CSIRO PUBLISHING
www.publish.csiro.au/journals/wr Wildlife Research, 2007, 34, 253–261

Penetration of remnant edges by noisy miners (*Manorina melanocephala*) and implications for habitat restoration

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Abstract. The noisy miner (*Manorina melanocephala*) is a large, communally breeding colonial native honeyeater renowned for aggressively excluding virtually all other bird species from areas they occupy. In the woodlands of southern and eastern Australia, numerous studies have identified the domination of remnants by noisy miners as having a profound negative effect on woodland bird communities. Despite this, very little is known about the habitat characteristics that make domination of a site by noisy miners more likely. This study investigated the depth from edges that noisy miners penetrated into large woodland remnants (>48 ha) within Victoria and attempted to identify habitat characteristics that influenced the depth to which they penetrated. Penetration depth differed significantly across four broad habitat types but commonly ranged from 150 m to more than 300 m from the remnant edge. If noisy miners colonise a site, their capacity to penetrate in from a remnant edge has implications for the size that remnants need to be (>36 ha) to contain any core 'noisy-miner-free' habitat and the width that habitat corridors need to be to avoid domination by noisy miners (>600 m). Broad differences in habitat type and the abundance of noisy miners at a site were the most powerful predictors of penetration distance. The density of canopy trees on a site was the only other habitat variable contributing to the most parsimonious model of penetration depth. Decreasing density of trees was associated with increasing penetration depth by noisy miners.

Introduction

Most natural ecosystems are experiencing major declines in biodiversity due primarily to habitat loss and fragmentation caused by humans: mainly clearing for agricultural production (United Nations Environment Program 1995; Trzcinski *et al.* 1999). Australia's temperate woodlands have been extensively cleared for agricultural use since European colonisation. It is estimated that 80–90% of the woodland habitat has now been cleared (Hobbs and Hopkins 1990; Robinson 1993*b*; Major *et al.* 2001), and much of what remains is severely fragmented and becoming increasingly degraded by livestock grazing and other human activities (Hobbs and Hopkins 1990). Vegetation clearance and fragmentation have been responsible for the widespread decline of many Australian woodland birds (Saunders 1989; Robinson 1991; Garnett 1992; Robinson 1993*a*; Barrett *et al.* 1994; Mac Nally 1999; Major *et al.* 2001). Many studies have examined the influence on declining woodland bird communities of a range of landscape and habitat features such as patch size and shape, landscape configuration (i.e. isolated or connected), vegetation structure and effects of livestock grazing (Loyn 1987; Catterall *et al.* 1991; Trzcinski *et al.* 1999; Mac Nally *et al.* 2000; Ford *et al.* 2001; Major *et al.* 2001).

In the woodlands of southern and eastern Australia, numerous studies have identified the domination of remnants by the noisy miner (*Manorina melanocephala*), a hyperaggressive native honeyeater, as having a profound negative effect on woodland bird communities (Dow 1977; Loyn 1987; Catterall *et al.* 1991; Catterall *et al.* 2002; Barrett *et al.* 1994; Grey *et al.* 1997, 1998; Mac Nally *et al.* 2000; Major *et al.* 2001; Catterall 2004). Piper and Catterall (2003) suggested that the noisy miner may be acting as a 'reverse keystone species' (*sensu* Simberloff 1998) throughout most of its range. The fragmentation of the landscape and sharp increase in the amount of edges allows 'open country' species such as the noisy miner to move into once-forested areas, to the detriment of 'forest interior' species (Lovejoy *et al.* 1986; Hobbs 1993; Murcia 1995; Luck *et al.* 1999). These 'open country' species (which include both native and introduced species) are able to thrive in these human-altered environments because they tend to have more general habitat requirements than 'forest-interior' species (Garrott *et al.* 1993; Harrison and Bruna 1999). It has been predicted that, in the future, populations of native species that have benefited from human disturbance will increase and overwhelm more sensitive species, leading to the 'biotic impoverishment' of many of the world's ecosystems (Noss 1990; Garrott *et al.* 1993).

The noisy miner is a large (~60 g), sedentary, communally-breeding, native honeyeater that forms colonies sometimes numbering hundreds of individuals (Dow 1977). They are considered unique among birds for their indiscriminate aggressive exclusion of most other birds from areas they occupy (Dow 1977; Loyn *et al.* 1983). There is substantial evidence that other small insectivorous birds are less abundant in remnants occupied by noisy miners (Ford and Bell 1981; Ford 1985; Loyn 1985, 1987; Catterall *et al.* 1991; Grey *et al.* 1997, 1998; Mac Nally *et al.* 2000; Major *et al.* 2001). Their preferred habitat

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is open eucalypt woodland, and they have been observed to preferentially occupy small remnants under 20 ha in size, as well as the edges of larger forest blocks (Dow 1977; Ford and Bell 1981; Loyn 1987; Barrett *et al.* 1994; Mac Nally *et al.* 2000; Catterall *et al.* 2002; Catterall 2004; Martin *et al.* 2006). Consequently, the noisy miner is one native species that has benefited from the clearing and fragmentation of the landscape. The noisy miner's domination of remnant vegetation within parts of its range is believed to have increased substantially (Low 2002). Across its distribution in New South Wales there has been an increase in reporting rate of noisy miners by an average of 15% over the 20 years to 2001 (Barrett and Silcocks 2002). Some authors suggest that the domination of remnant woodland by noisy miners is now the most important process threatening woodland bird communities in southern and eastern Australia (Major *et al.* 2001; Piper and Catterall 2003).

Since the early 1990s, large-scale revegetation projects have been carried out in Australia in an attempt to restore some of the native vegetation that was extensively cleared for agricultural production. Although the goal of much revegetation work is to improve habitat for wildlife (Wilkins *et al.* 2003), current revegetation guidelines are fairly general and concentrate on landscape-scale factors such as landscape connectivity, and optimal patch size, shape and configuration (Lambeck 1997; Watson *et al.* 2001; Brooker 2002). Despite the evidence highlighting the substantial impact of the noisy miner on bird communities in remnant eucalypt woodlands, very little is known about the habitat characteristics that make domination of a site by noisy miners more likely. It is important to ensure that current restoration efforts are not simply creating additional habitat for noisy miners.

A study by Piper and Catterall (2003) in subtropical Queensland found that noisy miners were occasionally penetrating remnants to depths of ~200 m from the edges of large woodland fragments. We aimed to replicate Piper and Catterall's (2003) study in a range of broad habitat types along the edges of large remnants (>48 ha) in the southern part of the species' range. The aim was to see whether, at sites where noisy miner colonies occurred, there were differences in penetration depth among different habitat types, and to provide insights into how wide remnants and corridors might need to be for them to include habitat that will not be dominated by noisy miners. We also aimed to identify habitat features that were influencing the depth to which noisy miners penetrated remnants.

Methods

Study sites and design

Each study site ($n = 28$) was centred on a separate and discrete noisy miner colony located on the edge of a large forest or woodland remnant (195–29 000 ha) (Fig. 1). Some sites were located within the same large remnant. However, the minimum distance between sites was 385 m, and most sites (71.4%) were at least 1 km apart. The mid-point of the noisy miner colony along the remnant edge was defined as the point at which a standard playback of noisy miner territorial vocalisations elicited the greatest response from the noisy miners present. The distance that noisy miners penetrated into the fragment from the edge was determined by conducting point counts along four transects running perpendicular to the remnant edge at each study site, placed 25 m and 75 m either side of the colony mid-point. Counts were carried out at the remnant edge, then every 50 m to a depth of 300 m into the remnant (Fig. 1).

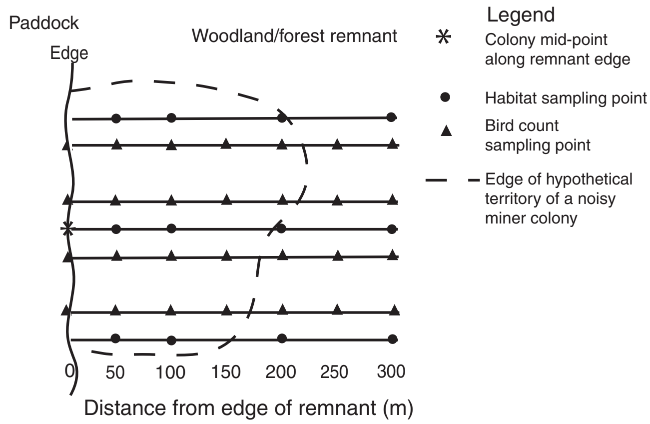

Fig. 1. The sampling layout of transects and survey points at a site.

Piper and Catterall (2003) had reported noisy miners penetrating up to 200 m into subtropical woodlands. Therefore, in an attempt to ensure the detection of noisy miners at maximum distances from the edge, transects were run to a depth of 300 m from the edge of the remnant. In order for all survey points to be at least 300 m from any other clearing other than the remnant edge, remnants had to be at least 600 m deep and 800 m long, and hence the study focussed on remnants at least 48 ha in area. Two minutes were spent at each sample point ($n = 28$) within a site, recording the number of individuals of all species of bird seen or heard within a 20-m radius of the observer. Birds that were seen flying more than 3 m above the canopy were classified as being outside the sampling area. All surveys were conducted by the same observer (J. Oldland) to avoid interobserver variability. Surveys were conducted only during fine weather with no strong winds, and were confined to before 1130 hours and after 1530 hours in summer (depending on the weather conditions at the time), when birds were most active. During winter, counts were conducted throughout the day. Two rounds of surveys were conducted at all sites: once between late October 2004 and early March 2005 and again during August 2005.

The depth to which noisy miners penetrated into fragments was quantified in two different ways. The first method was based upon the mean number of noisy miners detected at point counts conducted at each distance from the remnant edge (0–300 m). In the second method a single penetration value was calculated for each study site, based on the maximum distance that noisy miners were detected away from the fragment edge along each of the four bird survey transects in either season. The median of the eight (four transects in each season) maxima was used as the estimate of the penetration distance of the colony into the remnant at a site. The distribution of values of penetration distance was scrutinised and found to be highly kurtotic and skewed. Several transformations were tried and a reflected arcsine-square-root transformation (Tabachnick and Fidell 1996) proved to be the best at reducing both kurtosis and skewness.

Habitat variables

Habitat characteristics of a site were recorded along three transects, placed at the mid-point of the noisy miner colony, and 100 m either side of the mid-point. Along each transect, habitat characteristics were surveyed at depths of 50, 100, 200 and 300 m into the remnant (Fig. 1). The basic sampling unit was the area within a 25-m radius of a fixed point along a transect. Nineteen habitat variables were measured at each sampling point (Table 1). Data were pooled for each site, giving a total of 28 replicates.

Multiple regression was used to identify habitat variables that accounted for variation in penetration depths of noisy miners across sites. Preliminary analysis of the habitat variables revealed that some variables lacked sufficient variability across the 28 study sites to be informative (e.g. flowering trees) and so were omitted from further analyses. Furthermore, some variables were so strongly correlated with one or more other habitat variables that their inclusion in multiple regression models would have violated collinearity assumptions (Quinn and Keough 2002). The six remaining variables (broad habitat type, noisy miner abundance, tree distance, tree species richness, shrub cover and ground cover) were examined to identify which violated the assumptions of multiple regression and required transformation. Consequently, tree distance was $\log_{10}$-transformed, the arcsine-square-root transformation was applied to shrub and ground cover values and the square-root transformation was applied to tree species richness. All 64 possible subsets of these six predictor variables were modelled using SPSS ver. 12. Hierarchical partitioning was used to identify which of the six variables made the most important contribution to the models generated to account for penetration depth (Chevan and Sutherland 1991). Second-order Akaike's information criterion corrected for small sample size (AIC_c) and Bayesian information criterion (BIC) were used to identify the most parsimonious of the 64 possible models (Quinn and Keough 2002; Burnham and Anderson 2004), i.e. smaller values of AIC_c or BIC indicate models that best fit the data with the fewest number of parameters.

Habitat descriptions

The study was carried out in four woodland and forest habitat types within Victoria where noisy miners are abundant: box–ironbark forests, low-rises grassy woodlands, riverine grassy woodlands and Gippsland Plains grassy woodlands. All sites were located on the edges of remnant forest blocks within an agricultural matrix. The land immediately adjacent to the remnant edges was dominated by pasture, with occasional scattered paddock trees throughout.

Box–ironbark forests

Seven sites were located in box–ironbark forest habitat, occurring within the Goldfields bioregion of Victoria in Whroo-Rushworth State Forest (36°39′39″S, 145°3′43″E), Heathcote-Greytown National Park (36°50′23″S, 144°48′27″E) and Redcastle State Forest (36°43′29″S, 144°47′51″E), all of which form part of the Rushworth Forest Block; and Wellsford State Forest (36°41′49″S, 144°24′4″E). Forest blocks ranged in area from 10 700 to 29 000 ha. These sites occurred on gently

Table 1. Summary of habitat variables measured at each sample point

Heights and distances (>3 m) were measured with an Optilogic laser rangefinder (accuracy ±1–2 m depending on reflectivity of object). Measurements for percentage cover were taken in four 1-m^2 quadrats placed 25 m to the north, south, east and west of the sampling mid-point. DBH is diameter at breast height

Habitat variable	Description
Canopy structure variables	
Tree distance	Distance from the sampling mid-point to the nearest five trees (with a DBH >10 cm) (m)
Tree girth	Girth of the five nearest trees (m)
Tree canopy distance	Distance between the canopy edges of the five nearest trees and the canopy edges of their nearest neighbours (m)
Tree height	Height of the five tallest trees (m)
Dead trees	Presence of dead trees (>10 cm DBH)
Plant diversity variables	
Tree species richness	Number of species of canopy tree (including *Acacia* spp. >5 m tall)
Tree species identity	Identity of all canopy species present recorded
Flowering trees	Number of flowering trees present (up to 10 trees)
Shrub species richness	Number of species of woody shrubs (>0.3 m and <2 m tall)
Understorey structure variables	
Shrub density	Number of woody shrubs (>0.3 m tall)
Stump density	Number of stumps
Log density	Number of logs (fallen timber >8 cm DBH)
Percentage cover variables	
Shrub cover	% woody shrub cover (>0.3 m tall)
Groundcover	% ground cover (grass, sedge or herb)
Rock	% rock cover
Bare	% bare ground cover
Litter	% litter cover
Lichens, bryophytes, mosses	% cover lichens, bryophyte and mosses
Height ground cover	Heights of ground cover (grass, sedge, herb) measured in four corners of each quadrat ($n = 16$ measurements) (cm)

undulating rises and low hills, at elevations of 148–205 m. Soils in these areas are typically fairly shallow and infertile, and the average rainfall is 400–600 mm per annum (Environment Conservation Council 2001). The dominant canopy species was *Eucalyptus microcarpa*, which commonly occurred with other species including *E. tricarpa*, *E. leucoxylon*, *E. macrorhyncha*, *E. polyanthemos*, *E. melliodora* and *E. camaldulensis*. The maximum height of trees at these study sites averaged 24 m. The understorey vegetation ranged from dense to open and supported a range of species including *Acacia* spp., *Hibbertia* spp., Asteraceae (e.g. *Cassinia arcuata*), Fabaceae, *Bursaria spinosa*, *Brachyloma ciliatum*, *Calytrix tetragona* and *Astroloma humifusum*, along with some native grass and herb species.

Low-rises grassy woodlands

An additional three sites were located within the Goldfields bioregion, which were distinct from the other seven box–ironbark forest habitat sites, being classified as low-rises grassy woodland habitat. These sites were located in Kamarooka State Forest (36°30′25″S, 144°27′34″E) and Runnymede Flora and Fauna Reserve (36°34′32″S, 144°43′48″E). Remnants and forest blocks ranged in area from 300 to 5400 ha. Sites in low-rises grassy woodland habitat occurred on gently undulating terrain and flood plains of minor creeklines, at elevations of 134–218 m. Soil fertility and water availability in these areas is generally moderate, supporting larger trees and a higher floristic diversity than box–ironbark forest (Muir *et al*. 1995; Environment Conservation Council 2001). The canopy was open, and was dominated by *E. microcarpa* or a mix of *E. microcarpa* and *E. albens*. The maximum height of trees at these study sites averaged 23.7 m. The shrub layer was generally sparse. However, *C. arcuata* was present at high densities at some sites. Various *Acacia* spp., *Dodonaea viscosa* and *B. spinosa* were also common. This habitat type supported a sparse, yet diverse ground layer of native grasses and herbs (Muir *et al*. 1995).

Riverine grassy woodlands

Ten sites were located within the Victorian Riverina bioregion, in riverine grassy woodland habitat. Seven sites were located along the Goulburn River, between Cooma Bend State Forest (36°18′31″S, 145°20′57″E) and Daunt's Bend State Forest (36°29′39″S, 145°21′5″E). The remaining three sites were located at Doctor's Swamp (36°18′31″S, 145°20′57″E) and Reedy Lake State Forest (36°43′13″S, 145°5′14″E). Remnants and forest blocks ranged in area from 263 ha to more than 4700 ha. Sites within riverine grassy woodland habitat occurred in the floodplains of major rivers and streams, and on flat or gently undulating plains at elevations of 114–155 m. Soils in this habitat type are fertile silts and sands, and the average rainfall is 300–700 mm per annum (Department of Sustainability and Environment 2004). The canopy was fairly open, consisting of *E. camaldulensis* and *E. largiflorens*. The maximum height of trees at these study sites averaged 29.2 m. The most common species occurring in the sparse shrub layer were *Acacia dealbata* and *C. arcuata*. The ground layer was typically dominated by native grass and sedge species, with amphibious species occurring in areas prone to inundation (Department of Sustainability and Environment 2004).

Gippsland Plains grassy woodland

Eight sites were located within the Gippsland Plains bioregion, in Gippsland Plains grassy woodland habitat, at Swallow Lagoon Nature Conservation Reserve (37°54′5″S, 147°9′35″E), Providence Ponds Flora and Fauna Reserve (37°56′40″S, 147°19′60″E) and Moormung Flora and Fauna Reserve (37°55′13″S, 147°31′11″E). Remnants and forest blocks ranged in area from 195 to 1650 ha. Sites within Gippsland Plains grassy woodland habitat occurred on fertile soils on flat or gently undulating plains at elevations of 34–134 m. The canopy was typically dominated by *E. tereticornis*. However, *E. polyanthemos*, *E. macrorhyncha*, *E. bosistoana*, *E. bridgesiana*, *E. melliodora* and *E. viminalis pryoriana* also occurred at various sites. The maximum height of trees at study sites averaged 23.8 m. The understorey was open and often included a small tree layer that supported species such as *Acacia implexa*, *A. melanoxylon*, *A. mearnsii*, *A. paradoxa*, *Allocasuarina littoralis*, *Melaleuca parvistaminea* and *Kunzea ericoides*. The ground layer in this habitat was diverse, and usually 'grassy', with herbs, sedges and lilies also present (Davies *et al*. 2002).

Results

The distance to which noisy miners penetrated into remnants (hereafter referred to as depth), among the 28 sites examined, ranged from 100 m to more than 300 m from the edge of the remnant (Fig. 2). Noisy miners penetrated significantly different depths in different habitat types (one-way ANOVA, $F = 4.94$, d.f. = 3,24; $n = 28$, $P = 0.008$). Penetration depth was greater in low-rises grassy woodland and riverine grassy woodland habitats than in box–ironbark forest and Gippsland

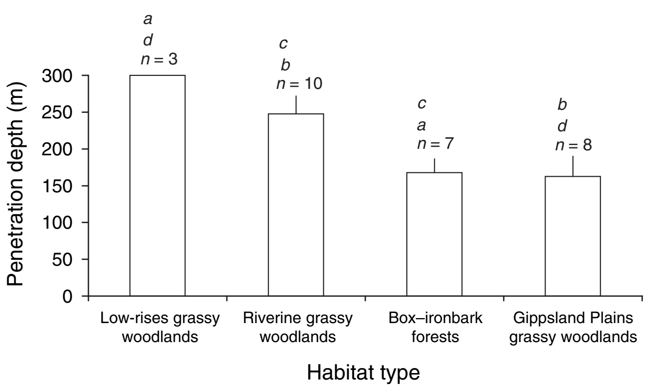

Fig. 2. Mean maximum penetration depth of noisy miners in each broad habitat type. The penetration depth at a site was determined by calculating the median of the maximum depth at which noisy miners were detected along each of the four bird survey transects in either season. A mean was calculated from these values for the sites, within each broad habitat type. Letters above columns indicate habitats that differed significantly according to an l.s.d. *post hoc* comparison test. For (*a*) (low-rises grassy woodlands) $P = 0.01$, for (*b*) (riverine grassy woodlands) $P = 0.015$, for (*c*) (box–ironbark forests) $P = 0.026$, and for (*d*) (Gippsland Plains grassy woodlands) $P = 0.007$. Error bars depict one standard error around the mean.

Plains grassy woodland habitats (Fig. 2). The mean number of noisy miners detected at each depth also differed significantly between regions (Fig. 3). Noisy miners were generally abundant at all depths in both low-rises grassy woodland habitat and riverine grassy woodland habitat (Fig. 3*a*, *b*). However, in box–ironbark and Gippsland Plains grassy woodland habitats there was a gradual decline in abundance of noisy miners with increasing depth (Fig. 3*c*, *d*).

Of the six variables considered, hierarchical partitioning revealed that broad habitat type and the abundance of noisy miners at a site made the greatest independent contributions to improving the fit of the 64 models considered (Table 2, Fig. 4). The remaining four variables (tree distance, tree species richness, shrub cover and ground cover) made comparatively small, but similar, contributions to improving the fit of models. AIC_c and BIC information criteria both identified a four-factor model, that included broad habitat type, the abundance of noisy miners, tree distance and percentage shrub cover as the most parsimonious model (Table 3) ($r^2 = 0.74$, $F = 9.8$, d.f. = 6,21, $P < 0.001$, $AIC_c = -32.4$, $BIC = -31.3$). The independent contribution of percentage shrub cover to this model was not significant, and an equivalent three-factor model, from which percentage shrub cover had been removed, had very similar explanatory power (Table 4) ($r^2 = 0.73$, $F = 11.74$, d.f. = 5,22, $P < 0.001$, $AIC_c = -28.8$, $BIC = -27.5$). Although the relationship between penetration depth and tree distance varied between regions (Fig. 5), and this influenced the slope in the final model, overall there was a positive, though non-significant relationship when data from all regions were pooled ($r = +0.23$, d.f. = 28, $P = 0.25$).

Total avian species richness and species richness of birds smaller than noisy miners were significantly lower ($n = 9$, median = 14.5 and 3.5 species respectively) at sites where noisy miners penetrated to depths of 300 m or more (Mann–Whitney U-test, total species richness U = 17.0, $P = 0.001$, birds smaller than noisy miners U = 4.0, $P < 0.001$),

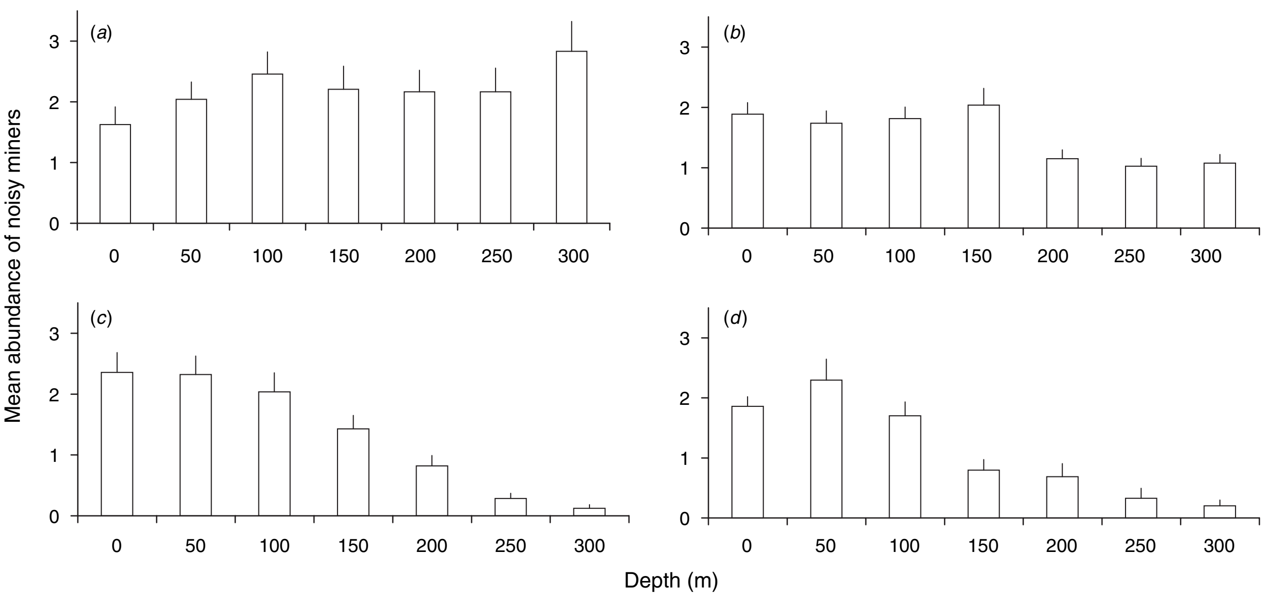

Fig. 3. Mean abundance of noisy miners per sampling point at each depth across all broad habitat types. (*a*) low-rises grassy woodlands ($n = 3$ sites); (*b*) riverine grassy woodlands ($n = 10$ sites); (*c*) box–ironbark forests ($n = 7$ sites); (*d*) Gippsland Plains grassy woodlands ($n = 8$ sites). Error bars depict one standard error around the mean. A mixed-plot repeated-measures ANOVA, in which subjects were sites and the within-subjects factor was distance from the remnant edge, revealed significant differences between the four regions in the number of noisy miners at different depths, $F = 4.7$, d.f. = 11,89 (with Greenhouse–Geisser adjustment) , $n = 28$, $P < 0.001$.

Table 2. Hierarchical partitioning of multiple regression results utilising all 64 possible model subsets

R is a measure of zero-order association between the dependent variable (penetration depth) and the independent variables (see below). I is the independent component of R, and J is the joint component of R (Chevan and Sutherland 1991). % I and % J indicate the percentage distribution of effects, i.e. the mean proportion of the explained variability accounted for by each variable

Regression effects	Independent variables: Broad habitat type	Tree distance	Tree species richness	High and low shrubs (%)	Ground cover (%)	Abundance of noisy miners	Total
R	0.43	0.05	0.20	0.06	0.08	0.50	1.31
I	0.33	0.05	0.08	0.06	0.05	0.29	0.84
J	0.10	0.00	0.12	0.00	0.03	0.21	0.47
% I	39.21	5.64	9.05	6.61	5.49	34.01	100.00
% J	21.18	0.97	26.19	−0.13	6.30	45.23	100.00

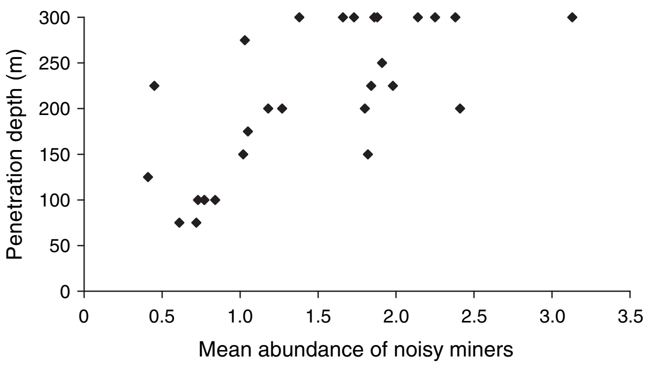

Fig. 4. Relationship between the mean abundance of noisy miners per sampling point at each site and the maximum penetration depth at each site.

compared with sites where noisy miners penetrated less than 300 m ($n = 19$, median = 21.5 and 10.5 species respectively) (birds were deemed smaller than noisy miners if their mean length, as reported in Pizzey and Knight (1997), was <25.5 cm). Not surprisingly, there were also significant negative correlations between the mean abundance of noisy miners and the total avian species richness at a site ($r = -0.44$, d.f. = 26, $P = 0.019$) and the species richness of bird smaller than noisy miners at a site ($r = -0.45$, d.f. = 26, $P = 0.008$).

Discussion

Noisy miners penetrated to depths of at least 300 m at some sites in each of the four broad habitats surveyed. This is considerably further than Piper and Catterall (2003) found in their study, where only 6% of noisy miner records occurred at depths greater than 200 m into the remnant. Because the four broad habitat types we chose to examine were unavoidably located in different regions of the state, the possibility exists that the difference in penetration depths we detected between the four habitat types (and the findings of Piper and Catterall (2003)) could be a result of regional differences in the characteristics of remnants, e.g. time since clearing, duration of noisy miner occupation of the region, total vegetation cover available in the region, soil fertility, or the nature of the surrounding agricultural matrix. Our study of various local habitat characteristics at each site shed little light on possible local factors influencing the distance noisy miners penetrated into a remnant. Apart from broad habitat type and the abundance of noisy miners at a site, the only remaining habitat variable that made a substantial contribution to the most parsimonious model of penetration depth was the density of canopy trees on a site. It is interesting to note that the shallower penetration depths reported by Piper and Catterall (2003) were for study sites in subtropical open forests with a higher tree density than in Victorian woodlands. This might account for the differences in penetration depths detected in the two studies.

The methodology of our study was built upon the penetration depths reported by (Piper and Catterall 2003), i.e. rarely greater than 200 m. We attempted to go well beyond this depth in our sampling design (to 300 m) to cater for the possibility that noisy miners might penetrate further in southern Australia. In hindsight, it would have been preferable to have surveyed even deeper at some sites and our findings should be viewed as potentially under-estimating maximum penetration distances by noisy miner colonies. However, if we had set the condition that sites had to be located in remnants at least 400–500 m from any other clearing (i.e. 800 m deep and 1000 m long), this would have so drastically constrained the number of locations suitable for study sites that our sample size would have been too small for meaningful analysis. We struggled to find just 28 sites with noisy miners that met the size criteria for 300 m in the whole of the state of Victoria.

In our attempt to determine maximum penetration distances from edges by noisy miners, we were also unavoidably compelled to study the edges of *remnant* vegetation, as opposed to patches of revegetation planted by humans. This was because too few revegetation patches had the appropriate shape or area (>48 ha) to accommodate our study design. Despite this limitation, we believe our findings are of relevance to current and future revegetation efforts for biodiversity, should such revegetation efforts be colonised by noisy miners.

Piper and Catterall (2003) estimated that with an edge effect 200 m deep, remnants less than 13 ha in size were likely to be totally dominated by noisy miners. However, since our study found that noisy miners can routinely penetrate to depths of 300 m in some habitats, a remnant colonised by noisy miners would actually need to be over 36 ha in area in order to avoid

Table 3. The most parsimonious multiple regression model explaining variation in maximum penetration depth of noisy miners

Since broad habitat type was a categorical variable with four categories, it had to be entered as three dummy variables, with one category (riverine grassy woodlands) arbitrarily designated as the reference category (Quinn and Keough 2002). $n = 28$

Variable	B	s.e.	β	t	P
Broad habitat type intercept (riverine grassy woodlands)	2.81	0.65		4.21	<0.001
Box-ironbark forests	−0.81	0.24	−0.72	−3.33	0.003
Gippsland Plains grassy woodlands	−0.73	0.22	−0.67	−3.27	0.004
Low-rises grassy woodlands	−0.11	0.23	−0.07	−0.47	0.640
Noisy miner abundance	0.33	0.11	0.46	2.97	0.007
Tree distance ($\log_{10}$-transformed)	−1.19	0.59	−0.37	−2.00	0.058
Shrub cover (arcsine-square-root transformed)	0.63	0.73	0.16	0.86	0.399

Table 4. Multiple regression of variables explaining variation in maximum penetration depths of noisy miners
$n = 28$

Variable	B	s.e.	β	t	P
Broad habitat type intercept (riverine grassy woodlands)	2.55	0.58	–	4.42	<0.001
Box–ironbark forests	−0.67	0.18	−0.59	−3.75	0.001
Gippsland Plains grassy woodlands	−0.65	0.20	0.60	−3.22	0.004
Low-rises grassy woodlands	−0.01	0.20	−0.01	−0.07	0.947
Noisy miner abundance	0.39	0.09	0.54	4.25	<0.001
Tree distance ($\log_{10}$-transformed)	−0.96	0.53	−0.30	−1.82	0.082

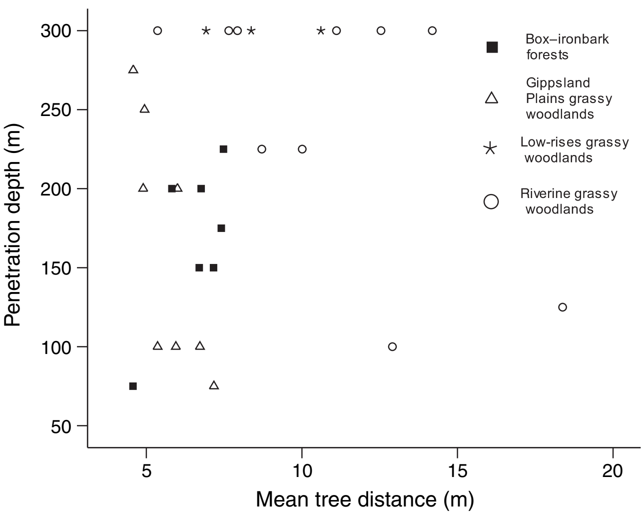

Fig. 5. Relationship between mean tree distance and maximum penetration depth at each site.

being entirely occupied by noisy miners in the habitats we examined.

Another possible implication from our study is that, given their width, many vegetation corridors currently being planted within the noisy miner's range are likely to become totally dominated by noisy miners. Major *et al.* (2001) also warned that a focus on creating landscape connections between fragments was likely to be creating additional habitat for noisy miners, hence strengthening their hold over the landscape. Our findings support this conclusion and suggest that corridors need to be much wider in order to avoid total occupation by noisy miners. For example, in Gippsland Plains grassy woodlands, where noisy miners typically penetrated remnants to depths of ~150 m, corridors would need to be more than 300 m wide in order to not consist entirely of 'edge habitat' and hence be dominated by noisy miners. Because noisy miners penetrate further into remnants in some habitats than in others, revegetation guidelines may need to be tailored to suit individual habitat types.

In many cases it will not be feasible to plant corridors with sufficient width to include noisy miner–free habitat. Nevertheless, it should be noted that although corridors <300 m wide may be of limited value to many birds, they can be extremely important for other biota such as small mammals and reptiles (Bennett 1990, 1999). Clearly a range of different revegetation strategies is needed to meet the needs of different taxa. We propose that revegetation projects aimed at the conservation of woodland birds need to place more emphasis on increasing the size of existing remnants, rather than on just establishing corridors.

It has been well established that noisy miners preferentially occupy and dominate small remnants less than 20 ha in size. There is also a considerable body of correlational evidence that suggests that where noisy miners occupy these small remnants, the species richness and abundance of other small birds is reduced (Loyn 1987; Ford *et al.* 1995; Catterall *et al.* 1997; Grey *et al.* 1997; Mac Nally *et al.* 2000; Major *et al.* 2001). However, the results of our study suggest that noisy miners also have negative impacts on woodland bird communities along the edges of *large* remnants in a range of habitat types within Victoria. Increased penetration depth of noisy miners into a remnant resulted in declines in both total avian species richness, and species richness of birds smaller than noisy miners at the site level. Our study suggests that noisy miners may actually be occupying and dominating a larger portion of the remnant vegetation left within their range, and hence pose an even larger threat to declining woodland birds than has previously been appreciated.

Although this study had only limited success identifying local habitat features that influence the depth to which noisy miners will penetrate into remnants, further research comparing sites with and without noisy miners may shed light on habitat features that increase the likelihood that a site will be colonised by noisy miners. Insights from such research (e.g. Hastings and Beattie 2006) then need to be incorporated into future restoration efforts to ensure that current and future good intentions do not further facilitate the domination of the landscape by noisy miners.

Acknowledgements

This research was supported by a Discovery Grant from the Australian Research Council to Dr Michael Clarke. We are grateful to Rick Taylor for providing unpublished data. David and Ruth Read provided vital local knowledge and warm hospitality in Gippsland. Thank you to Merilyn Grey and Ian Davidson for their knowledgeable input. This paper also benefited greatly from the constructive comments of Damon Oliver, Carla Catterall and Ralph Mac Nally. Finally, we are grateful to Daniel Hesketh, Lauren Brown and Monica Hersburgh for assisting in the field.

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Manuscript received 11 October 2006, accepted 23 May 2007