Chapter 6

Effective control of the common myna

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Understanding basic species population dynamics for effective control: a case study on
community-led culling of the common myna (*Acridotheres tristis*).

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6.1 Abstract

Population manipulation of introduced species can be difficult and many widespread eradication or reduction attempts have failed. Understanding the population dynamics of a species is essential for undertaking a successful control program. Despite this, control attempts are frequently undertaken with limited knowledge of the species population dynamics. For example, in Australia, concern over the impact of the introduced common myna (*Acridotheres tristis*) has led to community members culling the species. In this paper, we assessed the impact of a community-led common myna culling program over broad and fine-scales in Canberra, Australia. We utilised a basic population model to enhance understanding of common myna population dynamics and the potential influence of various culling regimes. We found a significant negative relationship between common myna abundance and culling at fine-scales (1 km²). However, over broad-scales the relationship between common myna abundance and culling was not significant. Our population model indicated culling at a rate of 25 birds per km² per year would reduce common myna abundance, regardless of initial density. Our results suggest that currently too few individuals are being removed from the Canberra population, and natural reproduction, survival and/or immigration is able to replace the culled individuals. This highlights the value of undertaking basic population modelling to assess if potential control measures are capable of achieving desired outcomes. Our work provides information for researchers, government and community groups interested in controlling not only the common myna, but also other introduced species.

6.2 Introduction

Increased trade and transport has led to the introduction of numerous species worldwide and the number of these introductions is increasing (Lockwood *et al.* 2005; Hulme 2006; Ascunce *et al.* 2011; Ricciardi 2012). The impact of introduced species can be varied with some species having devastating impacts while others are relatively benign (Davis 2003; Gurevitch and Padilla 2004; Gurevitch *et al.* 2011; Ricciardi 2012; Ruhren 2012). Due to the negative impacts of some introduced species, often significant effort is made to manage them.

Introduced species management has traditionally focused on eradication (Newton 1998; Davis *et al.* 2011). However, many widespread eradication or reduction measures have failed, leading to a waste of resources (Caughley 1977; Ward *et al.* 1979; Feare 1991; Newton 1998; Shine and Doody 2010; Davis *et al.* 2011). Examples include the failure of 26 of 30 attempted plant eradications in the Galapagos Islands since 1996 (Davis *et al.* 2011) and the expenditure of more than \$AUD20 M that has largely failed to eradicate cane toads (*Rhinella marina*) from Australia (Shine and Doody 2010).

Eradicating an introduced species can be difficult and often only isolated populations or newly established species can be effectively eradicated (Hulme 2006; Zabala *et al.* 2010). The

traits that make a species abundant are often the same traits that make a species difficult to control (for example, a high reproductive rate) (Newton 1998; King and Powell 2011). A population can compensate for culling losses through density dependent changes in reproduction, survival or immigration (Nichols *et al.* 1984; Conroy and Kremenetz 1990; Feare 1991; Nichols 1991; Shine and Doody 2010; Lebreton 2005; King and Powell 2011). Therefore, some species can be culled heavily year after year without achieving long-term reductions in their abundance or the damage they cause (Newton 1998; Shine and Doody 2010; King and Powell 2011).

Due to the difficulties associated with introduced species eradication, many programs have now shifted toward impact reduction measures, as they tend to be a more economical and effective way of impact mitigation (Newton 1998; Shine and Doody 2010; Melero *et al.* 2010). Numerous methods are available to reduce the impact of introduced species, such as the use of physical protection (fencing, netting, exclusion device), repellents (chemical, scaring device), resistance measures (development of resistant crops), and management (changing cropping time, location or planting ‘lure crops’) (Feare 1991; Newton 1998; Tracey *et al.* 2007). However, the impact of an introduced species is often related to their abundance and therefore, population control may still be necessary (Braysher 1993). Strategies should aim to reduce species abundance to below a threshold where their ‘damage’ is minimised (Braysher 1993; Shea and NCEAS 1998; Beggs and Rees 1999; Eiswerth and Johnson 2002). However, defining such thresholds can be extremely difficult (Hone 1995; Choquenot and Parkes 2001; Eiswerth and Johnson 2002; Buckley *et al.* 2004; Edwards *et al.* 2004). For detailed assessments of setting thresholds for pest control see Choquenot and Parkes (2001) and Bonesi *et al.* (2007).

To reduce the size of a population, the number of individuals culled annually needs to be higher than the maximum rate of population growth (Caughley 1977; Sutherland *et al.* 2004; Zuberogitia *et al.* 2010; King and Powell 2011). This rate of culling also needs to be sustained year after year or numbers may recover (Shine and Doody 2010; Zuberogitia *et al.* 2010; King and Powell 2011). Therefore, to successfully manipulate the population size of a species, its biology and population dynamics need to be understood (Newton 1998; Zuberogitia *et al.* 2010; Zabala *et al.* 2010; King and Powell 2011; Louette *et al.* 2013). An effective method for population reduction also needs to be available (Newton 1998; Melero *et al.* 2010; Zabala *et al.* 2010; Louette *et al.* 2013; Tobin *et al.* 2014).

A key example of a control program, where little is known about species biology and population dynamics, is the culling of the common myna (*Acridotheres tristis*) in Australia. Using this case study, we demonstrated how knowledge of a species population dynamics and biology can be employed to enhance culling outcomes.

Globally, concern over the impact of the common myna is growing rapidly due to it competing with native species for territory and nest cavities (Feare and Craig 1998; Lowe *et al.* 2000; Dhami and Nagle 2009). In 2005, the Australian community voted the common myna as

the ‘most significant pest’, ‘the pest problem seen to be increasing most’, and the top ‘pest problem that needs more control’ (ABC 2011). Concern about the common myna was greater than species that have devastating impacts in Australia such as the cane toad, red fox (*Vulpes vulpes*), feral cat (*Felis catus*) and European rabbit (*Oryctolagus cuniculus*) (Reddiex and Forsyth 2006). As a result, community-led trapping programs for the common myna have been initiated (CIMAG 2013). These trapping programs often focus on culling as many individuals possible, but are not guided by an understanding of the number of individuals that need to be culled to reduce species abundance. Therefore, it is timely that the effectiveness of community-led culling is assessed.

In this paper, we used culling data and bird survey data to investigate whether community-led culling is reducing common myna abundance in the city of Canberra, Australia. We analysed changes in common myna abundance over fine-scale (1 km²) and broad-scale (approximately 70 km²) regions, in relation to culling by a community group. We hypothesised that community-led culling in Canberra is responsible for apparent reductions in common myna abundance. We then utilised a basic population model to enhance understanding of common myna population dynamics and the potential impact of various culling rates on the abundance of this introduced species.

6.3 Method

6.3.1 Background on the common myna and community-led culling

The common myna is listed in the world’s worst invasive species (Lowe *et al.* 2000) and the species has been observed dominating natural nesting cavities, evicting native birds, killing the chicks and destroying eggs (Byrd 1979; Jones 1996; Pell and Tidemann 1997a, b; Feare and Craig 1998; Harper *et al.* 2005).

In April 2006, the Canberra Indian Myna Action Group (CIMAG) was formed (CIMAG 2013). CIMAG is a community group that aims to reduce the impact of the introduced common myna, through public education (to reducing feeding and breeding opportunities) and a humane culling program. The group uses pet food as bait to lure individuals into valve traps, before humanely euthanising them (CIMAG 2013). Traps are designed to permit entry to the common myna by means of a valve through which individuals can enter freely but are unable to locate the exit (Tidemann 2010). CIMAG has approximately 1,400 members that have euthanised over 45,000 individuals from the Canberra area over 7 years (CIMAG 2013). The common myna was once the third most common bird species in Canberra but since culling began it has dropped to the thirteenth most common bird species (Canberra Ornithologists Group 2011). The culling program is considered to be ‘successful’ by CIMAG. Other community groups around Australia and overseas have sought advice and support from CIMAG to implement their own control programs (B Handke 2013 pers. comm. 20 Jan.).

6.3.2 Study site

Canberra is the capital city of Australia with a population of approximately 368,000 people. The city is located in south-east Australia, 150 km inland and at an elevation of 605 m above sea level (Gibbs *et al.* 2011) (Figure 6.1). The city experiences relatively warm to hot summers [mean maximum temperature 27.6°C (81.7°F)] and cool to cold winters [mean minimum temperature 0.9 °C (33.6°F)], when frost and fog are common (BOM 2013). The average annual rainfall is 632 mm (BOM 2013). Canberra is a planned city known as the ‘bush capital’ due to the large number of nature reserves and undeveloped hills in and around the city. The main vegetation types in nature reserves are dry sclerophyll forest, open savannah, and woodland.

Figure 6.1: Study area and location of the four regions and 15 fine-scale culling areas in the suburbs of Canberra, South East Australia

6.3.3 Broad-scale common myna abundance and impact of culling

We used long-term survey data (see Garrock *et al.* 2013a) to document common myna abundance over 29 years in Canberra. The Canberra Ornithologists Group (COG) started the Garden Bird Survey (GBS) in 1981. Observers surveyed an area of 3.1 hectares, recording the

maximum number of each bird species seen or heard in a site at any one time over a 7-day period. The duration of each observation was unspecified in the GBS procedures. In our analysis, we accounted for this potential variability by estimating average common myna abundance per region per season (see below). A total of 74,492 surveys were undertaken in Canberra from July 1981 to 2010. We used ArcGIS 10[®] (ESRI 2010) to define four geographic regions in Canberra (Figure 6.1). We based regions primarily on geographic location and development history of the city. This enabled grouping of survey sites over a broad area to assess the impact of common myna culling.

Due to seasonal variation in common myna abundance (Pell and Tidemann 1997a), we divided each year into two time periods: Spring–Summer (September to February) and Autumn–Winter (March to August). Common myna abundance is highest in Spring–Summer when the species breeds and young birds fledge. By contrast, common myna abundance declines over Autumn–Winter due to natural mortality (Pell and Tidemann 1997a). We broadly termed these two periods as ‘breeding’ (Spring–Summer) and ‘non-breeding’ (Autumn–Winter) seasons.

GenStat 15[®] (VSN International 2012) was used to conduct all analyses. We fitted a hierarchical generalised linear model (Lee *et al.* 2006) to raw common myna counts using a quasi-Poisson model with a logarithmic link function. Region (1–4), year (1–29), season (breeding, non-breeding), and their interactions were treated as fixed effects. GBS sites were treated as a random effect with a log-gamma distribution. For each combination of region, year and season, we estimated the average common myna abundance per site, reducing the data from 74,492 individual surveys to 232 estimates. We calculated the bi-annual abundance of the common myna per km² per region.

CIMAG provided information on the location and number of common myna birds trapped and euthanised. Recording of cull numbers commenced in July 2006, with some isolated culling occurring prior to this time. CIMAG send monthly requests to members asking them to report the number of individuals culled. However, some culling may go unreported and no attempt is made to record data on ‘trap effort’ (the amount of effort required to trap birds). Therefore, we were unable to factor this into our analysis. We defined the yearly culling period as November to October, as we wanted to calculate the number of birds culled prior to November (the peak period of fledgling emergence) (Pell and Tidemann 1997a).

We entered data from CIMAG into ArcGIS 10[®] (ESRI 2010) with the location and date of each bird caught. We calculated the total numbers of the common myna euthanised in each of the four regions over 12 month culling periods. This provided three culling periods (Nov 2006–Oct 2007, Nov 2007–Oct 2008 and Nov 2008–Oct 2009).

We used estimates of common myna abundance per km² per region in the breeding season to investigate the impact of common myna culling. We focused on the breeding season (Spring–Summer) due to greater abundance and visibility of the species at this time.

We observed that there were some reductions in common myna abundance prior to April 2006, when culling commenced (Figure 6.2). Therefore, we wanted to avoid mistakenly identifying culling as reducing common myna abundance when it could potentially be due to other influences (for example, natural population limitation). Therefore, we extended our analysis to include common myna abundance in the 3 years prior to common myna culling (when common myna culling was approximately equal to zero). We used linear regression, to investigate the relationship between changes in common myna abundance per km² and birds culled per km².

6.3.4 Fine-scale common myna abundance and impact of culling

We selected 15 sites in Canberra to survey the abundance of the common myna. We located each site in a residential suburb that adjoined a nature reserve. Nature reserves ranged from dense woodlands to open grassy woodlands and were dominated by Eucalyptus species. We ensured that residential sub- divisions were constructed more than 20 years ago so that the vegetation was well established. Sites were situated at least 2 km apart, as the common myna rarely travels further than 2 km per day (Feare and Craig 1998; Dhami and Nagle 2009).

We surveyed our 15 sites in the breeding season in September, November and January, commencing in September 2008 and concluding in January 2011. We used experienced bird observers to identify the common myna by both sight and call, using line transect surveys. Transects were located through a suburb at a right angle to an adjacent nature reserve. Transects were 1 km in length and 60 m wide. Observers walked transects for a period of 20 min, within 3 hours of sunrise. Each transect was walked two to three times per survey month, by two or more observers. We ensured surveys were undertaken in good weather conditions with little or no rain or wind to limit weather effects on detectability (Lindenmayer *et al.* 2009). Observers attempted to sight all birds heard calling and take into account the movement and direction of birds to minimise the chance of double counting.

A total of 311 transect surveys was conducted by 15 observers. We calculated the mean common myna abundance per km² in each of the 15 sites, each breeding season (2008–2009, 2009–2010, 2010–2011). From this, we calculated the change in common myna abundance from one time period to the next. This provided two measures of the yearly change in common myna abundance for each of the 15 sites.

We used ArcGIS 10[®] (ESRI 2010) to identify a 1 km buffer around each of the 15 survey transects. We defined this buffer zone as the fine-scale culling area (Figure 6.1). We then calculated the number of birds euthanised within each of the 15 culling areas for the two culling

periods (Nov 2008–Oct 2009 and Nov 2009–Oct 2010). To assess the impact of culling, we performed linear regression analysis using the change in common myna abundance per km² against birds culled per km².

6.3.5 Common myna population model

We used a logistic growth model to investigate how a common myna population would react to various levels of culling at different population densities. For simplicity, we utilised a model with no stochastic dynamics (environmental change). We used the generalised logistic model outlined by Sutherland *et al.* (2004) and derived from Caughley (1977). The change in population from one time period to the next was governed by the following equation (Sutherland *et al.* 2004):

$$N_{t+1} = N_t + r_{\max}N_t(1 - N_t/K) - c_tN_t$$

where N_t is population size at time t , $r_{\max}$ is the maximum growth rate, K is the carrying capacity, and, c_t is the culling rate for the same time period.

We used the average estimates from Grarock *et al.* (2013a) for the maximum growth rate and carrying capacity for the common myna over four regions in Canberra. Grarock *et al.* (2013a) used a composite 41-year data set to reconstruct the invasion sequence of the common myna in Canberra since 1968. The average maximum rate of population growth was 24.1 (± 6.4) birds per km² per year and the average maximum population size was 205.9 (± 34.6) birds per km².

We used the logistic growth model to establish an understanding of the number of individuals that need to be culled to reduce the population size. From this model, we identified the level of cull required (per km² per year) at various population sizes to produce a significant (>10%) reduction in common myna abundance. Ultimately culling strategies should aim to reduce common myna abundance below a predefined threshold, where their ‘damage’ is minimised (Braysher 1993; Shea and NCEAS 1998; Eiswerth and Johnson 2002). However, in the absence of such research on the common myna, we used a 10% reduction as a guide to assist with understanding the number of individuals that need to be culled to reduce species abundance.

We then estimated the total annual rate of common myna culling per km² in Canberra required to reduce species abundance. Using ArcGIS 10[®] (ESRI 2010) we calculated the total area of Canberra suburbs as 326.92 km². This area includes only the suburban area of Canberra and excludes nature reserves as well as surrounding urban areas of Queanbeyan, Hall and Hume. We then multiplied the required culling rate per km² per year, by the total area of Canberra suburbs.

Figure 6.2: Long-term abundance of the common myna (*Acridotheres tristis*) in four regions in Canberra, south-eastern Australia. Black arrows indicate the commencement of a community run common myna culling program.

Figure 6.3: Relationship between the changes in common myna abundance (*Acridotheres tristis*) and the number of individuals culled in four regions in Canberra, South East Australia. (a) Common myna abundance only in the years where culling occurred ($F_{1,10}=1.30$, $P=0.282$), (b) common myna abundance in the three years prior to common myna culling (when common myna culling was equal to zero) ($F_{1,22}=0.67$, $P=0.423$).

6.4 Results

6.4.1 Broad-scale common myna abundance and impact of culling

In all four regions in our study, common myna abundance fluctuated over the 29 year period (Figure 6.2). In some regions there appeared to be a decline in common myna abundance prior to the commencement of community-led culling (Figure 6.2).

We found no significant relationship between the yearly change in common myna abundance per km² and yearly culling per km² ($F_{1,10}=1.30$, $P=0.28$) (Figure 6.3a). When we included the 3 years of data prior to common myna culling, the relationship further weakened ($F_{1,22}=0.67$, $P=0.42$) (Figure 6.3b). Two of the largest observed reductions in common myna abundance (41 and 49 birds per km²) occurred prior to culling (Figure 6.3b). This finding indicates that community-led culling in Canberra is not responsible for reductions in *broad-scale* common myna abundance.

Figure 6.4: Relationship between the change in common myna abundance (*Acridotheres tristis*) and the number of individuals culled in 15 fine-scale sites (1 km²) in Canberra, South East Australia ($F_{1,28}=20.67$, $P<0.001$).

6.4.2 Fine-scale common myna abundance and impact of culling

We found a significant negative relationship between yearly change in common myna abundance per km² and yearly culling per km² ($F_{1,28}=20.67$, $P<0.001$). This finding indicates that community-led culling in Canberra is capable of reducing fine-scale common myna abundance.

6.4.3 Common myna population model

The expected outcomes of various culling rates on the abundance of common myna in Canberra are provided in Table 6.1. The level of cull required to create a significant reduction (>10%) in population density is highlighted with an asterisk (Table 6.1). Italic values are used to highlight populations that have reduced in size (Table 6.1).

According to our model, the natural rate of population growth is at its greatest when the population is at half carrying capacity (103 birds per km²). A cull effort of approximately 40 birds per km² is required to create significant (>10%) reductions in the population size at this density (Table 6.1). Fewer birds need to be culled to significantly (>10%) reduce the population size at densities below 103 birds per km², as the natural rate of population growth decreases. For example, at 20 birds per km², only 15 birds per km² need to be culled to achieve significant (>10%) reduction in population size. As a population approaches carrying capacity (205.9 birds per km²), the natural rate of population growth slows. When the population is above carrying capacity, the population size reduces naturally without culling. This natural decrease in population size may be responsible for the population reductions observed in some Regions prior to the commencement of the culling program in Canberra (Figure 6.2).

Our model indicated that culling at a rate of 25 birds per km² would result in reductions in the population size regardless of initial density (Table 6.1). This corresponds closely with what we observed for culling in Canberra. The culling rate in regions ranged from 0 to 15 birds per km² (Figure 6.3) and is lower than the model estimate of 25 birds per km². This helps to explain the lack of observed correlation between culling and common myna abundance at broad-scale sites. However, culling at fine-scale sites was occasionally >25 birds per km² and we observed a significant negative relationship between culling and common myna abundance (Figure 6.4), as would be predicted by our model.

Using this figure of 25 birds per km², we estimate that approximately 8,173 ($25 \times 326.92 \text{ km}^2 = 8,173$) birds would need to be culled each year in Canberra to reduce common myna abundance.

Table 6.1: Expected common myna (*Acridotheres tristis*) abundance after various culling regimes and initial population densities, using a logistic growth model[#]

Culling rate (birds per km ² per year)	0	2	5	10	15	20	24	25	30	40
Initial population size (birds per km ²)										
10	14.5	12.5	9.5	4.5*	0.0	0.0	0.0	0.0	0.0	0.0
20	28.5	26.5	23.5	18.5	13.5*	8.5	4.5	3.5	0.0	0.0
30	42.0	40.0	37.0	32.0	27.0*	22.0	18.0	17.0	12.0	2.0
40	55.1	53.1	50.1	45.1	40.1	35.1*	31.1	30.1	25.1	15.1
50	67.7	65.7	62.7	57.7	52.7	47.7	43.7*	42.7*	37.7	27.7
60	79.9	77.9	74.9	69.9	64.9	59.9	55.9	54.9	49.9*	39.9
80	102.9	100.9	97.9	92.9	87.9	82.9	78.9	77.9	72.9	62.9*
100	124.1	122.1	119.1	114.1	109.1	104.1	100.1	99.1	94.1	84.1*
150	169.1	167.1	164.1	159.1	154.1	149.1	145.1	144.1	139.1	129.1*
200	202.7	200.7	197.7	192.7	187.7	182.7	178.7*	177.7*	172.7	162.7
250	224.9*	222.9	219.9	214.9	209.9	204.9	200.9	199.9	194.9	184.9

[#]Italic values indicate populations would be expected to reduce in size and an asterisk indicates populations that would be expected to reduce significantly (>10%).

6.5 Discussion

6.5.1 Study Overview

We assessed the impact of community-led culling on common myna abundance, using bird survey data and culling at different spatial scales in Canberra, Australia. To the best of our knowledge, it is the first investigation to demonstrate that community-led culling is capable of reducing the local abundance of the common myna. We also utilised a basic population model to enhance the understanding of population dynamics and the potential influence of various culling regimes. Our study provided a unique opportunity to demonstrate how knowledge of species population dynamics and biology can be employed to enhance culling outcomes. This knowledge will enable culling targets to be set and guide planning of methods to achieve these targets. However, thresholds for common myna abundance still need to be determined and should be a priority. This paper provides valuable approaches and information for researchers, government and community groups interested in controlling not only the common myna but also other introduced species.

We observed that high-intensity culling appears to reduce common myna abundance at a fine-scale (1 km²). This indicates that community-led culling *is* capable of reducing the *local* abundance of the common myna. However, this high-intensity community-led culling may not be achievable over larger scales. Our results indicate that broad-scale culling of the common myna falls short of the levels required to reduce population numbers. The graphs of broad-scale common myna abundance (Figure 6.2) and culling (Figure 6.3) and the common myna culling model (Table 6.1) suggest that too few individuals are currently being removed from the population to achieve a significant (>10%) reduction in the population size. Therefore, we reject our hypothesis that community-led culling in Canberra is responsible for reductions in common

myna abundance. However, the apparent success of fine-scale culling indicates that broad-scale culling could become more effective if a greater number of individuals could be trapped. Our findings indicate that both the level of culling and the spatial extent of control operations influence the success of introduced species management.

According to our model, culling at a rate of 25 birds per km² per year should reduce the population size regardless of initial density (although not significantly for population densities between 60 and 150 birds per km²). Culling at 25 birds per km² per year is greater than the maximum rate of increase of 24.1 birds per km² and may be a suitable target rate for culling in Canberra.

Our model and analysis did not take into account environmental stochasticity, nor did it take into account populations in the area surrounding Canberra (nature reserves and other urban areas). Nature reserves are used extensively by the common myna in Canberra (Pell and Tidemann 1997a) and they may act as a source for individuals to reinvade the city areas. Therefore, the actual number of individuals that need to be culled annually would be greater than our estimated 8,173 birds. When culling first began CIMAG culled over 8,000 birds for two consecutive years, but more recently culling rates have ranged from 2,800 to 6,600 birds per year (B Handke 2013 pers. comm. 14 Jun.). These figures fall short of our conservative estimate of 8,173 birds that need to be culled annually.

Our basic population model provided valuable insight into the population dynamics of the common myna. It is by no means a comprehensive assessment of common myna population dynamics and should be seen as providing an initial assessment. This model could be further refined, if required, as more information is gathered, such as estimates of age structure, sex ratio, survival rates and reproduction rates. Additionally, more complex models exist that incorporate the links between life-history dynamics, culling rates and environmental variables (Sutherland *et al.* 2004).

Species demographics (age structure, sex ratio, birth rate and death rate) are critical in driving a species population size (Hengeveld 1989; Neubert and Caswell 2000; Simberloff and Gibbons 2004). Species are also influenced by density-dependent and density-independent factors. For example, density-dependent population regulation may occur as a population reaches carrying capacity and resources (e.g. food or territory) become scarce (due to overpopulation) (Hengeveld 1989; Rutz 2008). This may result in increased mortality or reduced breeding success and therefore the population size will reduce (Hengeveld 1989). As the population size reduces the pressure on available resources will ease and the population may eventually increase again. This process can lead to fluctuations in population size over time (Hengeveld 1989; Lensink 1998; Rutz 2008). Density-independent factors also influence population size and fluctuations. Examples of density-independent factors include weather, climate, and events such as drought or fires that can cause mortality and affect resource availability (Sher and Hyatt 1999; Richardson and Pysek 2006). Population size can also be

influenced by the existence of natural enemies (predators, parasites or disease) that increase mortality (Keane and Crawley 2002; Shea and Chesson 2002).

Due to these natural population fluctuations it can be difficult to clearly identify the influence of culling on a population. Unfortunately we were unable to obtain data on natural population fluctuations from sites without common myna culling. Therefore, without this knowledge we cannot say with certainty that observed changes are due to community-led culling. However the relationship for both broad and fine scale trapping is consistent with data from our population model, indicating that it is an accurate representation of this system and that high intensity culling is capable of reducing common myna abundance.

6.5.2 Understanding basic species population dynamics for effective control

For the successful management of vertebrate populations, managers need to have an informed and targeted strategy (Beggs and Rees 1999; Zabala *et al.* 2010; Louette *et al.* 2013; Tobin *et al.* 2014). Understanding the factors that influence the population dynamics of a species will assist with making informed decisions and make management more effective. These factors include life history strategy, population density, timing of cull, vulnerability of individuals to culling and species distribution and movements.

6.5.2.1 Life history strategy

The life history strategy of a species can have a big impact on the effects of a culling program, with some species capable of compensating for culling losses (Conroy and Kremenetz 1990; King and Powell 2011; Sandercock *et al.* 2011). For example, the common myna is able to produce up to seven eggs per clutch and up to three clutches per year (Grarock *et al.* 2013b). This enables the species to compensate for mortality- related losses from culling, through rapid reproduction.

6.5.2.2 Population density

As seen from our population model, population density had a large influence on the rate of population growth and, therefore, the impact of various culling rates (Eiswerth and Johnson 2002; Zuberogitia *et al.* 2010; King and Powell 2011; Sandercock *et al.* 2011). For example, species population growth tends to be slow initially, reaching a maximum growth rate as the population approaches half the carrying capacity; growth then slows as the density reaches carrying capacity. A species may also be more difficult to cull at lower densities due to difficulty locating or trapping individuals. This increased difficulty in trapping at lower densities has been observed in previous studies on the common myna (Griffin 2008; Griffin and Boyce 2009; King 2010; Tidemann 2010).

6.5.2.3 Cull timing

The timing of culling can be critical to the success of a management program (Newton 1998; Boyce *et al.* 1999; Kokko 2001; Ratikainen *et al.* 2008; Zabala *et al.* 2010). However, culling is often simply undertaken when the species is easiest to kill (Newton 1998; Shine and Doody 2010). Our data suggests that culling should be undertaken prior to the breeding season when the population is naturally at its lowest point. Common myna cull numbers are currently highest during and just after the breeding season, when young birds are easily trapped. This is not the most effective time for culling, as many of these individuals may be the ‘doomed-excess’ (individuals that may not have survived over winter) (Clark 1987; Hudson *et al.* 1997; Newton 1998). We suggest that the lack of reduction in common myna numbers over broad-scales may be caused by culling the ‘doomed-excess’. Therefore, common myna culling may be more effective if undertaken prior to the breeding season (approximately September, October and November). Arguably, the common myna has its largest impact during the breeding season (Pell and Tidemann 1997b; Garrock *et al.* 2013b). Therefore, culling from September to December may also reduce competitive pressure for nest cavities with native species.

6.5.2.4 Species ecology

Understanding the ecology of a species can help identify traits that may assist with species control (Newton 1998; Melero *et al.* 2010; King and Powell 2011; Tobin *et al.* 2014). Culling can target certain individuals in a population and this may reduce the effectiveness of culling operations (Dufour *et al.* 1993; Lebreton 2005; Melero *et al.* 2010). For example, valve trapping of the common myna may target young inexperienced birds (King 2010), potentially leaving individuals that cause the most damage and/or are strong breeders (Newton 1998; Brook *et al.* 2003; Melero *et al.* 2010). Culling operations that target breeding females may have greater success. Therefore, it is important to understand the age, sex and fitness of individuals being targeted by culling operations (Zabala *et al.* 2010; King and Powell 2011; Sandercock *et al.* 2011). Identifying and exploiting vulnerable aspects of a species life history will enhance culling numbers and the impact of culling. For example, in the evening, common myna individuals group together forming overnight roosts (sometimes containing hundreds of individuals) (Feare and Craig 1998; Tidemann 2010). Therefore, targeting roosts for management actions may be a good strategy. Alternatively, the female common myna remains on the nest over night and if culled, the male will often return with a new female to begin breeding (Tidemann *et al.* 2011). A highly effective method for euthanising breeding females in nest boxes and removing their eggs or chicks was developed by Tidemann *et al.* (2011) (Appendix 9).

6.5.2.5 Spatial element

Understanding the spatial structure, dispersal ability and capacity of a species to move

from high to low density areas is important for successful management (Edwards *et al.* 2004; Hampton *et al.* 2004; Travis and Park 2004; McMahon *et al.* 2010; Melero *et al.* 2010; Louette *et al.* 2013). Attempts to control only a small proportion of the population may fail due to immigration from surrounding areas (Pulliam 1988; Hone 1995; Zuberogitia *et al.* 2010) and lead to a waste of management resources (Myers *et al.* 2000; Zabala *et al.* 2010). For successful control, management units must be clearly defined (Robertson and Gemmell 2004; Zuberogitia *et al.* 2010). Small islands are one example of a naturally occurring management unit (Courchamp *et al.* 2003; Towns and Broome 2003). Control over a widespread area is possible but logistically difficult (Taylor *et al.* 2000; Towns and Broome 2003; Courchamp *et al.* 2003; Zabala *et al.* 2010). For species that occur on larger islands or continents, management units need to be defined that are both logistically tractable and have a low risk of recolonisation (Parkes 1990; Bomford and O'Brien 1995; Hampton *et al.* 2004). Using molecular genetics can be a powerful tool to understand and identify areas where there is negligible movement of individuals and thus suitable management units for control operations across a landscape (Moritz *et al.* 1996; Robertson and Gemmell 2004; Sarre *et al.* 2013).

There are many examples in the literature of species that have been culled in large numbers where there has been little to no overall reduction in their population size (Feare 1991). In Africa, millions of dollars were spent culling hundreds of millions of red billed quelea (*Quelea quelea*) for many years, yet the population size was not affected by this culling (Newton 1998). The species was able to reproduce quickly and control operations (spraying and dynamiting communal roosts) reached only a relatively small number of sites. The species quickly immigrated to, and bred, in areas where there had been heavy culling (Newton 1998). Widespread culling was abandoned with efforts directed towards localised culling for crop protection and altering cropping procedures. Likewise, community-led culling of the cane toad has had “... millions of dollars and thousands of hours ...” dedicated to it (Shine and Doody 2010), with “... no evidence that physical removal of cane toads has slowed the invasion” (Peacock 2007). However, in New Zealand community-led trapping of invasive animals, in cooperation with government agencies, appears to be responsible for substantial increases in kiwi (*Apteryx mantelli*) abundance (Glen *et al.* 2012). The importance of community acceptance and assistance in control programs can be critical to their success (Bremner and Park 2007). Therefore, scientists need to communicate research effectively to ensure community enthusiasm for conservation is harnessed into meaningful projects based on good science.

6.5.3 Common myna management

Suitable management units and abundance thresholds for the common myna still need to be defined. Our study indicated that intense localised culling appears to be effective but that current cull efforts are not high enough to have significant widespread reductions on common myna abundance in Canberra. The common myna appears to be somewhat sedentary and slow

at spreading to new areas (Grarock *et al.* 2013a), potentially enhancing cull effectiveness in the medium term (Sandercock *et al.* 2011). Due to the species' broad distribution across Australia and other continents, perhaps management actions should be undertaken in localised areas where the species is deemed to have the greatest impact. For example, management of the common myna near threatened species breeding areas, such as the superb parrot (*Polytelis swainsonii*), may be a good strategy. The adaptable nature of the common myna indicates that complimentary methods for controlling the species (such as roost and nest box culling) will likely be required to successfully reduce the abundance of this species (Tidemann 2010; Tidemann *et al.* 2011) (Appendix 9).

6.6 Conclusion

Our case study shows the value of undertaking basic population modelling before carrying out control programs. A significant amount of effort is currently being directed towards a program that is not fully achieving the desired results. Understanding the population dynamics of a species is essential to undertaking a successful control program. Basic population models can be used to assess whether potential control measures are capable of achieving desired outcomes. These models can help avoid the wasteful allocation of resources to projects that are unlikely to be successful and/or drive innovation into alternative control measures. An effective control program needs to have an informed and targeted strategy with information on the number of individuals that need to be culled and the duration and timing of culling. Central to this understanding is defining the spatial distribution of management units and a target population size where the species impact is minimised.

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