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Towards effective management of an overabundant native bird: The noisy miner

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Abstract

Addressing threats to biodiversity from pest species is a global challenge. One such challenge is to mitigate the impact of an overabundant Australian songbird, the noisy miner *Manorina melanocephala,* on woodland birds. The overabundance of noisy miners is listed as a key threatening process under federal biodiversity legislation, but current understanding of where and how noisy miner populations can be managed to yield conservation benefits is unclear. We evaluated the effectiveness of noisy miner removal across 12 treatment areas totaling 3913 ha and nine control areas totaling 1487 ha important for the critically endangered regent honeyeater *Anthochaera phrygia*. Removal of noisy miners significantly reduced their densities in all but one of the treatment areas. In 10 of the 12 treatment areas, noisy miner densities remained below an impact threshold of 0.65–0.83 birds ha^{-1} for at least 3 to more than 12 months. The percentage of suitable noisy miner habitat in the surrounding landscape was not a strong predictor of noisy miner management success. Regent honeyeaters occupied six treatment areas, nesting successfully in four. The abundance of other songbirds increased post-miner removal in seven areas, decreased in three, and was mixed in two. Data from the control areas showed some variation in songbird numbers was independent of noisy miner management. We conclude that noisy miners can be managed in areas of high conservation value for a minimum cost of AUD $10 ha^{-1}. Larger treatment areas may be more important than the broader landscape context in maintaining long-term noisy miner suppression. Standardized, long-term monitoring is crucial to identify not only the drivers of pest species recolonization but also locations where threats from pests on endangered species can be addressed effectively while minimizing animal welfare and financial costs.

KEYWORDS

applied ecology, Australia, conservation, evidence synthesis, invasive species, nest survival, nomadic species, ornithology, pest management, population monitoring

1 | INTRODUCTION

Anthropogenic habitat change can alter the outcomes of interactions between co-occurring species to the benefit of one and the cost of others (Didham et al., 2007). Mitigating the impacts of pest species, whose populations and behaviors contribute to the decline of competitors, is a priority for preventing extinctions globally (Blackburn et al., 2004; Clavero & Garcia-Berthou., 2005; Doherty et al., 2016). With sufficient resource investment, suppression or eradication of such species can be extremely successful on islands (Holmes et al., 2015) but is harder to achieve and sustain in continental environments where source populations are often mobile and widespread (Kopf et al., 2017). Furthermore, while many pest species are introduced, this is not always the case (Brittingham & Temple, 1983). Managing native species whose populations have detrimental community-level impacts creates a management headache from an ethical perspective (Soulé et al., 2005), justifying a rigorous assessment of potential risks and benefits of culling for conservation (Kopf et al., 2017).

In South-Eastern Australia, the noisy miner *Manorina melanocephala*, a native songbird, has benefited greatly from widespread woodland clearance and fragmentation over the past century (Commonwealth of Australia, 2021; Mac Nally et al., 2012). Noisy miners are edge specialists and prefer to occupy sparsely to moderately timbered habitats with minimal understorey vegetation, or the edges of denser vegetation patches (Barati et al., 2016; Piper & Catterall, 2003; Val et al., 2018). Living in complex cooperative social groups (Higgins et al., 2001), noisy miners can reduce the abundance of co-occurring songbirds through interference competition, such as mobbing, within their territories (Commonwealth of Australia, 2014). These territories can be identified based on noisy miners exceeding a density of *c*0.65–0.83 birds ha^{-1} (Mac Nally et al., 2012; Thomson et al., 2015). Even at lower densities, noisy miners pose a threat to some nesting birds through disturbance or nest destruction (Crates et al., 2019). In response to the risk they pose to a suite of threatened woodland birds (Ford et al., 2001), noisy miners are listed as a key threatening process under federal biodiversity legislation (Commonwealth of Australia, 2014).

Some recent studies have attempted to suppress noisy miner populations via culling, but with highly variable outcomes. In the Northern Tablelands of New South Wales (NSW), Davitt et al. (2018) found little evidence that removing 3552 noisy miners from 12 woodland remnants reduced their abundance relative to control sites. At some treatment sites, noisy miners commenced recolonization just hours after culling ceased. Despite a negligible drop in noisy miner numbers, Davitt et al. (2018) detected an increase in songbird abundance and species richness at treatment sites. This was potentially due to culls disrupting noisy miner social structure, impacting their ability to exclude co-occurring bird species. Beggs et al. (2019) removed 538 noisy miners during repeat culls at 8 × *c*13 ha patches in the South-West slopes of NSW but observed their immediate recolonization. Despite a 22% decline in noisy miner abundance at treatment compared to control sites and a sustained increase in small birds, Beggs et al. (2019) concluded that noisy miner culling was uneconomical in such highly modified landscapes because, despite the substantial financial investment, post-cull noisy miner densities remained well above the 0.65 miners ha^{-1} impact threshold previously published (Thomson et al., 2015).

Other studies have reported successful reductions in noisy miner densities, and consequent improvement in measures of smaller birds, using an affordable management regime. Grey et al. (1997, 1998) reported noisy miner removal led to sustained reductions in noisy miner populations lasting at least 16 months and an associated increase in bird abundance and species richness at most treatment sites. Debus (2008) reported the successful suppression of noisy miners and subsequent recovery of bird species richness following a small-scale noisy miner cull, combining repeat culls with revegetation efforts in a single 15-ha woodland remnant. Crates et al. (2018) removed 350 noisy miners from 430 ha of woodland in the NSW Blue Mountains. This led to an increase in songbird abundance relative to a control area, allowing regent honeyeaters *Anthochaera phrygia* to nest there successfully (Crates et al., 2018). The regent honeyeater is a Critically Endangered songbird whose population has been impacted negatively by the spread of noisy miners (Commonwealth of Australia, 2016). There may now be fewer than 300 wild regent honeyeaters, so reducing the impact of noisy miners in regent honeyeater breeding areas will be crucial in saving the species from extinction (Heinsohn et al., 2022). Follow-up monitoring revealed that noisy miner numbers at one cull area remained suppressed for more than 1 year following the original cull (Crates et al., 2020). Other targeted culls in NSW have also led to a sustained reduction in noisy miner numbers (BirdLife Australia, unpublished data). These results provide evidence that noisy miner management can be effective in some landscapes.

To minimize the ethical costs associated with culling noisy miners and to maximize conservation returns on investments, there is a need for a better understanding of the factors determining the success of noisy miner culls. Predicting how and where noisy miner culling is most likely to lead to sustained noisy miner suppression can help inform spatial and logistical prioritization of

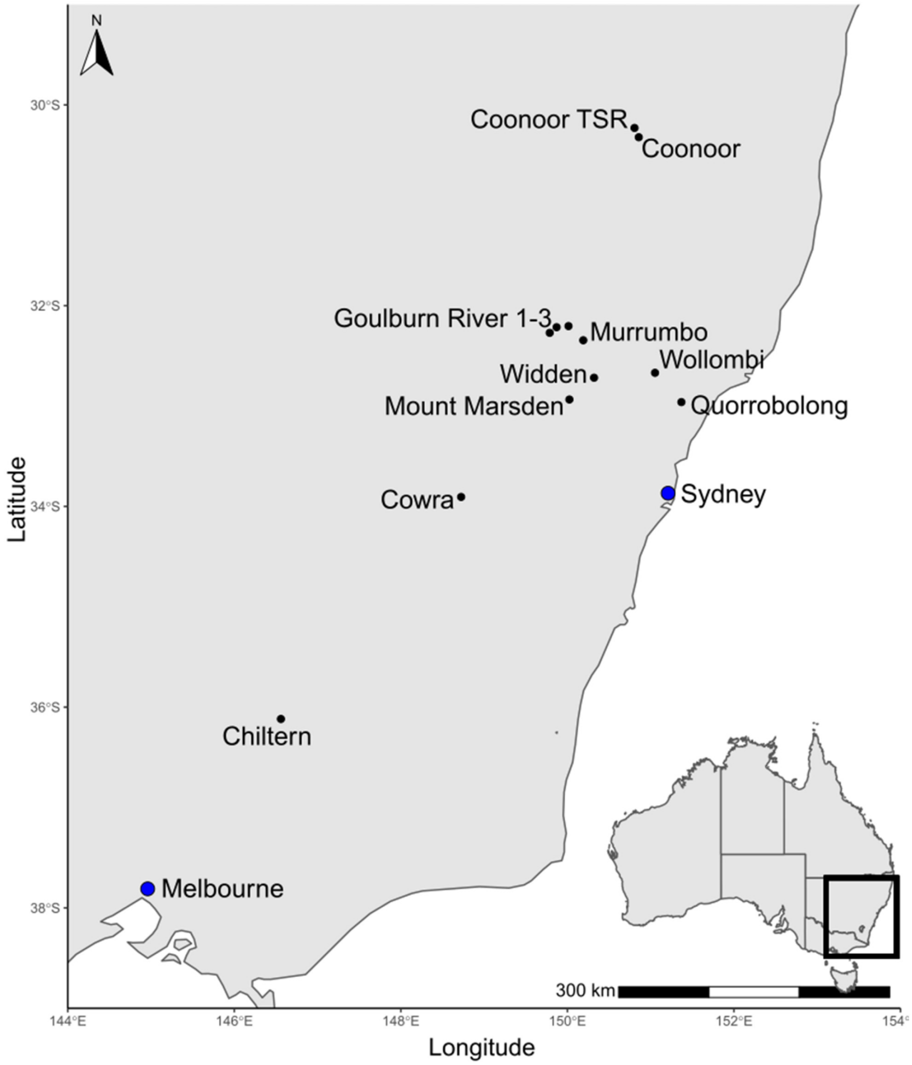

FIGURE 1 Location of noisy miner management areas. Locations of Sydney and Melbourne are shown for reference

resource investment, potentially helping prevent the extinction of the most at-risk species (Crates et al., 2019). The use of different culling and monitoring methodologies and a general lack of spatial replication of management effort has to date hindered the capacity to draw broad inferences on the determinants of successful noisy miner management (Melton et al., 2021).

Across 12 treatment and nine control areas, we aimed to identify factors affecting the success of noisy miner suppression and the magnitude of the response of songbirds to noisy miner suppression. Given the noisy miner's preference for more open woodland landscapes (Piper & Catterall, 2003; Val et al., 2018), we assume the recolonization potential of noisy miner source populations in the landscape surrounding our study areas should be negatively correlated with forest cover extent (Zeller et al., 2012). We therefore predicted that the percentage of the non-forested landscape surrounding the management areas—a proxy for noisy miner source population size and dispersal capacity—would predict noisy miner management success. We defined management success in terms of declines in noisy miner populations and increases in songbird populations (see methods), predicting greater success in management areas with a higher percentage of surrounding forest cover.

2 | METHODS

2.1 | Study location and design

The locations of 12 noisy miner management areas are shown in Figure 1 and described in Table 1.

The management areas ranged in size from 75 to 1200 ha and were selected based on their known or potential importance as regent honeyeater breeding areas and to span a range of landscape contexts (Figure 2). Habitats within the management areas are primarily riparian box-gum woodland, dominated by *Casuarina cunninghamiana* gallery forest and/or semi-cleared flats

TABLE 1 Summary of the 12 noisy miner management areas included in the study

	Treatment		Control							
Management area	Size (ha)	N[a] (sites)	Size (ha)	N* (sites)	Cull date	Person days culling	Cost ($AUD)	N (miners removed)	Monitoring duration (months)
Coonoor	250	76	100	36	May 19	5	2500	300	12
Coonoor TSR	300	48	100	21	May 20	4	2000	151	12
Goulburn river 1	430	144	N/A	N/A	August 17	12	6000	350	12
Goulburn river 2	120	52	N/A	N/A	August 18	8	2000	318	12
Goulburn river 3	100	36	N/A	N/A	August 19	6	1500	238	3
Wollombi	150	45	190	26	February 20	4	1000	81	12
Widden	1200	150	370	33	May 20	14	7000	1200	12
Mount Marsden	680	135	80	18	April 21	4	2000	955	3
Quorrobolong	75	16	130	21	June 19	4	2000	253	12
Cowra	88	56	167	48	July 21	5	2500	149	3
Chiltern	300	33	350	22	April 20	5	2500	200	12
Murrumbo	220	73	130	32	September 21	6	3000	150	3

Note: Note that cost estimates assume a mean daily pest management consultant rate (2017–2021) of AUD $500 and do not include costs associated with monitoring, reporting, or administration.

[a]Number of bird monitoring sites were established within each treatment and control area.

of *Eucalyptus melliodora, E. blakelyi, E. albens, E. molluccana* and *Angophora floribunda*. At Quorrobolong, the dominant vegetation community comprises *Corymbia maculata, E. fibrosa*, and *E. molluccana*. At nine of the management areas, we established nearby control areas (0.2–5 km from the respective treatment areas), from where no noisy miners were removed but habitats and landscape context were otherwise similar (Table 1). Within management and control areas, we established a set of fixed bird monitoring sites (the mean density of sites was one per 4.3 ha, range of 1.6–9.1 ha). The minimum distance between adjacent sites was 140 m, calculated as a trade-off between maximizing the independence of bird and habitat data at adjacent sites and minimizing the probability of missing regent honeyeaters that may be sparsely distributed in small breeding territories within the treatment areas (Crates et al., 2018).

2.2 | Bird and habitat surveys

In the week preceding the noisy miner culls in each management area (treatment and control), we surveyed each bird monitoring site once, recording the maximum count of all songbirds (defined as species of the order Passeriformes, excluding corvids, magpies, and choughs, Table S1) detected either visually or aurally by a single observer within 50 m of the point location over 5 min. We also recorded a eucalypt and mistletoe blossom score from 0 (none) to 4 (very heavy)—a proxy for the availability of nectar; a key food source for regent honeyeaters and many other songbirds (Bennett et al., 2014). The detectability of noisy miners using this survey protocol is high (0.83, Crates et al., 2018). The set of bird surveyors differed across areas but remained largely consistent within areas across the sampling periods (Table S2).

We repeated the pre-cull bird survey protocol in week-long periods commencing 2 days, 1 month, 3 months, and (apart from Cowra, Goulburn River 3, Mount Marsden, and Murrumbo) 1 year after the cull (Table S2). At Widden, Murrumbo, and Goulburn River 2 and 3, we did not conduct the 1-month post-cull surveys, as previous results showed the post-1-month and post-3-month bird data to be very similar in such areas (Crates et al., 2018). Where we detected regent honeyeaters, we conducted follow-up searches for more birds nearby and nest monitoring following the methodology described in Crates et al. (2019).

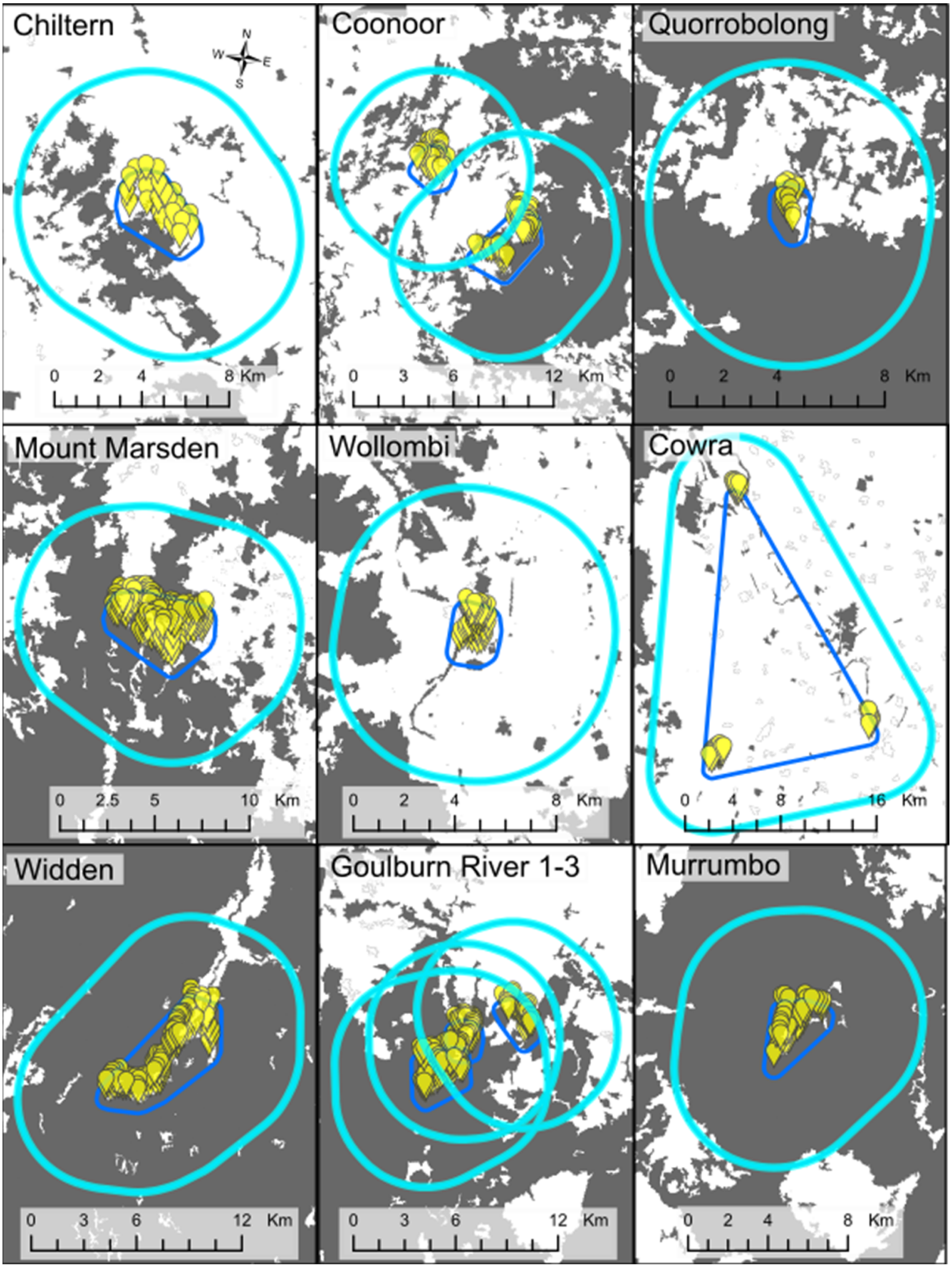

FIGURE 2 Location of noisy miner management areas and the proportion of suitable (white) or unsuitable (dark gray) noisy miner habitats in the surrounding landscape. Yellow pins denote monitoring sites within each management area. Thin dark blue and thick light blue polygons denote 500 m and 5 km buffers around each management area, respectively. Due to the spatial scale of the mapping, data from 20 km buffer areas are not shown

During one set of bird surveys in each area, we recorded a set of static habitat covariates following Crates et al. (2018), describing site-level vegetation cover, structure, and species composition (Table S3). The same observer recorded the vegetation data at all bird monitoring sites within each area, but different observers collected the habitat data in different areas (Table S2).

2.3 | Noisy miner culls

A noisy miner song broadcast was used to attract noisy miners within the 12 treatment areas, which were shot under license using 12-gauge shotguns and size 8 shot by professional pest management consultants. See Supporting Information for additional details on the culls and the acknowledgments section for details of all relevant permits. The clean kill rate (immediate death) of noisy miners using this methodology was 98%. The remaining 2% of animals were euthanized within 10 s via a second shot or cervical dislocation. Each management area was divided into smaller sections, which were then systematically searched and cleared of as many noisy miners as possible. On the final days of each cull, a follow-up search of the entire area was undertaken to remove as many remaining noisy miners as possible. The duration of culling effort in each area ranged from 4 to 14 person-days, broadly proportional to the size of each area and the density of noisy miners therein (Table 1). The aim of the culls was to reduce as much as possible the number of noisy miners occupying each management area for the subsequent 3-month breeding season and preferably up to 1 year.

2.4 | Response variables and management success metrics

We used noisy miner counts and songbird counts at monitoring sites within each area as the primary response variables. For each management area at each available time period, we calculated six success metrics. These were changes in:

1. *Area-level noisy miner occupancy rate*: The proportion of sites in each area where at least one noisy miner was detected during surveys at each post-cull time period, relative to the pre-cull time period.
2. *Area-level mean noisy miner density*: Mean number of noisy miners per hectare. Since noisy miners are a largely sedentary, colonial species (Higgins et al., 2001), we used the pre-cull noisy miner data as our reference for each area. We used an impact threshold density of 0.65 noisy miners ha^{-1}, which is the mean threshold density modeled by Thomson et al. (2015) across all bioregions, and a range of 0.44–0.83—the threshold density estimates from bioregions within which our management areas were located. We deemed culls successful if (i) the pre-cull noisy miner density was above 0.83 noisy miners per hectare, and (ii) the noisy miner density was below 0.83 (and ideally 0.65) birds per hectare at 3 months or 1-year post-cull.
3. *Site-level noisy miner abundance*: Beta coefficient size prediction derived from the period term in generalized linear models (GLMs) of noisy miner abundance for each area, relative to noisy miner abundance at the pre-cull time period. More negative beta effect sizes denote greater decreases in post-cull noisy miner abundance.
4. *Noisy miner impact on songbirds*: We used beta coefficient predictions for management area × noisy miner abundance interaction terms in GLMs of songbird abundance for each time period, to assess how the relationship between noisy miner abundance and songbird abundance differed within management areas over time. The response variables were (i) overall songbird abundance and (ii) small (<63 g and therefore smaller than noisy miners) resident songbird abundance ($n = 42$ species, Table S1). We used small residents as a second response metric because we assumed migrant species would show temporal variation in their abundance that was independent of any response to noisy miner management (Crates et al., 2020), such as time of year, drought, and eucalypt blossom (Ford et al., 2001; Mac Nally et al., 2012). More negative beta coefficients denote a greater impact of noisy miner abundance on songbird abundance, while less negative beta coefficients denote less impact of noisy miner abundance on songbird abundance.
5. *Site-level songbird abundance*: Beta coefficient prediction derived from "period" term in GLMs of (i) overall and (ii) small resident songbird abundance for each area. Larger positive beta values denote greater post-cull increases in songbird abundance.
6. *Differences in noisy miner and songbird abundance between treatment and control areas*. Beta coefficient predictions derived from the treatment × time period interaction term for each area at each post-cull time period. Relative to control data, in the nine areas where control data were available.

2.5 | Landscape context and noisy miner recolonization potential

We defined the spatial extent of each noisy miner management area (treatment only) by creating a minimum convex polygon (MCP) around monitoring sites. We then buffered these MCPs by 500 m, 5 km, and 20 km as three spatial scales predicted to affect the rate of noisy miner recolonization within the post-cull study periods (Figure 2, Barati et al. under review; Beggs et al., 2019; Crates et al., 2020; Davitt et al., 2018). Within each buffered MCP, we used a continental assessment of forest and woodland structure (Joint Remote Sensing Research Project, 2018; Scarth et al., 2014) to calculate the proportion of 30 m × 30 m raster cells that contained suitable noisy miner habitat. We reclassified remotely sensed vegetation structure data into a suitable habitat for noisy miner occupancy or dispersal (scattered trees, open woodland, or woodland with canopy height under 17 m) versus habitat unsuitable for noisy miner occupancy or dispersal (i.e., > 17 m closed-canopy woodland, open forest and closed forest, Piper & Catterall, 2003, Thomson et al., 2015).

2.6 | Noisy miner response models

We used R v3.4.3 (R Core Team, 2017) for all data analysis. For each study area, we fitted a saturated GLM to the abundance of noisy miners at each bird monitoring site. We used a negative binomial link function to reflect the distribution of the noisy miner count data. Predictors were the habitat covariates to control for variation in habitat features between monitoring sites (Table S3) and a time period factorial term (pre-cull, then 2 days, 1 month, 3 months, and 1-year post-cull). Where available, we ran additional models to include data from the control areas as a two-level factor (treatment vs, control), modeled as an interaction with the time period term. We used the package *lme4* v1.1-21 (Bates et al., 2015) to run the GLMs and the "dredge" function in *MuMIn* v.1.42.1 (Bartoń, 2018) to test all combinations of covariates in the saturated model and automatically identify the most parsimonious model for each area (i.e., a subset of covariates from the saturated model) based on lowest Akaike's Information Criterion (corrected for small sample size AICc) value and highest Akaike weight (Burnham & Anderson, 2004).

To explore potential spatial autocorrelation in the data, we repeated the noisy miner models described above using generalized additive models (GAMs) via package *mgcv* v1.9-23 (Wood, 2018). The GAMs included a smoothed spatial term for each survey site location s (*Lat, Long*). We regressed the beta-coefficients of the time period terms of interest in the noisy miner models derived from the GLMs and the GAMs for each area. Because the predictions from both models showed strong positive correlation across time periods (noisy miner $R = 0.99$–1, songbird $R = 0.93$–0.99, Figure S1), we focused on GLMs for the analysis.

To examine the relationship between the percentage of suitable noisy miner habitat in the wider landscape on the success of noisy miner suppression (i.e., potential rate of noisy miner recolonization across study areas), we fitted a series of GLMs with the response variables being area-level noisy miner "metrics of success" numbers 1–4, that is, changes in: (*1*) *mean noisy miner occupancy rate*; (*2*) *area-level mean noisy miner density*; (*3*) *site-level noisy miner abundance* (*effect size of the time period term in the noisy miner GLMs*); *and* (*4*) *the impact of noisy miner abundance on songbird abundance* (*effect size of the noisy miner term in the songbird GLMs*). Each management area therefore contributed a single data point for each response metric at each post-cull time period. We regressed the response metrics from the 12 management areas (treatment only) for each post-cull period against the estimated percentage of suitable noisy miner habitat in the surrounding landscape at the three spatial buffer scales of 500 m, 5 km, and 20 km. To check for potential landscape threshold effects on noisy miner management success, we also used a nonlinear (loess) model fit.

2.7 | Songbird response models

We used the same modeling approach for the songbird response variables as described above for the noisy miner response variables, using a Poisson link function to reflect the distribution of the songbird count data. We used the two songbird response metrics: all songbirds and then only small-bodied (<63 g) resident songbird species (Table S1). We defined residents following Crates et al. (2018) as those species not known to be seasonal migrants or nomads within the study area based on observer experience.

2.8 | Noisy miner impact on songbird models

To assess how the relationship between noisy miner abundance and both overall and small resident songbird abundance differed between the pre and post-cull time periods within each management area and overall (cf., Figure 6), we also fitted GLMs including (1) the interaction between noisy miner abundance × treatment area at each time period (described under cull success metric four above); and (2) an overall measure across all treatment areas at each time period including noisy miner abundance as a single term without the interaction with the treatment area.

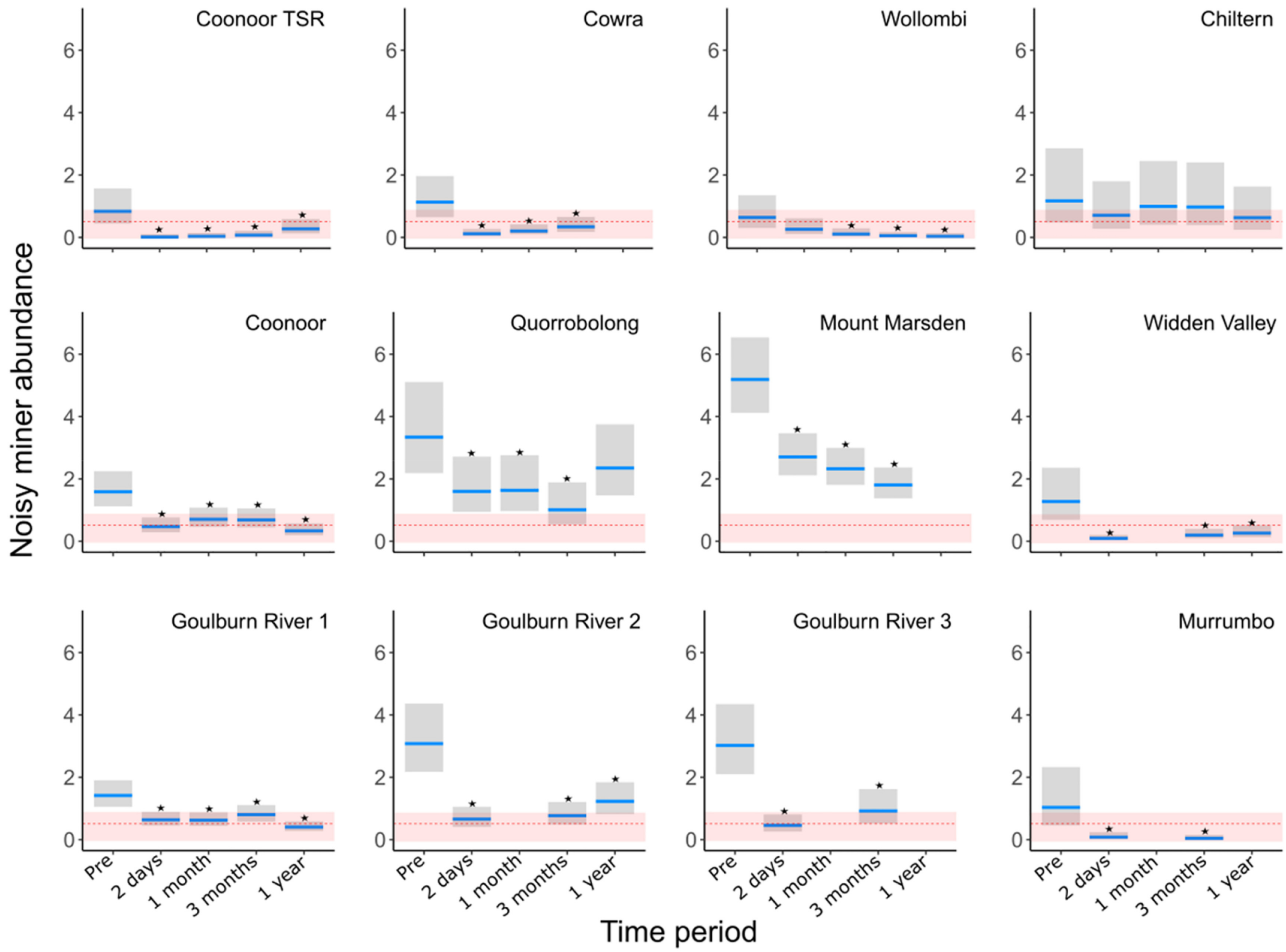

FIGURE 3 Modeled changes in noisy miner abundance across monitoring sites over time within each noisy miner treatment area. Predictions and 95% confidence intervals (solid lines and gray shading) are derived from the top-ranked (lowest AICc) generalized linear model for each treatment area. Stars denote post-cull time periods with statistically significant decreases in noisy miner abundances relative to pre-cull counts in each area. Density threshold (dashed red line) denotes a noisy miner impact threshold of 0.65 miners ha^{-1} or 0.51 miners per monitoring site. Shaded red area denotes the noisy miner impact threshold range 0.44–0.83 miners ha^{-1} within the bioregions of the study areas per Thomson et al. (2015). See Table S4 for summary statistics

3 | RESULTS

3.1 | Noisy miner removal

A total of 4345 noisy miners were removed across the 12 study areas (Table 1). Pre-cull mean noisy miner density was above the threshold impact of 0.65 miners ha^{-1} in all 12 areas but was below the upper threshold density estimate of 0.83 miners ha^{-1} in two areas (Wollombi and Coonoor TSR, Figure 3). Noisy miner density and occupancy rates declined in all treatment areas following noisy miner culls, but the magnitude and duration of the declines varied across areas (Figures 3 and S2). Mean noisy miner densities were below the mean impact threshold of 0.65 miners ha^{-1} at 3 months and/or 1-year post-cull in seven areas (Coonoor TSR, Cowra, Wollombi, Coonoor, Goulburn River 1, Murrumbo and Widden, Figure 3). Noisy miner densities remained below the upper threshold impact density of 0.83 miners ha^{-1} in further three areas (Chiltern, Goulburn River 2 and 3, Figure 3). Only in two areas (Mount Marsden and Quorrobolong) did noisy miner densities remain well above the upper threshold impact density (Figure 3). The top noisy miner models for each area and their summary statistics are shown in File S1.

Data from control areas, adjacent to nine of the treatment areas from where no miners were removed, showed some of the variation in noisy miner numbers was independent of noisy miner management (Figure S2). This was particularly the case for Mount Marsden and Wollombi (Figure S2). However, there were significant declines in noisy miner abundances at the 3-month

TABLE 2 Estimated percentage of suitable noisy miner habitat surrounding the 12 noisy miner management areas at spatial buffers of 500 m, 5 km, and 20 km. See Figure 2 for further information

	Buffer scale		
Management area	**500 m**	**5 km**	**20 km**
Coonoor	47	43	61
Coonoor TSR	80	76	63
Goulburn river 1	8	19	49
Goulburn river 2	20	33	51
Goulburn river 3	30	51	56
Wollombi	76	83	46
Widden	18	8	8
Mount Marsden	44	45	44
Quorrobolong	28	38	20
Cowra	94	92	83
Chiltern	76	78	77
Murrumbo	9	3	25

and/or 1-year post-cull time periods in six of the nine treatment areas with comparable control areas (Table S5).

3.2 | Landscape predictors of management success

The estimated percentage of suitable noisy miner habitat in the landscape surrounding the management areas varied from 3% (Murrumbo 5 km) to 94% (Cowra 500 m, Table 2). Only two metrics of management success were associated with the landscape context of the management areas: the change in mean area-level noisy miner density 3 months post-cull was positively associated with the percentage of suitable noisy miner habitat within 20 km, and the effect of noisy miner abundance on songbird abundance 2 days post-cull was positively associated with the percentage of suitable noisy miner habitat within 500 m and 5 km of management areas (Figure 4 and Table S6). Trends were broadly consistent across the three buffer scales of 500 m, 5 km, and 20 km (Figure 4) and there was no evidence for any threshold effects of landscape context on management success (Figure S3).

3.3 | Songbird responses to management

The response of songbird populations to noisy miner management was mixed (Figure 5 and S4). Relative to before noisy miner culls, total songbird abundance increased post-cull in seven of the treatment areas, declined in three areas (Mount Marsden, Widden, and Murrumbo), and was mixed in two (Chiltern and Goulburn River 3, Figure S4, Table S7). Restricting the analysis to small resident species showed broadly similar trends, although small resident songbird abundance was lower post-cull in Goulburn River 3 (Figure 5, Table S8). Relative to within control areas at the same time periods, small resident and overall songbird abundance was significantly higher in six of nine treatment areas, but control data suggested some variation in bird abundance was independent of noisy miner management (Figures S5 and S6 and Tables S9 and S10). The top songbird models for each area and their summary statistics are shown in File S1.

3.4 | Changes in noisy miner impact on songbirds

Generally, the impact of noisy miners on total songbird abundance and small resident songbird abundance was reduced across management areas after the culls. Relative to pre-cull, post-cull songbird curves shifted higher on the y-axis, indicating higher songbird abundances and reduced noisy miner impacts on songbirds, at comparable noisy miner abundances (Figures 6 and S7).

3.5 | Benefits for regent honeyeaters

We detected regent honeyeaters occupying habitat in six of the treatment areas in the breeding season following noisy miner management (Goulburn River 1, Quorrobolong, Widden, Murrumbo, Coonoor TSR, and Chiltern). Regent honeyeaters nested in five of these areas (all except Coonoor TSR) and successfully fledged juveniles in four of them (Goulburn River 1, $n = 5$ juveniles; Widden, $n = 5$; Murrumbo, $n = 2$, and Chiltern, $n = 3$). We did not detect regent honeyeaters in any of the nine control areas adjacent to management areas.

4 | DISCUSSION

Managing pest species is a global conservation challenge (Didham et al., 2007; Maxwell et al., 2016). Lethal management of any overabundant or exotic species is always undesirable from an ethical perspective (Ramp & Bekoff, 2015; Wallach et al., 2018). However, it is unfortunately often the only practical way to achieve large-scale management aims, such as the conservation of

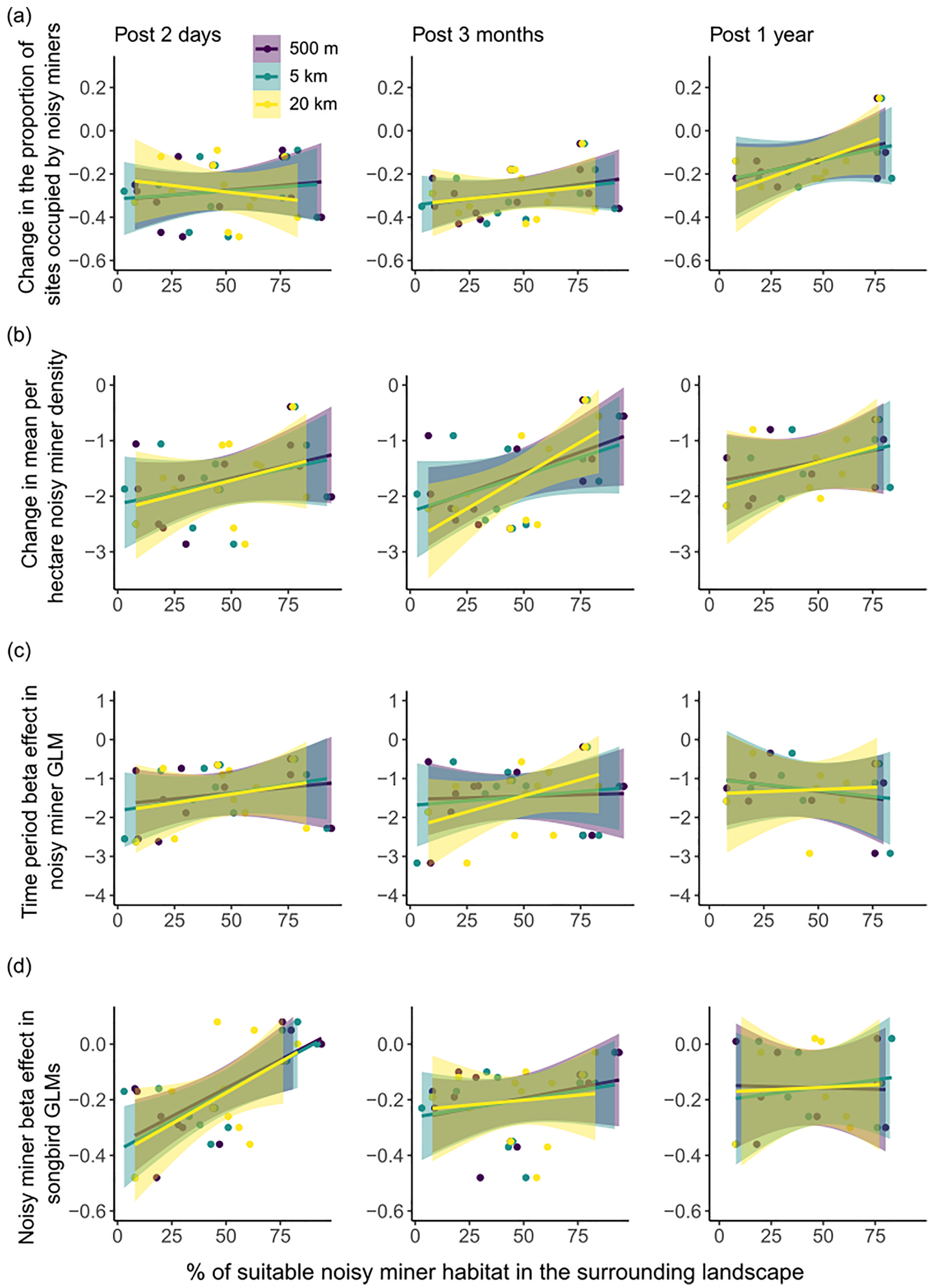

FIGURE 4 Linear regressions estimating the relationship between the percentage of suitable noisy miner habitat in the landscape surrounding 12 noisy miner management areas on measures of noisy miner management success. Columns denote time periods post-management, rows denote changes in (a) proportion of monitoring sites occupied by noisy miners; (b) mean per hectare noisy miner density; (c) beta effect of the time period term in the noisy miner generalized linear models; and (d) beta effect of the noisy miner term in the songbird models. See Table S6 for summary statistics

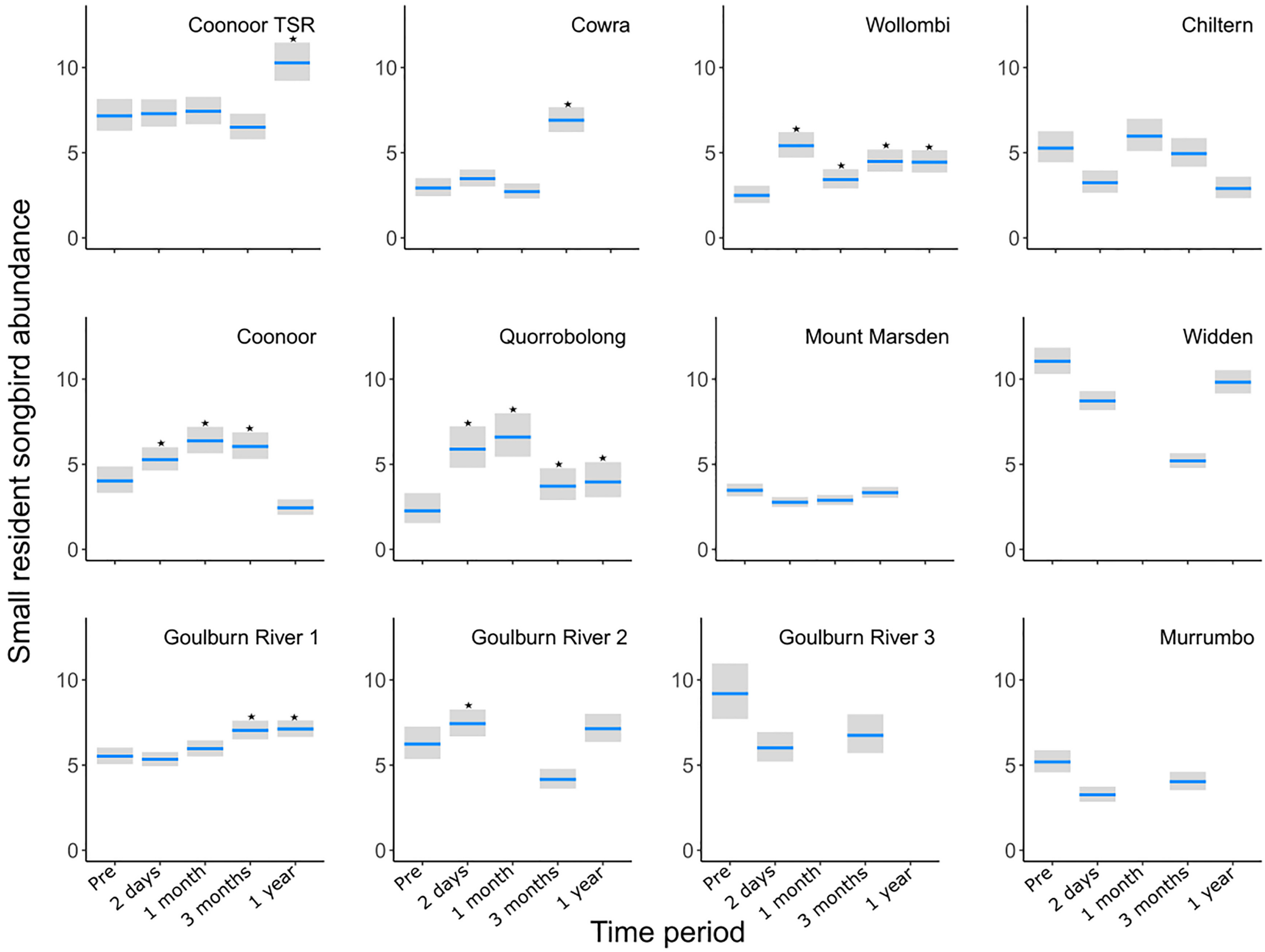

FIGURE 5 Modeled changes in small resident songbird abundance across monitoring sites over time within each noisy miner treatment area. Predictions and confidence intervals (solid lines and gray shading) are derived from the top-ranked (lowest AICc) small resident songbird generalized linear model for each treatment area. Stars denote post-cull time periods with statistically significant increases in small resident songbird abundances relative to pre-cull counts in each area. See Table S7 for model summary statistics

species on the brink of extinction, in an affordable and timely manner that minimizes individual suffering (Driscoll & Watson, 2019). In Australia, predicting the success of noisy miner management is a conservation priority to help address declines in a suite of threatened woodland birds. We undertook the most comprehensive assessment to date of the effectiveness of large-scale noisy miner management as a measure to suppress noisy miner populations and boost songbird populations in areas of high conservation value. Noisy miner management successfully suppressed noisy miner populations for three to over 12 months in 83% of the management areas. The response of songbird populations to noisy miner suppression was less clear-cut, although critically endangered regent honeyeaters subsequently occupied six of the 12 management areas and nested successfully in four of them. Contrary to our predictions, the broader landscape context was a poor predictor of whether noisy miners recolonized management areas within 1 year of their removal.

4.1 | Responses of noisy miners to management

A single management regime totaling 4–14 days at a cost of approximately AUD$ 10 ha^{-1} was sufficient to suppress noisy miner populations to below a threshold impact density of 0.65–0.88 birds ha^{-1}, in 10 of 12 management areas. Below this density, noisy miner impacts on co-occurring songbird populations are minimal (Thomson et al., 2015). Given recent findings of lack of success in reducing miner numbers through culling (Beggs et al., 2018; Davitt et al., 2018; Melton et al., 2021), noisy miners were successfully suppressed in a higher proportion of areas than might have been

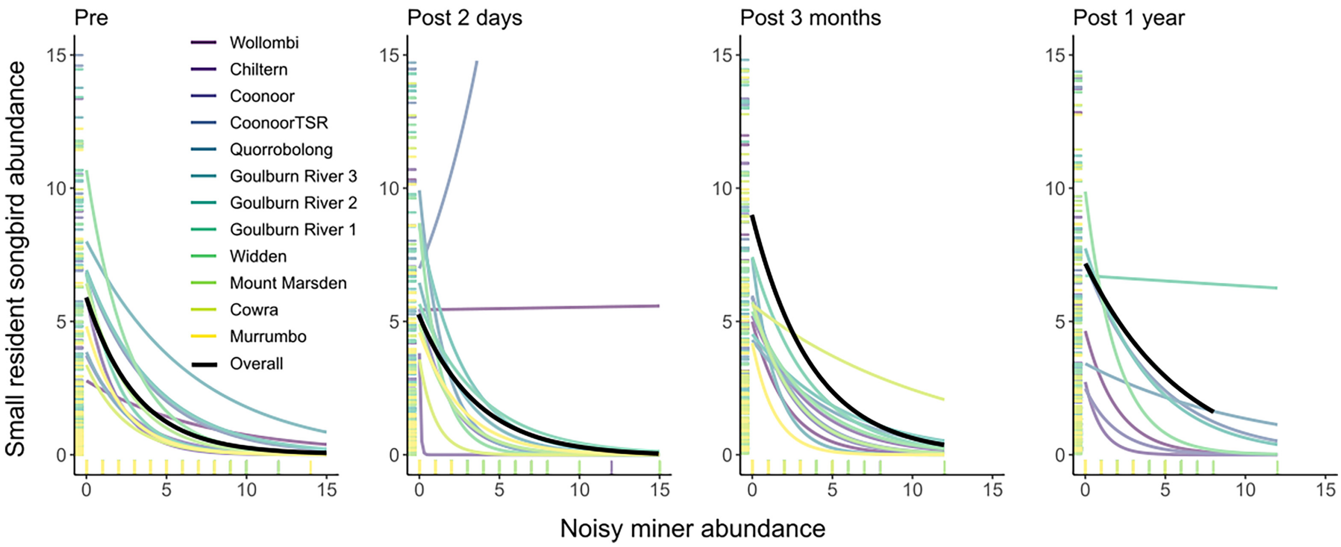

FIGURE 6 Modeled relationships between noisy miner abundance and small resident songbird abundance pre-noisy and at multiple time periods post-noisy miner management in the treatment areas. Predictions for each area are derived from "noisy miner × management area" interaction terms in small resident songbird generalized linear models for each time period. Solid black lines show the overall relationships between noisy miner and small resident songbird abundance at each time period, combining data from all treatment areas. See, Figure S7 for effects on total songbird abundance and Tables S11 and S12 for model summary statistics

expected. Furthermore, a number of the management areas were in lightly timbered landscapes with abundant noisy miner habitats nearby. Such landscapes are more similar to those of Beggs et al. (2019) and Davitt et al. (2018) where noisy miner control was deemed ineffective (notwithstanding subsequent songbird increases), than the forested landscape of Crates et al. (Crates et al., 2018; Crates et al., 2020) where control successfully achieved conservation goals.

What factors could have explained why noisy miner management was broadly successful in this study? Management areas included here were on average over 10 times larger, at 326 ha than those of Davitt et al. (2018, mean area 16–49 ha) and Beggs et al. (2018, mean area 13 ha). Larger treatment areas have a lower edge-to-area ratio, thus reducing the potential for immediate recolonization of noisy miners from nearby. Larger treatment areas also increase the chances of eliminating entire noisy miner colonies or significantly fragmenting "super-colonies," minimizing opportunities for re-establishment. Management was least successful at Quorrobolong—the smallest of the 12 treatment areas in this study at 75 ha. Mount Marsden was the second-largest treatment area, yet management still failed to suppress the noisy miner population below the target density there. We suggest this may be because the relative management effort (hectares treated per day) was lowest at Mount Marsden at 170 ha day^{-1}. Coupled with the relatively high pre-cull noisy miner density there, management effort was probably insufficient at Mount Marsden. The relatively high post-cull noisy miner density recorded is likely due to a high percentage of remaining birds, rather than rapid recolonization from surrounding areas.

Year effects could also explain broad-scale noisy miner population dynamics, and thus the potential for noisy miner management to achieve sustained reductions in noisy miner populations. Management efforts at 50% of our study areas occurred during the drought years of 2019 and 2020. Drought conditions may reduce noisy miner recolonization potential by suppressing breeding productivity and large-scale dispersal dynamics of individuals and/or colonies (Davitt et al., 2018). Data from nine control areas suggested that local noisy miner populations fluctuated more dramatically than may be expected for a sedentary, colonial species with high detectability, indicating that biotic factors such as drought also likely impacted the birds in these relatively dry years. Most culls that occurred in favorable conditions were still successful, however, suggesting that factors other than drought presence/absence were important. Such patterns were also observed by Beggs et al. (2019), suggesting the scale, frequency, and drivers of naturally occurring noisy miner movements requires further study. Timing management to occur in late winter or early spring, as was the case in 75% of management areas in this study, could also suppress breeding activity in that year, though in a wider review Melton et al. (2021) found no discernible effect of time of year on the effectiveness of noisy miner culling.

4.2 | Responses of songbirds to management

The success of noisy miner management in terms of the response of songbird populations was less clear. Although songbird populations increased post-cull in 75% of management areas, some of this increase can be attributed to the timing of the management, with migratory species returning to management areas in the months following culls in nine areas and departing post-cull in three. We attempted to account for variation in songbird life histories by establishing control areas where possible, and by including in a second songbird response metric only species we considered to be small-bodied residents. Data from control areas showed that some spatio-temporal variation in songbird occupancy patterns—even within species we considered resident—is likely to be independent of any effects of noisy miner management (Belder et al., 2021).

In some treatment areas, songbird populations were likely suppressed in the post-management monitoring periods due to rapid habitat degradation. For example, significant mistletoe mortality at Widden likely contributed to observed post-management declines in songbird numbers in this area (Crates et al., 2022). Cessation of mass Eucalypt flowering explained post-cull declines in songbird populations at Murrumbo, while the opposite led to large increases in songbird populations at Coonoor TSR. More generally, drought effects in 2019 and 2020 may have also suppressed songbird populations. These results highlight the challenges of monitoring songbirds at a sufficient scale to detect responses to noisy miner management with statistical confidence (Lindenmayer et al., 2020), particularly in environments such as Australia where spatio-temporal variability in climate and resource availability is high (Reside et al., 2010).

4.3 | Responses of regent honeyeaters

Many of the management areas included in this study were selected as known habitat for the regent honeyeater. Contemporary breeding success in this species is at an all-time low (Crates et al., 2019), contributing to the predicted extinction of the wild population within two decades without enhanced conservation efforts (Heinsohn et al., 2022). Implementing actions to improve regent honeyeater breeding outcomes is therefore an urgent requirement (Heinsohn et al., 2022). Regent honeyeaters have a contemporary range of over 300,000 km^2, an estimated population of fewer than 300 individuals, and nomadic movement patterns (Commonwealth of Australia, 2016). We, therefore, consider it a positive outcome that regent honeyeaters occupied six of the noisy miner management areas within 3 months of the noisy miner culls, nesting successfully in four. This shows that informed by spatially extensive monitoring data, management actions can be targeted in space and time to benefit the most at-risk species (Crates et al., 2018). Ongoing noisy miner management to remove small numbers of recolonizers thus represents a way to rapidly and substantially increase the availability of functional breeding habitat not only for regent honeyeaters but also many other threatened species in remaining core breeding areas.

4.4 | Effects of broader landscape context

Contrary to our predictions given results of previous studies (Beggs et al., 2018; Davitt et al., 2018), the percentage of suitable noisy miner habitat in the surrounding landscape was a poor predictor of noisy miner recolonization. Many of the response metrics showed weakly positive relationships with the proportion of suitable noisy miner habitats in the surrounding landscape, so our sample size of management areas ($n = 12$) may lack statistical power to detect broader trends. The results were largely consistent across the three buffer scales of 500 m, 5 km, and 20 km, offering little insight into whether noisy miners mainly recolonize from nearby or further afield when recolonization does occur. It is possible that broader landscape effects impact recolonization rates more than 1 year after culls is implemented, or that the percentage of available noisy miner habitat in the surrounding landscape is a poor proxy for the actual abundance of this widespread, but patchily distributed colonial species (Kopps et al., 2013). Alternatively, variation in the size of the management areas may have clouded any effects of landscape context, emphasizing the importance of standardizing the size of management areas in future efficacy assessments.

4.5 | Management and research implications

Our study provides evidence that short-term noisy miner management regimes, initially costing AUD \$2000–12,000 per area annually or AUD \$10 ha^{-1}, can lead to longer-term noisy miner suppression in areas of high conservation value. Where noisy miner numbers remain suppressed for 12 months or more, follow-up culls in such areas are expected to be less ethically and financially costly because many fewer noisy miners will require removal in subsequent years. Combining our results with other recent findings (Melton et al., 2021), we suggest that noisy miner suppression is more likely to result in sustained declines in noisy miner populations in larger treatment areas. Culling effort should reflect the size of the treatment area and the size of the noisy miner population therein. Management should occur in areas where noisy miner populations pose a demonstrable risk to threatened species, such as the regent honeyeater, but should ideally occur in a precautionary manner (Leung et al., 2002), such that noisy miner populations have not yet reached population densities sufficient to result in the local extinction of other threatened songbirds. In this way, songbird populations have the best chance to recover.

Our results suggest, however, that clear positive responses of songbird populations to noisy miner management should not be expected in the short term. Longer-term monitoring lasting multiple years post-management is required to quantify the full conservation returns on investment in noisy miner management. Long-term monitoring should also occur as much as possible in adjacent control areas, to partition effects of noisy miner management from often drastic fluctuations in songbird populations in response to other factors related to environmental variation (Reside et al., 2010).

Minimizing ethical costs associated with culling a native species is a vital consideration (Hampton et al., 2019). In the longer term, habitat restoration remains the least contentious solution to managing noisy miner populations (Law et al., 2014), but it is unlikely that large-scale revegetation could occur within a time-frame (<20 years) that would address noisy miner threats to the most vulnerable species such as the regent honeyeater. Revegetation is also not always successful in reducing miner densities either (Melton et al., 2021). In addition, many remnant areas of high biodiversity value occurring on productive soils (Watson, 2011) are private agricultural properties. In such places, large-scale habitat restoration is not currently a financially viable option due to the costs of setting aside commercial land for revegetation. This makes it crucial from an ethical perspective that noisy miner culling occurs in areas where there is scope for it to form part of a long-term management regime. In subsequent years, noisy miner numbers can then be kept low by removing small numbers of recolonizers, culling efforts can be expanded to surrounding areas, and, where possible, complemented with habitat restoration (Crates et al., 2020).

Above all, our study emphasizes the importance of standardized monitoring regimes in overcoming the challenges associated with identifying the predictors of successful pest species management (Melton et al., 2021). In this way, conservation returns can be maximized for the least ethical and financial cost.

AUTHOR CONTRIBUTIONS

Conceptualization: Ross Crates. **Data collection:** Ross Crates, Emily Mowat, Max Breckenridge, and Liam Murphy. **Data analysis:** Ross Crates. **Manuscript writing and revision:** Ross Crates, Paul G McDonald, Martine Maron, Courtney B Melton, and Robert Heinsohn. **Administration and supervision:** Ross Crates, Dean Ingwersen, Robert Heinsohn, and Emily Mowat.

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CONFLICT OF INTEREST

The authors declare no conflicts of interest.

DATA AVAILABILITY STATEMENT

An annotated R script and all data are available in the Supporting Information File S2. Due to data sensitivity, we have offset the spatial location data.

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SUPPORTING INFORMATION

Additional supporting information can be found online in the Supporting Information section at the end of this article.

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