediately released at the point of capture.

Nocturnal spotlighting was carried out once at each site, with a sampling effort that equated to a 20 minute spotlight by two observers along the 200 m site transect. Observers surveyed within 10 m either side of the transect and all observed reptiles were identified to species (where possible) and recorded. In addition to pitfall trapping and spotlighting, diurnal searches were undertaken along each site transect for 20 minutes (one person). Other incidental observations of reptiles were also recorded. However, only data from pitfall trapping, spotlighting and diurnal transect searches were combined and used in analyses.

3.2.5. Birds

Birds were recorded at each site using aural and visual surveys at two stationary points in each site between dawn and midday. The two stationary points were situated 50 m and 150 m along each site transect. At each stationary point, birds were surveyed over a 20 minute period. All species observed within 50 m of the stationary point were identified to species and their abundance recorded. Bird surveys were repeated at each site by a second observer in reverse order to ensure that sites that were surveyed later in the morning by observer 1 were surveyed earlier by observer 2. Each bird species was later assigned to a bird abundance class (0=absent, 1=1-2 individuals, 2=3-4 individuals, 3=>4 individuals) prior to analysis.

3.2.6. Non-volant mammals

Camera traps were used to survey non-volant mammals at each site. A single wildlife camera (Reconyx model Hyperfire HC600) was deployed ~100 m along the transect at each site along with a lure to attract mammals from within the immediate area of the site. The lure used was a peanut butter, honey and oats bait, together with a sesame oil soaked pad, both of which were placed inside a hair tube and secured in the ground using a tent peg. Cameras were set at a height of 1 m on a tree at each site, while the lure was placed at a distance of 1.5 m from the base of the tree. Cameras were programmed to have a high sensitivity, a RapidFire picture interval with 5 pictures per trigger, a 1 minute delay (quite period) between triggers and a high quality night mode. The sampling period for all sites was 14 days, with images recorded onto SD cards. At the conclusion of the sampling period, SD cards were downloaded into a corresponding site directory and then imported into Google Picasa for processing. Processing of images involved tagging those images if an animal was visible. Tags were generally made to species however there were occasions when an animal was unable to be identified. Species identifications were made using appropriate guides (e.g., The Mammals of Australia – Van Dyck and Strahan 2008). Once all images had been processed, species tags were extracted for each image using a script and ExifTool (Harvey 2013), which extracts metadata associated with each image. Extracted metadata were then imported into a spreadsheet for further processing. For each set of 5 consecutive images ('an event'), a species name was assigned if that species was tagged in any of the images that comprised an event. The total number of events assigned with a species name at each site was collated to provide a record of those species observed at each site and also their abundance.

3.2.7. Bats

Bats were surveyed at each site for 2-3 nights using two acoustic Anabat detectors (Titley Scientific – Brendale QLD) which were deployed 50 m and 150 m along each site transect, respectively. Detectors were positioned with microphones set at a height of 1 m and oriented at a 45° angle to the ground to face gaps within vegetation at each site. This was done to ensure detectors at each site were sampling the same volume of airspace in each site, since vegetation structure is known to influence the vertical stratification of open- and edge-space aerial-foraging bats (Adams et al. 2009). Also, gaps within vegetation at each site were selected for sampling to reduce any potential bias associated with a reduced volume of airspace in sites with dense vegetation (particularly unthinned sites). All files were downloaded from detectors and analysed using automated software, AnaScheme (Adams et al. 2010) in association with a key for bats of the Pilliga (unpublished – B. Law). Bat calls with fewer than three valid pulses (i.e. minimum of six data points and model quality of 0.8) were not analysed by AnaScheme. Because multiple bat species may call simultaneously, calls were assigned to a species only if >50% of pulses within the sequence were attributed to that species and only passes with a minimum of three pulses classified to the same species were identified. All calls that could not be assigned to a bat taxon were included in counts of total bat activity but were labelled as 'unidentified'. Since it is not possible to distinguish between calls of certain species (e.g., *Nyctophilus corbeni*, *N. geoffroyi* and *N. gouldi,* or *Mormopterus planiceps* and *M. petersi*, or *Scotorepens greyii* and *S.* sp.), these calls were assigned to a species group (e.g., *Nyctophilus* spp., or *Mormopterus planiceps/petersi*, or *Scotorepens greyii*/sp.) by AnaScheme. For each detector and each night, the number of bat passes for each species and total bat activity (all identified and unidentified bat calls) was tabulated.

3.2.8. Data analyses

For all response variables (e.g., stem density, dead stem density, CWD volumes and diversity metrics), mean ± SE was calculated for each treatment and plotted. All plots were visually assessed and trends are described. No statistical testing was undertaken to examine the response of variables to both thinning treatments as this will be done following resampling of all sites at a second time point post-thinning.

3.3. RESULTS

3.3.1. Vegetation structure and habitat values

3.3.1.1. Stem density

The density of all stems in the early thin control and early thin (pre-thinning) treatments was >3600 stems ha^{-1}. Post-thinning, stem density was reduced by ~50 % and was comparable to levels recorded in the early thin reference (2842 stems ha^{-1}) treatment (Fig. 3.3.1). The greatest reduction in stem density was recorded for the 5-10 cm size class, with a 75 % reduction in stem density post-thinning (Fig. 3.3.1). Reductions in stem density were also observed for the <5 cm and >20-30 cm size classes (Fig. 3.3.1).

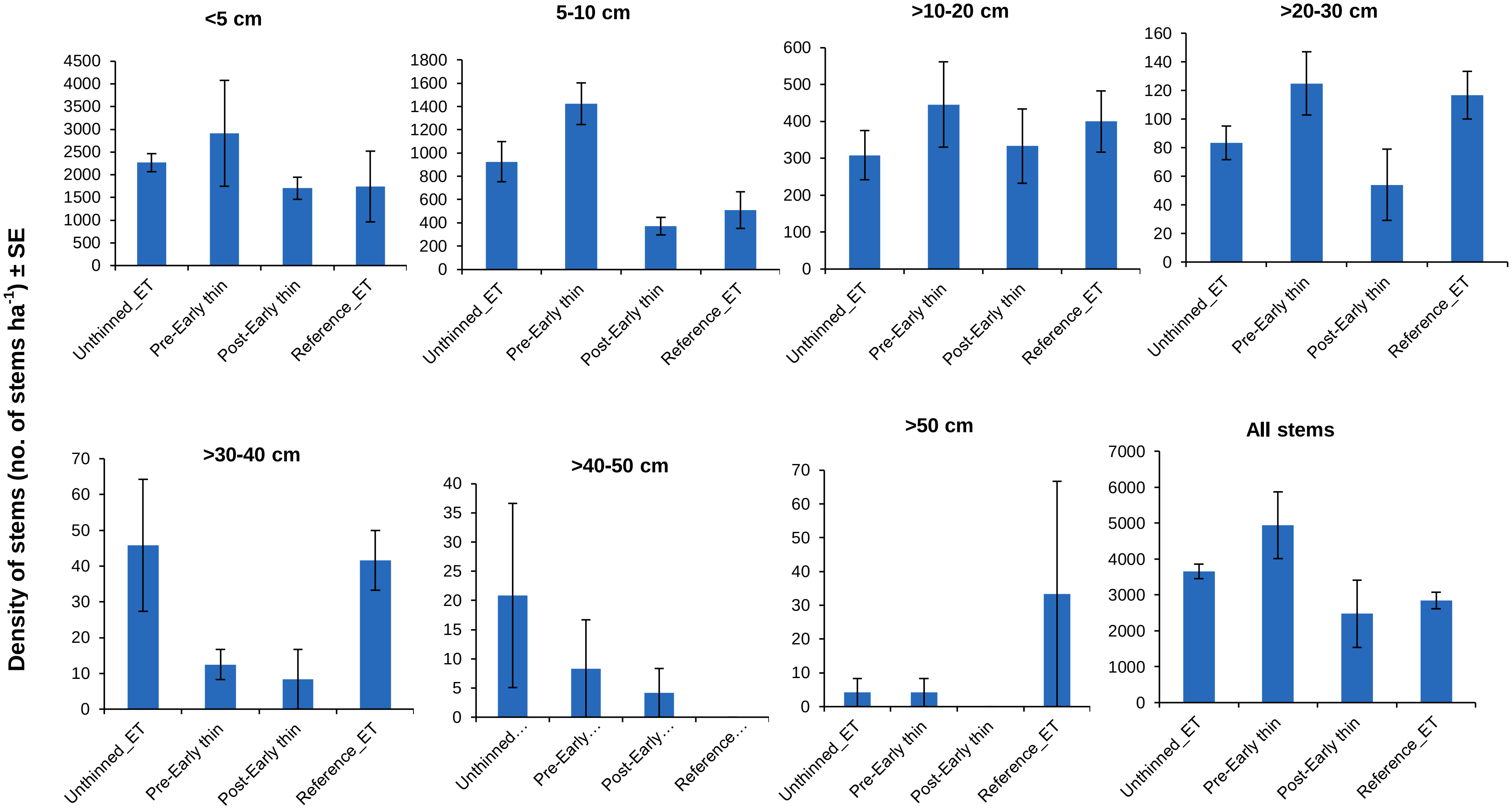


Fig. 3.3.1. Density of stems of varying size classes recorded in early thin control, early thin (pre- and post-thinning) and early thin reference treatments.

The density of all stems in the second thin control, second thin (pre-thinning) and second thin reference treatments ranged between 1175-1992 stems ha^{-1}. Post-thinning, the density of stems reduced by 25 % (Fig. 3.3.2). The greatest reduction in stem density was recorded for the >20-30cm size class, with a 83 % reduction in stem density post-thinning. Reductions in stem density were also observed for the 5-10 cm and >50 cm size classes (Fig. 3.3.2).

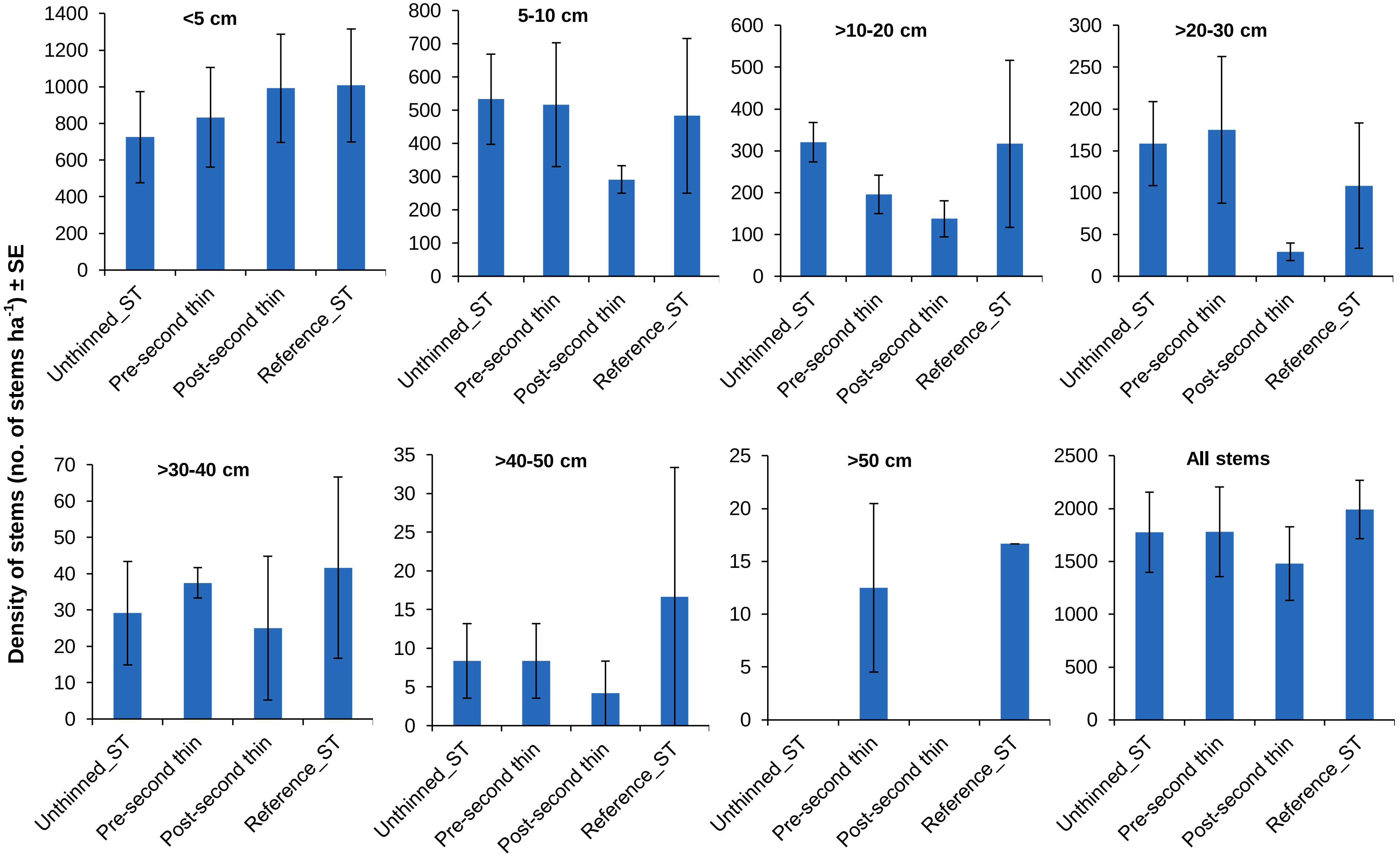


Fig. 3.3.2. Density of stems of varying size classes recorded in second thin control, second thin (pre- and post-thinning) and second thin reference treatments.

The density of cypress and buloke stems in the early thin treatment post-thinning reduced by ~60-70 %, respectively, with greatest reductions observed for the <5 cm and 5-10 cm size classes (Fig. 3.3.3). Stem density of cypress post-thinning was comparable to the early thin reference treatment whereas buloke stem density was almost 50-times greater than the early thin reference treatment. The density of Acacia stems in the early thin treatment post-thinning increased by ~400 %, with the increase dominated by stems in the <5 cm size class (Fig. 3.3.3). Acacia stem density was almost double that of the early thin reference treatment. The density of eucalypts remained relatively unchanged in the early thin treatment post-thinning, though there was a small reduction in stems of the 5-10 cm size class (Fig. 3.3.3). This and other smaller size classes were rarely recorded in the early thin reference treatment. The density of other stems remained unchanged in the early thin treatment post-thinning and was similar to the early thin reference treatment (Fig. 3.3.3). The density of dead stems in the early thin treatment was relatively unchanged post-thinning, though there was a reduction in the 5-10 cm size class and an increase in the <5 cm size class which may be associated with misclassification of stems in the 5-10 cm size class (Fig. 3.3.3).

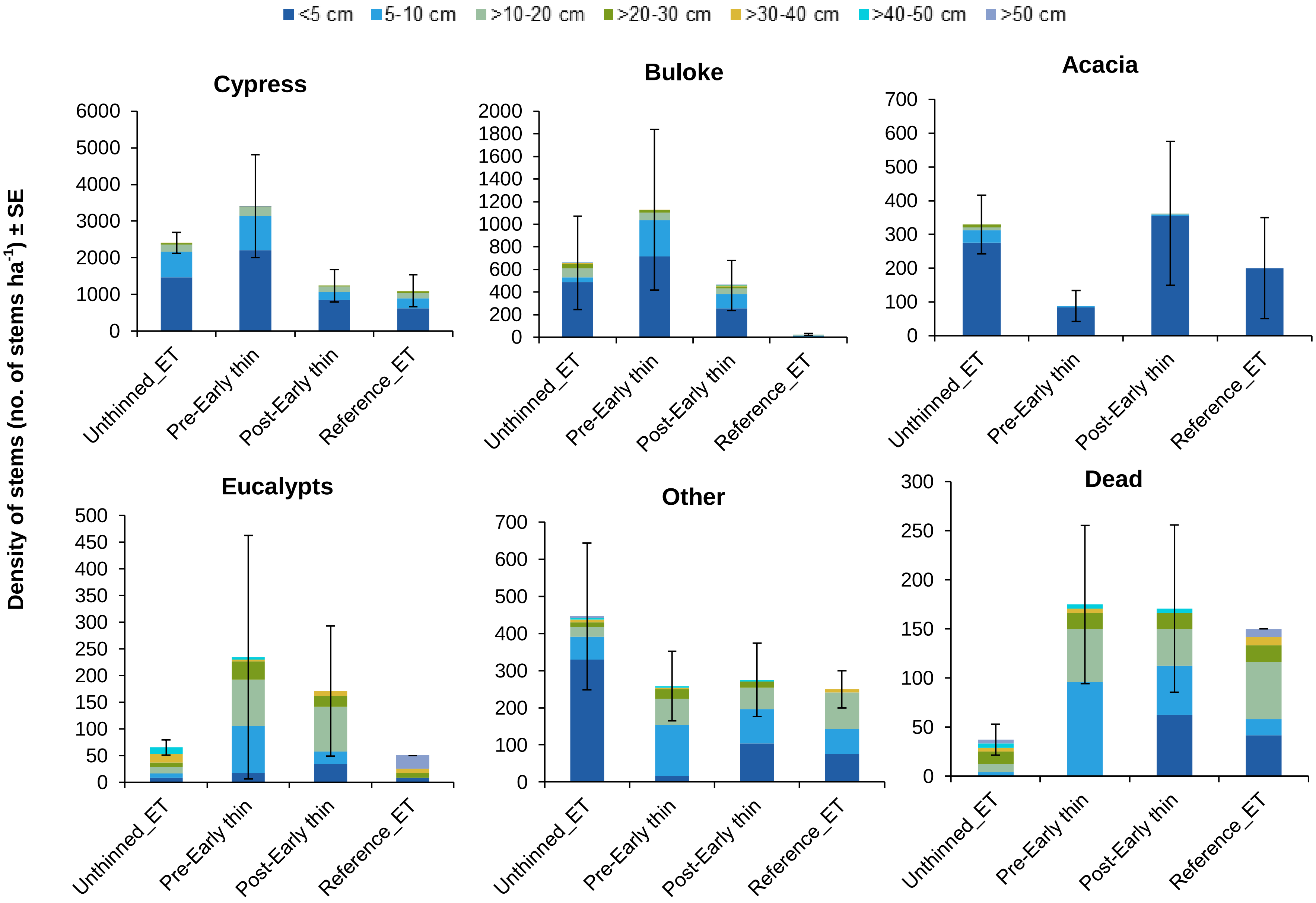


Fig. 3.3.3. Density of stems of varying size classes recorded for major taxa and dead trees in early thin control, early thin (pre- and post-thinning) and early thin reference treatments.

The density of cypress stems in the second thin treatment post-thinning reduced by ~20-70 %, with greatest reductions observed for the >20-30 cm size class (Fig. 3.3.4). Stem density of cypress post-thinning was comparable to the second thin reference treatment. The density of buloke in the second thin treatment was significantly affected by thinning with no buloke stems recorded post-thinning (Fig. 3.3.4). The density of Acacia stems remained relatively unchanged in the second thin treatment post-thinning, though there was a small reduction for the 5-10 cm size class (Fig. 3.3.4). Similar densities of Acacia were recorded in the second thin reference treatment. Acacia stem density post-early thinning was almost double that of the second thin reference treatment. There was a great diversity in size classes of eucalypts across all treatments (Fig. 3.3.4). The density of other stems in the second thin treatment showed a trend for a reduction post-thinning, with densities of stems in the 5-10 cm, >10-20 cm and >20-30 cm size classes being affected (Fig. 3.3.4). There was a reduction in the density of dead stems in the second thin treatment post-thinning, with greatest reduction seen in the 5-10 cm and >20-30 cm size classes (Fig. 3.3.4).

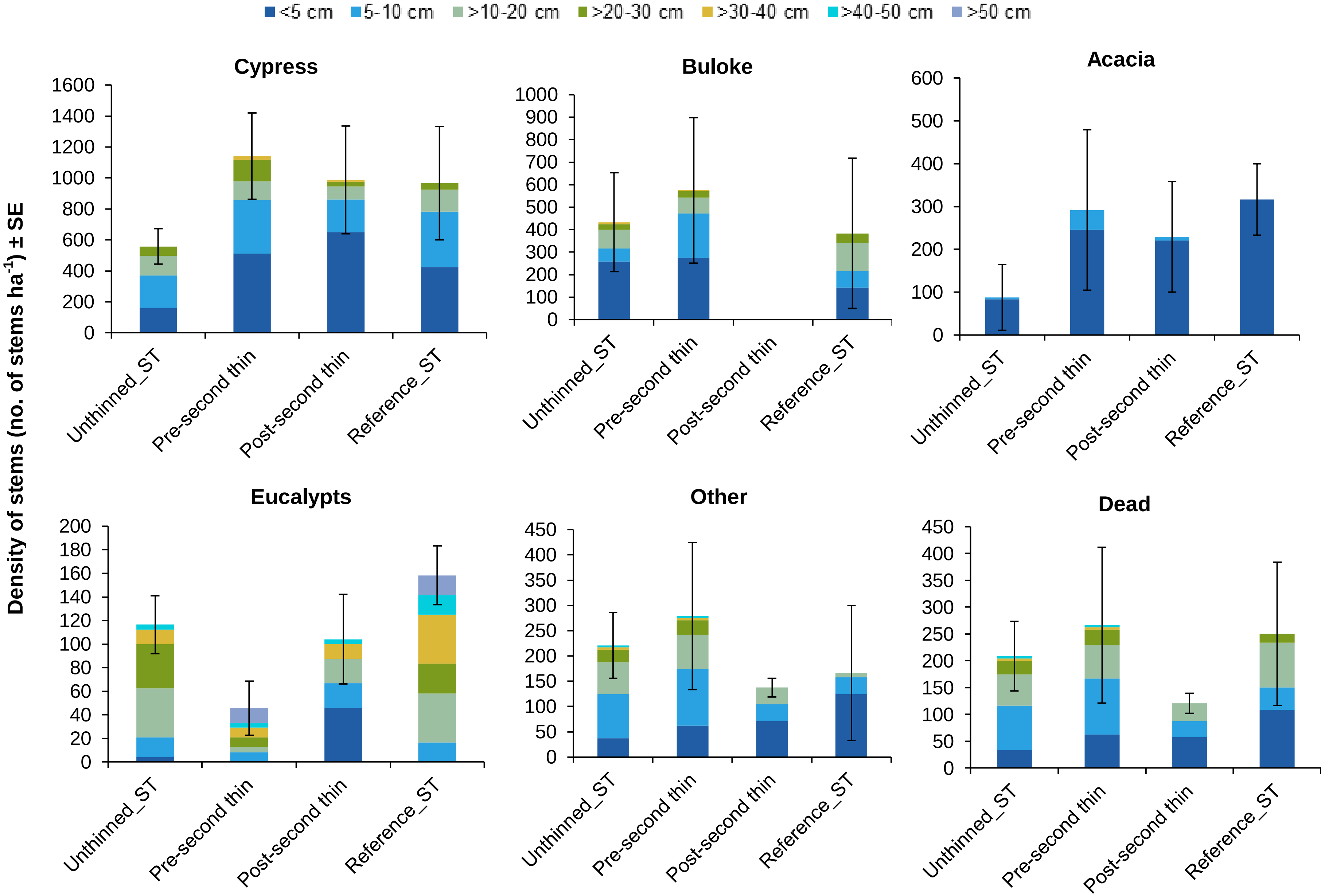


Fig. 3.3.4. Density of stems of varying size classes recorded for major taxa and dead trees in second thin control, second thin (pre- and post-thinning) and second thin reference treatments.

3.3.1.2. Stump density

Prior to thinning, stump density (from past harvesting) in early thin (221 stumps ha^{-1}) and early thin control (184 stumps ha^{-1}) treatments was greater than the few stumps recorded in the early thin reference (25 stumps ha^{-1}) treatment (Fig. 3.3.5). Post-thinning, the density of stumps in the early thin treatment increased by 83 %. Increases in the density of stumps post-thinning were found for the >10-20 cm and >20-30 cm size classes and to a lesser extent the >40-50 cm size class (Fig. 3.3.5). No increases in the density of stumps were observed for the <5 cm and 5-10 cm size classes, presumably because machinery rolled over the top of these and these were not cut.

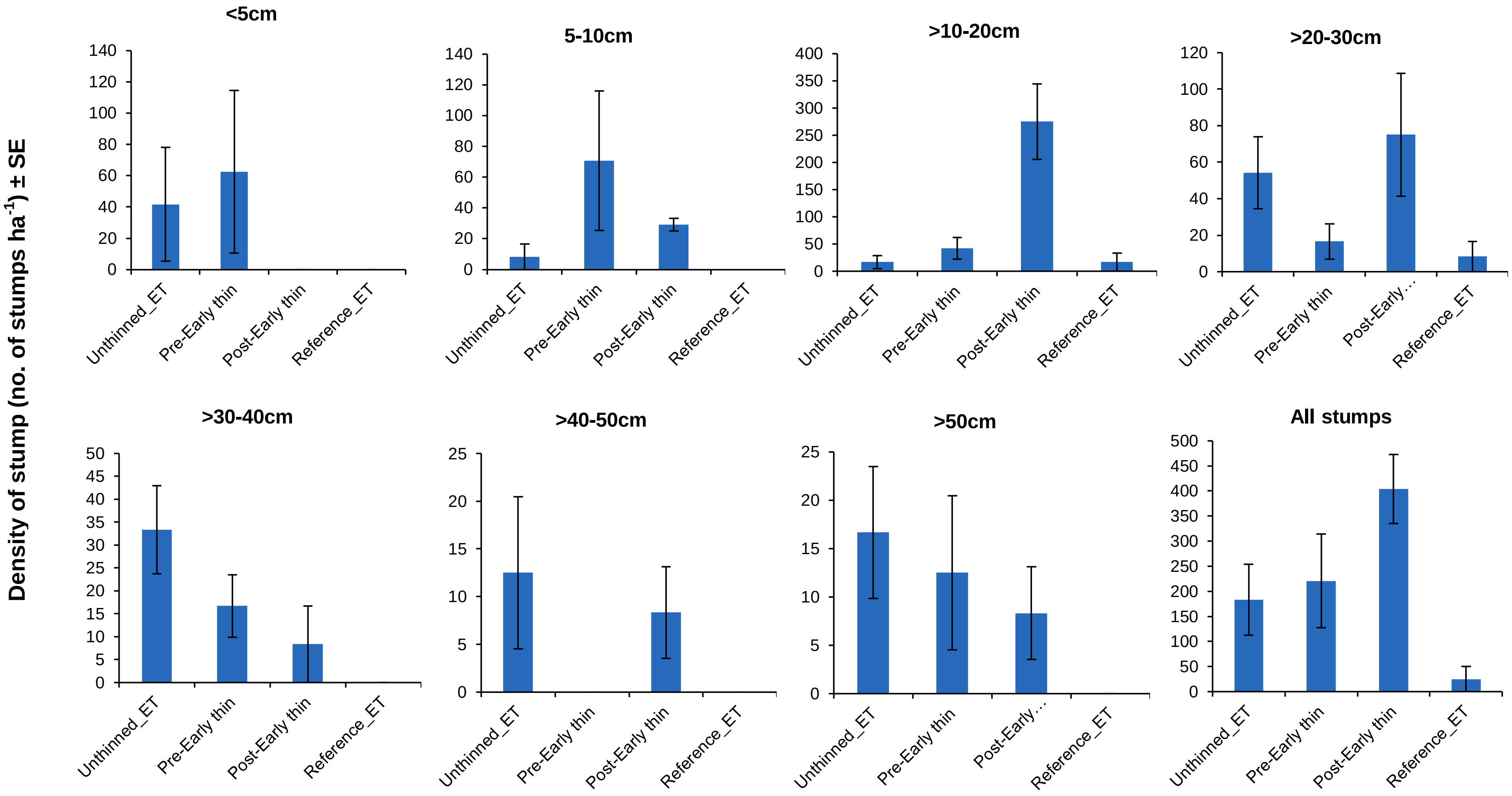


Fig. 3.3.5. Density of stumps of varying size classes recorded in early thin control, early thin (pre- and post-thinning) and early thin reference treatments.

The density of stumps in the second thin treatment (221 stumps ha^{-1}) was moderately greater than the second thin control (79 stumps ha^{-1}) and second thin reference (125 stumps ha^{-1}) treatments prior to thinning (Fig. 3.3.6). Post-thinning the density of stumps in the second thin treatment increased by 129 %. Increases in stump density post-thinning were found for the >10-20 cm size class, and to a lesser extent in the <5 cm, 5-10 cm, >20-30 cm and >40-50 cm size classes (Fig. 3.3.6).

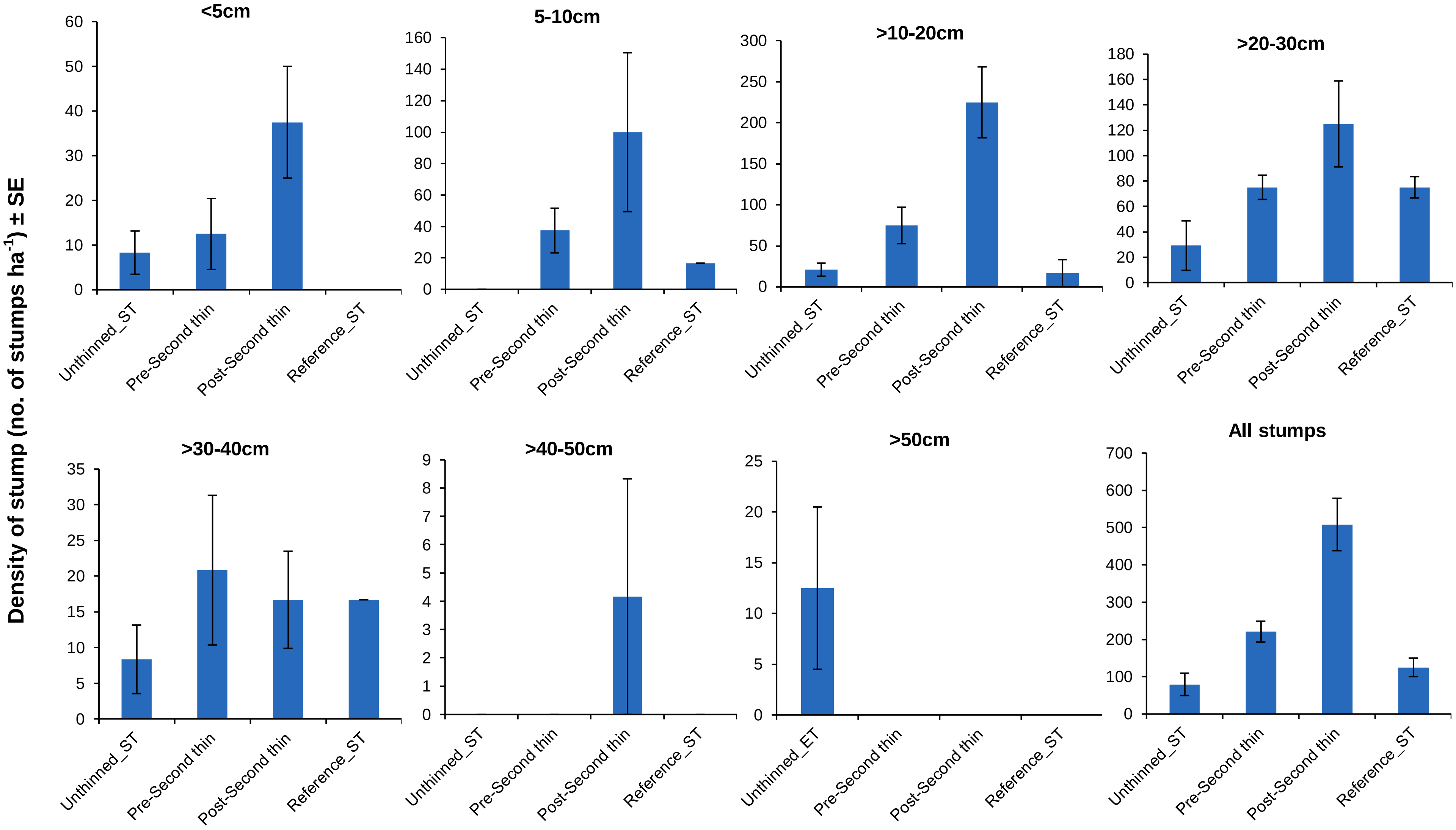


Fig. 3.3.6. Density of stumps of varying size classes recorded in second thin control, second thin (pre- and post-thinning) and second thin reference treatments.

3.3.1.3. Cover

Ground cover in the early thin control, early thin (pre-thinning) and early thin reference treatments was dominated (>66 %) by litter and this did not change with thinning (Fig. 3.3.7). Plant cover was low in the early thin reference (<2 %) treatment and absent from the early thin control and early thin treatments, both pre- and post-thinning (Fig. 3.3.7). Dung and cryptogam cover were also low across all treatments (<1.5 %) (Fig. 3.3.7). Bare ground cover in the early thin reference (25 %) treatment was 2-3-times greater than other treatments (Fig. 3.3.7). Post-thinning, there was a small increase in the cover of fine woody debris (FWD) in the early thin treatment.

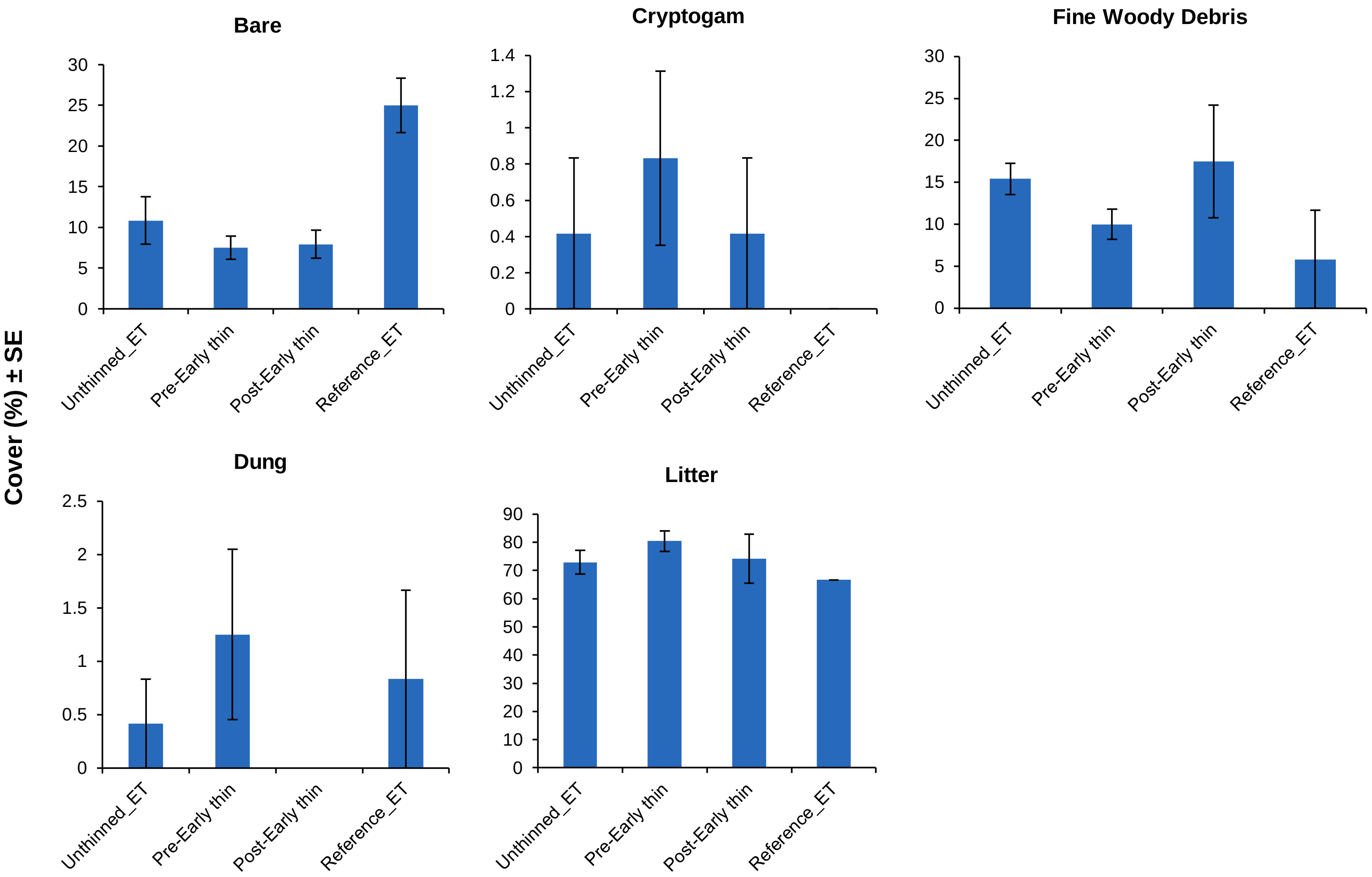


Fig. 3.3.7. Ground cover recorded in early thin control, early thin (pre- and post-thinning) and early thin reference treatments.

Similarly, ground cover in the second thin control, second thin (pre-thinning) and second thin reference treatments was dominated (>66 %) by litter and did not change with thinning (Fig. 3.3.8). Plants were absent from all treatments whereas dung and cryptogam were present at low levels (<4 %). Bare ground cover was greatest in the second thin reference (25 %) treatment and there was a trend for an increase in the second thin treatment post-thinning (Fig. 3.3.8). Post-thinning, there was a doubling of the amount of FWD (Fig. 3.3.8).

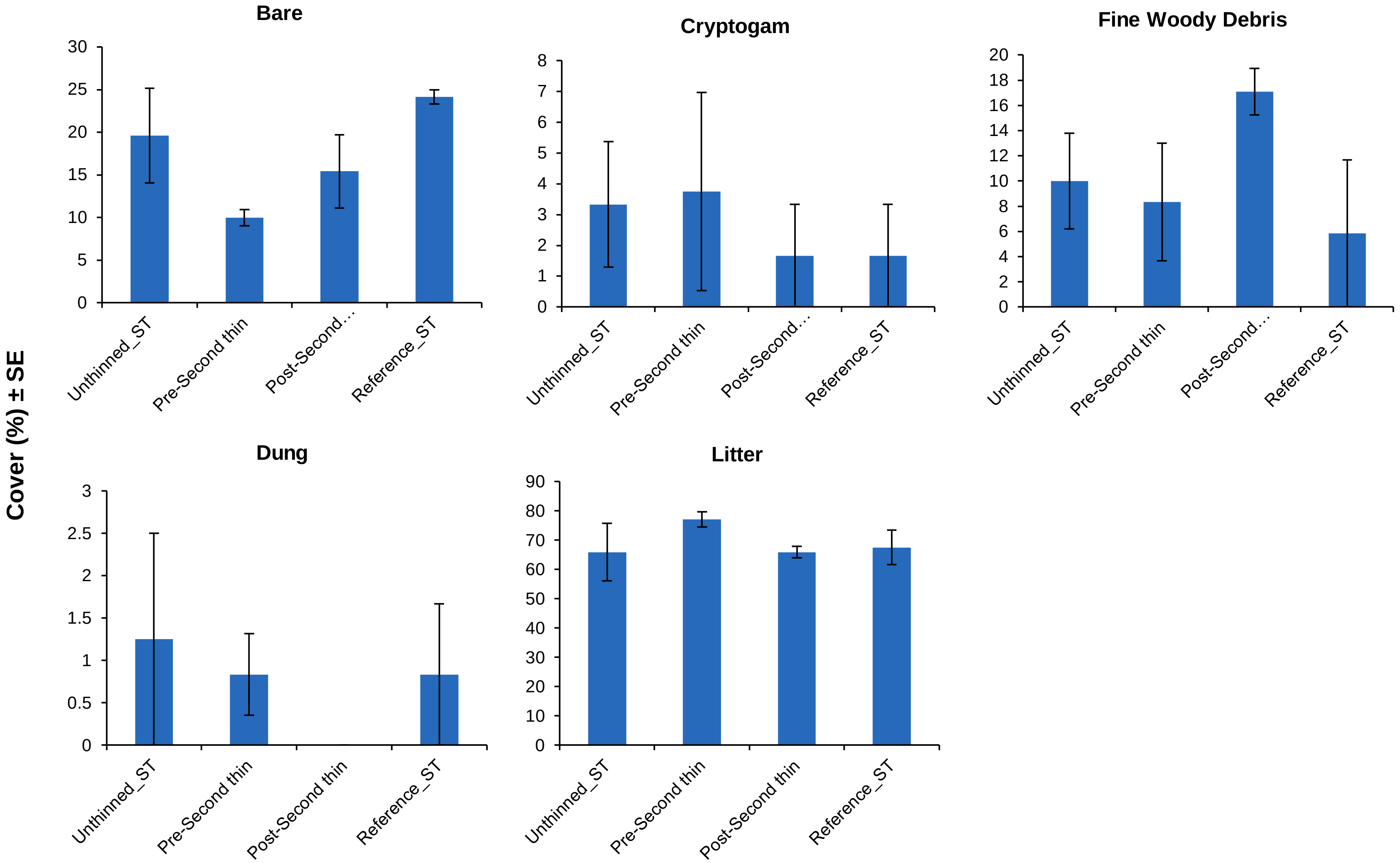


Fig. 3.3.8. Ground cover recorded in second thin control, second thin (pre- and post-thinning) and second thin reference treatments.

Understorey cover in the early thin treatment was similar to the early thin reference treatment prior to thinning (Fig. 3.3.9). Post-thinning, cover was 8-times greater in the early thin treatment (Fig. 3.3.9). Thinning reduced midstorey and sub-canopy cover by approximately 50 %. For the former, this reduction resulted in cover that was comparable to the early thin reference treatment. For the latter, cover in the early thin treatment was 45 % lower than the early thin reference treatment. Canopy cover remained unchanged with thinning in the early thin treatment (Fig. 3.3.9).

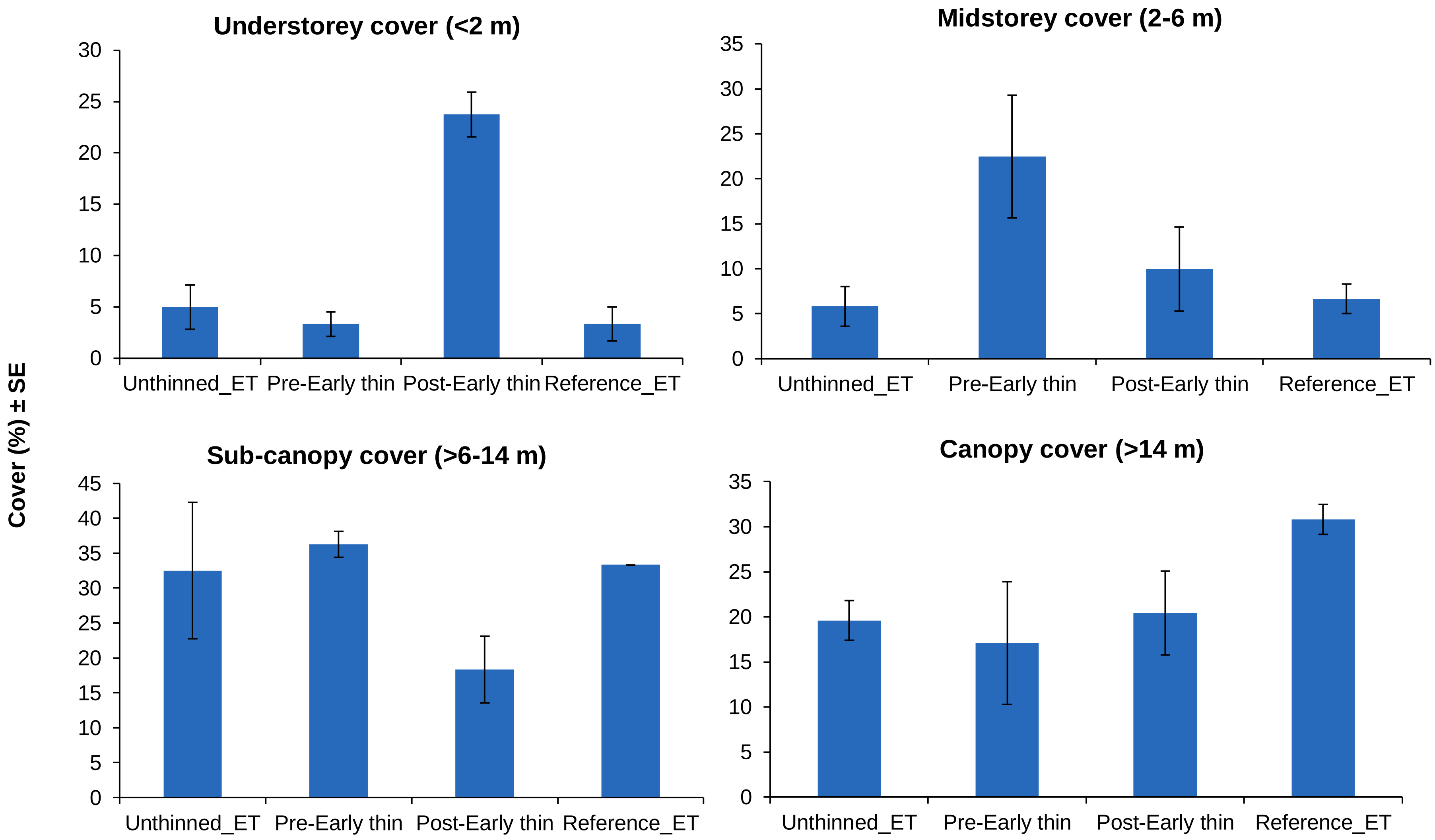


Fig. 3.3.9. Cover recorded for various strata in early thin control, early thin (pre- and post-thinning) and early thin reference treatments.

Understorey cover and midstorey cover remained relatively unaffected by thinning in the second thin treatment (Fig. 3.3.10). Sub-canopy cover reduced by ~50 % in the second thin treatment post-thinning, whereas canopy cover remain unchanged (Fig. 3.3.10).

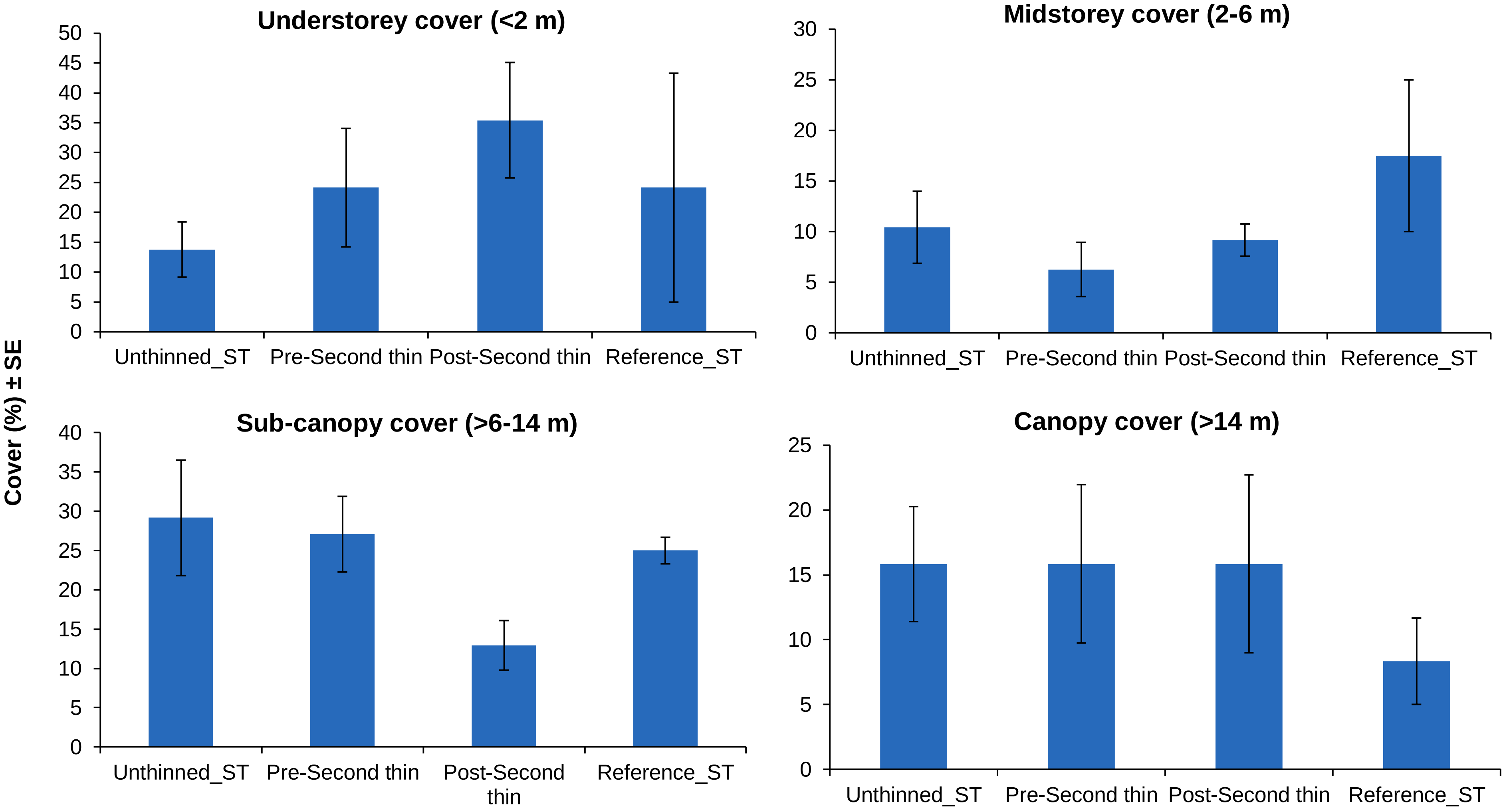


Fig. 3.3.10. Cover recorded for various strata in second thin control, second thin (pre- and post-thinning) and second thin reference treatments.

3.3.1.4. Coarse woody debris

The volume of CWD in the early thin treatment did not differ from pre- (62 m^3 ha^{-1}) to post-thinning (76 m^3 ha^{-1}) and was comparable to the volume of CWD in the early thin control (79 m^3 ha^{-1}) but was 2-3 times greater than the early thin reference (26 m^3 ha^{-1}) treatment (Fig 3.3.11). There was a trend for greater volumes of hollow-bearing CWD post-thinning in the early thin treatment (23 m^3 ha^{-1}) relative to other treatments (12-14 m^3 ha^{-1}).

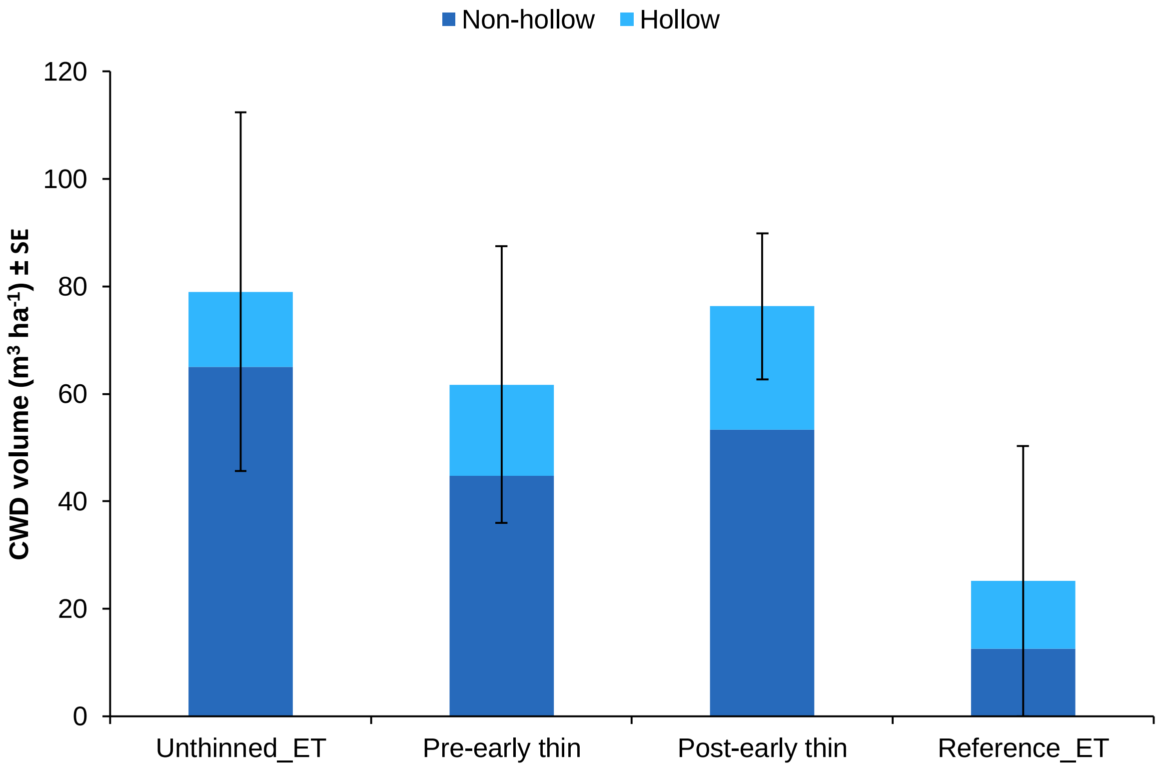


Fig. 3.3.11. Volume of CWD (hollow and non-hollow) recorded across all early thin treatments.

The volume of CWD in the second thin treatment increased from pre- (13 m^3 ha^{-1}) to post-thinning (46 m^3 ha^{-1}) and was 3-4 times greater than the second thin control (20 m^3 ha^{-1}) and the second thin reference (22 m^3 ha^{-1}) treatment (Fig 3.3.12). No hollow-bearing CWD was recorded in the second thin control and second thin treatments prior to thinning. Post-thinning the volume of hollow-bearing CWD in the second thin treatment (20 m^3 ha^{-1})

increased and was comparable to levels recorded in the second thin reference treatment (17 m^3 ha^{-1}).

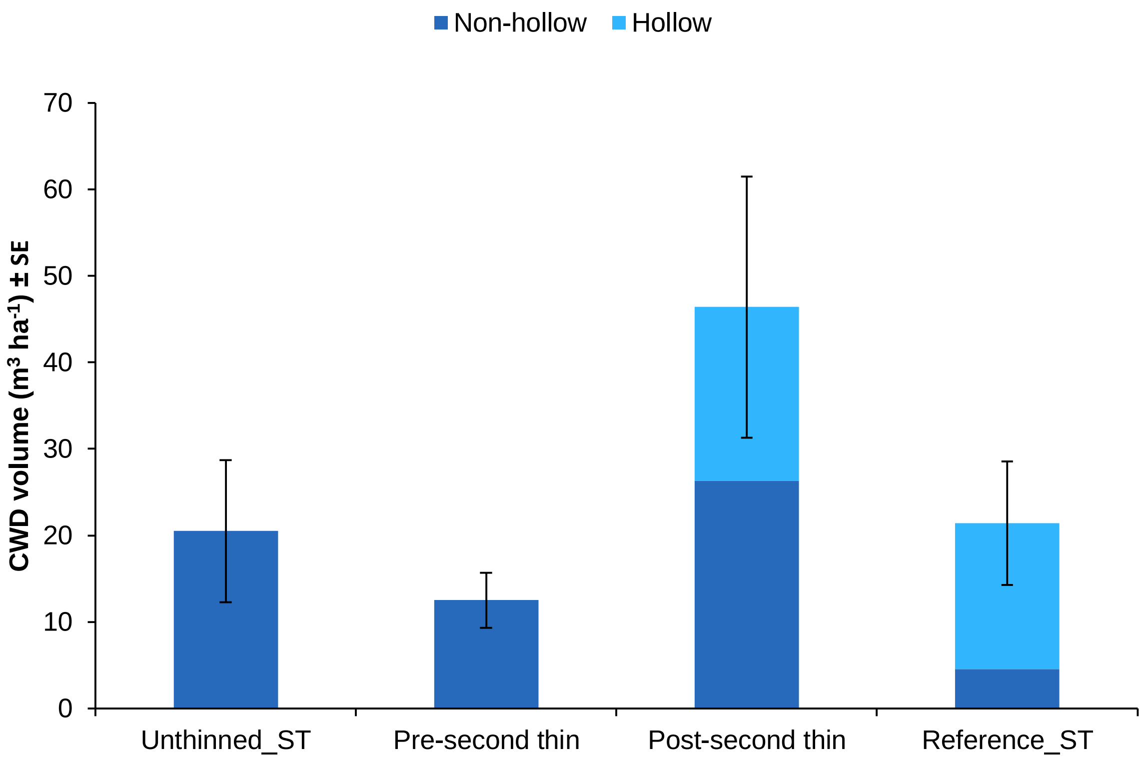


Fig. 3.3.12. Volume of CWD (hollow and non-hollow) recorded across all second thin treatments.

3.3.2. Biodiversity responses

3.3.2.1. Bats

In all, 32,328 bat calls were recorded representing 11 bat taxa pre- and post-thinning (Table S3.3.1). *Vespadelus vulturnus* (Little Forest Bat), *Mormopterus* spp. (Freetail Bats) and *Saccolaimus flaviventris* (Yellow-bellied Sheathtail Bat) represented most bat activity (28 %, 17 % and 15 %, respectively). Bat activity doubled in the early thin treatment post-thinning, but showed little change over the same period in the early thin control and early thin reference treatments (Fig. 3.3.13a). Bat diversity increased in the early thin control (doubled) and early thin treatments (tripled) from pre- to post-thinning, though the magnitude of change was greater in the latter, whereas diversity halved in the early thin reference treatment from pre- to post-thinning (Fig. 3.3.13b).

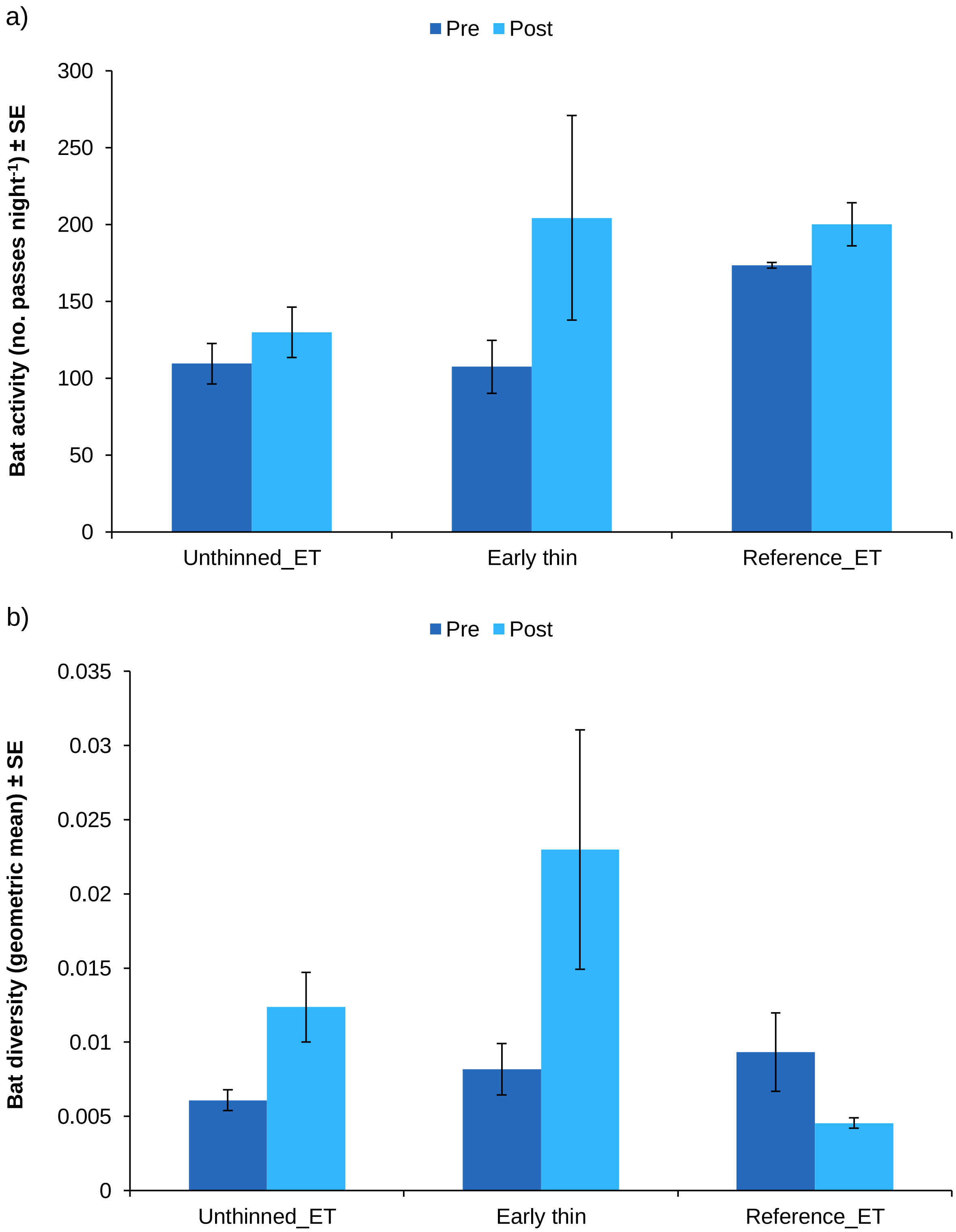


Fig. 3.3.13. Bat (a) activity and (b) diversity pre- and post-thinning in early thin control, early thin and early thin long undisturbed reference treatments.

Bat activity increased equally in the second thin control and second thin treatments, while there was a marginal increase in activity in the second thin reference treatment (Fig. 3.3.14a). Bat diversity increased in all treatments post-thinning, though the magnitude of change in the second thin treatment was five and 1.5-times greater than the second thin control and second thin reference treatments, respectively (Fig. 3.3.14b).

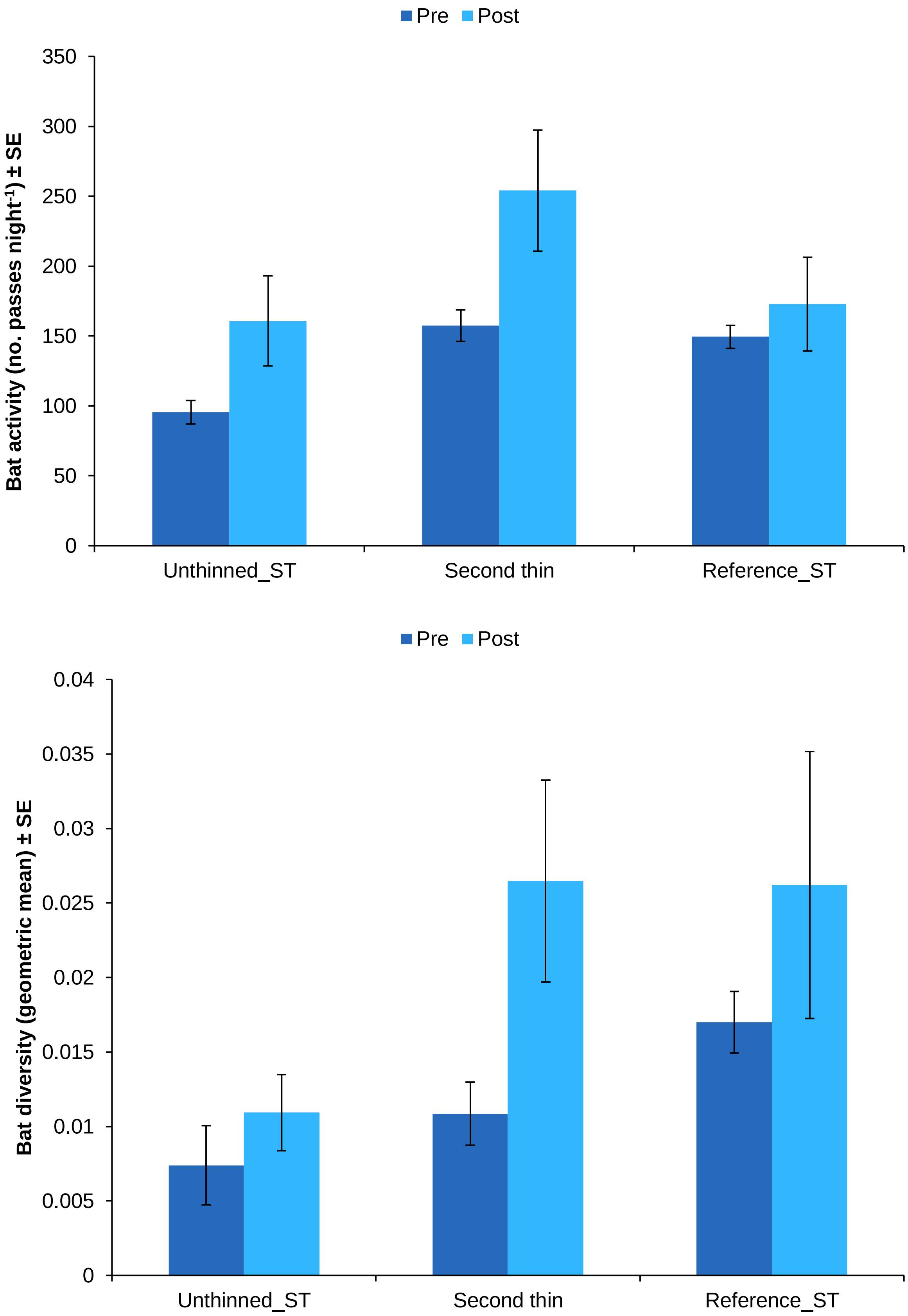


Fig. 3.3.14. Bat (a) activity and (b) diversity pre- and post-thinning in second thin control, second thin and second thin long undisturbed reference treatments.

3.3.2.2. Birds

In all, 70 bird species were recorded across all treatments (Table S3.3.2). *Eopsaltria australis* (Eastern Yellow Robin), *Pachycephala rufiventris* (Rufous Whistler) and *Rhipidura albiscapa* (Grey Fantail) were the most commonly detected taxa (77.5 %, 75 % and 72.5 % of all sites, respectively). Bird diversity was similar in the early thin control and early thin reference treatments from pre- to post-thinning, whereas there was an 80 % increase in diversity at the early thin sites post-thinning (Fig. 3.3.15).

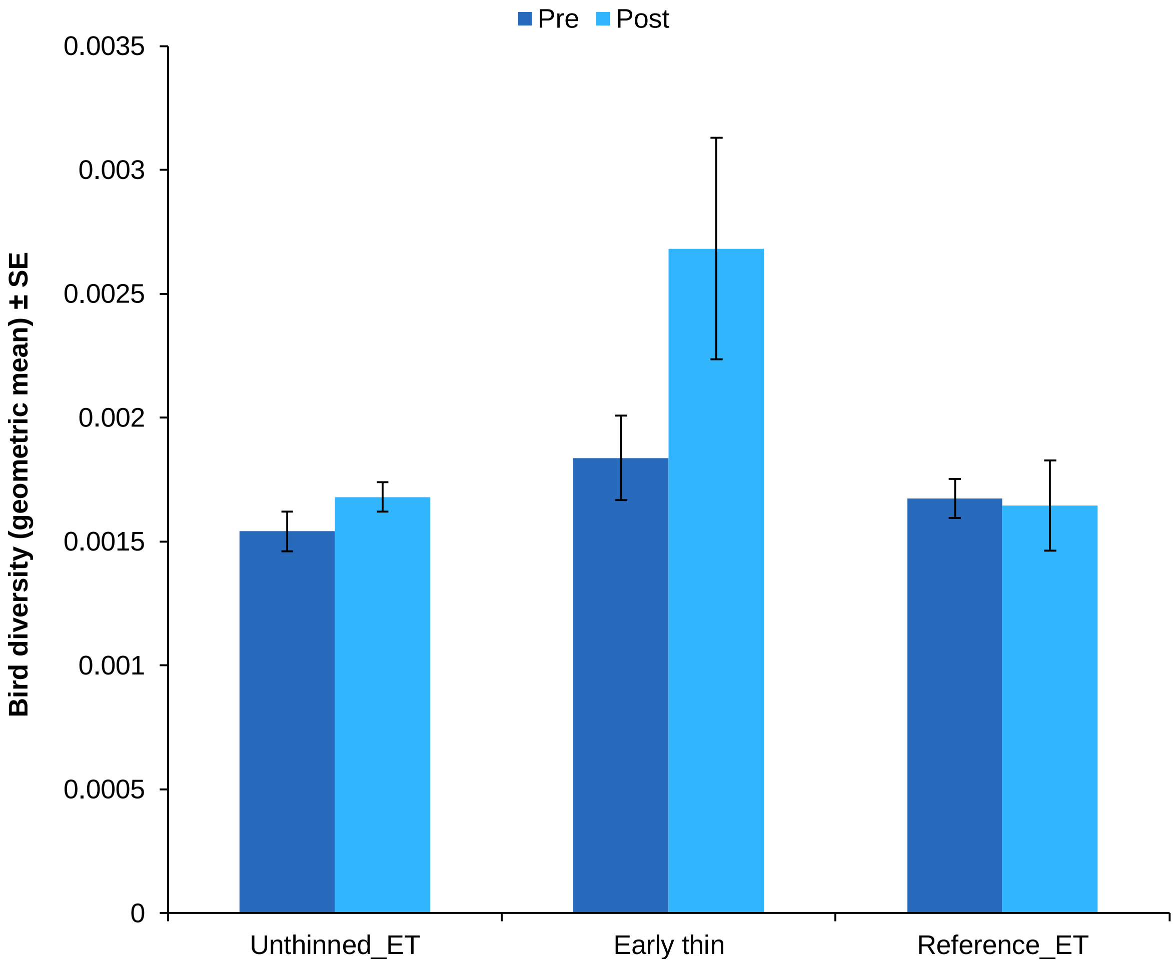


Fig. 3.3.15. Bird diversity pre- and post-thinning in early thin control, early thin and early thin reference treatments.

Bird diversity increased post-thinning in the second thin control and second thin treatments, though the magnitude of change was greater for the latter and double that of the second thin reference treatment (Fig. 3.3.16).

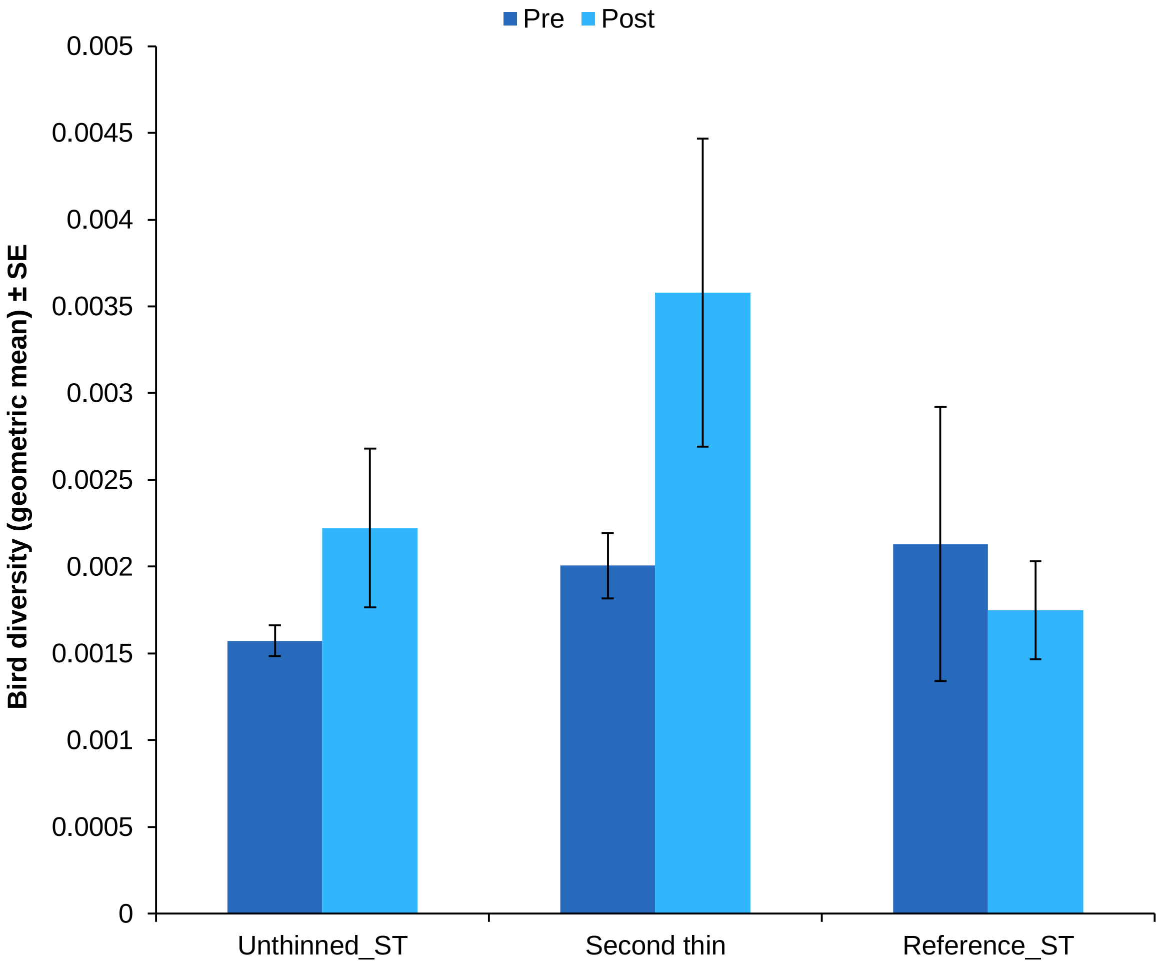


Fig. 3.3.16. Bird diversity pre- and post-thinning in second thin control, second thin and second thin reference treatments.

3.3.2.3. Non-volant mammals

In all, 59 events were recorded across all treatments representing 12 mammal species (7 native and 5 introduced) (Table S3.3.3). *Macropus giganteus* (Eastern Grey Kangaroo), *Capra hircus* (Goat) and *Antechinus flavipes* (Yellow-footed Antechinus) represented most mammal activity (31 %, 15 % and 10 %, respectively). Native mammal diversity was similar pre- and post-thinning in early thin control, early thin and early thin reference treatments (Fig. 3.3.17).

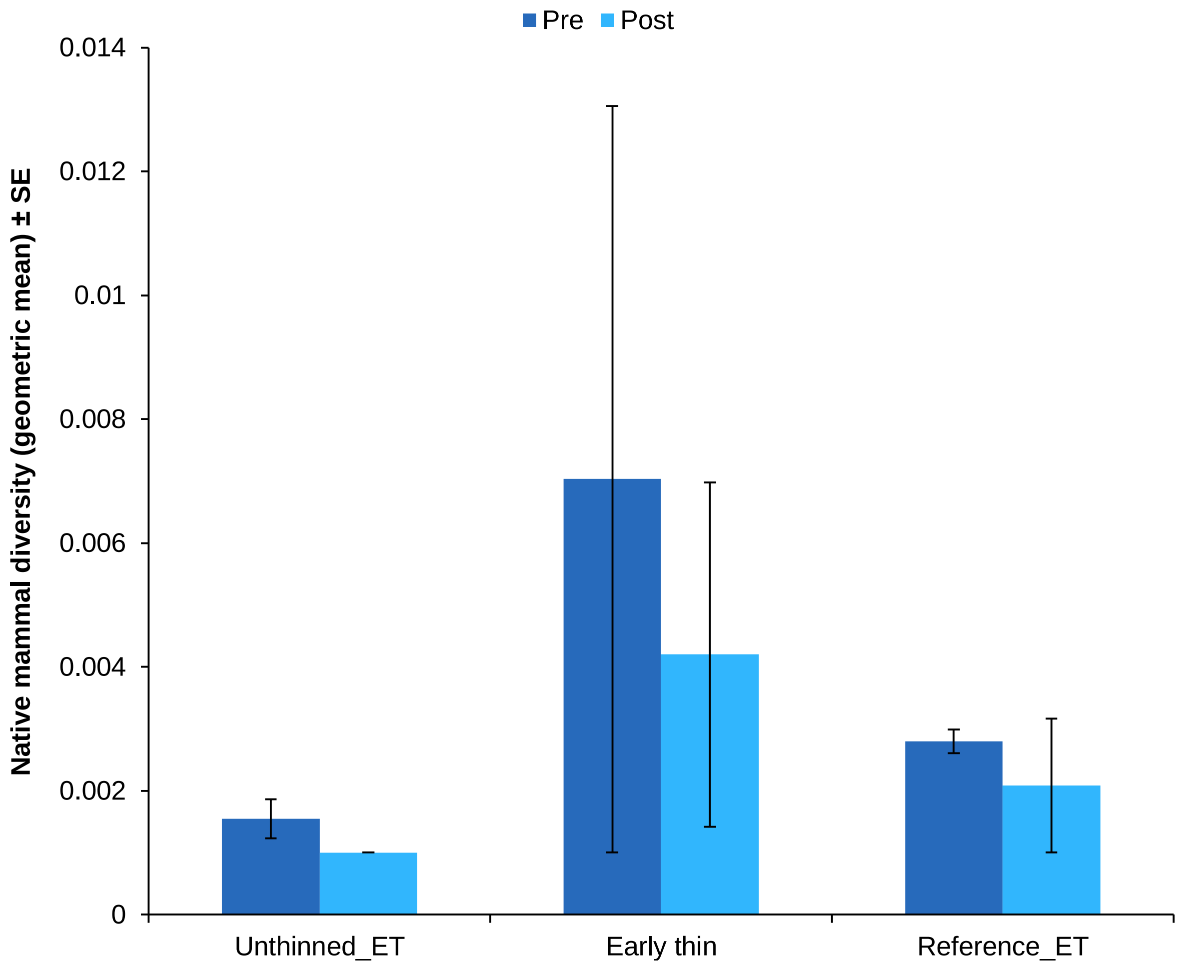


Fig. 3.3.17. Native mammal diversity pre- and post-thinning in early thin control, early thin and early thin reference treatments.

Native mammal diversity halved post-thinning in the second thin control treatment, whereas diversity tripled post-thinning in the second thin treatment. Despite the increase in the second thin treatment, diversity was two-times greater in the second thin reference treatment (Fig. 3.3.18).

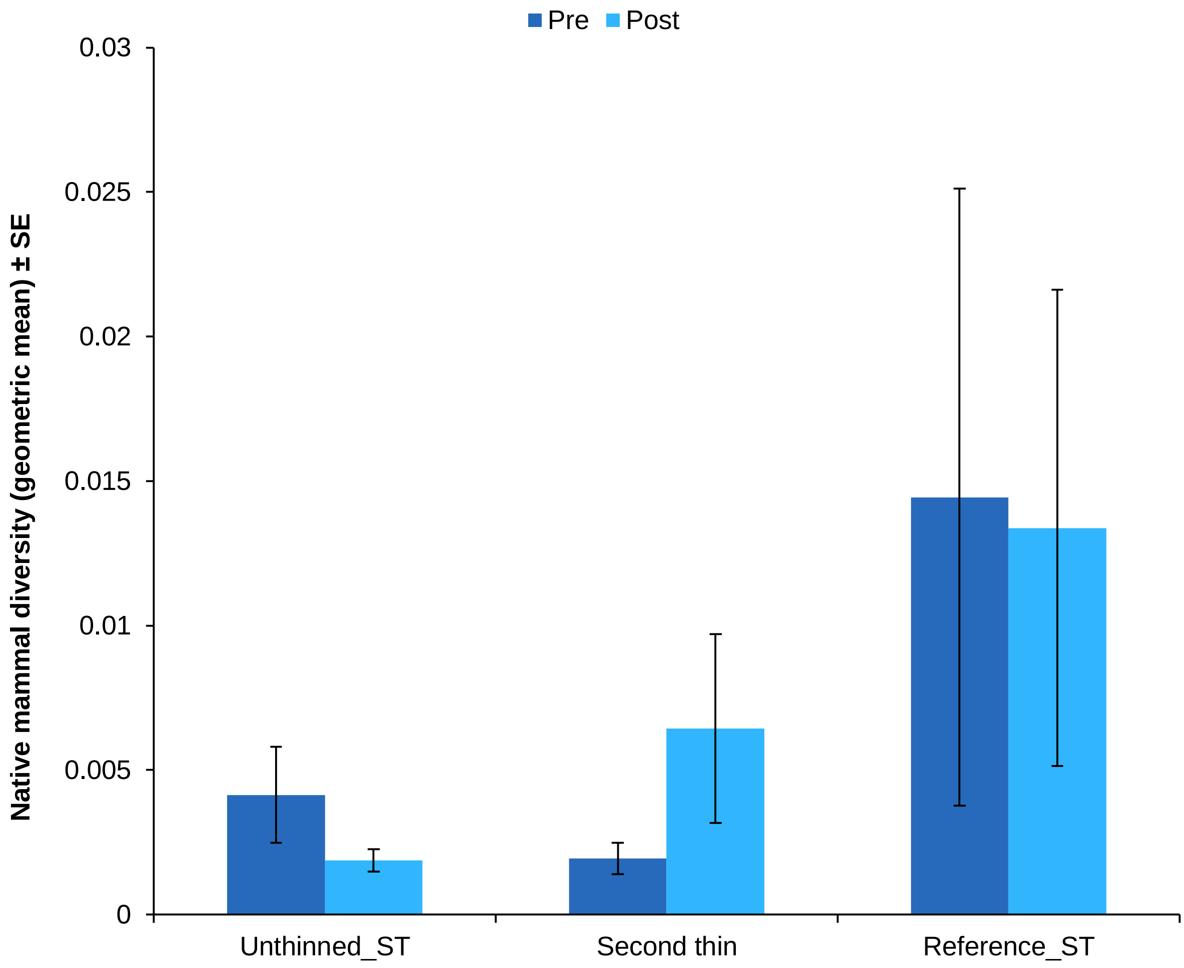


Fig. 3.3.18. Native mammal diversity pre- and post-thinning in second thin control, second thin and second thin reference treatments.

3.3.2.4. Reptiles

In all, 328 reptiles representing 12 reptile taxa were recorded across all treatments pre- and post-thinning (Table S3.3.4). *Morethia boulengeri* (Boulenger's Snake-eyed Skink), *Gehyra* spp. (Dtellas) and *Diplodactylus vittatus* (Eastern Stone Gecko) were the most frequently detected taxa (78 %, 10 % and 4 % of all reptile records, respectively). There was a trend for an increase (3-4 times) in reptile abundance from pre- to post-thinning in the early thin control, early thin and early thin reference treatments (Fig. 3.3.19a). Reptile diversity was similar in early thin control and early thin treatments from pre-to post-thinning, whereas diversity doubled in the early thin reference treatment (Fig. 3.3.19b).

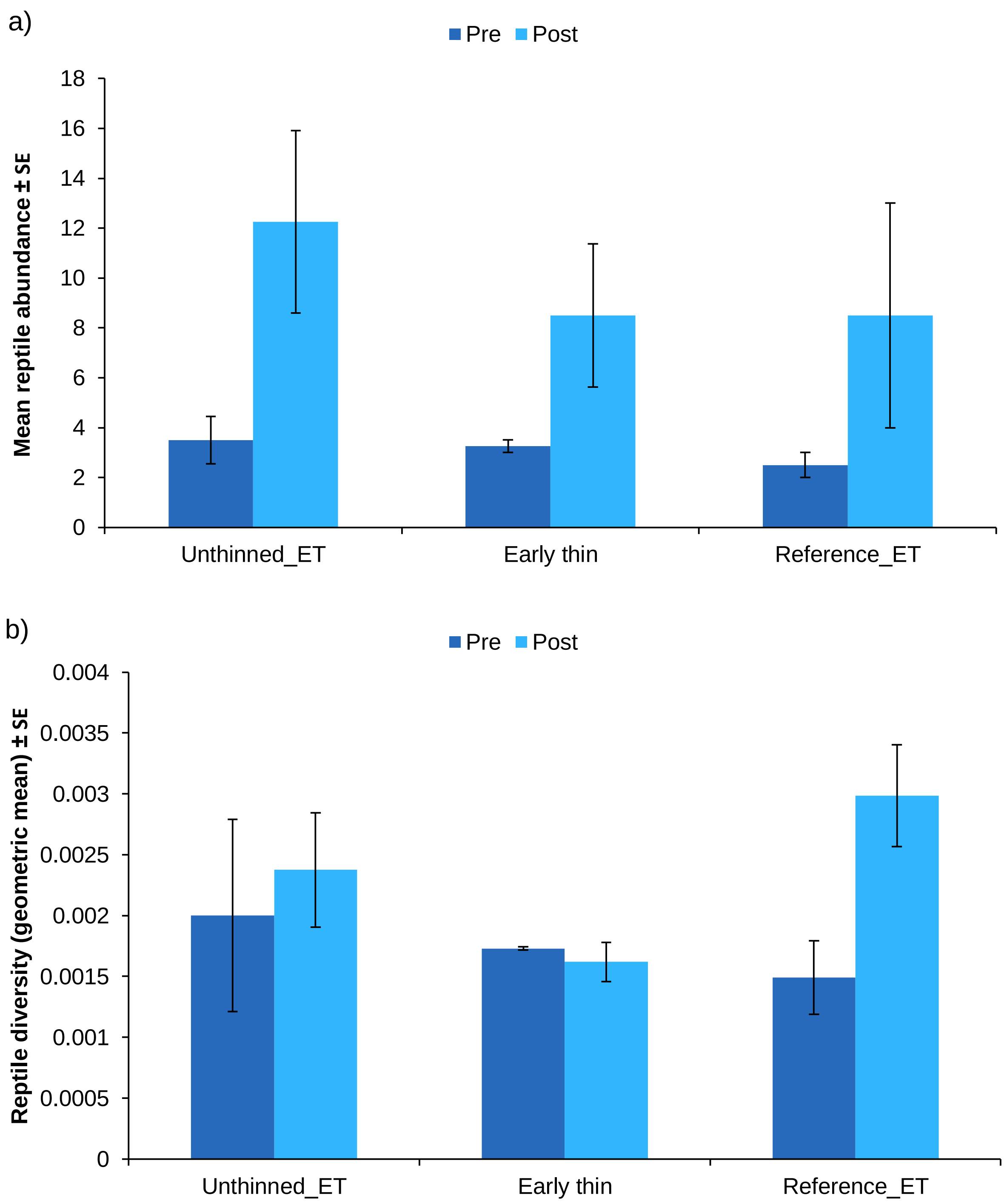


Fig. 3.3.19. Reptile (a) abundance and (b) diversity pre- and post-thinning in early thin control, early thin and early thin reference treatments.

Reptile abundance post-thinning increased in the second thin control (2.5-times greater) and the second thin treatments (4-times greater) though the magnitude of change was greater for the latter (Fig. 3.3.20a). There was no change in reptile abundance from pre- to post-thinning surveys in the second thin reference treatment. Reptile diversity was relatively similar (given overlapping error bars) from pre- to post-thinning in the second thin control, second thin reference treatments and the second thin treatment post-thinning (Fig. 3.3.20b).

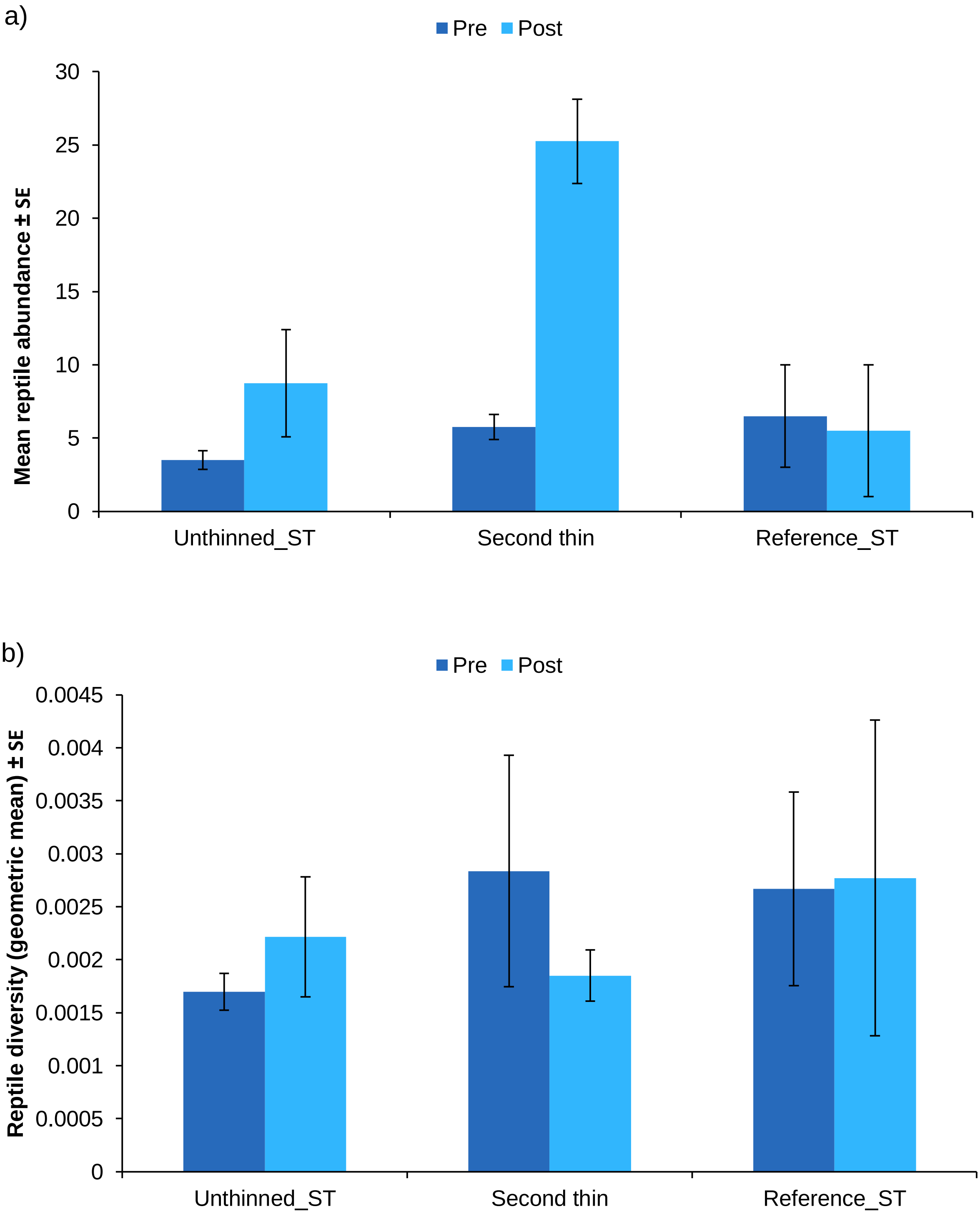


Fig. 3.3.20. Reptile (a) abundance and (b) diversity pre- and post-thinning in second thin control, second thin and second thin reference treatments.

3.3.2.5. Understorey plant diversity

In all, 114 plant taxa were recorded pre- and post-thinning across all treatments (Table S3.3.5). Plant diversity increased from pre- to post-thinning in the early thin control (77 % increase) and early thin treatment (133 % increase), though the magnitude of change was greater for the latter, whereas diversity in the early thin reference treatment remained relatively unchanged (Fig. 3.3.21).

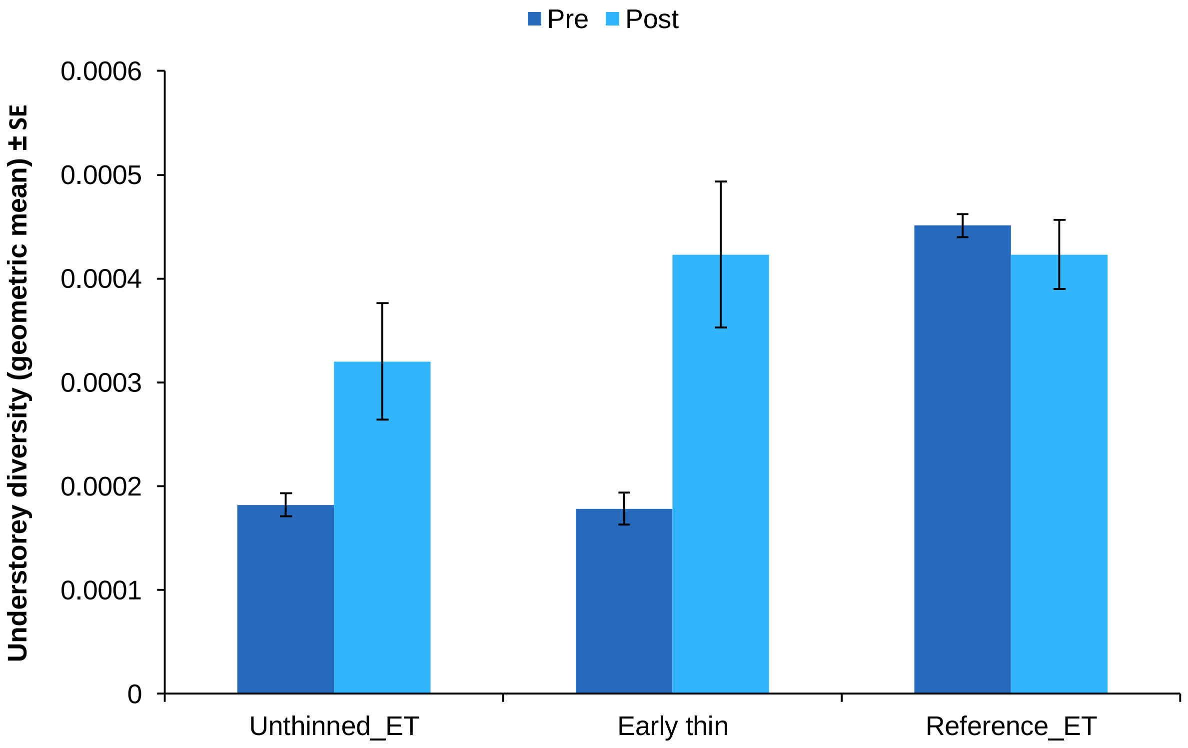


Fig. 3.3.21. Plant diversity pre- and post-thinning in early thin control, early thin and early thin reference treatments.

Plant diversity increased from pre- and post-thinning in the second thin control (130 % increase), second thin (95 % increase) and second thin reference (65 % increase) treatments (Fig. 3.3.22).

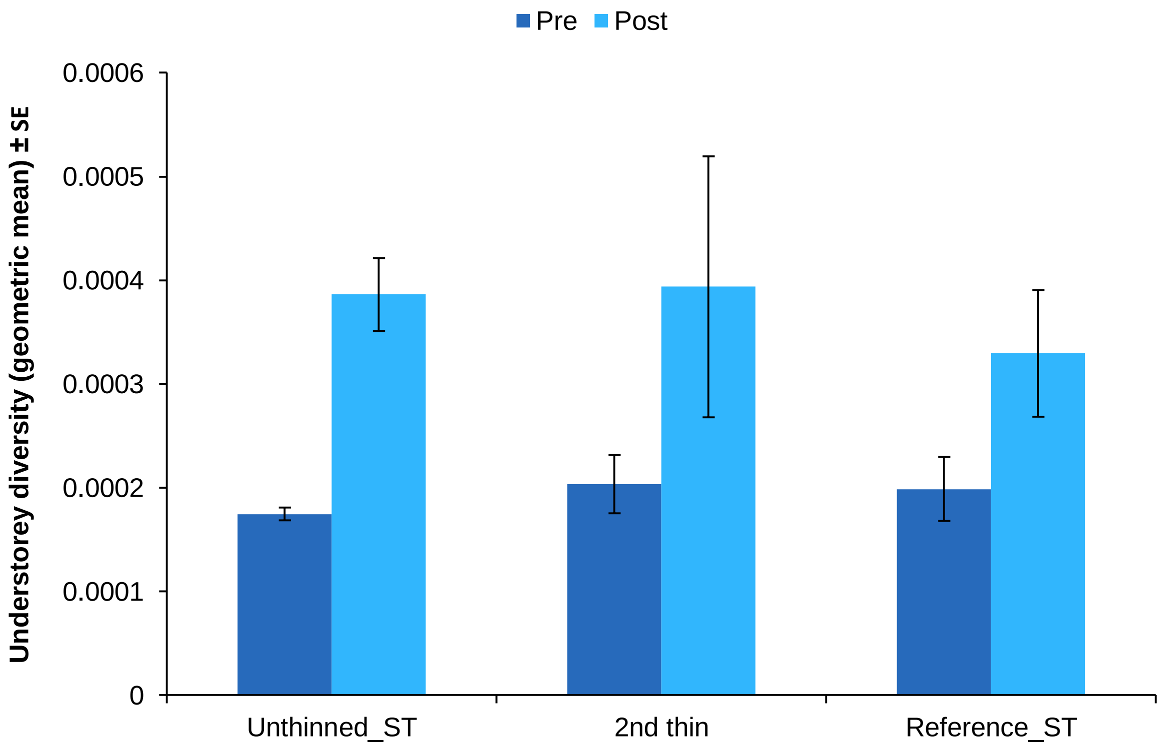


Fig. 3.3.22. Plant diversity pre- and post-thinning in second thin control, second thin and second thin reference treatments.

3.3.2.6. Composite diversity

In all, 230 taxa were recorded pre- and post-thinning across all treatments. Composite diversity increased from pre- to post-thinning in the early thin control (50 % increase) and early thin treatments (90 % increase), though the increase in the thinned treatment was almost double that of the control. Diversity remained stable in the early thin reference treatment (Fig. 3.3.23).

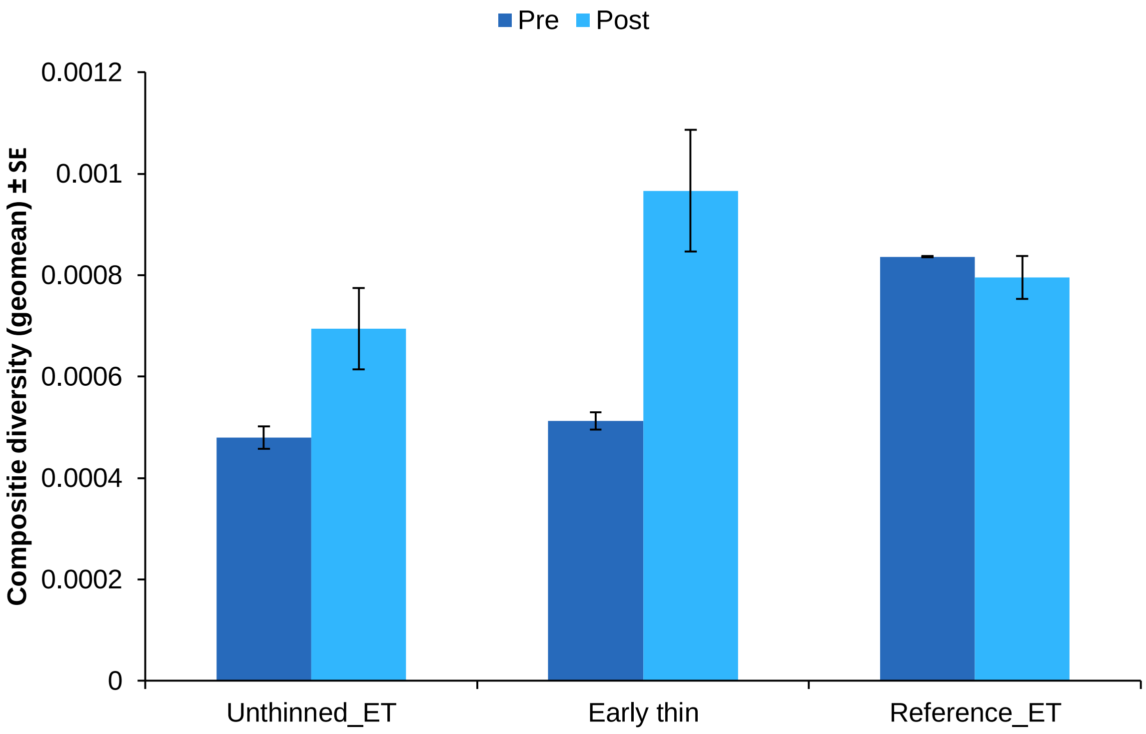


Fig. 3.3.23. Composite diversity pre- and post-thinning in early thin control, early thin and early thin reference treatments.

Composite diversity also increased from pre- to post-thinning in all second thin treatments (75 %, 75 % and 30 % increase in second thin control, second thin and second thin reference treatments, respectively) (Fig. 3.3.24).

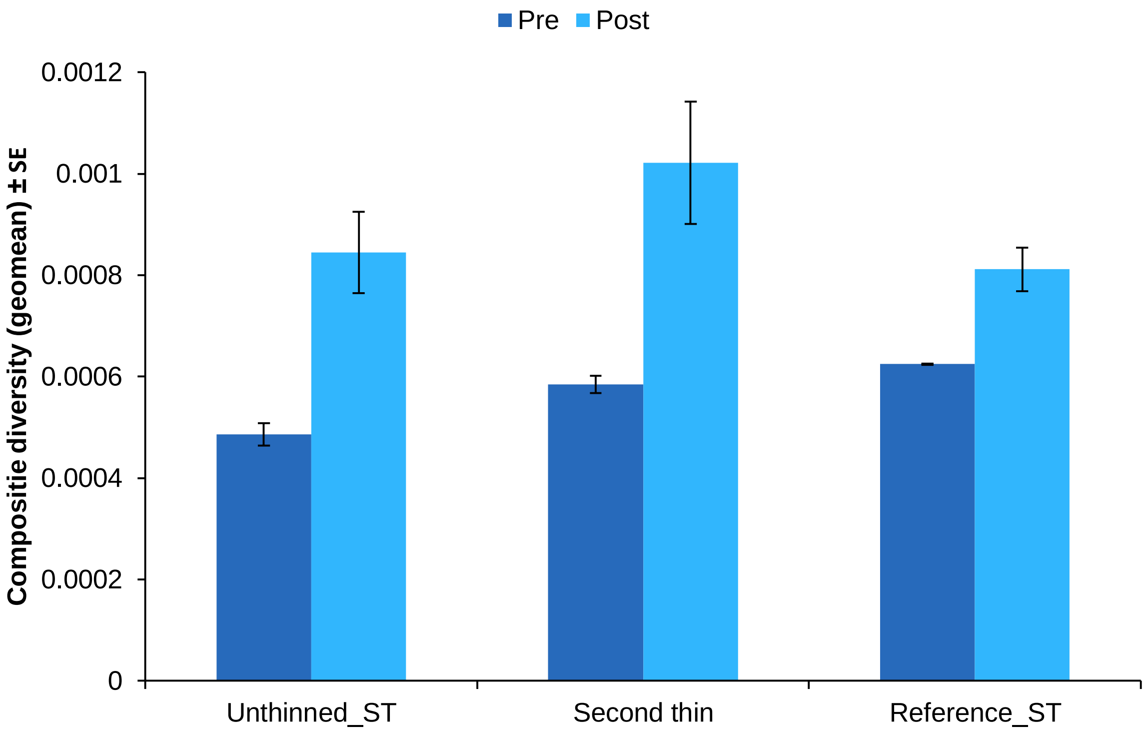


Fig. 3.3.24. Composite diversity pre- and post-thinning in second thin control, second thin and second thin reference treatments.

3.4. DISCUSSION

3.4.1. Vegetation structure and coarse woody debris

As expected, stem density reduced with experimental thinning for both treatment types (i.e., early thin and second thin). Overall, reductions in stem density were greater for the early thin treatment than the second thin treatment. However, the size class of stems that were targeted in early thin and second thin operations varied, with the greatest reductions in stem density recorded for the 5-10 cm size class in the early thin and the >20-30 cm size class in the second thin treatment. These results were expected given the aim of the early thin operation was to reduce the number of small stems in the stand in order to retain the most commercially viable young cypress and allow them to grow under reduced competition for resources. Conversely, second thinning is carried out when cypress trees have grown to a size that can be milled for sawn timber (14-24 cm).

The main tree species affected by thinning operations were commercial species (cypress, buloke and ironbark), with reductions in stem density of 20-70 %. For the early thin treatment greatest reductions were observed for the <5 cm and 5-10 cm size classes. Although stem density of cypress post-thinning was comparable to the early thin reference treatment, buloke stem density was almost 50-times greater than the early thin reference treatment. In the case of buloke, no stems were recorded post-thinning in the second thin treatment.

The density of Acacia stems in the early thin treatment post-thinning increased by ~400 %, with the increase dominated by stems in the <5 cm size class, whereas little change was observed for Acacia post-thinning in the second thin treatment. Furthermore, Acacia stem density was almost double that of the long undisturbed treatment. The positive response of Acacia to thinning was expected given it exhibited a similar response in our chronosequence assessment (See Chapter 2). However, we expect that any dominance from Acacia regrowth after thinning will be relatively short-lived and future resampling of experimental plots will allow Acacia cover to be tracked over time.

Thinning did not appear to affect the density of eucalypts in the early and second thin treatment, with a great diversity of size classes recorded. It has been suggested that high densities of *Allocasuarina luehmannii* and *Callitris* spp. may act to competitively exclude *Eucalyptus* spp. (see chapter 2 and Waters et al. 2018). As our experimental plots continue to be resampled into the future, data will become available to examine this hypothesis.

The density of dead stems was not affected by early thinning, with similar densities recorded pre- and post-thinning. However, a number of fallen dead trees (10-20 cm dbhob) were observed at one of the experimental thinned plots (ET2). Furthermore, a reduction in the density of dead stems (mostly buloke) was recorded for the second thin treatment post-thinning, with greatest reduction seen in the 5-10 cm and 20-30 cm size classes. Trees with these characteristics were found to be used by maternity colonies of bats during the radio-tracking component of our project (see Chapter 4). Therefore it is critical that these trees are retained during thinning operations, particularly in clumps as they can often form a network of roost trees between which bats regularly switch.

Post-thinning, the density of stumps in the early thin and second thin treatments increased as expected. The increases in number of stumps for particular size classes in thinning treatment (early thin or second thin) reflected the size classes of stems targeted by both types of thinning operations. For the early thin treatment, there was also a small increase in the number of stumps in the >40-50 cm size class and this represented the removal of a small number of ironbarks during the thinning operations.

Ground cover across all treatments was was dominated by litter, with little plant cover and this did not change with thinning. Low levels of cover were also recorded for cryptogams and fine woody debris (branches and twigs), though there was a small increase in the cover of fine woody debris (FWD) in the early thin treatment and a doubling in the second thin treatment. These results are consistent with findings from our chronosequence assessment where there was no effect of thinning on ground cover. Moderate levels of herbivores (native and introduced) may be associated with a negligible change in plant and cryptogam cover and further research is required to examine impacts of herbivores on ground cover.

Understorey cover responded positively (increased by 8-times) to thinning in the early thin treatment, whereas no change was detected in the second thin treatment. This result likely reflects the lesser effect of thinning in the second thin treatment as treated sites were relatively more open than early thin sites prior to thinning. Thinning reduced midstorey and sub-canopy cover by approximately 50 % in the early thin treatment. This level of cover was comparable to the early thin reference treatment in the case of midstorey cover. This reduced level of cover resulted in a marginal increase in reptile diversity in the early thin treatment. Sub-canopy cover of the early thin treatment was 45 % lower than the early thin reference treatment. This strata was also reduced significantly in the second thin treatment, reflecting that this strata is a key target of second thin operations. Canopy cover remained unchanged with thinning in both the early thin and second thin treatments.

The effect of thinning on the volume of CWD varied between thinning operations. There was no effect of thinning on CWD volumes in the early thin treatment, although CWD volumes were still 2-3 times greater than the early thin reference treatment. In the second thin treatment, the volume of CWD was 3-4 times greater than the second thin control and second thin reference treatments. Thinning provided a pulse in hollow-bearing CWD post-thinning in the second thin treatment, even though thinnings were removed from the forest as part of the commercial operation. This increase in hollow-bearing CWD may be associated with the reduction in dead tree density, particularly larger dead trees in the second thin treatment. Reptiles were a fauna group that responded positively to thinning in the early thin treatments and the provision of hollow CWD via thinning may provide habitat that is otherwise depleted in the unthinned forest.

3.4.2. Biodiversity

Bat activity and diversity responded positively to thinning in the early thin treatment, but exhibited a neutral response in the second thin treatment. The significant positive response to thinning in the early thin treatment likely reflects the dramatic reduction in stem density in this treatment, relative to the second thin treatment. Bats are known to respond positively to reduced stem densities and a threshold of stem density has been identified above which bat activity declines (Blakey et al. 2017).

Birds responded positively to both types of thinning operations, though the effect size was greatest for the early thin treatment. Again, the greater effect of thinning in the early thin treatment likely reflects the dramatic reduction in stem density in this treatment relative to the second thin treatment. The second thin treatment was generally more open (lower understorey cover and fewer stems) and may provide a lower diversity of habitat structure for birds. Despite this, noisy miners were infrequently detected in this treatment, with the species detected at two thinned sites and one control site in Baradine State Forest. Nevertheless, both types of thinning did result in greater bird diversity than unthinned forest.

Non-volant native mammal diversity exhibited a neutral response to thinning in the early thin treatment, whereas diversity tripled post-thinning in the second thin treatment. Despite an increase in the second thin treatment, driven primarily by macropods, diversity was also two-times greater in the second thin reference treatment. The low level of ground cover across the study area may not permit a rapid response of mammals to thinning despite the availability in thinned areas of greater volumes of hollow-bearing CWD that ground mammals may use for shelter. The presence (though at low levels of activity) of introduced predators may also diminish the response of ground mammals that are thought to be relatively rare in the study area (see Chapter 2 and Date and Paull 2000).

There was a trend for an increase in reptile abundance post-thinning, though the increase was recorded across all treatments except the second thin reference treatment, possibly reflecting heavy rainfall in the 6-months prior to post-thinning surveys. Reptile diversity however showed a neutral or negative response to thinning in the second thin treatment, possibly because this treatment was relatively open prior to thinning.

Plant diversity increased in treatments and controls post-thinning in both early thin and second thin treatments, though the magnitude of change in the second thin treatment was lower than the second thin control treatment. Similarly for overall (composite) diversity, the response to thinning was positive, though there was no difference between second thin and second thin control treatments which had more than double the diversity of the second thin reference treatment.

3.4.3. Management recommendations

Together these results demonstrate that immediate effects of thinning are generally positive or neutral for habitat values and biodiversity, though this varies with the type of thinning operation (early thin or second thin). This suggests that the mechanical disturbance created during the thinning operations had limited immediate impacts on the biodiversity present. Though the immediate effects of thinning were mostly neutral or positive, species from various groups of biodiversity will exhibit species-specific responses (see Gonsalves et al. in press and Chapter 2). Furthermore, unthinned regrowth was found to represent similar value to thinned treatments for some components of habitat structure and diversity. As such, it is important that management at the landscape-scale maintains a mosaic of thinned and unthinned regrowth. It is critical to resample our experimental plots into the future when biodiversity will have had more time to respond to the structural changes and to track the changing habitat structure.

4. DOES THINNING HOMOGENOUS AND DENSE REGROWTH BENEFIT BATS? EVIDENCE FROM RADIO-TRACKING, ULTRASONIC DETECTION AND TRAPPING

4.1. INTRODUCTION

Renewal ecology is a newly proposed concept that promotes both the creation and/or enhancement of landscapes that support biodiversity *and* provide ecosystem services for human communities (Bowman *et al*. 2017). Renewal ecology recognizes the need to harmonize biodiversity with humans, for the benefit of both, and it emphasises active management, using targeted interventions. It differs from restoration ecology that typically aims to return ecosystems to a previous, indigenous, state (Montoya et al. 2012). There are many forms of active management that potentially fit this concept, including certain forestry activities.

Timber production forests throughout the world not only provide a valued timber resource, but are important for maintaining biodiversity and irreplaceable ecosystem services (Beaudry *et al*. 1997; Gustafsson 2012). Silvicultural thinning is often employed in dense secondary forest regrowth to reduce tree competition and encourage faster growth of mature, harvestable trees (Law *et al.* 2016a). This is particularly the case when the secondary regrowth that dominates stands post-harvesting comprises trees which are often uniform in age and size, creating a high density of stems with few canopy breaks (Bauhus *et al*. 2009). In addition to influencing tree growth, thinning has both direct and indirect effects including altering physical structure and composition of vegetation and modifying soil properties and microclimates (see Verschuyl *et al*. 2011 for review). Given that biological responses to thinning are complex, with changes being detrimental to some taxa and beneficial to others (Verschuyl *et al.* 2011; Fuller 2013; Eyre *et al*. 2015; Blakey *et al*. 2016), it is essential to more fully explore the effects of this practice on a range of taxa.

Insectivorous bats are a diverse group of mammals whose morphology, including body size and wing shape, is often related to vegetation structure (McKenzie and Rolfe 1986; Aldridge and Rautenbach 1987; Norberg and Rayner 1987; Crome and Richards 1988; Bullen and McKenzie 2001; Hanspach *et al*. 2012). In particular, one guild of bats, closed-space bats (Denzinger and Schnitzler 2013), that possess specialised traits favouring slow, manoeuvrable flight within vegetation clutter, especially in forests, is considered especially prone to extinction (Jones *et al*. 2003; Safi and Kerth 2004). While these bats can also forage within open space (Brigham *et al.* 1997; Pavey *et al*. 2001), they are often forest dependent and are predicted to be particularly sensitive to vegetation loss and potentially altered vegetation structure, for instance due to thinning of dense forests.

Forest thinning can be beneficial for a range of bat species by reducing clutter levels and presumably increasing foraging efficiency by echolocation (Humes *et al*. 1999; Patriquin and Barclay 2003; Law *et al*. 2016a; Blakey *et al*. 2016). Yet, closed-space species typically have higher activity levels in forests with higher stem densities (Müller *et al.* 2012; Fuentes-Montemayor *et al*. 2013; Blakey *et al.* 2017), highlighting the need for a better understanding of their response to thinning. Despite roosts being a critical part of the bat life cycle (Kunz and Lumsden 2003), it is poorly understood how forest thinning affects roost selection, especially for closed-space bats. One threatened, closed-space species (*Nyctophilus corbeni*) prefers roosting where total stem density and the density of dead stems is high and appears to avoid roosting in thinned areas (Law *et al.* 2016b). In comparison, a less manoeuvrable edge-space species (*Lasiurus borealis*) roosts in open conditions created by thinning (Perry *et al.* 2007).

Our study aimed to contrast flight activity and roost selection by a small suite of bat species to thinning in a dry, but dense, forest of white cypress pine *Callitris glaucophylla* that had been recently experimentally thinned in four separate patches. *Callitris glaucophylla* is geographically widespread in Australia and has history of silvicultural thinning in parts of its distribution. The species is slow growing and long-lived, tolerating intense intra-specific competition at high densities (Lacey 1972; Lacey 1973; FCNSW 1988), with little evidence of self-thinning (Thompson and Eldridge 2005a) for up to 200 years in low rainfall environments (Read 1995). We chose to radio-track multiple bat species to contrast the roost preferences of different closed-space species that possess differing conservation status (*Nyctophilus corbeni, N. gouldi* and *N. geoffroyi*), with one small, edge-space species (*Vespadelus vulturnus*). We predicted the three closed-space species would prefer to roost and be more active in dense unthinned forest (Law *et al.* 2016b), but we also expected some partitioning of roost tree attributes between the species. In contrast, the small, edge space species was predicted to roost and be more active in the thinned forest (Blakey *et al*. 2017). We based our assessments on a combination of radio-tracking, ultrasonic detection and systematic trapping. Trapping with an acoustic lure (Hill and Greenaway 2005) was used to assess the activity of *Nyctophilus* in the thinned-unthinned mosaic, because it is currently not possible to distinguish echolocation calls of these three *Nyctophilus* species (Reinhold *et al*. 2000).

4.2. METHODS

4.2.1. Study species

The south-eastern long-eared bat *N. corbeni* is listed nationally as a threatened species in Australia (Vulnerable, *Environment Protection and Biodiversity Conservation Act 1999*). In contrast, two sympatric members of the *Nyctophilus* genus (*N. gouldi* and *N. geoffroyi*) are abundant and widespread (Turbill and Ellis 2006; Churchill 2008), though all are morphologically similar, belonging to the same closed-space guild with steep, broad band echolocation calls. The main threatening processes for *N. corbeni* are considered to be extensive loss of habitat through clearing for agriculture, grazing and altered fire regimes (Duncan et al. 1999), with different forestry activities having contrasting effects (Law et al. 2016b). The fourth study species, the little forest bat *V. vulturnus*, weighs just 4 g and it was selected because individuals were expected to forage within a 1 km radius (Campbell et al. 2005; Law et al. 2011; Gonsalves et al. 2013b), thus making the species highly suited, on a local scale, to revealing relative use of thinned and unthinned forest. It is a common species belonging to the edge space guild with relatively slow and manoeuvrable flight due to low wing aspect ratio and loading (O'Neill and Taylor 1986; Rhodes 2002). These ecomorphological traits indicate it is agile enough to follow the edges provided by well-separated crowns of trees, but less likely to regularly forage in very dense unthinned forest (Law et al. 2011; Blakey et al. 2017). All research was approved by the DPI FCNSW Animal Ethics Committee (Authority Number 21/15-17).

4.2.2. Study area

The study was undertaken in State Forests of the Pilliga area (-30°44'45', 149°6'41') in the Southern Brigalow Belt bioregion of north-west New South Wales (NSW), Australia. The area experiences a hot and dry climate, with the mean annual rainfall for nearby Baradine being 633 mm (1944-98). The Pilliga area has a long history of European use and consequently considerable changes to forest structure have been inferred. Rolls (1981) and van Kempen (1997) suggest that much of the Pilliga area was originally open woodland with a similar plant species composition to today, but with different extent and density. Increasing tree density is thought to have displaced sheep grazing ventures. Subsequently, to prevent unrestricted exploitation of timber resources, the first forest ranger was appointed in 1877 and the first local forestry office opened in 1911 (Forestry Commission of NSW 1986). In particular, *Callitris* regeneration over this time appears to have been strongly influenced by a combination of changed burning regimes, introduction of rabbits and interactions with drought and flood years.

The vegetation today is characterised as a forest that varies in tree height with soil type. White cypress pine *Callitris glaucophylla* and Narrow-leaved Ironbark *Eucalyptus crebra* dominate the tree species (Binns and Beckers 2001; Whipp et al. 2012). Dense stands of *C. glaucophylla*, black cypress pine *C. endlicheri*, buloke *Allocasuarina luehmannii* and *Acacia* spp. are scattered throughout the forest, but are treated silviculturally to maximise tree growth for timber production (Forestry Commission of NSW 1986). Locally, our study area was dominated by dense, unthinned stands of regrowth *C. glaucophylla*. In June-July 2016, experimental thinning was undertaken with small machinery at four 12 ha replicate patches as part of a broader study on the effects of thinning on biodiversity. Thinning at these patches was considered 'early' thinning that targeted 4-6 m tall cypress and aimed to achieve tree spacing of approximately 6 m (280 stems ha^{-1}). Each replicate was separated from the nearest thinned patch by at least 200 m. Four designated unthinned control patches, each 12 ha in size, were used for comparison.

4.2.3. Bat activity

Bat activity was assessed in November (spring) 2016 and March (autumn) 2017 in the thinned-unthinned mosaic using ultrasonic Anabat detectors (Titley Scientific – Brendale QLD) for two nights in the centre of each thinned or unthinned replicate. In the autumn sampling session, two detectors were positioned ~100 m apart within each treatment, while a single detector was located in the centre of each treatment in the spring sampling session. Detectors were positioned on the ground but with microphones set at a height of 1 m and oriented at a 45° angle to the ground to face gaps within vegetation at each site. This was done to ensure detectors at each site were sampling a similar volume of airspace. All files were analysed using automated software, AnaScheme (Adams et al. 2010), in association with a key for bats of the Pilliga (unpublished – B. Law). Only bat passes with more than three valid pulses, minimum of six data points and model quality of 0.8 were analysed by AnaScheme. Because multiple bat species may call simultaneously, calls were assigned to a species only if >50% of pulses within the sequence were attributed to that species and only passes with a minimum of three pulses classified to the same species were identified. All calls that could not be assigned to a bat taxon were included in counts of total bat activity but were labelled as 'unidentified'. Since it is not possible to distinguish between calls of certain species (e.g., *N. corbeni*, *N. geoffroyi* and *N. gouldi*, or *Mormopterus planiceps* and *M. petersi*, or *Scotorepens greyii* and *Scotorepens* sp.), these calls were assigned to a species group (e.g., *Nyctophilus* spp., or *Mormopterus planiceps/petersi*, or *Scotorepens greyii*/sp.) by AnaScheme. For each detector and each night, the number of bat passes for each species (nightly activity) and total bat activity (all identified and unidentified bat calls) was tabulated.

In November 2016/17, we used harp-traps (Tidemann and Woodside 1978) to quantify activity in the thinned-unthinnned mosaic for the three species of *Nyctophilus*, given that their echolocation calls cannot be identified to species-level. To reduce bias from trapping in the open thinned forest and dense, unthinned forest we used an acoustic lure (Sussex Autobat; Hill and Greenaway 2005) with the aim of luring foraging bats directly into the harp-trap from a distance of < 50 m. We systematically trapped the thinned-unthinned mosaic using multiple pairs of traps, one with an acoustic lure and one without. One trap pair was positioned in each of the four thinned and four unthinned replicates, typically for two nights, with the acoustic lure rotated between traps each night. Traps were set near the centre of replicate patches to avoid luring individuals from outside the treated area. Additional trapping was undertaken on any flyways that were present, but these results are not presented here. The number of bats trapped per trap-night on flyways or away from flyways was summed for each thinned and unthinned replicate.

4.2.4. Radio-tracking

To maximise the opportunity of detecting bats roosting and foraging in thinned patches (which were proportionately small in comparison to the surrounding untreated forest), only bats trapped within thinned forest were radio-tagged. If bats were trapped in the first half of the night, they were processed immediately and released on the same night of capture. All other bats were retrieved the following morning and released the following evening. Since the major aim of this study centred on maternity roost preferences, female bats that were pregnant or lactating were selected for radio-tracking. Radio-transmitters with an aerial length of 12 cm (Holohil, Model LB-2X, 0.27 – 0.42 g – < 6.8 % of body mass) were attached mid-dorsally (between shoulder blades) to bats using an adhesive (Urobond, Urocare). Pregnant females weighed 3.75 g more than lactating females, but we did not observe difficulty flying or use of unusual roosts with the additional mass of transmitters. After radio-transmitters were attached, bats were placed into calico bags and held for a minimum of 10 mins prior to release at the site of capture.

Bats were radio-tracked in the maternity season of two years (November 2016/17). Roost trees were located by homing in on the signals of radio-tracked bats. Once a roost tree was identified, a GPS waypoint was taken for the tree. An inspection of the roost tree was made using binoculars in an attempt to identify the roost type (e.g., hollow, fissure, etc.). For each roost tree, diameter at breast height over bark (dbhob, 1.3 m) was measured while tree height was visually estimated. Each roost tree was also assigned an age-class (0-8; following Gibbons et al. 2000), hollow abundance (0-4; 0=no apparent hollows, 4=abundant hollows) and canopy cover was estimated visually or using the 'Habitapp' Android application. An emergence count was conducted at roost trees on dusk to confirm hollow-type and to count the number of bats that exited the roost. Counts usually continued for 30 mins after the exit of the first bat, which generally took place when there was enough ambient light to discern flying bats.

To quantify the amount of time radio-tracked bats spent in thinned plots, two Australis 26K scanning receivers (Titley Scientific, Brendale QLD) fitted with remote RF data loggers (Titley Scientific, Brendale QLD) and omni-directional whip antennas (Titley Scientific, Brendale QLD) were set in the centre of two thinned plots (ET2 and ET4) in 2017. Antennas were secured 3 m above the ground and on a tree. Scanning receivers actively scanned through radiofrequencies of tagged bats and RF data loggers logged the presence of a signal ("Entry") every four seconds if a pre-determined signal strength was achieved and a minimum of 3 pulses was detected in that period. If a signal was not detected, this was also logged ("Exit"). The time spent by each radio-tracked bat in each thinned plot was tallied and averaged across nights for each bat. Gain settings on receivers were calibrated using an active transmitter to record bats within 200 m of the logger. This was done to minimise the chance of logging bats that were flying outside of thinned patches.

4.2.5. Roost tree availability

We estimated roost tree availability at a local scale and in relation to the surrounding thinned-unthinned mosaic. At the local scale, we estimated tree density around each roost tree by measuring the distance to the nearest tree (>10 cm Diameter at breast height over bark, dbhob) in each quadrant (Pollard 1971). The point quarter technique was then used to calculate the local density of trees (including those with a hollow) for all roosts pooled across years. The standard error for the tree density estimate was calculated following Pollard (1971). At the patch scale, we calculated stem density (plus hollow stem density) from three 20 x 10 m plots located along a 200 m transect in each thinned or unthinned replicate. Both the number of and dbhob, of all trees and shrubs within each plot was measured and allocated to one of four categories (< 10 cm; >10 cm to < 30 cm; >30 to < 50 cm; and >50 cm). The number of dead trees was also recorded within each plot but not allocated to size classes. These data were used to calculate mean tree density for thinned and untinned treatments.

4.2.6. Data analyses

A canonical analysis of principal coordinates (CAP) was used to examine and contrast characteristics of trees used for roosting and those available in the local neighbourhood around roost trees. Prior to analysis, a square-root transformation was applied to continuous variables (tree dbhob and height) and all data were then normalised. A resemblance matrix (Euclidean distance) was constructed and a CAP analysis was undertaken. Vectors representing tree characteristics were overlaid onto the CAP output. CAP analyses were conducted using the PERMANOVA+ add-on package in Primer 6 (PRIMER-E Ltd, Plymouth, UK; Clarke and Gorley 2001).

The number of captures of focal species (*N. corbeni*, *N. geoffroyi*, *N. gouldi*, *V. vulturnus*) in the unthinned-thinned mosaic was compared between thinning treatments using Wilcoxon signed-rank tests after accounting for the number of trap nights in each treatment.

Bat activity of focal species (*Nyctophilus* spp. and *V. vulturnus*) and total bat activity (all species combined) was compared between treatments and season using a generalized linear mixed model (GLMM). Thinning treatment, season and the interaction of treatment by season were fixed effects while site was used as a random effect. Response variables (bat activity) were $\log_{10}$-transformed prior to analysis.

Wilcoxon signed-rank tests and GLMMs were run using SPSS 19.0 (IBM).

4.3. RESULTS

4.3.1. Roosting

In all, 22 bats representing four species (*Nyctophilus corbeni*, *N. geoffroyi*, *N. gouldi* and *Vespadelus vulturnus*) were radio-tracked across both years. Of these, 21 were either in the late stages of pregnancy or were lactating, while a single male was also tracked (Table 4.3.1). However, no roosts were located for the male so results represent roosting preferences of maternity females.

Table 4.3.1. Summary of trapping and roost-tree data collected for four species during radio-tracking over two maternity seasons (2016 & 2017).

Trapping/roost data	2016	2017
No. of bats trapped	141	94
No. of bats tracked	12	9
Species tracked	Nc (3), Nge (2), Ngo (4), Vv (3)	Nc (3), Nge (2), Ngo (2), Vv (2)
No. of roost trees	21	32
Roost tree (cm dbhob)	27.3±1.9	31.6±4.4
Roost tree height (m)	15.1±1.4	11.9±1.2
Dead roost trees	76.2	93.1

Nc = *Nyctophilus corbeni*, Nge = *Nyctophilus geoffroyi*, Ngo = *Nyctophilus gouldi*, Vv = *Vespadelus vulturnus*. Trapping data includes trap sites that were located in experimental thinned and unthinned treatments, including traps on flyways that were not paired for comparison with traps set with an acoustic lure.

A total of 52 roost trees were located across both years of radio-tracking. Of these 24, 18, 6 and 4 were used by *N. gouldi*, *N. corbeni*, *V. vulturnus* and *N. geoffroyi*, respectively. Across both years, >75 % of roost trees were dead (Table 4.3.1; Figure 4.3.1).

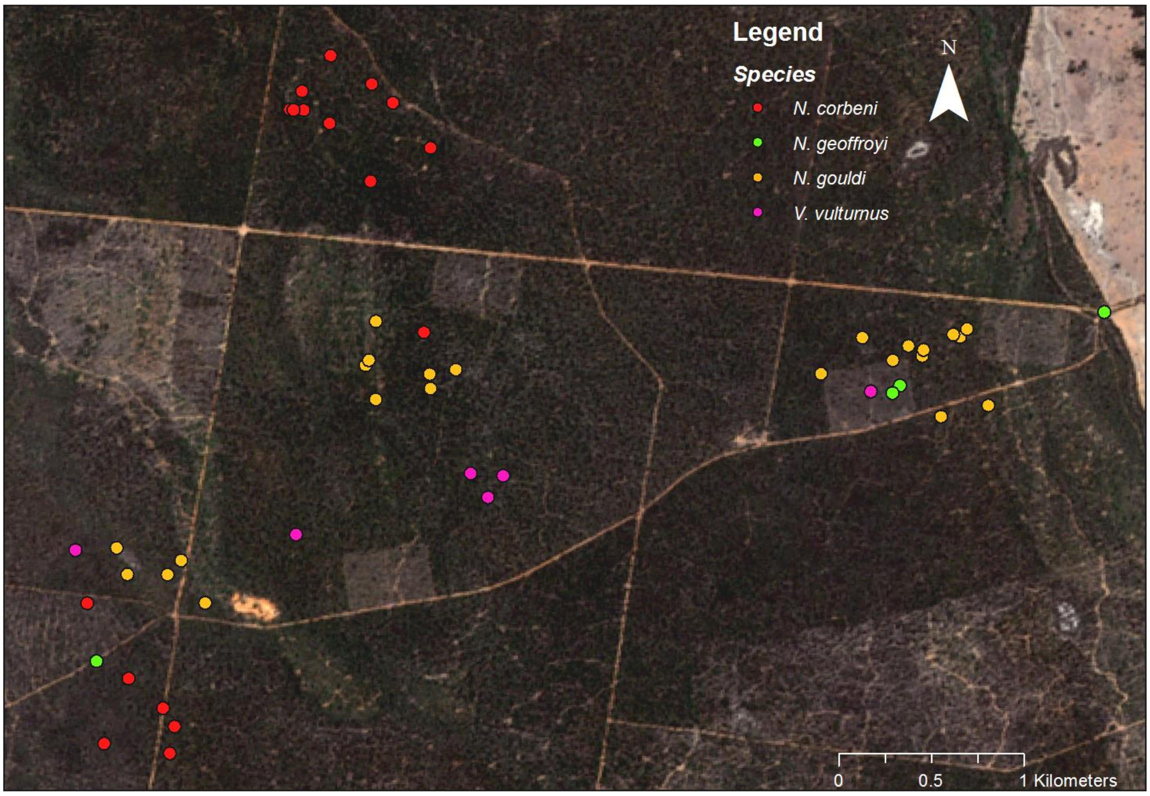


Fig. 4.3.1. Satellite image (Sentinel-2) showing the location of roost trees and four 12 ha thinned blocks where radio-tagged bats were trapped. Larger areas of commercial timber harvesting are also apparent in the south-east and north-west.

Buloke and eucalypts together represented more than two-thirds of roost trees for each species (Fig. 4.3.1) and these were typically dead (>75 %). For *N. corbeni*, living and dead eucalypts represented most roost trees followed by dead buloke, cypress and other unknown dead trees. Only a small number (n=4) of *N. geoffroyi* roosts were located before transmitters were detached. Three roost trees were buloke whereas one individual roosted in a living Pilliga Box in a riparian zone on the forest-paddock interface <1 km from the capture site. For *N. gouldi*, roosts were equally spread across dead buloke and living and dead eucalypts, with a small percentage of dead tree roosts unable to be identified to species. Dead eucalypts represented most roost trees of *Vespadelus vulturnus*, with dead buloke and cypress also used (Fig. 4.3.2).

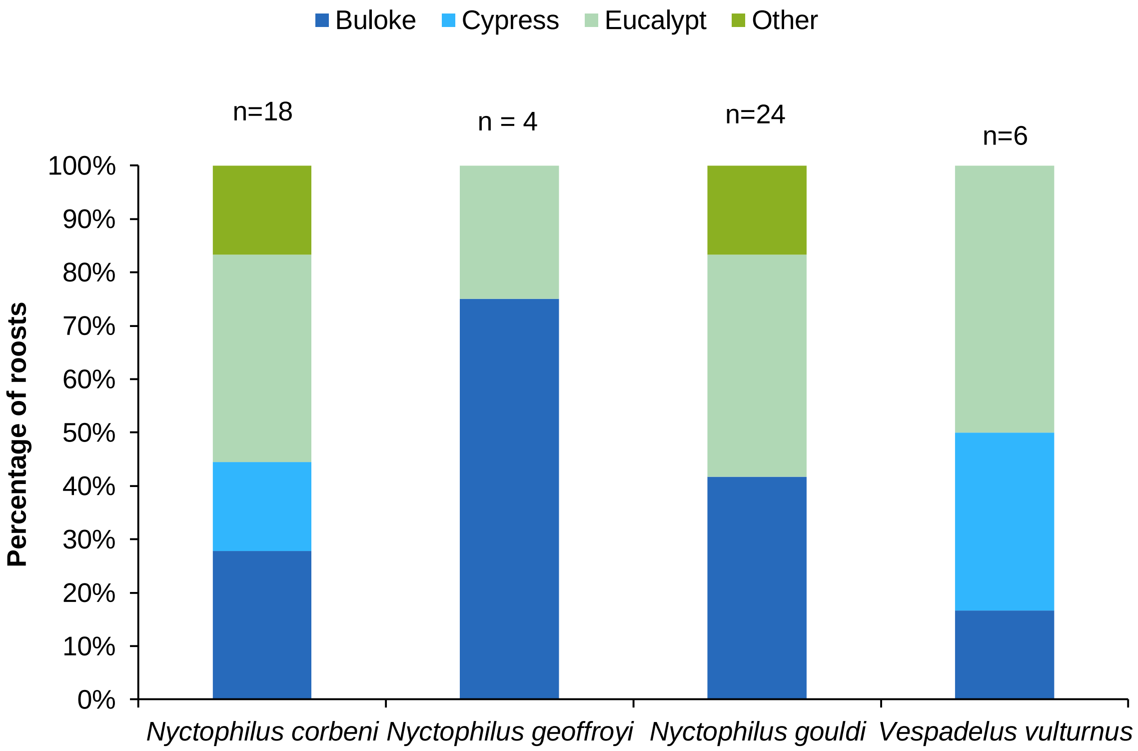


Fig. 4.3.2. Percentage of roost trees of each species that were represented by buloke, cypress, eucalypts and other trees (2016 & 2017).

For all species, ≥ 50 % of all roosts were located in a hollow or a fissure (Fig. 4.3.3). *Nyctophilus corbeni* and *N. gouldi* also roosted under bark (Fig. 4.3.3). *Vespadelus vulturnus* roosts were exclusively in hollows (Fig. 4.3.3).

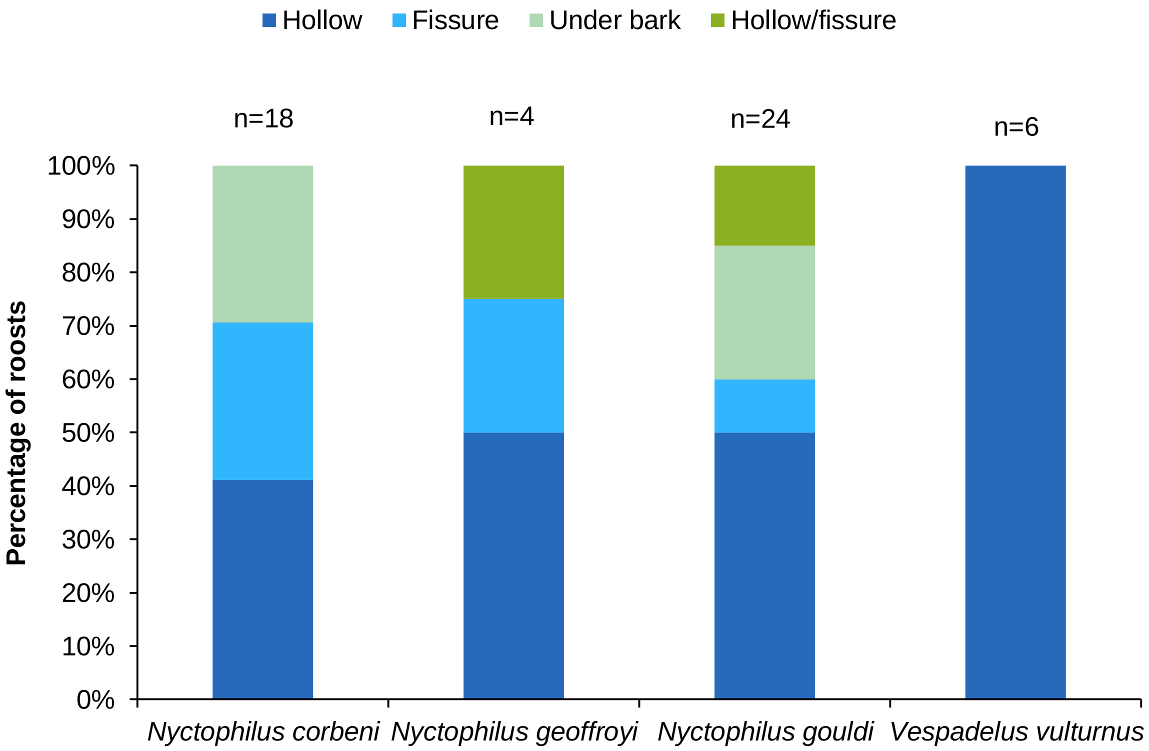


Fig. 4.3.3. Percentage of roosts of each species that were located in a hollow, fissure, under bark and hollow/fissure (classification uncertain) (2016 & 2017).

A CAP revealed that roost trees were distinct from available living trees in the local area (Fig. 4.3.4). Although available living trees covered a broad range of heights, sizes (dbhob) and extent of canopy cover, they all generally lacked hollows and received low scores for degree of senescence (Figure 4). Dead available trees were similar to a subset of trees used for roosting by *Nyctophilus* spp. and *V. vulturnus* and were characterized by a greater degree of senescence and more hollows than living available trees (Fig. 4.3.4). A subset of roost trees were larger (>dbhob) and contained more hollows than available dead trees. Roost trees used by all species were very similar based on measured attributes, indicating considerable overlap between species.

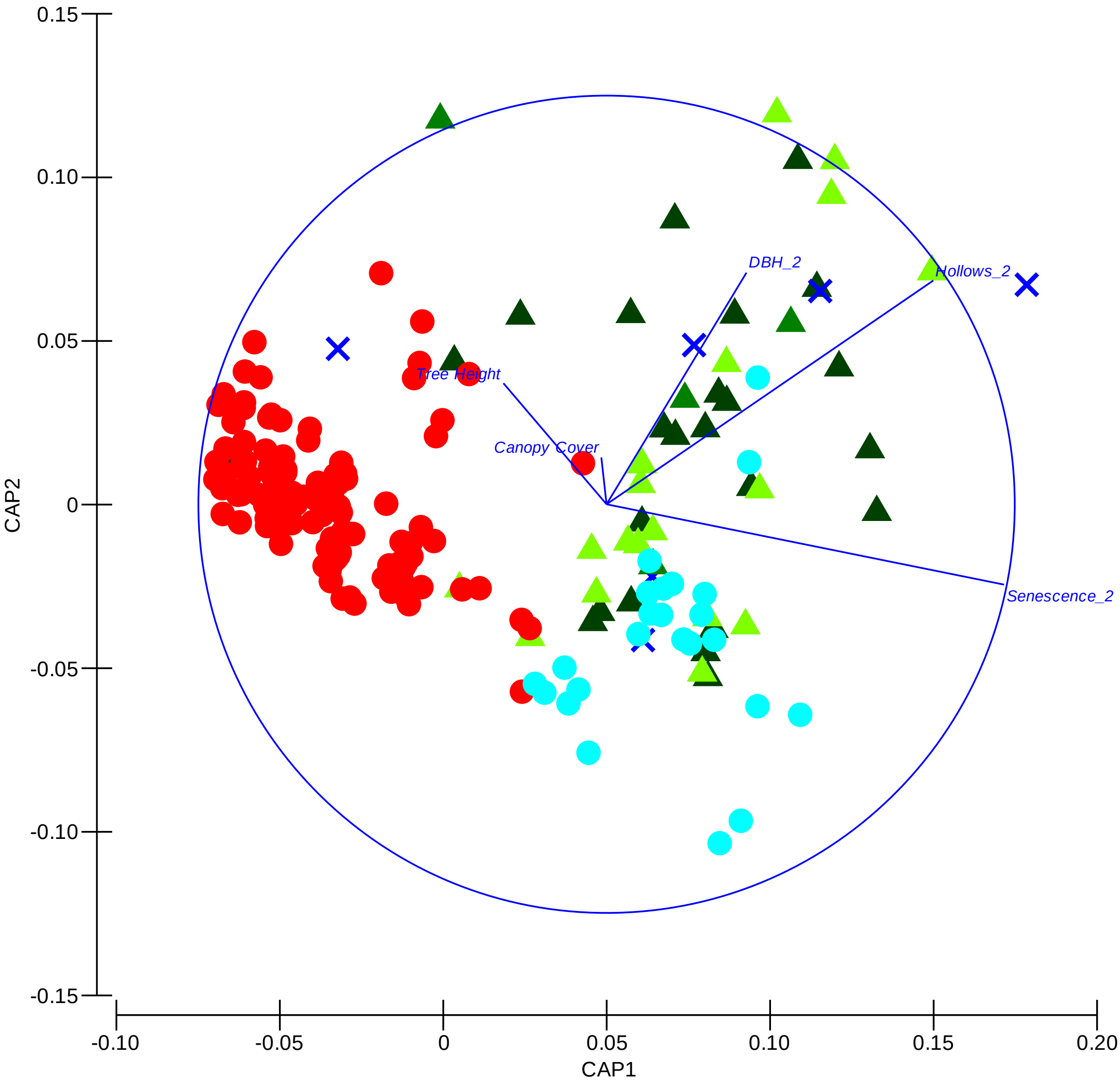


Fig. 4.3.4. Canonical analysis of principal coordinates illustrating characteristics of trees used for roosting and local available trees. Solid red and cyan circles indicate living and dead available trees, respectively. Blue crosses indicate trees used for roosting by *V. vulturnus*. Light green, dark green and black triangles indicate trees used for roosting by *N. corbeni*, *N. geoffroyi* and *N. gouldi*, respectively.

4.3.2. Stem density

Although all tagged bats were caught in thinned treatments, only 6 % of roosts were located there (n=52). This was reflected in a relatively high stem density surrounding roost trees. Although there was considerable overlap in roost tree characteristics of *Nyctophilus* spp. (Fig. 4.3.4), areas that *N. geoffroyi* roosted in tended to be more open (thinned patch, riparian zone adjacent to paddock and along a 4WD track) than those used for roosting by conspecifics. As a result, stem density in patches around roost sites was calculated for *Nyctophilus* spp. with and without data for *N. geoffroyi* roosts. The density of stems (>10 cm dbhob) around *Nyctophilus* spp. roosts was 366±4 stems ha^{-1} (382±4 stems ha^{-1} excluding *N. geoffroyi* roosts). The density of stems surrounding roosts was intermediate to stem density in thinned (358±89 stems ha^{-1}) and unthinned (463±76 stems ha^{-1}) sites. However, this ignores stems < 5 cm dbhob, which had a much higher density in unthinned (3654±199 stems ha^{-1}) than thinned (2475±230 stems ha^{-1}) sites (see Fig. 4.3.5). The density of hollow stems (>10 cm dbhob) in patches around roost trees of *Nyctophilus* spp. was 28±1 stems ha^{-1}, which was less than unthinned (50±13 stems ha^{-1}), but comparable to thinned (25 ±16 stems ha^{-1}) sites. Hollows were also present in a small number of stumps > 1 m in height, but these were excluded from calculations of hollow stems. Density of stems in patches around roost sites could not be calculated for *V. vulturnus* using the point-quarter technique (Pollard 1971) as there were too few roosts located for the species.

Fig. 4.3.5. Photographs illustrating the density of small stems (<10cm dbhob) in (a) unthinned control sites, (b) thinned sites, (c) areas around roost trees used by *N. corbeni* and *N. gouldi*, and (d) areas around roost trees used by *N. geoffroyi*

and *V. vulturnus* located in thinned sites. Pink flagging tape (c) and arrow (d) indicate roost tree.

4.3.3. Night foraging

Data loggers in the centre of two thinned plots (ET2 and ET4) provided preliminary data on time spent in these plots by radio-tagged bats. Data logged for two individuals (one lactating *N. corbeni* and one lactating *N. gouldi*) over 4-5 nights in 2017 revealed that both individuals spent >25 % of the logged period per night (~6.5 h due to battery drainage) in these two thinned plots. Both bats were only logged in the thinned plot (37±1 % and 26±9 % of the logged period for *N. gouldi* and *N. corbeni*, respectively) in which they were captured and roosted approximately 0.4 km (*N. gouldi*) and 1.3 km (*N. corbeni*) from the centre of these plots. Bats captured in other thinned plots (ET1 and ET3) were not recorded on either logger.

4.3.4. Bat activity

In all, 8546 bat passes were recorded across all sites and both seasons (1326 - spring 2016; 7220 - autumn 2017). Of these, 4987 were identified to one of 10 taxa. *Vespadelus vulturnus* was most frequently recorded (37 % of identified calls), followed by *Mormopterus planiceps/Mormopterus petersi* (20 %), *Chalinolobus gouldii* (10 %), *Scotorepens greyii/Scotorepens* sp. (9 %), *Saccolaimus flaviventris* (8 %) and *Austronomus australis* (6 %). Calls from other taxa represented <5 % of all identified calls. Total nightly activity (no. calls night^{-1}) differed significantly between thinning treatments ($F_{11,1}$=6.810, P=0.024) and seasons ($F_{11,1}$=5.699, P=0.036), while there was no significant interaction effect of treatment by season ($F_{11,1}$=1.373, P=0.266). Total nightly activity was almost two-times greater in the thinned (183±43 calls night^{-1}) treatment compared to the unthinned (97±15 calls night^{-1}) treatment and ~64 % greater in autumn (167±35 calls night^{-1}) compared to spring (102±29 calls night^{-1}).

The activity of *Nyctophilus* spp. did not differ between thinning treatments (F11,1=1.095, P=0.318) or seasons (F11,1=1.932, P=0.192), but was significantly affected by the interaction of treatment by season (F11,1=7.055, P=0.022). *Nyctophilus* spp. activity was 5-times greater in the thinned treatment compared to the unthinned treatment in spring, whereas activity was 2-times greater in the unthinned treatment compared to the thinned treatment in autumn (Fig. 4.3.6).

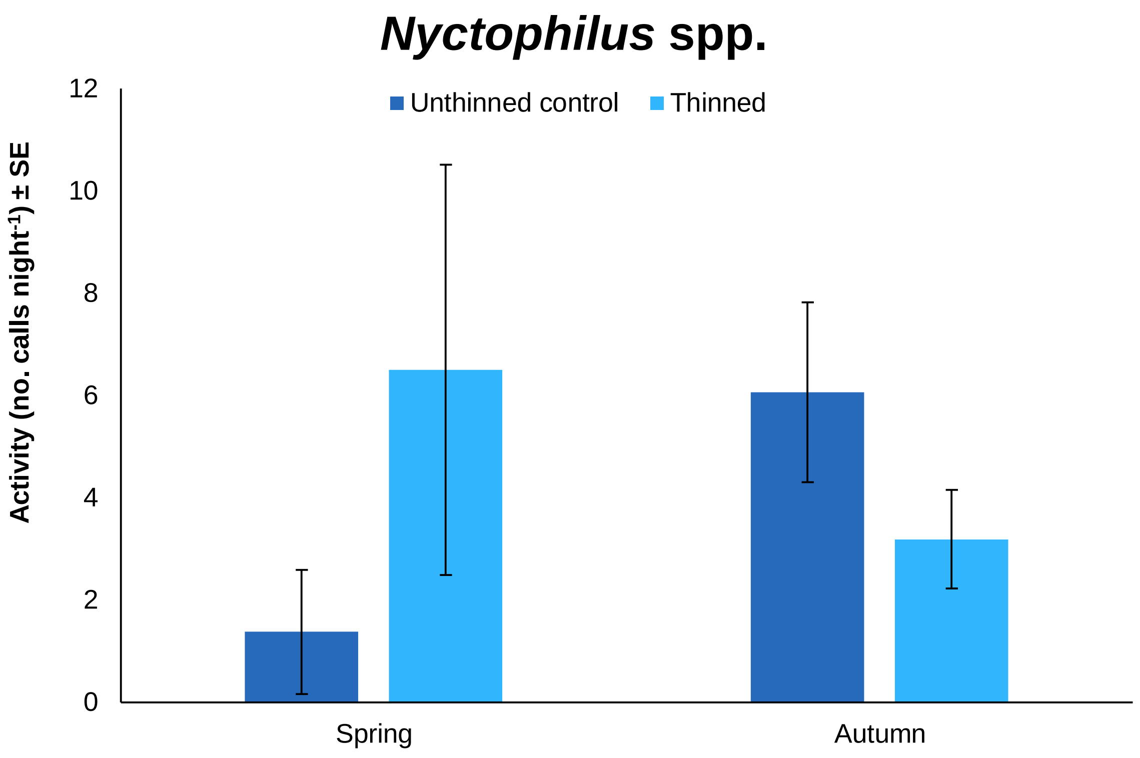


Fig. 4.3.6. Mean total nightly activity ± SE in unthinned and thinned treatments in spring 2016 and autumn 2017.

The activity of *V. vulturnus* was ~ 4 times greater in the thinned treatment compared to the unthinned treatment (F=4.555, P=0.056), but did not differ between seasons (F=0.028, P=0.871) and was not affected by the interaction of treatment by season (F=0.003, P=0.955).

4.3.5. Harp trapping captures

In all, 236 bats (165 in thinned and 71 in unthinned) were captured in thinned and unthinned sites across both years (Table 4.3.2). The number of bats captured in the unthinned treatment was 2.0 bats per trap night, whereas 2.6 bats per trap night were captured in the thinned treatment.

Table 4.3.2. Harp trapping data for experimental unthinned and thinned treatments over two years of trapping (2016 & 2017). Teal shading indicates species that were radio-tracked.

Species	**Unthinned (n=35 trap nights)**	**Thinned (n=65 trap nights)**
Chalinolobus gouldii	7	12
Chalinolobus picatus	0	3
Mormopterus petersi	0	1
M. planiceps	1	2
Nyctophilus corbeni	3	8
N. geoffroyi	7	36
N. gouldi	43	23
Saccolaimus flaviventris	0	1
Scotorepens balstoni	0	5
S. greyii	3	27
Scotorepens sp.	0	6
Vespadelus vulturnus	7	42
All bats	71	166

To evaluate whether capture rates differed between thinned and unthinned treatments, only trapping data for Autobat traps were included in analysis. Captures per trap-night for *N. corbeni* (Wilcoxon W=16.000, P=0.554), *N. geoffroyi* (Wilcoxon W=13.500, P=0.163), *V. vulturnus* (Wilcoxon W=13.000, P=0.144) and all bats (Wilcoxon W=15.000, P=0.386) were evenly spread across both thinning treatments, whereas capture rates were higher in the unthinned treatment for *N. gouldi* (Wilcoxon W=10.000, P=0.021).

4.4. DISCUSSION

Our study of day-roosting and nocturnal activity of multiple bat species in the dry cypress forests of the Pilliga revealed a complex pattern of habitat use across the thinned-unthinned mosaic that was only partially predicted by ecomorphology. Two closed-space (*N. corbeni* and *N. gouldi*) species in the genus *Nyctophilus* avoided roosting in relatively open, thinned areas as predicted. Few roosts were located for *N. geoffroyi* and these were located in thinned patches or in more open areas (e.g., along a forest road and in a riparian zone), suggesting that this species may be less sensitive to thinning. The edge-space species (*V. vulturnus*) was found to roost in the unthinned and thinned patches, though sample sizes were low for this species. In contrast, flight activity of *Nyctophilus* spp. was evenly spread across the forest mosaic, suggesting they actively foraged in both cluttered and uncluttered forest. Systematic trapping allowed this response to be distinguished at a species level with *N. gouldi* being captured more frequently in unthinned forest, while no difference was found for *N. geoffroyi* and *N. corbeni*. The activity, but not capture rate, of *V. vulturnus* (edge-space) was higher in thinned than unthinned forest as predicted by ecomorphology. Such varied patterns of habitat use have implications for managing the forest landscape and emphasise the value of heterogeneous landscapes for biodiversity conservation.

4.4.1. Roost selection

At the scale of individual trees, small maternity colonies (<10 bats) of all *Nyctophilus* spp. were typically in trees with a small diameter (<30 cm) and that were usually dead, especially *A. luehmannii* and *Eucalyptus*. Maternity colonies were located in hollows and fissures. All three *Nyctophilus* species overlapped substantially in the attributes of their roost trees. Near identical patterns of roost selection were found for *N. corbeni* elsewhere in the Pilliga forests (Law *et al.* 2016b) and also for *N. gouldi* in taller, wetter forests in other regions, though larger tree diameters are often preferred (Lunney *et al.* 1988; Webala *et al.* 2010; Threlfall *et al.* 2013; Burgar *et al.* 2015). Dead trees are also selected for roosts by *N. geoffroyi,* though again larger diameter trees are often used (Lumsden *et al.* 2002a; Law *et al.* 2011). Only a small sample of roosts was found for *V. vulturnus* and these were similar to those used by *Nyctophilus* spp.. Roosts of *V. vulturnus* are also commonly located in dead trees of variable diameter (Campbell 2005; Law *et al*. 2011; Rueegger *et al*. 2018). Clearly, the retention of dead trees to protect bat roosts is critical during forest thinning (Waters et al. 2018), yet collateral damage can result due to lack of knowledge of the importance of smaller dead trees with hollows for roosting bats. This in part resulted in half the hollow tree density in thinned plots compared to unthinned plots. Elsewhere in the Pilliga forests, thinning reduced dead tree density, but had no effect on hollow density or large tree density (Waters *et al*. 2018).

A final unexpected observation for roost trees was the frequency with which many had been ring-barked in the past (26 % of roosts). In one portion of our study area (which overlapped a thinned block), ringbarking of small ironbark and buloke was extensive and took place 40 years previously in 1977 leading to an abundance of dead trees with hollows that *N. gouldi* commonly used. It was only in this 12 ha thinned block that a small number of bat roosts (2 *N. geoffroyi* and 1 *V. vulturnus*) were located, potentially because of the higher dead hollow stem density. Two other roosts located along the northern boundary of this thinned block were used by *N. gouldi*. Elsewhere in the study area, ringbarked trees were used by *N. corbeni*. The longevity of ring-barked trees is in stark contrast to dead conifers in North America, where only 4 % of stags used as bat roosts remained standing after 10 years (Lacki *et al*. 2012). In that study, stags with large diameter and short height persisted for longer.

At a landscape scale, two (*N. corbeni* and *N. gouldi*) of the three *Nyctophilus* species avoided roosting in the open thinned areas and this has been found previously for *N. corbeni* in the Pilliga forests (Law et al. 2016b). These results are also consistent with *N. gouldi* avoiding roosting in narrow and open bushland strips in urban areas (Threlfall *et al*. 2013). Roosts in unthinned areas are typically surrounded by a high density of stems and hollow-bearing trees, which is consistent with previous studies of these species (Threlfall *et al*. 2013; Law *et al*. 2016b). In contrast, *N. geoffroyi* tended to roost in more open areas within the forest mosaic, including within one thinned block with a high density of ringbarked dead hollow-bearing trees. Other studies have found the species roosted in large forest blocks, with roosts surrounded by a lower density of stems (176 stems ha^{-1}; Lumsden *et al*. 2002b) than what has been found for conspecifics (251 stems ha^{-1} and 400 stems ha^{-1}; Threlfall *et al*. 2013; Law *et al*. 2016b). *Vespadelus vulturnus* roosts often have reduced canopy cover (Campbell 2005; Law *et* *al*. 2011; Ruegger *et al*. 2018), but only a single roost was located in thinned areas in our study. Considerably more data are needed on the patterns of roost selection by different bat species in relation to different management treatments (Law et al. 2016a). We are aware of just one other study of bat roosts in relation to thinning, which found a less manoeuvrable edge-space species (*Lasiurus borealis*) roosting in open conditions created by thinning (Perry *et al.* 2007). Based on our results we predict that roosts of other closed-space species will be most sensitive to practices that reduce clutter around roost trees.

4.4.2. Nocturnal activity

We found total bat activity was greater in thinned than unthinned areas and that this trend was maintained for *V. vulturnus*, as measured by number of echolocation calls and capture-rate. Forest thinning is known to be beneficial for a range of bat species due to increased foraging efficiency by echolocation where clutter levels have been reduced (Humes *et al*. 1997; Patriquin and Barclay 2003; Law *et al*. 2016a; Blakey *et al*. 2016). In contrast, flight activity of *Nyctophilus* spp. was evenly spread across thinned and unthinned forest. Closed-space species typically have higher activity levels in forests with higher stem densities (Müller *et al.* 2012; Fuentes-Montemayor *et al*. 2013; Blakey *et al.* 2017), though they are not restricted to clutter (Fenton 1990; Brigham *et al*. 1997). For example, *N. gouldi* has an aspect ratio of 5.77, is slow flying and occasionally uses forest openings (Brigham *et al*. 1997). In dense, wet forests it uses track and off-track sites equally (Law and Chidel 2002). Interestingly, Brigham *et al.* (1997) found habitat use by light tagged bats was indistinguishable for *N. gouldi* and *N. geoffroyi*.

Despite very similar morphology and echolocation calls, we found *N. gouldi* was captured more than expected in unthinned than thinned regrowth, while *N. geoffroyi* and *Nyctophilus corbeni* were captured equally in both treatments. These species-based trapping results (which aren't available from ultrasonics due to call similarity) from areas with different amounts of clutter are consistent with other studies of broad habitat use by these species. *Nyctophilus geoffroyi* can forage among scattered trees in open farmland (Lumsden et al. 2002a; Law et al. 2011) and is less sensitive to urbanisation than *N. gouldi* (Threlfall et al.

2013). Similar differences in habitat use have been documented for a number of morphologically similar species (Arletazz 1999; McConville *et al*. 2013).

4.4.3. Management implications

As an active management practice aiming for an ecological outcome, silvicultural thinning of dense regrowth is a form of 'renewal ecology' where goals include both the creation and/or enhancement of landscapes for biodiversity and ecosystem services for humans (Bowman *et al*. 2017). Yet the varied patterns of habitat use by multiple species that we identified emphasise the value of heterogeneous landscapes. In particular, heterogeneity should include a mosaic of thinned areas, but also recognize the value of dense unthinned patches both to fulfil the requirements of different species, but also for different aspects of a species ecology (roosting vs foraging areas; Law and Dickman 1998). The thinned patches in our study (12 ha) were of sufficient size to result in increased flight activity, but the size of unthinned patches required for roosting remains poorly known, though *N. corbeni* roosts in alleys of unthinned regrowth adjacent to thinned strips (Law et al. 2016b).

The importance of retaining dead trees as roosts for bats, even those that are relatively small (~ 20 cm dbhob), is a further clear management action emerging from our study (see also Law et al. 2016a). Unexpectedly, we also identified ring-barking as an important contributor to the abundance of dead trees in our study area. This practice was historically deployed for land-clearing or to remove unwanted trees in forestry areas. Applied as a strictly regulated technique on select species that can occur in high abundance (e.g. buloke) it may be useful for accelerating hollow development in areas where hollow trees are currently scarce, especially for threatened species such as *N. corbeni*. This is particularly the case as nest boxes often have limited life spans and effectiveness for threatened species of bats (Kavanagh et al. 2010; Rueegger 2016).

5. RECOMMENDATIONS ABOUT THE SUITABILITY OF THINNING FOR BIODIVERSITY IN CYPRESS FORESTS OF THE PILLIGA

This project assessed whether thinning of forest regrowth restored habitat for biodiversity in the cypress pine- and buloke-dominated forests of the Pilliga. Using a chronosequence of time since thinning and establishing a thinning trial, we have reported on the immediate and longer-term effects of thinning on vegetation structure, habitat values and broad groups of biodiversity. Furthermore, we examined species-specific responses of a small suite of echolocating bats, including a threatened species that may be sensitive to thinning (Law et al. 2016). Based on the findings from each component of the project, below we provide recommendations about the suitability of thinning for biodiversity in the cypress forests of the Pilliga.

The short- and longer-term responses of biodiversity to thinning of *C. glaucophylla* regrowth were mostly positive or neutral. However, individual species responded idiosyncratically across a chronosequence landscape and unthinned forest represented habitat of similar value to thinned forest for some taxa. For example, radio-tracking of focal bat species identified that although thinned areas had greater amounts of flight activity by bats, few maternity colonies were located within thinned plots. This suggests that it is important to retain some unthinned regrowth forest to provide a mosaic forest structure suitable for a diverse suite of flora and fauna. We recommend that broad-scale application of thinning should be avoided to ensure that heterogeneity of forest structure is maintained across the landscape (Date et al. 2002; Hunter 2013; Kay et al. 2016; Law et al. 2016; Eyre et al. 2015). Instead, we recommend a patchwork or mosaic of different vegetation densities would be of most value to biodiversity. Similar principles have been recommended for the management of invasive native scrub (Central West Local Land Services 2014). However, the degree or extent of forest regrowth to retain within the broader landscape is unclear and characterising patch size was beyond the scope of our study. Further research and monitoring is needed to identify a target for landscape retention of forest regrowth.

Although most responses of biodiversity to thinning were positive or neutral, the chronosequence component of our project identified a negative effect of thinning on dead tree densities. Although dead tree density was not significantly reduced from pre-to post-thinning in our thinning trial, small dead trees had been knocked over, probably as collateral damage, presumably by machinery during the thinning operation. Dead trees, even those that are relatively small (e.g. 10-30 cm dbhob), are important habitat elements in forests and in our study were found to be used for roosting by maternity colonies of several bat species, including the threatened *N. corbeni*. Hollows most commonly formed in small dead bulokes and were much less likely to be present in small dead cypress, probably because of termite resistance. As such, it is critical that these elements are retained within the local landscape. Any application of thinning should highlight the need for retention of dead trees, in both thinned and unthinned stands. This should also include small dead trees (15-30 cm dbhob) with small hollow entrance diameters that can be overlooked but represent habitat for bats and reptiles. Consideration should be given to protecting dead trees from damage by thinning machinery wherever possible. Retention of these habitat elements in clumps may be beneficial for some taxa, particularly maternity colonies of bats that routinely switch between a network of tree roosts. Our radio-tracking study demonstrated that different bat species had broad roosting areas, with little overlap among species. Therefore it is important that clumps of dead trees are retained across the landscape to provide a variety of roosting/nesting areas for various taxa.

The value of thinning for increasing CWD volumes was variable and related to the type of thinning that was undertaken. Increases in CWD were observed for non-commercial thinning in our chronosequence assessment, whereas no change was observed for the commercial thinning that represented our intermediate time since thinning treatment. A similar result was found for CWD volume in our early thin treatments where most thinnings were removed from the site, which is more akin to a commercial thin. If thinning is applied more intensively, specific recommendations for retention of thinnings as CWD may be required. We also highlight the value of larger pieces of CWD as likely providing proportionately greater habitat value than smaller pieces of debris (Date and Paull 2000).

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Table S2.3.1. Mean abundance of shrub (<2 m) and understorey plant species recorded in 0.02 ha plots in all treatments.

Species	Unthinned	Recently thinned	Intermediate thinned	Old thinned	Long undisturbed
Abutilon sp.1	0±0	0±0	0±0	0±0	0.6±0.6
Acacia mariae	0±0	0.1±0.1	0±0	0±0	0±0
Acacia pravifolia	0±0	0±0	0±0	0.1±0.1	0±0
Acacia sp.1	0.1±0.1	0.2±0.1	0.2±0.2	0.1±0.1	0.6±0.2
Acacia sp.2	0.1±0.1	0±0	0±0	0±0	0.1±0.1
Ajuga sp.1	0±0	0.7±0.7	0.8±0.8	0.7±0.7	0.6±0.6
Allocasuarina sp.1	0.8±0.4	0.4±0.3	0.1±0.1	0.2±0.1	0.6±0.2
Alternanthera sp.1	0±0	0±0	0±0	0±0	0.2±0.2
Amaryllidaceae sp.1	0±0	0.1±0.1	0±0	0.1±0.1	0.8±0.8
Aristida sp.1	1.6±0.8	3.8±0.6	2.4±0.4	2.2±0.6	2.5±0.5
Aristida sp.2	0.3±0.3	0±0	0±0	0.4±0.4	0.2±0.2
Austrodanthonia sp.1	0.2±0.1	0.4±0.3	0.4±0.4	0.2±0.1	0.1±0.1
Austrodanthonia sp.2	0±0	0±0	0.3±0.3	0±0	0±0
Austrostipa sp.1	1.4±0.3	5.3±2.4	7±1.6	5.2±1.8	3.6±1.2
Austrostipa sp.2	0±0	0±0	0.3±0.3	0±0	0±0
Bidens sp.1	0±0	0±0	0±0	0.1±0.1	0±0
Boerhavia sp.1	0±0	0±0	0±0	0.1±0.1	0.1±0.1
Brachyloma daphnoides	0±0	0±0	0±0	0.1±0.1	0±0
Brachyscome sp.1	0.9±0.6	0.4±0.4	0.2±0.2	1±0.8	0.9±0.7
Broadleaf grass	0±0	0±0	0±0	0.2±0.2	0±0
Bursaria sp.1	0.1±0.1	0±0	0±0	0±0	0±0
Callitris sp.1	0±0	0.1±0.1	0.1±0.1	0.1±0.1	0.2±0.2
Calotis sp.1	0.8±0.4	2.7±1.6	3.2±2.2	3±2	0.9±0.9
Calytrix sp.1	0±0	0.1±0.1	0±0	0±0	0±0
Carex inversa	0±0	0±0	0±0	0±0	0.8±0.8
Cassinia arcuata	0±0	0±0	0±0	0.1±0.1	0±0
Cassinia sp.	0±0	0±0	0.2±0.2	0.1±0.1	0±0
Chamaesyce sp.1	0±0	0±0	0.6±0.6	0±0	0±0
Cheilanthes sp.1	3.6±2	5.5±2.2	5.9±1.8	1.4±0.9	2.7±1.6
Cheilanthes sp.2	0±0	0±0	1.1±1.1	0.4±0.4	0±0

Chenopodium sp.1	0±0	0±0	0±0	0.1±0.1	0±0
Chenopodium sp.2	0±0	0±0	0±0	0.1±0.1	0±0
Chloris sp.1	0±0	0±0	0±0	0.1±0.1	0±0
Chrysocephalum sp.1	0±0	0±0	0.5±0.4	0±0	0.1±0.1
Convolvulus sp.1	0±0	0±0	0.1±0.1	0±0	0.1±0.1
Cyperus sp.1	2.7±1.8	0.5±0.3	1.3±1.1	2.1±1.4	2±1.3
Cyperus sp.2	0±0	0±0	0±0	0.1±0.1	0±0
Daviesia ulicifolia	0±0	0.1±0.1	0±0	0±0	0±0
Desmodium sp.1	0±0	0±0	1.2±1.2	0.6±0.6	0.3±0.3
Dianella sp.1	0.1±0.1	0.1±0.1	0.1±0.1	0.1±0.1	0±0
Dichondra sp.1	0±0	0±0	0±0	1.9±1.9	0±0
Digitaria sp.1	0±0	0±0	0±0	0±0	0.1±0.1
Dodonaea sp.1	0.2±0.2	0±0	0.1±0.1	0.3±0.2	0.2±0.1
Einadia sp.1	0.6±0.6	0.3±0.3	0.8±0.6	2.2±1	0.2±0.1
Einadia sp.2	0±0	0.1±0.1	0.2±0.2	0±0	1.2±1.1
Enteropogon sp.1	0.1±0.1	1.1±0.8	0±0	0.8±0.8	0.8±0.7
Epacridaceae sp.1	0±0	0.1±0.1	0±0	0±0	0.1±0.1
Eragrostis sp.1	0.4±0.3	0.5±0.4	0.4±0.1	1.6±0.9	0.9±0.3
Eragrostis sp. 2	0.1±0.1	0±0	0±0	0±0	0±0
Eremophila sp.1	0±0	0±0	0±0	0.1±0.1	0.1±0.1
Eriochloa sp.1	0±0	0±0	0±0	0±0	0.1±0.1
Geijera sp.1	0±0	0±0	0±0	0±0	0.1±0.1
Glycine sp.1	0.4±0.3	0.1±0.1	0.5±0.5	1.4±1.4	0.4±0.3
Glycine sp.2	0±0	0±0	0±0	0±0	0.3±0.3
Gnephosis sp.1	0±0	0±0	0±0	0±0	0.4±0.4
Goodenia sp.1	0.8±0.6	0.9±0.4	0.5±0.4	0.2±0.2	0.2±0.2
Helipterum sp.1	0±0	0±0	0±0	0±0	0.2±0.2
Laxmannia compacta	0±0	0±0	0±0	0±0	0.1±0.1
Leucopogon muticus	0±0	0.1±0.1	0±0	0±0	0±0
Lomandra filiformis	0±0	0±0	0±0	0±0	0.1±0.1
Lomandra multiflora	0±0	0±0	0±0	0.3±0.3	0±0
Lomandra sp.1	2.1±0.9	1.2±0.4	0.7±0.3	1.1±0.4	0.7±0.3
Lomandra sp.2	0±0	0.1±0.1	0.3±0.3	0±0	0±0
Lomandra sp.5	0±0	0±0	0±0	0±0	0.5±0.5

Lomandra sp.6	0±0	0±0	0±0	0±0	0.2±0.2
Lotus sp.1	0±0	0±0	0±0	0±0	0.1±0.1
Macrozamia sp.	0±0	0±0	0±0	0±0	0.1±0.1
Malacocera sp.1	0±0	0±0	0±0	0±0	0.2±0.2
Mirbelia sp.	0±0	0±0	0±0	0±0	0.2±0.2
Opuntia sp.1	0±0	0±0	0±0	0.1±0.1	0.1±0.1
Orchidaceae sp.1	0.1±0.1	0±0	0±0	0±0	0±0
Oxalis sp.1	0.4±0.4	0.1±0.1	1±1	0±0	0±0
Panicum sp.1	0.1±0.1	0±0	0±0	0±0	0±0
Paspalidium sp.1	0.1±0.1	0.4±0.4	0±0	0±0	0±0
Pimelea sp.1	0±0	0±0	0±0	0±0	0.1±0.1
Poaceae sp.1	1.6±0.7	0.9±0.4	2.1±1.5	3.3±2.2	1.2±0.8
Poaceae sp.2	0.3±0.2	0.1±0.1	0±0	0.7±0.5	0±0
Poaceae sp.3	0.1±0.1	0±0	0±0	0.1±0.1	0±0
Pomax sp.1	0.1±0.1	0±0	0±0	0±0	2.3±2.3
Pomax umbellata	0.1±0.1	0.1±0.1	0±0	0±0	0.5±0.4
Pultenaea setulosa	0±0	0±0	0.1±0.1	0±0	0±0
Pultenaea sp.	0±0	0.1±0.1	0±0	0±0	0.1±0.1
Rhodanthe sp.1	0.9±0.4	1.1±0.5	0.7±0.3	1.7±1.1	1.5±0.5
Sclerolaena sp.	0±0	0±0	0±0	0±0	0.1±0.1
Sclerolaena sp.1	0±0	0±0	0.1±0.1	0.1±0.1	0±0
Sclerolaena sp.2	0±0	0±0	0.1±0.1	0.3±0.3	0±0
Senna sp.1	0±0	0±0	0±0	0.1±0.1	0±0
Sida sp.	0±0	0±0	0±0	0±0	0.1±0.1
Sida sp.1	0.1±0.1	0±0	0.3±0.3	0.6±0.4	0.2±0.1
Sida sp.2	0±0	0±0	0±0	0.1±0.1	0±0
Solanum ferocissimum	0±0	0.2±0.2	0±0	0.1±0.1	0±0
Solanum shrub	0.1±0.1	0±0	0±0	0±0	0±0
Solanum sp.1	0±0	0.3±0.3	0.4±0.3	0.3±0.2	0.1±0.1
Sporobolus sp.1	0.3±0.2	0.2±0.1	0.1±0.1	0.2±0.1	0.5±0.4
Swainsona sp.1	0±0	0±0	0.1±0.1	0.2±0.2	0.2±0.2
Thyridolepis sp.1	0±0	0.7±0.7	0.3±0.3	0.1±0.1	0±0
Tricoryne elatior	0.3±0.3	0±0	0±0	0±0	0±0
Trifolium sp.1	0±0	0±0	0.1±0.1	0.2±0.2	0±0

Trifolium sp.2	0±0	0±0	0.1±0.1	0±0	0±0
Tripogon sp.1	0±0	0±0	0±0	0.4±0.4	0±0
Unknown herb 1	1.2±0.6	2.8±1.9	0.9±0.4	1.3±0.3	2±0.7
Unknown herb 2	0.1±0.1	0.5±0.2	0.3±0.1	0.5±0.4	0.6±0.4
Unknown herb 3	0±0	0.3±0.2	0.1±0.1	0.2±0.1	0.1±0.1
Unknown herb 4	0±0	0.2±0.1	0±0	0.1±0.1	0.3±0.3
Unknown herb 5	0±0	0±0	0±0	0.1±0.1	0±0
Unknown herb 6	0±0	0±0	0.2±0.2	0.1±0.1	0±0
Unknown herb 7	0±0	0±0	0.1±0.1	0±0	0±0
Unknown herb 8	0±0	0±0	0.1±0.1	0±0	0±0
Unknown herb 9	0±0	0.1±0.1	0±0	0±0	0±0
Unknown shrub 1	0±0	0.2±0.1	0.1±0.1	0.1±0.1	0.4±0.4
Unknown shrub 2	0±0	0.1±0.1	0±0	0±0	0±0
Unknown shrub 3	0±0	0.2±0.2	0±0	0±0	0±0
Urochloa sp.1	0±0	0.2±0.2	0±0	0±0	0±0
Verbena sp.1	0±0	0±0	0.1±0.1	0±0	0±0
Vittadinia sp.1	0±0	0.1±0.1	1.2±1.2	0.5±0.5	0±0
Wahlenbergia sp.1	0±0	0±0	0.3±0.2	0.6±0.6	0.1±0.1

Table S2.3.2. Summary of generalized linear mixed models relating plant diversity, bat activity and diversity, bird diversity, native mammal activity and diversity, introduced herbivore and predator activity, reptile abundance and diversity and composite biodiversity to forest management treatments.

Response	Model term	Coefficient	SE	T	*P*
Plant diversity (gamma – identity link)	Intercept (Long undisturbed)	2.264	0.246	9.183	<0.001
	Unthinned	-0.625	0.288	-2.173	0.039
	Recent thinning	-0.366	0.290	-1.263	0.218
	Intermediate thinning	-0.357	0.281	-1.269	0.216
	Old thinning	0.033	0.453	0.073	0.942
Nightly bat activity (normal – log link)	Cluster[a]	0.014	0.010	1.366[b]	0.172
	Intercept (Long undisturbed)	0.698	0.063	11.077	<0.001
	Unthinned	-0.136	0.061	-2.224	0.030
	Recent thinning	0.071	0.056	1.259	0.213
	Intermediate thinning	-0.047	0.058	-0.809	0.421
	Old thinning	0.056	0.055	1.023	0.310
Bat diversity (gamma – identity link)	Intercept (Long undisturbed)	0.007	0.001	6.981	<0.001
	Unthinned	-0.002	0.001	-1.478	0.144
	Recent thinning	0.003	0.002	1.827	0.072
	Intermediate thinning	0.002	0.002	0.908	0.367
	Old thinning	0.000	0.002	-0.061	0.951
Bird diversity (gamma – identity link)	Intercept (Long undisturbed)	0.020	0.001	18.567	<0.001
	Unthinned	-0.002	0.001	-1.306	0.203

	Recent thinning	0.002	0.002	1.485	0.150
	Intermediate thinning	0.001	0.002	0.744	0.464
	Old thinning	0.002	0.002	1.586	0.125
Native mammal activity (normal – log link)	Intercept (Long undisturbed)	0.375	0.578	0.649	0.523
	Unthinned	-0.274	0.877	-0.313	0.757
	Recent thinning	0.188	0.705	0.267	0.792
	Intermediate thinning	1.166	0.556	2.098	0.047
	Old thinning	0.288	0.663	0.435	0.668
Native mammal diversity (normal – identity link)	Intercept (Long undisturbed)	0.016	0.005	3.432	0.002
	Unthinned	0.001	0.006	0.001	0.999
	Recent thinning	0.001	0.006	0.144	0.887
	Intermediate thinning	0.012	0.006	1.911	0.068
	Old thinning	0.005	0.006	0.879	0.388
Introduced herbivore activity (normal – identity link)	Intercept (Long undisturbed)	0.167	0.512	0.325	0.748
	Unthinned	-0.167	0.725	-0.230	0.820
	Recent thinning	0.633	0.760	0.833	0.413
	Intermediate thinning	1.000	0.725	1.380	0.180
	Old thinning	1.167	0.725	1.610	0.120
Introduced predator activity (normal – identity link)	Intercept (Long undisturbed)	0.001	0.322	0.001	0.999
	Unthinned	0.667	0.456	1.462	0.157
	Recent thinning	0.200	0.478	0.418	0.679
	Intermediate thinning	-0.001	0.456	-0.001	0.999

	Old thinning	0.167	0.456	0.366	0.718
Reptile abundance (normal – log link)	Intercept (Long undisturbed)	-1.022	0.331	-3.091	0.005
	Unthinned	0.246	0.368	0.669	0.510
	Recent thinning	-0.447	0.548	-0.816	0.422
	Intermediate thinning	0.679	0.344	1.976	0.059
	Old thinning	0.021	0.417	0.050	0.960
Reptile diversity (gamma – identity link)	Intercept (Long undisturbed)	-3.827	0.096	-40.010	<0.001
	Unthinned	-0.072	0.100	-0.717	0.480
	Recent thinning	-0.115	0.119	-0.965	0.344
	Intermediate thinning	0.274	0.127	2.155	0.041
	Old thinning	-0.035	0.120	-0.294	0.771
Invertebrate biomass (normal – log link)	Intercept (Long undisturbed)	1.711	0.166	10.283	<0.001
	Unthinned	0.174	0.217	0.801	0.442
	Recent thinning	0.289	0.208	1.389	0.195
	Intermediate thinning	0.315	0.206	1.529	0.157
	Old thinning	0.203	0.215	0.994	0.368
Biodiversity (normal – identity link)	Intercept (Long undisturbed)	0.021	0.001	30.392	<0.001
	Unthinned	-0.002	0.001	-2.325	0.028
	Recent thinning	<0.0001	0.001	0.424	0.675
	Intermediate thinning	0.001	0.001	0.758	0.456
	Old thinning	0.001	0.001	1.256	0.221

[a] Denotes random effect.

[b] Denotes Z-score for variance

Table S2.3.3. Median abundance class for bird species in all treatments. 0=absent, 1=1-2 individuals, 2=3-4 individuals, 3>4 individuals.

Species	Unthinned	Recent thinning	Intermediate thinning	Old thinning	Long undisturbed
Australian Magpie	0	0	1	0	0
Australian Raven	0	0	0	0	0
Australian Ringneck Parrot	0	0	0	0	0
Black-eared Cuckoo	0	0	0	0	0
Black-faced Cuckoo Shrike	0	0	0	0	0
Brown Treecreeper	0	0	0	0	0
Brown-headed Honeyeater	0	0	0	0	0
Buff-rumped Thornbill	0	0	1	0	0
Common Bronzewing	0	0	0	0	0
Crested Bellbird	0	0	0	0	0
Crested Pigeon	0	0	0	0	0
Dusky Woodswallow	0	0	0	0	0
Eastern Rosella	0	0	0	0	0
Eastern Yellow Robin	1	1	1	1	1
Fantail Cuckoo	0	0	0	0	0
Fuscus Honeyeater	0	0	0	0	0
Galah	0	0	0	0	0
Grey Butcherbird	0	0	0	0	0
Grey Fantail	1	1	1	1	1
Grey Shrike Thrush	1	1	0	1	1

Grey-crowned Babbler	0	1	0	0	0
Horsefields Cuckoo	0	0	0	0	0
Inland Thornbill	0	1	0	1	1
Laughing Kookaburra	0	0	0	0	0
Leaden Flycatcher	0	1	0	1	0
Little Eagle	0	0	0	0	0
Magpie-lark	0	0	0	0	0
Mistletoebird	0	1	1	0	1
Noisy Friarbird	1	1	0	0	0
Noisy Miner	0	0	0	0	0
Olive-backed Oriole	0	0	0	0	0
Owlet Nightjar	0	0	0	0	0
Pallid Cuckoo	0	0	0	0	0
Pied Butcher Bird	0	0	0	0	0
Pied Currawong	0	0	0	0	0
Rainbow Bee-eater	0	0	0	0	0
Red-capped Robin	0	0	1	0	0
Restless Flycatcher	0	0	0	1	0
Rufous Whistler	1	1	2	2	1
Sacred Kingfisher	0	0	0	0	0
Scarlet Robin	0	0	0	0	0
Shining Bronze Cuckoo	0	0	1	0	0

Small Honeyeater	0	0	0	0	0
Speckled Warbler	0	0	0	0	0
Spiny-cheeked Honeyeater	0	0	0	0	0
Spotted Pardalote	0	1	0	0	1
Striated Pardalote	0	0	0	0	0
Striated Thornbill	0	0	0	0	0
Striped Honeyeater	0	0	0	0	0
Sulphur-crested Cockatoo	0	0	0	0	0
Superb Fairy-wren	0	0	0	0	0
Turquoise Parrot	0	0	0	0	0
Varied Sitella	0	0	0	0	0
Varied Triller	0	0	0	0	0
Weebill	1	1	1	1	1
Western Gerygone	0	0	0	0	1
White-bellied Cuckoo-shrike	0	0	0	0	0
White-browed Babbler	0	0	0	0	0
White-browed Treecreeper	0	0	0	0	0
White-browed Woodswallow	0	0	0	0	0
White-eared Honeyeater	0	0	0	0	0
White-plumed Honeyeater	0	0	0	0	0
White-throated Treecreeper	1	1	0	1	1
White-winged Chough	0	0	0	0	0

White-winged Triller	0	0	0	0	0
Willie Wagtail	0	0	0	0	0
Wood Duck	0	0	0	0	0
Yellow Thornbill	1	1	2	2	2
Yellow-faced Honeyeater	0	0	1	1	1

Table S2.3.4. Mean±SE nightly activity recorded for bat taxa in all treatments.

Species	Unthinned	Recent thinning	Intermediate thinning	Old thinning	Long undisturbed
Austronomus australis	4.6±1.3	2.4±0.7	6.8±1.8	2.8±0.9	12.7±7.9
Chalinolobus gouldii	2.8±1	12.7±4.5	3.7±1.1	7.7±2.5	7.7±2.2
Chalinolobus morio	0.1±0.1	0.2±0.2	0.2±0.1	0.2±0.1	0.2±0.2
Chalinolobus picatus	0.5±0.4	0.4±0.1	0.3±0.2	0.7±0.3	0.1±0.1
Mormopterus lumsdenae	0.1±0.1	0.3±0.2	0.7±0.5	0.3±0.2	0.3±0.3
Mormopterus petersi	0±0	0.3±0.2	0.1±0.1	0.1±0.1	0.2±0.1
Mormopterus planiceps	1.9±0.8	3.5±1.3	2.9±1.1	1.9±0.9	1.4±0.5
Mormopterus planiceps/petersi	2.5±0.9	5.8±1.8	7.2±3.2	2.1±0.6	7.3±2.3
Nyctophilus spp.	4.3±1	9.9±2.1	6.5±1.2	7.7±1.6	10.5±2.6
Saccolaimus flaviventris	8.6±2.1	15.8±5.7	5.5±1.2	9.2±1.6	11.9±4.5
Scotorepens balstoni	0.1±0.1	1.6±0.6	1±0.3	1.1±0.5	0.7±0.4
Scotorepens greyii/Scotorepens sp.	6.5±2.5	15.3±3.8	5.7±1.2	9.1±3.2	4.9±1.4
Vespadelus vulturnus	24.6±9	61.4±21.8	30.3±10.5	126±70.4	24.7±6.8

Teal shading indicates species or member of species group is listed as vulnerable under the NSW Biodiversity Conservation Act 2016.

Table S2.3.5. Mean±SE activity (no. of events per two weeks) for mammal species in all treatments.

Species	Unthinned	Recent thinning	Intermediate thinning	Old thinning	Long undisturbed
Capra hircus	0±0	0.6±0.4	0.3±0.2	0.8±0.7	0.2±0.2
Macropus dorsalis	0±0	0±0	0.2±0.2	0.2±0.2	0±0
Macropus giganteus	0.2±0.2	1.2±0.8	2.8±1.7	2.3±0.6	1.3±0.8
Macropus rufogriseus	0.5±0.5	0±0	0.8±0.5	0.2±0.2	0±0
Oryctolagus cuniculus	0±0	0.2±0.2	0.8±0.8	0.5±0.5	0±0
Tachyglossus aculeatus	0.3±0.2	0.2±0.2	0±0	0±0	0.2±0.2
Trichosurus vulpecula	0.2±0.2	0.4±0.4	0.5±0.3	0±0	0±0
Vulpes vulpes	0.7±0.7	0.2±0.2	0±0	0.2±0.2	0±0

Teal shading indicates species or member of species group is listed as vulnerable under the NSW Biodiversity Conservation Act 2016.

Table S2.3.6. Mean±SE abundance for reptile species trapped (over 12 days) or observed during spotlighting (20 mins; two observers) in all treatments.

Species	Unthinned	Recent thinning	Intermediate thinning	Old thinning	Long undisturbed	Survey method
Amphibolurus burnsi	0±0	0.2±0.2	0±0	0±0	0±0	B
Cryptoblepharus pannosus					✓	I
Ctenotus allotropis	0.2±0.2	0±0	0.3±0.2	0±0	0.3±0.3	P,B,I
Diplodactylus vittatus	1.5±0.8	0.2±0.2	0.8±0.4	0±0	0.2±0.2	P,B,S
Diporiphora nobbi	0±0	0±0	0.2±0.2	0.3±0.2	0.3±0.2	P,B,I
Egernia striolata		✓				I
Furina diadema	0±0	0±0	0±0	0±0	0.2±0.2	S
Gehyra variegata	0.3±0.3	0.2±0.2	0.5±0.5	0±0	0±0	P,B,S
Heteronotia binoei	0.2±0.2	0±0	0.8±0.3	0.7±0.3	0±0	P,B,S,I
Hoplocephalus bitorquatus	0±0	0±0	0±0	0.2±0.2	0±0	S
Lerista muelleri	0±0	0±0	0±0	0.2±0.2	0±0	P
Lygisaurus foliorum	0±0	0.2±0.2	0.2±0.2	0.2±0.2	0.2±0.2	P,B
Morethia boulengeri	0±0	0.2±0.2	0.8±0.3	0.2±0.2	0.3±0.2	P,B,I
Oedeura monolis	0±0	0±0	0.2±0.2	0±0	0±0	S
Oedeura robusta	0±0	0±0	0±0	0±0	0.2±0.2	B
Pogona barbata		✓	✓			I
Pygopus lepidopodus	0±0	0.2±0.2	0.2±0.2	0±0	0±0	P,S
Ramphotyphlops ligatus	0±0	0±0	0.2±0.2	0±0	0±0	P
Strophurus williamsi	0±0	0±0	0.3±0.3	0±0	0.2±0.2	P,B,S

Varanus gouldii	✓		I
Varanus varius		✓	I

P=Pipe; B=Bucket; S= Spotlighting; I= Incidental. Teal shading indicates species is listed as vulnerable under the NSW Biodiversity Conservation Act 2016. ✓ indicates at least one incidental record of a species.

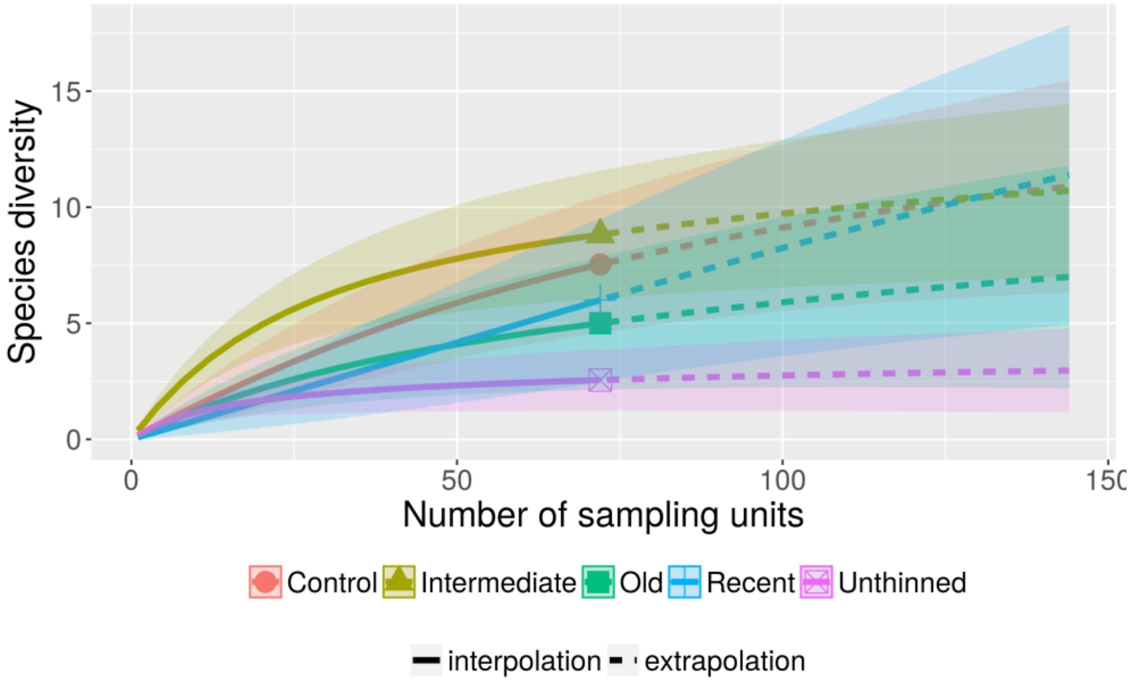

Fig. S2.3.1. Extrapolated reptile diversity (Shannon-Weiner) recorded across all forest management treatments.

Table S3.3.1. Mean±SE nightly activity (no. calls night^{-1}) recorded for bat taxa in all treatments pre- and post-thinning.

Species	**Pre-thinning**						**Post-thinning**					
	Early thin control	Early thin	Early thin reference	Second thin control	Second thin	Second thin reference	Early thin control	Early thin	Early thin reference	Second thin control	Second thin	Second thin reference
Austronomus australis	0.9±0.5	0.9±0.4	1.4±0.4	0.7±0.4	0.3±0.1	2.7±2.1	6±2.9	7.1±1.4	4.4±2.9	7.8±2.1	8.4±1.6	12±1.5
Chalinolobus gouldii	6.6±1.4	4.5±1.4	5±3.3	3.3±1.1	2.6±0.5	3.2±1.3	7.9±1	7.9±1.5	11.1±1.1	9.3±3.1	8.3±2.1	7.3±0.5
Chalinolobus morio	0±0	0.1±0.1	0±0	0.4±0.3	0.1±0.2	1±0.5	0.5±0.3	0.8±0.5	0±0	0.1±0.1	0.2±0.1	2.1±0.1
Chalinolobus picatus	0.1±0.2	0.5±0.4	0.6±0.6	0.3±0.2	0.7±0.4	0.1±0.1	2.1±0.9	3.1±2.2	0.1±0.1	2.3±1.9	1.4±0.5	0.5±0
Mormopterus lumsdenae	0.5±0.2	1±0.7	0.6±0.1	0.6±0.6	2±1.7	0.5±0.3	0.8±0.4	1±0.4	0.1±0.1	0.2±0.1	1±0.6	0.6±0.4
Mormopterus planiceps/petersi	9.2±1	13.1±3	18±6.2	3.9±1.2	22.1±6	5.3±2.8	12±1.9	19.4±6	21.9±12.7	7.7±1.1	17.6±5.1	6.9±1.6
Nyctophilus spp.	2.8±1.1	2.9±1	3.9±0.1	2.7±1.3	3.9±1.3	4.6±1.5	6.1±1.8	3.2±1	1.3±0.4	6.5±2.4	5.4±0.8	12±8.8
Saccolaimus flaviventris	8.8±2	8.8±0.9	17.3±10	8.6±1.2	13.3±2.2	4.9±0.4	4.9±1.5	8.3±1.8	17.5±13.2	11.1±2.7	43.9±20.6	4.6±1.9
Scotorepens balstoni	0±0.1	0.1±0.1	0.8±0.5	0.2±0.1	1.1±0.2	0.9±0.4	0.3±0.3	0.9±0.3	0±0	0.3±0.2	1.5±0.7	0.4±0.4
Scotorepens greyii/sp.	12.6±3.9	13.1±3.2	17.4±14.8	16.4±7.3	12.2±4.4	13.7±2.8	5.3±2	11.1±2.7	0.8±0.3	6.8±3.2	16.4±3.1	6.9±1.6
Vespadelus vulturnus	12.9±5.3	12.5±3.4	2.4±1.1	12.2±5.6	27.2±8	52.4±27.2	11.2±3.1	56.6±42.4	1.5±0.6	16.4±10.1	29.9±7.9	25.3±10.4

Table S3.3.2. Median abundance class for bird species in all treatments pre- and post-thinning. 0=absent, 1=1-2 individuals, 2=3-4 individuals, 3>4 individuals.

Species	**Pre-thinning**						**Post-thinning**					
	Early thin control	Early thin	Early thin reference	Second thin control	Second thin	Second thin reference	Early thin control	Early thin	Early thin reference	Second thin control	Second thin	Second thin reference
Australian Magpie	0	0	0	0	0	0	0	0	0	0	1	0
Australian Mapie-lark	0	0	0	0	0	0	0	0	0	0	0	0
Australian Raven	0	0	0	0	0	0	0	0	0	1	0	0
Australian Ringneck Parrot	0	0	0	0	1	0	0	0	1	0	1	0
Bar-shouldered Dove	0	0	0	0	0	0	0	0	0	0	0	0
Brown Thornbill	0	0	0	0	0	0	0	0	0	0	0	0
Brown Falcon	0	0	1	0	0	0	0	0	0	0	0	0
Brown Treecreeper	0	0	0	0	0	0	0	0	1	0	1	0
Brown-headed Honeyeater	0	0	0	0	0	0	0	0	0	0	0	0
Common Bronzewing	0	0	0	0	0	0	0	0	0	0	0	0
Crested Pigeon	0	0	0	0	0	0	0	0	0	0	0	0
Crested Shrike-tit	0	0	0	0	0	0	0	0	0	0	0	0
Double-barred Finch	0	0	0	0	0	0	0	0	0	0	0	0
Eastern Rosella	0	0	0	0	0	0	0	0	0	0	0	0
Eastern Yellow Robin	1	1	1	1	1	1	2	1	1	1	1	1
Fantail Cuckoo	0	0	0	0	0	0	0	0	0	0	0	0
Flycatcher	0	0	0	0	0	0	0	0	0	0	0	0
Galah	0	0	1	0	0	2	0	0	1	1	1	0
Gerygone	0	0	0	0	0	0	0	0	0	0	0	0
Glossy Black Cockatoo	0	1	0	0	0	0	0	0	0	0	0	0

Grey Butcherbird	0	0	0	0	1	0	1	1	1	0	1	2
Grey Fantail	1	1	1	1	1	1	1	1	1	2	1	2
Grey Shrike-thrush	0	1	1	0	1	0	1	1	1	0	1	1
Grey-crowned Babbler	0	0	1	0	0	0	0	0	0	0	1	0
Horsefield's Bronze Cuckoo	0	0	0	0	0	0	0	0	0	0	0	0
Inland Thornbill	1	0	0	0	1	0	1	0	0	0	0	1
Jacky Winter	0	0	0	0	0	0	0	0	0	0	1	0
Laughing Kookaburra	0	0	0	0	0	0	0	0	0	0	0	0
Leaden Flycatcher	0	0	0	0	0	0	0	0	0	0	0	0
Little Friarbird	0	0	0	0	0	0	0	0	0	0	0	0
Masked Woodswallow	0	0	0	0	0	0	0	0	0	0	0	0
Lewin's Honeyeater	0	0	0	0	0	1	0	0	0	0	0	0
Mistletoebird	1	1	1	1	1	1	1	1	1	0	1	1
Noisy Friarbird	0	0	0	0	0	1	1	1	0	1	1	1
Noisy Miner	0	0	0	0	0	0	0	0	0	0	1	0
Owlet Nightjar	0	0	0	0	0	0	0	0	0	0	0	0
Peaceful Dove	0	0	0	0	0	1	0	0	0	0	0	0
Pied Butcherbird	0	0	0	0	0	0	0	0	0	0	0	0
Pied Currawong	0	0	0	0	0	0	0	0	0	0	0	1
Quail	0	0	0	0	0	0	0	0	0	0	0	0
Red-capped Robin	0	0	0	1	0	0	0	1	0	0	1	0
Restless Flycatcher	0	0	0	0	0	0	0	0	0	0	0	0
Rufous Whistler	2	2	1	1	1	1	1	1	1	0	1	1
Shining Bronze Cuckoo	0	0	0	0	0	0	0	0	0	0	0	0
Silvereye	0	0	0	0	0	0	0	0	0	0	0	0

Singing Honeyeater	0	0	0	0	0	1	0	0	0	0	0	0
Speckled Warbler	0	0	0	0	0	0	0	0	0	1	0	1
Spiny-cheeked Honeyeater	1	1	1	1	0	1	1	1	1	1	1	0
Spotted Bowerbird	0	0	0	0	0	0	0	0	0	0	0	0
Spotted Pardalote	0	0	1	0	1	1	0	0	0	0	0	0
Striated Pardalote	0	0	1	0	0	0	0	0	0	0	0	0
Striped Honeyeater	0	0	0	0	0	0	0	1	0	0	0	0
Thornbill	0	0	0	0	0	0	0	0	1	0	0	0
Torresian Crow	0	0	0	0	0	0	0	0	0	0	0	0
Turquoise Parrot	0	0	0	0	0	0	0	0	0	0	0	0
Varied Sitella	0	0	0	0	0	0	0	1	0	0	0	0
Variegated Wren	0	0	0	0	0	0	0	0	0	0	0	0
Weebill	1	2	2	1	2	1	0	0	1	0	0	0
Western Gerygone	0	0	0	0	0	0	0	0	1	0	0	0
White-browed Treecreeper	0	0	0	0	0	0	0	0	0	0	0	0
White-browed Woodswallow	0	0	0	0	0	0	0	0	0	0	0	0
White-eared Honeyeater	0	0	0	1	1	1	0	1	0	1	0	1
White-plumed Honeyeater	0	0	0	0	0	0	0	0	0	0	0	0
White-throated Honeyeater	0	0	0	0	0	1	0	0	0	0	0	0
White-throated Treecreeper	1	1	1	1	2	1	1	1	1	1	0	1
Willie Wagtail	0	0	0	0	0	0	0	1	0	0	0	0
Yellow Thornbill	1	1	0	0	0	2	0	2	0	3	1	2
Yellow-cheeked Honeyeater	0	0	0	0	0	0	0	0	0	0	0	0
Yellow-faced Honeyeater	0	1	0	0	0	1	0	0	0	1	0	1
Yellow-rumped Thornbill	0	0	0	0	0	0	0	0	0	0	0	0

Table S3.3.3. Mean±SE activity (no. of events per two weeks) for mammal species in all treatments pre- and post-thinning

Species	**Pre-thinning**						**Post-thinning**					
	Early thin control	Early thin	Early thin reference	Second thin control	Second thin	Second thin reference	Early thin control	Early thin	Early thin reference	Second thin control	Second thin	Second thin reference
Antechinus flavipes	0±0	0.3±0.3	0±0	0.3±0.3	0.3±0.3	0.5±0.5	0±0	0±0	0±0	0±0	0±0	1±0
Capra hircus	0.3±0.3	0.8±0.8	0±0	0.3±0.3	0.8±0.8	0±0	1.5±1.5	0±0	20.5±20.5	0.8±0.8	1.3±1	0±0
Equus caballus	0±0	0±0	0±0	0±0	0.3±0.3	0±0	0±0	0±0	0±0	0±0	0±0	0±0
Macropus dorsalis	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0.5±0.5
Macropus giganteus	0.5±0.3	0.3±0.3	4.5±1.5	2±0.9	1±1	0.5±0.5	0±0	1.3±1	0±0	0.5±0.3	2±1.2	6±5
Macropus rufogriseus	0±0	0±0	0±0	0±0	0±0	1±1	0±0	0.3±0.3	0±0	0±0	0.8±0.5	0±0
Mus musculus	0±0	0±0	0±0	0±0	0.3±0.3	0±0	1.3±0.8	0.5±0.5	0±0	0.5±0.3	0.5±0.5	0.5±0.5
Sminthopsis murina	0±0	0.3±0.3	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0±0
Sus scrofa	0±0	0±0	0.5±0.5	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0±0
Tachyglossus aculeatus	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0.5±0.5	0±0	0±0	0±0
Vulpes vulpes	0±0	0±0	2±1	0±0	0.3±0.3	0.5±0.5	0±0	0±0	0.5±0.5	0±0	0±0	0±0
Wallabia bicolor	0±0	0±0	0±0	0.3±0.3	0±0	1.5±1.5	0±0	1±1	0±0	1.8±1.8	0.5±0.5	0±0

Table S3.3.4. Mean±SE abundance for reptile species trapped or observed in all treatments. Pre- and post-thinning.

Species	**Pre-thinning**						**Post-thinning**					
	Early thin control	Early thin	Early thin reference	Second thin control	Second thin	Second thin reference	Early thin control	Early thin	Early thin reference	Second thin control	Second thin	Second thin reference
Amphibolurus burnsi	0±0	0±0	0±0	0±0	0±0	0.5±0.5	0±0	0±0	0±0	0±0	0±0	0±0
Ctenotus allotropis	0.5±0.5	0±0	0±0	0±0	0±0	0.5±0.5	0±0	0.5±0.5	0±0	0.8±0.8	0±0	0.5±0.5
Diplodactylus vittatus	0.3±0.3	1±0	0±0	0.3±0.3	0±0	0±0	0.3±0.3	0±0	0±0	0.5±0.5	0±0	1.5±1.5
Diporiphora nobbi	0±0	0±0	0±0	0±0	0.3±0.3	0±0	0±0	0±0	0±0	0±0	0±0	0±0
Gehyra spp.	0±0	0±0	0±0	0±0	0±0	0±0	1.5±0.7	0±0	6±4	0.3±0.3	3.3±2	0±0
Heteronotia binoei	0±0	0±0	0.5±0.5	0.3±0.3	0.3±0.3	0.5±0.5	0.5±0.3	0.3±0.3	0±0	0±0	0±0	0±0
Lerista sp.	0±0	0±0	0±0	0.3±0.3	0.3±0.3	0±0	0±0	0±0	0±0	0±0	0±0	0.5±0.5
Morethia boulengeri	2.5±0.5	2.3±0.3	2±0	2.8±0.5	4.5±1.3	5±3	9.8±4.6	7.8±3.1	1.5±0.5	7±3.1	22±3.2	3±2
Parasuta dwyeri	0±0	0±0	0±0	0±0	0.3±0.3	0±0	0±0	0±0	0±0	0±0	0±0	0±0
Pogona barbata	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0.5±0.5	0±0	0±0	0±0
Strophurus williamsi	0.3±0.3	0±0	0±0	0±0	0.3±0.3	0±0	0±0	0±0	0.5±0.5	0±0	0±0	0±0
Vermicella annulata	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0.3±0.3	0±0	0±0

Table S3.3.5. Mean abundance of shrub (<2 m) and understorey plant species recorded in all treatments pre- and post-thinning.

Species	**Pre-thinning**						**Post-thinning**					
	Early thin control	Early thin	Early thin reference	Second thin control	Second thin	Second thin reference	Early thin control	Early thin	Early thin reference	Second thin control	Second thin	Second thin reference
Acacia deanei	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0.1±0.2	0±0	0±0
Acacia sp.	0.1±0.2	0.1±0.2	1±1.4	0±0	0.3±0.4	0.3±0.4	0±0	1.3±1.3	0.8±1.4	0.2±0.2	0.4±0.3	0.8±0.7
Acacia sp. (2)	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0.3±0.5	0±0
Allocasuarina luehmannii	0.2±0.3	0.3±0.3	0±0	0.4±0.3	0.1±0.2	0.2±0.3	1.1±0.8	1±1	0±0	0.7±0.5	0.1±0.2	0.2±0.3
Alternanthera sp.	0±0	0.3±0.5	0.3±0.4	0.3±0.5	0±0	0±0	0.3±0.6	0±0	0.7±0.6	0.9±1.1	0.1±0.2	0±0
Aristida sp.	1.9±1.3	1±1.1	1.3±2	8.4±5.3	3.7±2.3	3.2±1.8	1.4±1.2	6.6±5.4	0.3±0.6	18.8±15.8	5.1±3.6	1.2±1.3
Aristida sp. (1)	0±0	0±0	0±0	0±0	0±0	0.2±0.3	0±0	0±0	0±0	0±0	0±0	0±0
Aristida sp. (2)	0±0	0±0	0±0	0±0	0±0	0.2±0.3	0±0	0±0	0±0	0±0	0±0	0±0
Austrostipa sp.	8.8±3.5	10.9±5.5	5.7±4.9	3.3±2.2	11.8±7.1	9.5±16.5	5.9±3.8	38.9±20.6	7±4.7	0.9±0.6	23.1±15	41.8±30.2
Austrostipa sp. (1)	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0±0	1.9±1.9	0±0	0±0
Austrostipa sp. (2)	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0.8±0.8	0±0	0±0
Austrostipa sp. (a)	0±0	2.1±2.2	0±0	1.8±1.9	5±4.9	0±0	0±0	0±0	0±0	0±0	0±0	0±0
Austrostipa sp. (b)	0±0	0.3±0.5	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0±0
Austrostipa sp. (c)	0±0	0.6±0.8	0±0	3.6±4.2	1.6±2.3	0±0	0±0	0±0	0±0	0±0	0±0	0±0
Austrodanthonia sp.	0.1±0.2	0.2±0.2	0±0	0.3±0.3	0.2±0.3	0.8±0.8	0±0	0±0	0±0	0.2±0.3	0.2±0.3	0±0
Austrodanthonia sp. (1)	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0.5±0.9	0±0
Austrodanthonia sp. (2)	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0.1±0.2	0±0
Boerhavia dominie	0±0	0±0	0.3±0.4	0.2±0.3	0±0	0±0	0.2±0.3	1.3±1.5	0.2±0.3	0.6±0.7	0±0	0±0
Brachiaria sp.	0±0	0±0	1.3±1.6	0±0	0±0	0±0	1.7±1.3	1.8±2.3	1.3±1.6	0.4±0.6	6±8.1	1.8±2
Brachyscome sp.	0±0	0±0	0.5±0.6	0.1±0.2	0.3±0.5	0.3±0.4	2.5±1.7	2±2.3	3±1.8	10.3±6	4.9±4.4	0.5±0.8

Callitris sp.	0.2±0.3	0±0	0±0	0.1±0.2	0.3±0.3	0±0	0.4±0.6	1.2±1.1	0±0	0.2±0.3	0.2±0.2	0±0
Calotis sp.	1.3±1.4	7.3±6.9	19.2±14.1	9.3±5.9	13.4±8.9	9.7±4.7	55.9±15.2	122.4±26.1	45.2±20.2	11.9±8.3	10.7±9	39.8±12.2
Chamaesyce sp.	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0.5±0.5	0±0	0±0
Cheilanthes sp.	0±0	0±0	0±0	0.3±0.5	0±0	0±0	1.4±2.3	1.6±2.3	0±0	7.5±7.1	3.8±3.6	15.3±19.2
Chenopodium sp.	0±0	0±0	1.5±1.9	0±0	0±0	0±0	6.3±6.1	7.6±4.5	1.7±1.9	0±0	0.5±0.6	0±0
Chenopodium sp. (1)	0±0	0±0	0±0	0±0	0±0	0±0	0±0	2±3.5	0±0	0±0	0±0	0±0
Chenopodium sp. (2)	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0.3±0.5	0±0	0±0	0±0	0±0
Chenopodium sp. (3)	0.3±0.5	0±0	3±5.2	0.8±1	0.6±0.6	0±0	3.6±3.3	9.3±14.3	0±0	0±0	1±1	0±0
Convolvulus sp.	0±0	0±0	0.3±0.6	0±0	0±0	0±0	0.2±0.3	0.8±1	1.5±0.8	0±0	0±0	0±0
Conyza sp.	0±0	0±0	0±0	0±0	0±0	0±0	0.2±0.3	2.1±3.1	0±0	0±0	0.2±0.2	0.5±0.8
Craspedia sp.	0±0	0±0	0±0	0±0	0.1±0.2	0±0	0±0	0.8±1.3	0±0	0±0	0.3±0.4	0±0
Curry plant	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0±0	2.8±2.4	0.1±0.2	0.2±0.3
Cymopogon sp.	0±0	0.2±0.2	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0±0
Cyperus sp.	0±0	0.7±0.6	13.5±7.4	0±0	0.2±0.3	0.2±0.3	3.5±2.9	0.7±0.9	9.8±6.4	10.8±8.1	6.6±10.3	0.8±0.9
Daisy (1)	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0.2±0.3	0±0	0±0
Daisy (margin)	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0±0	5.5±6.6	0±0	0±0
Desmondium sp.	0.8±0.7	1.8±1.7	0±0	0±0	0.7±0.6	0±0	1.5±1.6	4.8±4	0±0	0±0	0.1±0.2	0±0
Desmondium sp. (1)	0±0	0.4±0.6	0.2±0.3	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0±0
Dianella sp.	0.7±0.6	0.2±0.3	0±0	0.9±0.6	0.3±0.3	0±0	0±0	0±0	0±0	0.1±0.2	1.3±1.6	5.5±8.5
Dichondra repens	0±0	0±0	0±0	0±0	0±0	0±0	7.4±11.8	0±0	0±0	0±0	3.4±3.9	0±0
Digitaria sp.	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0.4±0.7	0.2±0.2	0±0
Dodonaea sp.	0.5±0.6	0±0	0±0	0±0	0.2±0.2	0.2±0.3	0.3±0.3	0±0	0±0	0±0	0.1±0.2	0.5±0.8
Einadia sp.	7.1±4.8	9.5±4.5	0.5±0.6	0.3±0.3	3.1±2	2.2±3.1	3.4±2	9.6±8.6	0.8±0.9	0.3±0.3	0.1±0.2	4.3±3.4
Einadia sp. (1)	0±0	0±0	1.2±0.9	0±0	0±0	0±0	0±0	0±0	4.8±6	1.7±2	0.5±0.6	0±0
Einadia sp. (2)	0±0	0±0	0.3±0.4	0±0	0±0	0±0	0.3±0.6	0±0	0.7±0.6	0.1±0.2	0±0	0±0

Enteropogon acicularis	0±0	0.3±0.5	2.5±0.7	0.1±0.2	0±0	2.3±2.5	0±0	0±0	1±0.9	0.1±0.2	0±0	0±0
Eragrostis sp.	0.3±0.3	0.2±0.3	6.8±4	0.6±0.6	0.8±0.7	0.3±0.6	5.1±4.7	3.8±4.4	4.5±3.8	1.8±1.3	2±1.9	2±1.7
Eragrostis sp. (1)	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0±0	2.3±2.6	0±0	0±0
Eragrostis sp. (2)	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0±0	1.4±1.7	0±0	0±0
Eremophila sp.	0±0	0±0	0±0	0.1±0.2	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0±0
Eucalyptus sp.	0±0	0±0	0±0	0±0	0±0	0±0	0.2±0.3	0±0	0±0	0.3±0.5	0.4±0.7	0.2±0.3
Geijera parviflora	0.4±0.7	0±0	0±0	0±0	0±0	0±0	0.2±0.3	0.4±0.5	0.2±0.3	1.3±2.2	0±0	0±0
Glycine sp.	0±0	0.3±0.6	0.3±0.6	0.1±0.2	0.1±0.2	0.2±0.3	2.9±2.3	1.9±1.6	0±0	0±0	0.6±0.6	0.7±0.7
Goodenia sp.	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0.2±0.3	0±0	0±0
Ixiolaena sp.	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0.2±0.3	0±0	0±0
Juncus sp.	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0.5±0.6	0±0	0±0
Kennedia sp.	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0.2±0.3	0±0
Leiocarpa sp.	0±0	0±0	0.2±0.3	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0±0
Lomandra sp.	0.4±0.6	0.6±0.5	0.2±0.3	0.2±0.2	3.1±2.6	4.2±4.2	1.2±0.7	2±1.6	0±0	0.9±1.2	0±0	0.7±0.7
Lotus sp.	0±0	0±0	0.2±0.3	0±0	0±0	0±0	1.3±1.6	0±0	0.2±0.3	0±0	0±0	0±0
Monochather paradoxa	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0.1±0.2	0±0
Myoporum sp.	0±0	0±0	0±0	0±0	0±0	0±0	0.3±0.6	0±0	0±0	0±0	0±0	0±0
Nardoo sp.	0±0	0±0	0±0	0±0	0.1±0.2	0±0	0±0	0±0	0±0	0±0	0±0	0±0
Opuntia aurantiaca	0±0	0±0	0.2±0.3	0.1±0.2	0.4±0.4	0±0	0±0	0±0	0±0	0.3±0.6	0±0	0±0
Opuntia stricta	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0.2±0.3	0±0	0±0	0±0
Oxalis sp.	0±0	0±0	0±0	0±0	0±0	0±0	0.4±0.4	4.5±3.2	0.7±1.1	0.1±0.2	1.3±0.9	1.3±1.6
Panicum sp.	0.1±0.2	0.3±0.3	0±0	0.4±0.6	0±0	0.2±0.3	0±0	0±0	0±0	0.7±1	0.3±0.4	0.2±0.3
Paspalidium sp.	0±0	0±0	1.7±2.3	0±0	0±0	0±0	0±0	2.3±3.5	7.3±12.7	0±0	0.1±0.2	1.7±2.5
Pimelea sp.	0±0	0±0	0.2±0.3	0±0	0.2±0.2	0±0	0±0	0±0	0±0	0±0	0±0	0±0
Psoralea sp.	0±0	0±0	0±0	0±0	0±0	0±0	0.2±0.3	0±0	0±0	0±0	1.4±2.1	0±0

Rhagodia sp.	0±0	0.1±0.2	1.8±2.2	0±0	0.2±0.2	0±0	0±0	1.3±2.3	0±0	0±0	0±0	0.5±0.8
Rhagodia sp. (1)	0±0	0±0	0.3±0.6	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0±0
Rhodanthe sp.	0.3±0.4	0±0	0.2±0.3	3.3±2.7	2.3±2.1	0.8±1.1	0.7±0.7	0.3±0.6	0.8±1.1	3.3±4.1	4.8±3.2	0.3±0.6
Sclerolaena sp.	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0.7±1.1	0±0	0±0	0±0
Sida sp.	0±0	0±0	0±0	0±0	0±0	0±0	0.5±0.6	1.1±1.3	0±0	0±0	0.2±0.3	0.8±1.4
Sida sp. (1)	0.1±0.2	0±0	4.3±3.1	0.1±0.2	0±0	0±0	0±0	8.5±11.7	1±1.4	0±0	0±0	0.2±0.3
Sida sp. (2)	2.3±1.3	0.6±0.9	5±3.3	0±0	0.1±0.2	0±0	0.3±0.4	0±0	7.2±9.8	0±0	0±0	0.2±0.3
Sida sp. (purple)	0±0	0±0	0±0	0±0	0±0	0±0	0.4±0.7	0±0	0±0	0±0	0±0	0±0
Solanum ferocissimum	0.2±0.3	0±0	0.2±0.3	0±0	0.1±0.2	1.2±1.7	0.4±0.5	2.1±2.2	0.2±0.3	0.3±0.3	0±0	0±0
Solanum sp.	0.1±0.2	0±0	0±0	0±0	0±0	0±0	0.1±0.2	0.8±1.3	0±0	0±0	0.2±0.3	0±0
Sporobolus sp.	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0.2±0.3
Styphelia sp.	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0.3±0.6	0±0	0±0
Thyridolepis mitchelliana	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0.3±0.6	0±0	0±0	0±0	0±0
Trifolium sp.	0±0	0±0	0±0	0±0	0.3±0.6	0±0	0±0	0.3±0.6	0±0	0±0	0±0	0±0
Tripogon loliiformis	0±0	0±0	2.2±2.8	0±0	0±0	0±0	0±0	0±0	1.7±2.9	2±2.4	0±0	0.7±1.1
Unknown Daisy	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0±0	1±1.2	0±0	0±0
Unknown Daisy (1)	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0±0	3.3±3.9	0±0	0±0
Unknown Daisy (2)	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0.3±0.5	0±0	0±0
Unknown Grass	0.4±0.5	0.6±0.9	0±0	0.3±0.5	0.2±0.3	0.3±0.6	0.4±0.5	2.3±2.7	0±0	2.4±2.6	0.2±0.3	1.8±1.5
Unknown Grass (1)	0±0	0±0	0.5±0.8	0±0	0±0	0±0	0±0	0±0	0.5±0.8	0.6±0.9	0±0	0±0
Unknown Grass (2)	0±0	0±0	0.5±0.6	0±0	0±0	0±0	0±0	0±0	0.5±0.6	0.1±0.2	0±0	0±0
Unknown Grass (3)	0±0	0±0	0.3±0.4	0±0	0±0	0±0	0.1±0.2	0±0	0.3±0.4	0±0	0±0	0±0
Unknown Grass2	0.2±0.3	0.2±0.2	2.3±2.1	0±0	0±0	0.2±0.3	0.8±0.7	3.4±3.3	3.3±2.1	0±0	0.8±1.1	0±0
Unknown Herb	0±0	0.2±0.3	0±0	0±0	0.1±0.2	0±0	0±0	0±0	0±0	0.5±0.6	0±0	0±0
Unknown Herb (1)	4.4±1.5	4.7±3.1	2.3±2.8	0.9±1.2	0.8±0.8	2±3.5	1.4±0.9	4.8±2.7	11.3±13.3	1.4±1.6	1±0.7	1.5±1.1

Unknown Herb (2)	1±1.3	0.1±0.2	12.5±7.6	0±0	1.8±2.2	0±0	4.4±2.5	3.8±2.9	7±5.7	0.3±0.4	0.4±0.5	1.3±1.6
Unknown Herb (3)	0.3±0.3	0±0	1.7±2.2	0±0	0±0	0±0	4.4±1.9	3.3±3.2	1.5±2.3	0.3±0.4	1.8±2.1	0.3±0.6
Unknown Herb (4)	0±0	0±0	3±4.9	0±0	0±0	0±0	0.3±0.5	2.3±2.2	0±0	0.2±0.2	0.2±0.3	0±0
Unknown Herb (5)	0±0	0±0	0±0	0±0	0±0	0±0	0.2±0.3	3.5±4.3	0±0	0.1±0.2	0.1±0.2	0±0
Unknown Herb (6)	0±0	0±0	0±0	0±0	0±0	0±0	0.4±0.6	1.3±2.2	0±0	0±0	0.1±0.2	0±0
Unknown Herb (7)	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0.5±0.9	0±0	0±0	0.3±0.5	0±0
Unknown Herb (8)	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0.3±0.5	0±0	0±0	0.1±0.2	0±0
Unknown Herb (9)	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0.3±0.5	0±0	0±0	0±0	0±0
Unknown Herb (10)	0±0	0±0	8.7±15	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0±0
Unknown Purple Herb	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0.8±1.1	0±0	0±0
Unknown ligon	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0.5±0.8	0±0	0±0	0±0
Unknown native legume	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0.1±0.2	0±0
Unknown Shrub	0.6±0.6	0±0	0±0	0.3±0.6	0±0	0.3±0.4	0±0	0±0	0±0	0±0	0±0	0±0
Unknown Shrub (1)	2.6±3.6	0±0	0±0	8.3±8	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0±0
Unknown Shrub (2)	1.5±2	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0±0
Vittadinia sp.	0±0	0.5±0.6	0.3±0.6	0±0	1.3±1.6	0±0	1.9±1.2	11.1±9.6	0.3±0.6	0±0	1.3±1.5	0.5±0.6
Wahlenbergia sp.	0±0	0±0	0.2±0.3	0±0	0.2±0.2	0±0	0.2±0.3	1.7±2.4	0.2±0.3	0±0	0.8±0.9	1.8±1.5
Yellow shrub	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0±0	2.1±2.9	0±0	0±0
Yellow star	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0±0	0.1±0.2	0±0