Conservation Science W. Aust. 8 (3) : 367–381 (2013)

Feral cat control as part of *Rangelands Restoration* at Lorna Glen (Matuwa), Western Australia: the first seven years

DAVE ALGAR, MIKE ONUS AND NEIL HAMILTON

Department of Parks and Wildlife, Science and Conservation Division, P.O. Box 51, Wanneroo, Western Australia 6946.

ABSTRACT

The *Rangelands Restoration* program at the Lorna Glen Conservation Reserve aims to achieve the successful reconstruction and conservation of biodiversity for the area. It forms part of the potential expansion of the Department of Parks and Wildlife's Western Shield fauna conservation program into the rangelands. However, reintroduction of native species to strategic areas can only occur if an effective, sustained feral cat control can be achieved. We tested the hypothesis that an effective and sustained feral cat control strategy (less than 10 cats per 100 km of track-transect) was achievable with an annual winter baiting program using the *Eradicat*® feral cat bait. Six of the seven baiting programs conducted at Lorna Glen over the period 2003–09 resulted in significant reductions in indices of cat activity. The efficacy of these programs confirms previous findings from other sites in the arid and semi-arid zone that baiting for feral cats in the winter months, when prey availability is low, can effectively control cat numbers. Rainfall will affect baiting outcomes and supplementary trapping campaigns can be used to augment effective cat control.

INTRODUCTION

The Australian arid zone has experienced a high rate of native mammal decline following European settlement. Since the 1920s, approximately 33% of all mammals and about 90% of medium-sized mammals (35–5500 g adult body-weight range) have either suffered dramatic range contractions or are extinct (Burbidge & McKenzie 1989). Many of these species are now restricted to several offshore islands and others, due to small population sizes and restricted geographic ranges, are vulnerable to extinction. A number of causes have been proposed to explain this decline. These include changed fire regimes, competition from introduced herbivores, disease, variability in weather and site fertility and predation by introduced predators, specifically the feral cat, *Felis catus*, and fox, *Vulpes vulpes* (see Burbidge & McKenzie 1989; Dickman 1996a, 1996b; Environment Australia 1999; Johnson et al. 1989; Morton 1990). Predation by feral cats has also been demonstrated to threaten the continued survival of many other native species that persist at low population densities (e.g. Risbey et al. 2000; Smith & Quin 1996) and has been identified as one of the major obstacles to the reconstruction of faunal communities, as it has prevented the successful reintroduction of a number of species to parts of their former range (Christensen & Burrows 1995; Dickman 1996b; Environment Australia 1999; Gibson et al. 1995). The suppression of introduced predators is therefore a critical component of successful reintroduction, recovery or maintenance of populations of small to medium-sized native fauna (Christensen & Burrows 1995; Fischer & Lindenmayer 2000; McKenzie et al. 2007).

Lorna Glen Conservation Reserve, a former pastoral lease, was acquired by the Department of Parks and Wildlife (DPaW) in 2000, with the objective of restoring the natural ecosystem by: (1) controlling introduced predators, in particular feral cats as foxes are rare or absent in this arid environment; (2) controlling introduced herbivores; (3) introducing ecologically appropriate fire regimes; and (4) reintroducing native mammals to the area. The site is typical of the arid zone rangeland ecosystems from which many Australian medium-sized native mammals have declined or become extinct. The project, known as *Rangelands Restoration*, aims to reintroduce 11 species of mammals to Lorna Glen over the next ten years (Dunlop & Morris 2009). The 11 species are: bilby (*Macrotis lagotis*), brushtail possum (*Trichosurus vulpecula*), rufous hare-wallaby (*Lagorchestes hirsutus*), burrowing bettong (*Bettongia lesueur*), golden bandicoot (*Isoodon auratus*), western barred bandicoot (*Perameles bougainville*), numbat (*Myrmecobius fasciatus*), red-tailed phascogale (*Phascogale calura*), chuditch (*Dasyurus geoffroii*), Shark Bay mouse (*Pseudomys fieldi*) and pale field-rat (*Rattus tunneyi*). If successful, this will not only improve the conservation status of these species, but also reinstate some of the ecological processes that these animals once provided to the rangelands (Dunlop & Morris 2009). The project forms part of the potential expansion of DPaW's *Western*

© The Government of Western Australia, 2013

Shield program, which aims to achieve the successful reconstruction and conservation of faunal biodiversity of this area, in the rangelands. However, reintroduction of native species to strategic areas can only occur if effective and sustained feral cat control can be achieved.

Poison baiting is recognized as the most effective method for controlling feral cats on mainland Australia by most practitioners (Algar & Burrows 2004; Algar et al. 2007; Environment Australia 1999; Short et al. 1997). As part of *Rangelands Restoration*, an ongoing campaign to bait feral cats has been conducted, in order to develop and prove baiting strategies that provide for sustained and effective feral cat control at Lorna Glen. The project follows research programs and operational feral cat baiting trials conducted in the interior arid zone at the Gibson Desert and Wanjarri Nature Reserves (Algar & Burrows 2004; Burrows et al. 2003). This earlier research in the arid and semi-arid zones indicated that the effectiveness of baiting programs for feral cats is maximized by distributing baits during the cool, dry winter periods (Algar & Burrows 2004). At this time, the abundance and activity of all prey types, in particular predator-vulnerable young mammals and reptiles, is at its lowest, and bait degradation due to rainfall, ants and hot, dry weather is significantly reduced.

The hypothesis tested was that an effective and sustained feral cat control strategy was achievable with an annual winter baiting program that distributed feral cat baits at a density of 50 baits km^{-2}. A benchmark of 'reducing and maintaining cat numbers to less than 10 per 100 km of track-transect (hereafter referred to as 10 cats per 100 km) had previously been proposed as the level at which reintroductions of native species could potentially occur (Morris et al. 2004). Thus the efficacy of the baiting strategy on feral cat activity could be assessed against this predetermined benchmark. Monitoring the rate of cat reinvasion and distribution provided an indication of the effectiveness of this strategy over time or whether there was a need for additional control effort (e.g. trapping and on-track baiting) to maintain low numbers of feral cats. In this paper we report the impact of this feral cat control strategy over the first seven years (2003–2009) of the project.

METHODS

Site description

Lorna Glen pastoral station was purchased by DPaW in 2000 for the creation of a conservation reserve; its current status is Unallocated Crown Land. Lorna Glen was destocked over a period of three years. The reclaimed pastoral lease is situated approximately 180 km east-north-east of Wiluna at 26° 13' S, 121° 33' E and is 2350 km^2 in area. The station has common boundaries with Wongawol, Yelma, Millrose and Granite Peak pastoral leases as well as a small area of Unallocated Crown Land that separates Lorna Glen from Earaheedy pastoral lease. The lease straddles the boundary between the Murchison and the Gascoyne Bioregions (IBRA; Thackway & Cresswell 1995). The study site comprises two main land systems: (1) Bullimore—sand plains and dunes dominated by spinifex (*Triodia* spp.); and (2) Sherwood—breakaways and stony plains dominated by mulga and other acacia shrublands. The vegetation unit most common across the station is the hummock grasslands, shrub steppe (Beard 1974): *Acacia aneura* (mulga) and *Eucalyptus kingsmillii* over *Triodia basedowii* (hard spinifex). Smaller areas of low *A. aneura* woodland are also present. The geology and geomorphology of the area is described by Mabbut et al. (1963).

The climate of the area is characterized by low and erratic rainfall with the annual average of 255 mm (Bureau of Meteorology records 1940–2003). Annual rainfall in 2003, 2007 and 2008 approximated the average (255 mm; 1940–2002), in 2004 and 2006 it was significantly greater than average, and in 2005 and 2009 annual rainfall was significantly less than the average (Table 1). Average maximum daily temperatures range from 19.4 °C in winter to 39 °C in summer.

Baits and baiting programs

The feral cat baits (*Eradicat*®) used during this and earlier baiting campaigns in the arid zone (Algar & Burrows 2004; Burrows et al. 2003) were manufactured at DPaW's bait manufacturing facility at Harvey, Western Australia. The bait is similar to a chipolata sausage in appearance,

Table 1

Monthly rainfall data (mm) over the study period. Missing values are where data were not supplied to the Bureau of Meteorology for the month and rainfall is presumed to be zero.

Year	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Annual
2003		8.4	39.0	163.4	0.2	3.4	0.0	11.8	0.0		19.4	6.8	252.4
2004	50.8	92.0	28.6	77.0	35.4	1.2	20.2	0.0	7.2	0.0	9.8	15.8	337.8
2005	0.0	0.4	13.2	7.8	2.8	4.3	36.3	10.6	0.0	0.5	8.8	37.1	121.8
2006	203.7	124.6	64.8	48.2	3.6	0.0	1.4	0.6	10.7	67.8	29.4	80.7	633.5
2007	110.0	3.0	59.2	71.4	10.2	0.0	5.2		0.0	1.2	2.0	34.7	296.9
2008	15.2	91.5	5.0	7.8	0.4	19.6	1.4	16.4	2.1	1.0	66.7	32.2	259.3
2009	7.6	11.2	101.6	15.4	0.0	8.2	0.4	2.6	2.4	0.4	12.1	5.8	167.7

approximately 20 g wet-weight, dried to 15 g, blanched and then frozen. This bait is composed of 70% kangaroo meat mince, 20% chicken fat and 10% digest and flavour enhancers (Patent No. AU 781829). Toxic feral cat baits were dosed at 4.5 mg of sodium monofluoroacetate (compound 1080) per bait. Prior to being laid, feral cat baits were thawed and placed in direct sunlight. This process, termed 'sweating', causes the oils and lipid-soluble digest material to exude from the surface of the bait. All feral cat baits were sprayed, during the sweating process, with an ant deterrent compound (Coopex®) at a concentration of 12.5 g l^{-1} as per the manufacturer's instructions. This process is aimed at preventing bait degradation by ant attack, which may also deter bait acceptance by cats because of the physical presence of ants on and around the bait medium.

The baiting programs were conducted from a dedicated baiting aircraft which deployed the baits at predetermined drop points. The baiting aircraft flew at a nominal speed of 130 knots and 500 feet above ground level and a GPS point was recorded on the flight plan each time bait left the aircraft. The 'bombardier' released 50 baits into each 1 km map grid, along flight transects 1 km apart, to achieve an application rate of 50 baits km^{-2}. The ground spread of 50 baits is a cluster of approximately 200 × 50 m (D Algar, unpub. data).

The multi-faceted umbrella program *Rangelands Restoration* evolved under an adaptive management framework. As a consequence, the area baited, and monitoring techniques used, were not consistent throughout the years of this study, which is a reality for large-scale management programs such as this. However, the baiting strategy remained the same. In 2007, there was a major shift in the emphasis of the program with commencement of the native species reintroduction phase of *Rangelands Restoration* (Dunlop & Morris 2009). Unfortunately in 2006, no decline in cat activity indices had occurred following baiting and thus to increase the likelihood of survivorship of reintroduced species the management committee for the project requested that short-term, small-scale trapping programs be integrated into the control regime. At the conclusion of each survey period, excluding the pre-baiting survey, monitoring data were examined to assess whether this option was warranted.

In 2003, a small-scale pilot baiting trial was conducted. Two sites of similar habitat were selected, each 625 km^2 in area (25 km × 25 km quadrat) separated by a distance of at least 5 km. One site was the treatment (baited) site and the other was the control (non-baited) site. In 2004, the baited area was increased to cover an area of 1725 km^2, which encompassed the entire Lorna Glen lease but excluded the control area that was not baited in 2003. Baiting programs conducted from 2007 on covered the whole lease, an area of 2350 km^2.

Surveys of cat activity

Monitoring the abundance of feral cats, as for many other mammalian carnivores, is difficult because they occur at low densities, have large home ranges and tend to be secretive and cryptic (Long et al. 2007; Marks et al. 2009; Saunders et al. 1995; Witmer 2005). Capture–recapture studies to estimate abundance are usually impractical because mammalian carnivores are difficult to trap, leading to low capture rates and recapture probabilities (Saunders et al. 1995). Consequently, most monitoring schemes are reliant on indices of abundance derived from data such as track (footprint) counts. We acknowledge that the number of tracks encountered during a track-based survey is likely to be a function of population density as well as activity levels of individuals in the population. Activity levels may be influenced by factors such as seasonal changes in behaviour during breeding and dispersal and the availability of resources including food and shelter (Berry et al. 2012; Edwards et al. 2000; Engeman et al. 2002; Wilson & Delahay 2001). Baiting is also likely to influence the activity patterns of feral cats, as movements may be influenced by removal of neighbours and the imposed changes in population density, or as a consequence of disrupted social structures. Despite these potentially confounding factors, track-count indices were used at Lorna Glen as an index to abundance because mark-recapture methods were unlikely to yield estimates due to low predicted recapture rates. Here, the track-based surveys were simple to implement and time-efficient as extensive lengths of track needed to be surveyed rapidly. The intention was not to estimate population size but to compare activity indices immediately prior to and immediately following baiting programs. Apart from the pilot study, surveys were conducted for four (or five, depending on year) consecutive days immediately prior to the day of baiting (pre-baiting survey period) and for four or five consecutive days approximately ten days directly after baiting (post-baiting survey period). Over this short time period, the population was assumed closed and therefore the difference in indices pre- and post-baiting was a measure of baiting impact. To monitor broad trends over time, cat activity was also surveyed at approximately seasonal intervals (October, December, April and the baiting period of July/August) to provide information on the rate and magnitude of reinvasion into the baited site. The surveys to derive indices of activity were initially conducted along lengths of continuous track (2004–2007). In 2007, survey tracks were repositioned to focus on the area that encompassed the proposed reintroduction site. Not all these tracks were suitable to conduct continuous track counts so sand plots were also included in the survey method to monitor feral cat activity over the period 2007–2009.

Continuous track counts

An extensive network of tracks and roads was present throughout the study area. Many of these tracks consisted of a sandy surface substrate that enabled cat footprints to be observed and thus daily cat activity to be monitored along their length. The monitoring methods used were similar to that outlined in Edwards et al. (2000) and Burrows et al. (2003). Prior to the commencement of each survey period, each track was cleared of previous cat activity by towing a heavy iron drag behind a 4WD vehicle. Track counts were conducted each survey by two highly experienced observers, driving all-terrain vehicles (ATVs) at a speed of 10–15 km h^{-1}. Tracks were inspected for cat tracks each morning one hour after sunrise, and cleared of animal activity by towing a light-weight chain iron drag. To reduce potential observer bias, inspection of tracks between the observers was rotated on a daily basis. Similarly, survey periods were alternated between two different skilled observer groups. Footprints of individual animals were differentiated on the basis of location on the track. While cats usually followed the dragged tracks for some distance, occasionally they would wander on and off the track. To reduce the possibility of double counting, footprints were assigned to an individual animal if no other foot prints were present on at least the previous 1 km of track. Subsequent footprints were also assigned to that individual unless at least 1 km was traversed with no new foot prints present or the imprint could be clearly differentiated on the basis of size or the direction of travel or the direction of entry/exit to and from the track. Each time new cat footprints were encountered along the track, data were recorded on the direction of movement (i.e. whether the animal walked along the track or crossed it), distance of the footprints from the start of the track, approximate size of imprints and whether more than one animal was present. Using these rules and the same skilled observers, the technique provided a reliable and repeatable survey protocol.

Sand plots counts

From 2007–09, 10 permanently marked sand plots, located at 1 km intervals along the survey tracks, were also used to monitor feral cat activity. Each sand plot, 1 ×

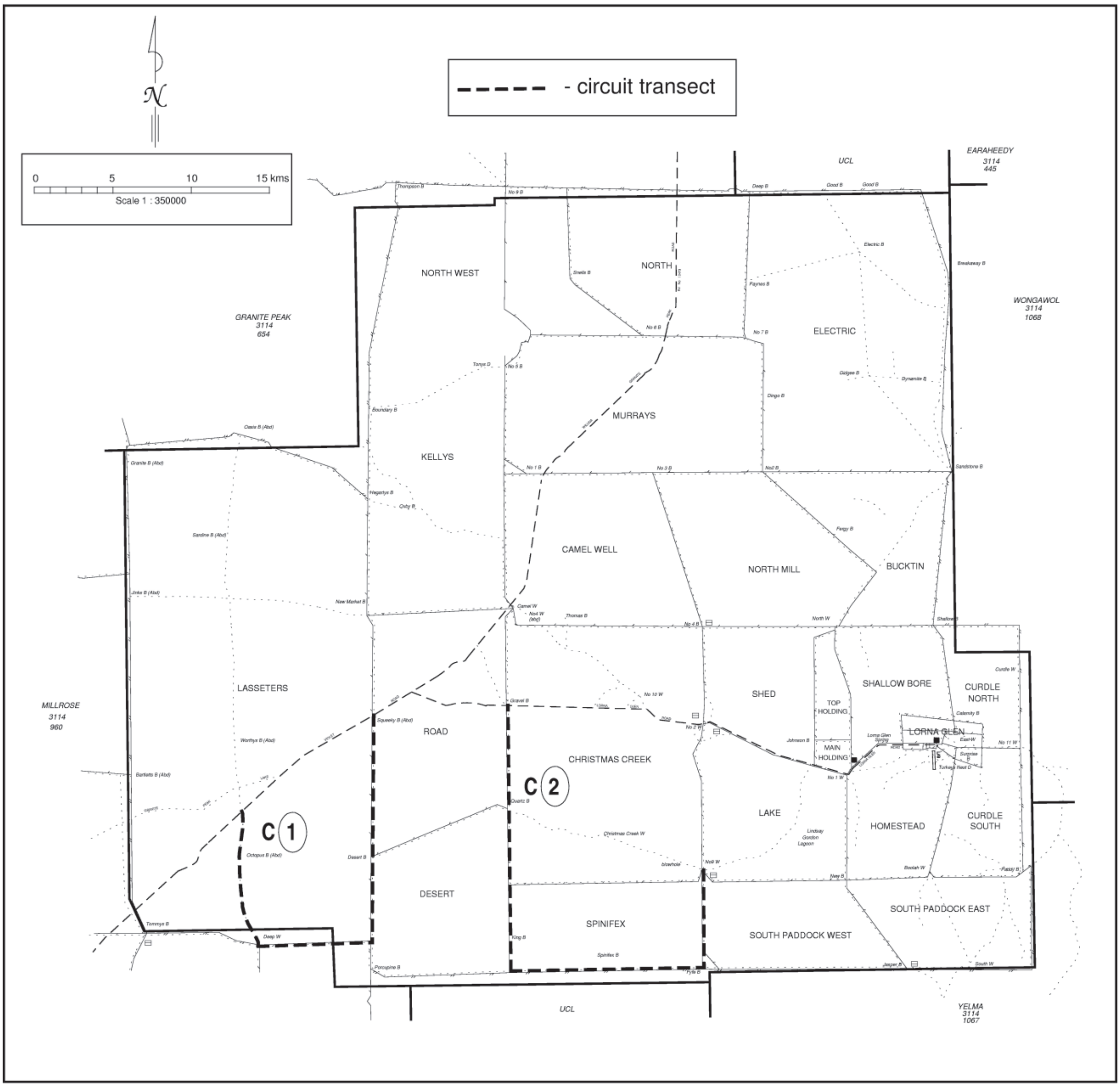

Figure 1. Location of the two circuit survey tracks (2003); C1 in baited site and C2 in non-baited site.

1 m in size, was positioned on the track edge and cleared into a bush or manufactured with brush/fallen trees. Lures were used at each sand plot to increase the likelihood of detecting animals, particularly at low density, and to maximize the number of plot incursions for each survey period. Plots that have no attracting lure generate sample sizes that are too low to adequately monitor population changes (Fleming et al. 2001). In this study, plots contained an audio lure (Felid Attracting Phonic, Westcare Industries, Western Australia) and a non-toxic *Eradicat*® bait to attract cats to the sand plots. The use of a non-toxic *Eradicat* ® bait also enabled us to monitor non-target bait take through the year (M Onus & D Algar, unpub. data). The audio lure was located at the back of the plot, either concealed under leaf litter or hidden within the bush. The *Eradicat* ® bait was located approximately halfway into the plot from the entrance and sprayed with an ant deterrent compound (Coopex®). *Eradicat*® baits were replaced following removal. Both lures were removed outside the survey periods. Each plot was observed for the presence or absence of tracks, as it was not possible to determine the number of intrusions by individual animals onto the plot. Each day, the plots were swept to clear evidence of previous activity.

Phase 1: 2003

In 2003, feral cat activity was monitored along two track circuits, each 36 km in length, with one track circuit located in the baited site and the other in the non-baited site (Fig. 1). A baiting program was conducted on 13 July with each circuit monitored daily for three consecutive days prior to the baiting program (10–12 July) and similarly for 13 days two days after the baiting program (15–27 July).

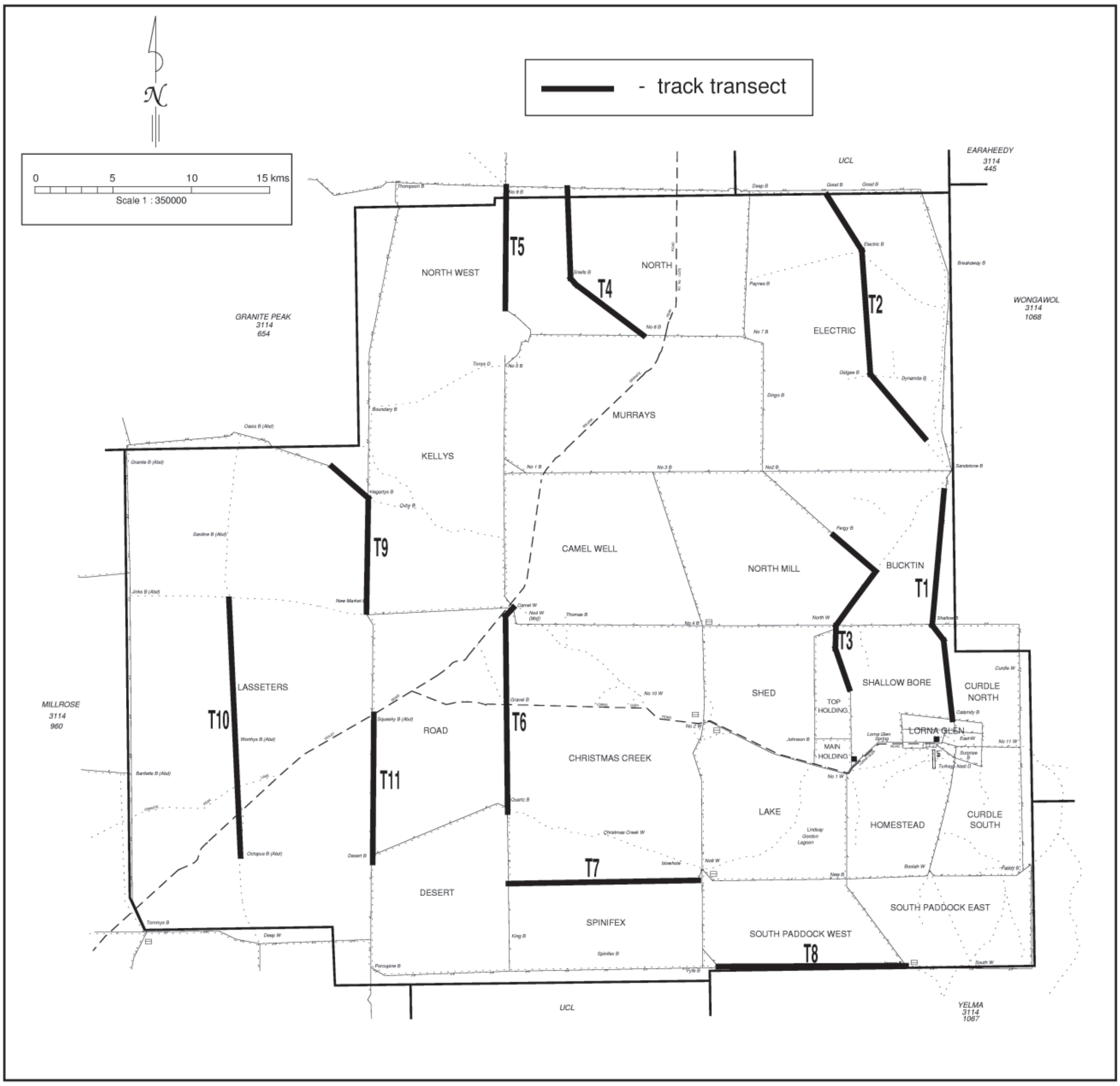

Figure 2. Location of the 11 permanent survey tracks (2004–2006). Survey tracks T1, T2, T3, T4, T5, T9, T10 and T11 were in the baited site and survey tracks T6, T7 and T8 were in the non-baited site.

Phase 2: 2004–06

Over the period 2004–06, the efficacy of baiting programs and subsequent reinvasion was assessed by comparing the track activity of feral cats in the baited and non-baited sites. To be able to make valid statistical comparisons of cat activity indices over time, it was essential to have data from a number of tracks rather than a single continuous circuit. It was also necessary to separate the tracks by sufficient distance so that the tracks were spatially-independent sampling units, thus minimizing the probability of a single animal being recorded on more than one track in any single survey period (e.g. Beier & Cunningham 1996; Edwards et al. 2000; Kendall et al. 1992; Sargeant et al. 1998; Wilson & Delahay 2001; Zielinski & Stauffer 1996).

In 2004, 11 permanent survey tracks were established across the study site. Eight tracks were located within the baited site and three tracks were placed in the non-baited site (see Fig. 2). The tracks varied in length from 10 to 16.5 km. To ensure independence (Harrison et al. 2002), tracks were separated by a minimum distance of at least 5 km, a distance that has been adopted at other sites in the arid zone (Edwards et al. 2000; Burrows et al. 2003) and was considered to be larger than the average diameter of a feral cat home range for the area. The survey tracks chosen aimed to provide a broad coverage of the entire study area and an efficient and representative sampling of the population using the surrounding habitat. As a number of surveys were conducted over time on the same area, the same locations were used (Engeman et al. 2002).

Counts of cat footprints were conducted along each

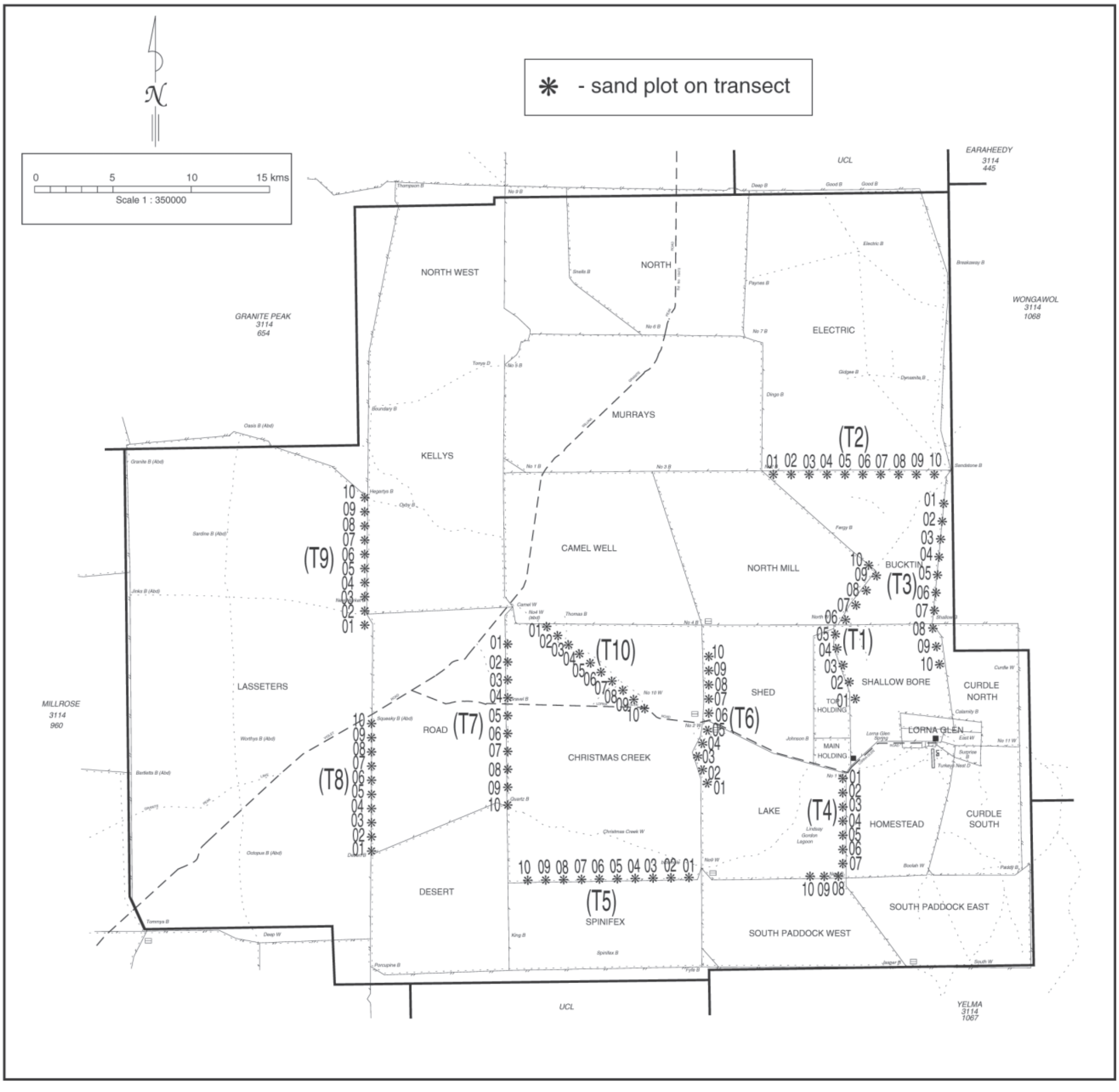

Figure 3. Location of the 10 permanent survey tracks (T1–T10) used in 2007–2009.

track for five consecutive days (where possible) during each survey period in 2004. Preliminary examination of this data was conducted using site level standard error, based on sub-sampling sets of 1, 2, 3, 4 or 5 consecutive days. Standard errors were 0.121, 0.088, 0.077, 0.074 and 0.076 for days 1 to 5 respectively and indicated that precision increased with increased number of sampling days up to four days, but the fifth day added virtually nothing to the precision. As such, surveys in 2005 and 2006 were conducted over four-day periods.

Phase 3: 2007–09

In 2007, the existing feral cat monitoring tracks were repositioned to focus on the area that encompassed the proposed reintroduction site. Ten permanent survey tracks were established across the reintroduction site (see Fig. 3). Each track was 10 km long and aimed to provide a broad coverage of the proposed fauna reintroduction area. The soil substrate along the length of three tracks was unsuitable (i.e. not continuous sand) to conduct continuous track counts. Continuous track counts were performed on tracks T1, T2, T3, T5, T7, T8 and T9 in conjunction with the introduction of sand-plot counts on all tracks to monitor feral cat activity. With the introduction of sand-plot counts, five-day survey periods were reinstated to monitor cat activity along the tracks.

Cat trapping

During Phase 3, a trapping program was implemented seasonally in 2007 (but not following the pre-baiting survey) and less frequently in subsequent years. Feral cats were trapped at locations around the track network, a circuit of 210 km in length. Trapping was conducted on an ad hoc basis and only where cat activity was routinely observed. Padded leg-hold traps, Victor 'Soft Catch'® traps No. 3 (Woodstream Corp., Lititz, Pa., USA) were used, with a mixture of cat faeces and urine as the attractant. A number of different trap sets were used; however, the main type of set employed open-ended trap sets, parallel to the track, with two traps positioned lengthwise (adjoining springs touching) and vegetation/sticks used as a barrier along the trap sides. Trapped cats were destroyed using a 0.22 calibre rifle. All animals captured were sexed and weighed; a broad estimation of age (as either kitten, juvenile or adult) was recorded using weight as a proxy for age. The pregnancy status of females was determined by examining the uterine tissue for embryos.

Calculation of indices and analyses

Continuous track count index

The Track Count Index (TCI) provided an estimate of the minimum number of individual animals responsible for footprints on the tracks. The TCI was calculated by summing the minimum number of cats responsible for the footprints based on the standard rules outlined earlier (TI_{min} of Edwards et al. 2000). Repeated counts of the same sampling unit are not independent because environmental conditions are related across days (Edwards et al. 2000; Engeman et al. 1998) and because some animals may follow the same pattern of movement from day to day. To avoid this problem of temporal non-independence in the track-based population indices, the footprint counts were summed over sampling days for each track (Edwards et al. 2000) and then standardized to the number of individual cats per 100 km. The mean TCI for all tracks was then calculated for each survey period.

Sand-plot counts

Because individuals cannot be identified on the basis of track characteristics, it is customary to just record whether an animal was detected at the station (Ray & Zielinski 2008). These presence/absence data are more robust to statistical analysis than the total number of detections recorded at a station or multiple-station sample units. Thus in this case, sand-plot counts have an index of usage expressed as the mean number of positive plots per night. The track Plot Activity Index (PAI) was formed by calculating an overall mean from the daily means for each survey period (Engeman 2005; Engeman et al. 1998). The VARCOMP procedure within the SAS statistical software package produced the variance component estimates. The PAI does not require assumptions of independence among plots to be made (Engeman et al. 1998). The PAI can be used to monitor populations at different times and statistical comparisons can be made between them (Engeman et al. 1998).

Data analysis

Over the period 2004–06, the program was a BACI design (before–after control–impact; Smith 2002), with a baited site (impact) and non-baited site (control). The data from 2007–09, where no control was available, followed a BA design (before–after; Smith 2002). The data were treated as independent samples and compared using two-sample tests ($\alpha = 0.05$), with any difference attributed to baiting impact. The data were tested for normality and homoscedasticity to satisfy the assumptions necessary for ANOVA and t-tests. Data for the years 2004–06 were analysed using an ANOVA. With removal of the non-baited control in 2007, the impact of baiting was assessed by comparing indices in the baited zone immediately prior to and following individual baiting programs. Comparison of activity changes before and immediately after the baiting program were analysed using a paired t-test. Pearson correlation analysis was used to examine the relationship between the two activity indices (Edwards et al. 2000)

Data for the baited site in 2004–05, 2005–06, 2007–08 and 2008–09 were examined for a reinvasion effect. Comparison of TCIs post-baiting with TCIs pre-baiting the following year were analysed using a paired t-test. Trend analysis was also performed on seasonal TCIs post-baiting through to pre-baiting the following year, using linear regression on log-transformed counts (Elzinga et al. 2001) to detect any consistent changes in TCIs over time.

RESULTS

Baiting programs

Of the seven baiting programs conducted at Lorna Glen to date, six resulted in significant declines in cat activity being recorded after baiting. Prior to any baiting program being conducted, the TCI summed over the period and standardized to number of cats per 100 km was 26.4. The average TCI (mean ± SE) since broad-scale baiting began in 2004 was 6.4 ± 0.7 and only once since 2007 has it been above 10 cats per 100 km.

Phase 1: 2003

During pre-baiting, the TCIs summed over the period and standardized to number of cats per 100 km were 25.9 for the non-baited site and 26.8 for site yet to be baited (Fig. 4). Over the survey period post-baiting, the TCI in the non-baited site was 31.8, while in the baited site the TCI decreased to 0 by day 10 following baiting, indicating a 100% decrease in cat activity.

Phase 2: 2004–06

Analyses indicated that the TCIs, including transformed data, were not normally distributed for combinations of period and site each year, as assessed by Shapiro Wilk's test ($P < 0.05$), and therefore two-way ANOVAs could not be conducted. This was most noticeable in the 2004 dataset, where part of the baited site had been baited the year before. As such, a one-way ANOVA was performed to determine whether there were differences between period–site groups using the non-parametric Kruskal–Wallis test (Allen & Bennett 2012), which does not require the assumptions of normal distribution and equal variances. Pair-wise comparisons were then conducted with a Bonferroni correction for multiple comparisons.

In 2004, TCIs were statistically significantly different between the period–site groups; $Chi^2(3) = 13.098$, $P = 0.004$. Post-hoc analysis revealed a statistically significant decline in TCIs following baiting in the non-baited site (*Mdn* = 28.30 pre-baiting) and (*Mdn* = 18.30 post-baiting; $P = 0.047$) and a highly significant decline in the baited site (*Mdn* = 14.30 pre-baiting) and (*Mdn* = 3.45 post-baiting; $P = 0.006$). The decline in TCIs in the non-baited site was of a much smaller magnitude than that observed in the baited site (Fig. 5). It does, however, suggest that there may have been some impact of baiting on cat activity in the non-baited site, which was adjacent to the much larger baited area.

In 2005, TCIs were statistically significantly different between the period–site groups; $Chi^2(3) = 10.136$, $P = 0.017$. Post-hoc analysis revealed no statistically significant difference in TCIs following baiting within the non-baited site (*Mdn* = 6.10 pre-baiting and *Mdn* = 6.80 post-baiting; $P = 0.827$), but a highly significant decline in the baited site (*Mdn* = 10.00 pre-baiting and *Mdn* = 1.50 post-baiting; $P = 0.006$ (Fig. 5).

In 2006, there was a statistically significant increase in TCIs post-baiting in both the baited and non-baited sites; $Chi^2(3) = 10.250$, $P = 0.017$. TCIs in the non-baited site were *Mdn* = 13.60 pre-baiting and *Mdn* = 35.40 post-baiting, and TCIs in the baited site were *Mdn* = 8.75 pre-baiting and *Mdn* = 15.85 post-baiting. A survey conducted in the baited site along five of the tracks a month later indicated no significant difference between the two

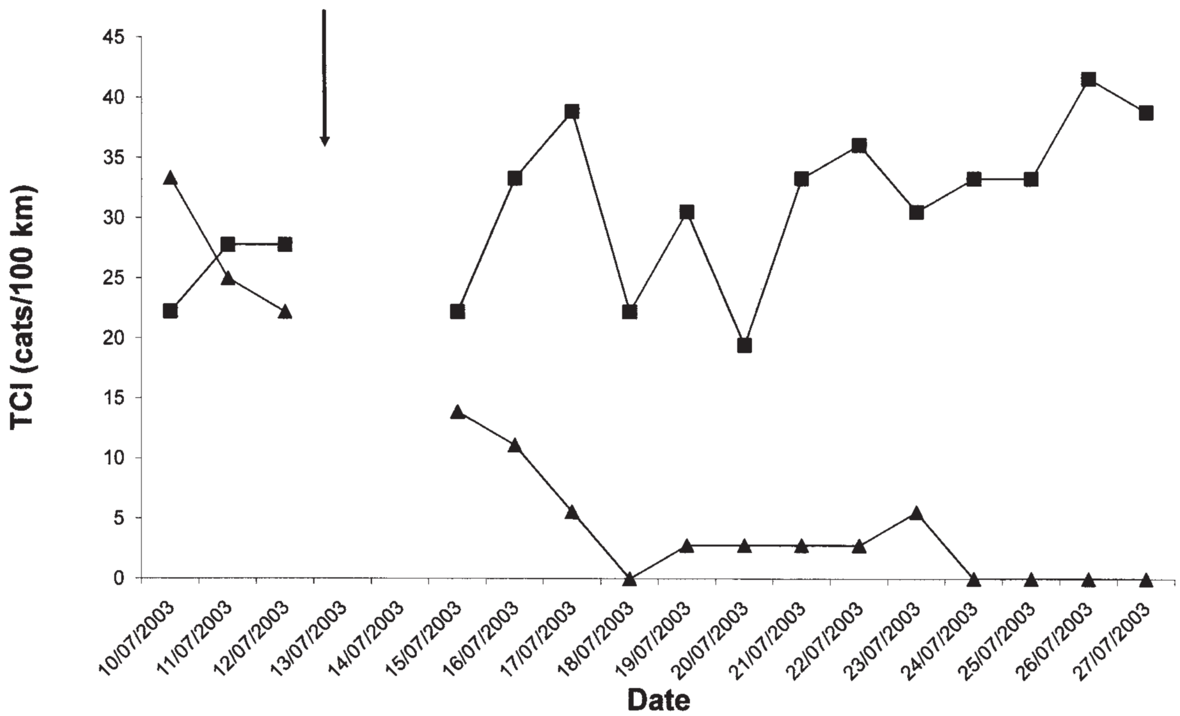

Figure 4. Pilot baiting trial in 2003 (Phase 1). Daily TCIs for the non-baited (■) and baited sites (▲) over the survey period. The arrow indicates the day baiting occurred (13 July 2003).

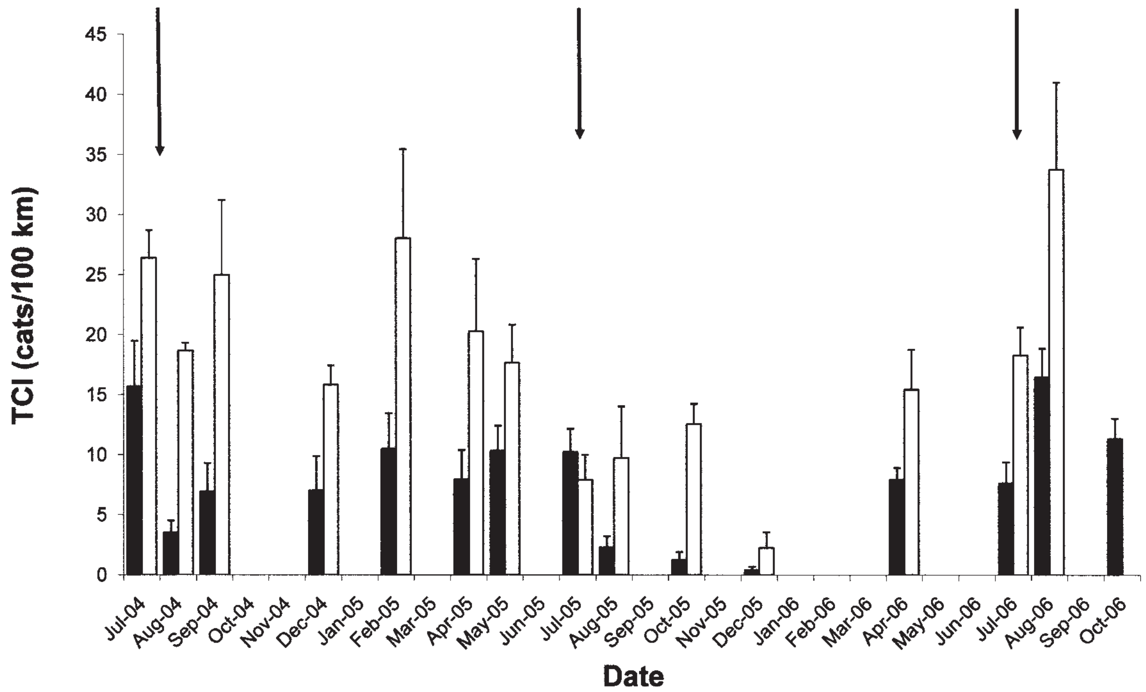

Figure 5. TCIs ± SE for each survey period across 2004–2006. Closed bars represent baited sites and open bars represent non-baited sites. No survey of the non-baited site was conducted in October 2006. Arrows indicate the timing of baiting programs.

survey periods (t = –1.33, P = 0.13). TCIs ± SE for these five tracks were 9.04 ± 2.06, 18.52 ± 2.96 and 11.34 ± 1.70 for pre-bait, post-bait and one month post-bait survey periods respectively (Fig. 5).

Phase 3: 2007–09

Paired t-tests were used to test whether there was a statistically significant mean difference between TCIs before and after baiting. The difference scores for pre- and post-baiting were normally distributed as assessed by the Shapiro Wilk's test ($P > 0.05$) for all years. In 2007, TCIs were lower post-baiting (6.00 ± 1.15) than pre-baiting (11.43 ± 1.89; Fig. 6); a statistically significant reduction of 5.43 (95% CI, 1.47 to 9.38), t(6) = 3.36, P = 0.015, d = 1.27. In 2008, TCIs were lower post-baiting (2.67 ± 1.12) than pre-baiting (9.67 ± 2.44; Fig. 6); a statistically significant reduction of 7.00 (95%

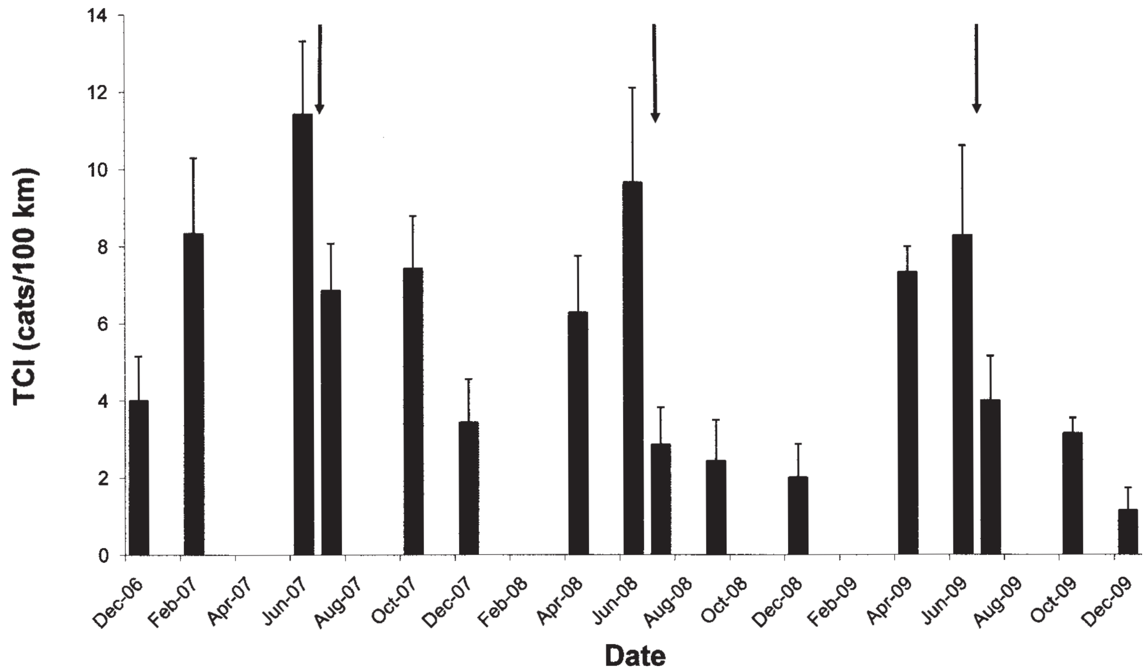

Figure 6. TCIs ± SE for each survey period across 2007–2009. Arrows indicate the timing of baiting programs.

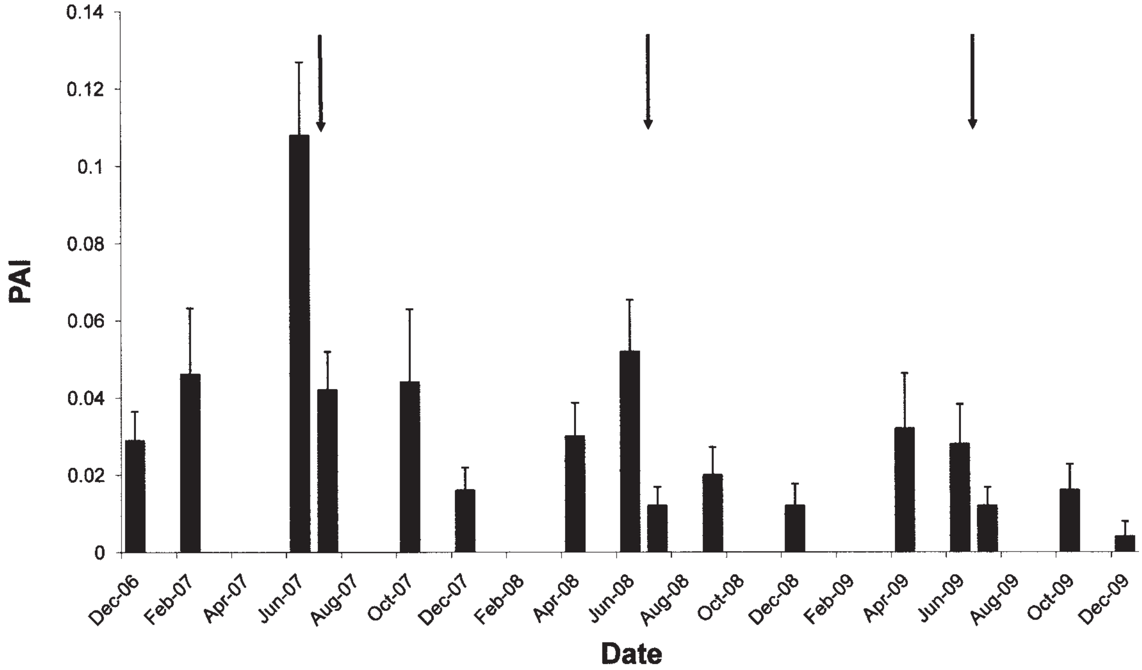

Figure 7. PAIs ± SE for each survey period across 2007–2009. Arrows indicate the timing of baiting programs.

CI, 0.13 to 13.87), t(5) = 2.62, $P = 0.047$, d = 1.07. In 2009, TCIs were lower post-baiting (3.67 ± 1.31) than pre-baiting (9.00 ± 2.62; Fig. 6); a statistically significant reduction of 5.33 (95% CI, 0.80 to 9.87), t(5) = 3.02, $P = 0.029$, d = 1.23.

Paired t-tests were used to test whether there was a statistically significant mean difference between PAIs before and following baiting. The difference scores for pre- and post-baiting were normally distributed as assessed by the Shapiro Wilk's test ($P > 0.05$) for all years. One of the data points in the 2007 pre-baiting dataset was found to be an extreme outlier (PAI = 0.48): rather than removing this value from the dataset it was modified by replacing the outlier's value with the next largest PAI (0.12). In 2007, PAIs were lower post-baiting (0.036 ± 0.011) than pre-baiting (0.072 ± 0.013; Fig. 7); a statistically significant reduction of 0.036 (95% CI, 0.003 to 0.069), t(9) = 2.48, $P = 0.035$, d = 0.78. In 2008, PAIs were lower post-baiting (0.012 ± 0.004) than pre-baiting (0.052 ± 0.014; Fig. 7); a statistically significant reduction of 0.040 (95% CI, 0.008 to 0.072), t(9) = 2.80, $P = 0.021$, d = 0.89. In 2009, there was a lowering of PAIs following baiting (Fig. 7) but it was not statistically significant, t(9) = 2.08, $P = 0.068$, with PAIs of 0.010 ± 0.005 post-baiting and 0.028 ± 0.009 pre-baiting, respectively. There was a strong positive correlation between TCIs and PAIs (r = 0.926, $P < 0.0005$).

Reinvasion

Paired t-tests were used to test whether there was a statistically significant mean difference between post-baiting TCIs in one year with pre-baiting TCIs in the following year. The difference scores for pre- and post-baiting were normally distributed as assessed by the Shapiro Wilk's test ($P > 0.05$) for all years. In 2004–05, TCIs were lower in the post-baiting period in 2004 (4.66 ± 1.33) than in the pre-baiting period in 2005 (10.24 ± 1.93); a statistically significant increase between periods of 5.58 (95% CI, 1.67 to 9.99), t(4) = –3.51, $P = 0.025$, d = 1.57. In 2005–06, TCIs were lower in the post-baiting period in 2005 (2.29 ± 0.94) than in the pre-baiting period in 2006 (7.84 ± 1.87); a statistically significant increase between periods of 5.55 (95% CI, 0.66 to 10.44), t(7) = –2.69, $P = 0.031$, d = 0.95. In 2007–08, there was a increase in TCIs in the post-baiting period in 2007 compared with the pre-baiting period in 2008, but it was not statistically significant t(5) = –0.93, $P = 0.394$, with TCIs post-baiting in 2007 of 7.00 ± 1.44 and pre-baiting in 2008 of 9.67 ± 2.44. The lack of a significant increase in cat activity between these two periods was probably because the majority of cats trapped (73% of the total, see below and Table 2) were captured over this period.

Table 2

Cat capture records over the period 2007–2009.

Trapping Period	No. Captures	
	No. males	No. females
February 07	11	8
July 07	2	2
October 07	10	6
December 07	1	1
February 08	1	2
October 08	2	1
December 08	1	2
February 09	4	2

In 2008–09, TCIs were lower in the post-baiting in 2008 (2.86 ± 0.96) than in the pre-baiting in 2009 (8.29 ± 2.33); a statistically significant increase between periods of 5.43 (95% CI, 0.12 to 10.74), $t(6) = -2.50$, $P = 0.046$, d = 0.95.

Trend analyses indicated that 2004–05 was the only period when there was a statistically significant increase in TCIs detected over time, $t(5) = 3.43$, $P < 0.05$, $\beta = 0.0171 \pm 0.0050$. In 2005–06, 2007–08 and 2008–09 there were no statistically significant increases in TCIs detected over time (2005–06: $t(3) = 2.65$, $P = 0.08$, $\beta = 0.0222 \pm 0.0084$; 2007–08: $t(3) = 0.48$, $P = 0.67$, $\beta = 0.0003 \pm 0.0006$; 2008–09: $t(3) = 2.61$, $P = 0.08$, $\beta = 0.0015 \pm 0.0006$). There was, however, a general tendency across years of increases in cat activity indices in late summer, through autumn and early winter prior to the baiting programs.

Trapping Programs

Over the period 2007–09, 56 cats were trapped comprising 32 males and 24 females (Table 2). Seventy-three per cent of these cats were trapped in 2007. Body weight (mean ± SE) for males was 3.7 ± 0.1 kg (range 2.5–5.1 kg) and 2.7 ± 0.1 kg (range 1.5–3.3 kg) for females. Using the yearling weight/age classes reported by Jones and Coman (1982), the population age structure of trapped cats at Lorna Glen were defined. Body weights for male cats indicated four were less than 3.0 kg and considered to be juvenile animals, 20 cats were between 3.0–4.0 kg, a weight that approximates that for sexual maturity and considered to be young adults of between 1–2 years of age and eight cats were >4.0 kg and were considered to be greater than two years of age. Body weights for female cats indicated eight were less than 2.5 kg and considered to be juvenile animals, nine cats were between 2.5–3.0 kg, a weight that approximates that for sexual maturity and considered to be young adults of between 1–2 years of age and seven cats were 3.0 plus kg and were considered to be greater than two years of age. The majority of adult females had kittens in utero during the spring, suggesting a main birth season in the late spring/early summer.

DISCUSSION

Baiting Effect

Prior to the implementation of baiting programs at Lorna Glen, the average cat activity index was 26.4 cats per 100 km. Following broad-scale baiting in 2004, the average cat activity index over the period of this program has been 6.4 ± 0.7 (mean ± SE) and only three times has it been much greater than 10 cats per 100 km (16.5 ± 2.4 in August 2006, 11.3 ± 1.9 in September 2006 and 11.4 ± 1.9 in June 2007). Reducing and maintaining cat numbers to less than 10 per 100 km was our benchmark, as this is the suggested level at which reintroductions of native species could potentially occur (Morris et al. 2004).

Six of the seven baiting programs conducted at Lorna Glen resulted in significant reductions in cat activity indices immediately after baiting. The baiting program conducted in 2006 was the only one that did not result in a significant decline in cat activity after baiting; in fact, an increase in cat activity occurred. The reason for this increase in the indices of cat activity in both the baited and non-baited site during this year is unclear. The following survey indicated that there was no difference in indices of cat activity between the two periods; indicating that baiting had no impact on cat activity that year. The lack of response to baiting was most likely due to the significant rainfall events that occurred over the summer months, which probably led to irruptions of small mammal populations that resulted in a major increase in prey availability (Dickman et al 1999; Morton 1990). The relationship between prey availability and bait consumption is discussed further below.

What is clear from this project is that, in most cases, the implementation of an annual baiting strategy can provide for the effective and sustained control of feral cats at the landscape level. This statement needs some qualification, as the imposition of a trapping program in 2007, following the lack of baiting impact in 2006, would have had contributed to the control of cat numbers throughout the year. In 2008 and 2009, the limited amount of cat trapping would have had little impact on the overall control achieved. The cat control strategy will benefit from the addition of supplementary, limited trapping programs, which may be required in years of high rainfall when baiting is less likely to have an impact because of the probable increase in prey resource.

The efficacy of these programs confirms previous findings from other sites in the arid and semi-arid zone (Algar & Burrows 2004; Burrows et al. 2003) that baiting for feral cats in the winter months, when prey availability is low, can achieve highly effective control. Similar results were also achieved following baiting programs in 2006 and 2007 at Mt Gibson in the southern rangelands, which resulted in significant reductions in cat activity recorded in both years. Since the 2006 baiting program at Mt Gibson, indices of feral cat activity did not recover to their original level with the implementation of an annual baiting regime (Algar & Richards 2010). A number of baiting programs conducted on islands have also been successful and have led to eradication of cats on Hermite Island, Montebellos (Algar et al. 2002) and Faure Island, Shark Bay (Algar et al. 2010), and will be the primary tool in the proposed eradication of cats on Dirk Hartog Island, Shark Bay (Algar et al. 2011a; Algar et al. 2011b).

The impact of broad-scale cat baiting at several other mainland sites has, however, been highly variable and/or less successful. There appear to be three factors that are critical to the outcome of baiting programs: 1) baiting intensity and bait encounter, 2) the abundance of prey items, and 3) weather conditions at the time of baiting. Cats, despite being opportunistic predators, will only consume a food item if they are hungry (Bradshaw 1992). If a cat encounters a bait when not hungry it may not be consumed regardless of the acceptability of the bait. The relationship between bait consumption and hunger can be extended to prey abundance, which is also a function of long-term weather conditions (season/rainfall). The likelihood of cats encountering baits when hungry is potentially diminished in the presence of an abundant prey population. Therefore bait uptake is invariably low when prey availability is high (Algar et al. 2007). The impact of baiting can also be substantially reduced if significant rainfall occurs immediately following the baiting program. Rain renders the baits less palatable to cats by washing away the oils and flavour enhancers that sweat to the surface of the bait. Bait longevity in the field is a critical component in developing successful baiting campaigns to target feral cats.

At Roxby Downs, South Australia, feral cat baiting programs using non-recommended bait application rates of 10–25 baits km^{-2}, have met with little success (Moseby et al. 2009). Movement patterns of radio-collared cats and inferred bait detection distances were used to suggest optimum baiting densities of at least 30 baits km^{-2} (Moseby et al. 2009). This baiting application rate, however, does not take into account that cats must encounter the bait when hungry and any reduction of bait availability due to non-target bait consumption.

When prey is plentiful—such as on the Peron Peninsula in Shark Bay, Western Australia, where there is an abundant rabbit (*Oryctolagus cuniculus*) population—cat baiting programs have met with varied success (Morris et al. 2004). Rabbits, when present, can form a substantial proportion of a feral cat's diet (e.g. Jones & Coman 1981; Martin et al. 1996; Risbey et al. 1999; Project Eden, unpub. data). The presence of such an abundant prey resource can impact upon bait acceptance by feral cats, as was found on the adjoining Heirisson Prong (Short et al. 1997). Subsequent studies on Peron Peninsula and with sites where rabbits were absent or in low numbers have shown that bait uptake by feral cats can be driven by the abundance of this primary prey (Algar & Burrows 2004; Algar & Angus 2008; Algar et al. 2007). Reduction of rabbit abundance on the peninsula could improve bait acceptance and extend the period of effective baiting.

Baiting programs conducted at Lorna Glen in 2004 and 2005 were likely to have been adversely affected by inclement weather, which may have resulted in a reduction in baiting efficacy. Cloudy conditions and rainfall hindered both bait preparation in the field and aerial deployment. Baiting programs conducted at Mt Gibson and Karara–Lochada in 2008 had no significant impact on indices of feral cat activity at either site (Algar & Richards 2010). This lack of response to baiting was most likely due to a major rainfall event that occurred immediately after baiting. Baiting outcomes could be improved if long-term weather forecasts are used to ensure that baiting programs are only conducted when prolonged periods of fine weather are assured. An operational protocol has now been established within DPaW to minimize the possibility of poor baiting outcomes due to adverse weather conditions (Algar & Richards 2010). Preparation of the baits prior to aerial delivery is also of critical importance to the success of the baiting program. In the field, baits must be permitted to sweat on racks under sunny conditions to allow the oils and lipid-soluble digest material to exude from the surface of the bait. If this process is prevented or interrupted due to adverse weather conditions, the baits may rapidly deteriorate and become either rancid or mouldy and, as a consequence, unpalatable to cats.

Reinvasion

The dispersal of young cats is not clearly understood; however, dispersal cannot occur before the permanent canines have erupted, which commences three and a half months after birth (Hemmer 1979). Females rarely venture as far afield as males, often establishing a territory close to that of their mother. Yearling/subordinate males tend to remain within their natal range until they are old and strong enough to establish themselves as dominants (Liberg 1981, cited in Liberg & Sandell 1988). As they grow they come under increasing attack from older males and in their second year they usually disperse (Liberg 1981).

At Lorna Glen, there was a general trend across years that indices of cat activity only tended to increase six months following baiting, that is, in the late summer, through autumn and early winter prior to the following baiting program. These results suggest that immediately after baiting there was little movement into vacated territories by resident animals that survived the baiting program. The majority of reinvasion appears to have come from natal recruitment or dispersal/immigration of young adult animals from outside the baited area. Assuming no bias in trappability, the body weights of the majority of trapped animals (73%) were within the first and second year cohort and therefore support this proposal. The impact of reinvasion by feral cats could be mitigated to some extent by increasing the size of the area baited. This will essentially provide a buffer zone around a core area of conservation significance.

This project has demonstrated that sustained control of feral cats can be achieved in the rangelands using an annual baiting strategy occasionally augmented with trapping. In the short term that this program has been operational, we have demonstrated sustained control of feral cats to levels of 10 cats per 100 km. It is anticipated that refinements to bait preparation and deployment, and trapping techniques, will further enhance the effectiveness of the control strategy and provide the sustained control required to allow reintroduction of a suite of native mammals.

ACKNOWLEDGEMENTS

DPaW staff from Manjimup and Kalgoorlie have contributed to this project over the past seven years and their assistance is gratefully acknowledged. We would also like to thank Neil Burrows, the former DPaW Science Director, for his support and encouragement during the course of this program. His comments on an earlier draft have improved this manuscript. The manuscript was further improved following the adoption of comments and suggestions by two anonymous reviewers. The maps were produced by Rob Doria, Geographic Information Services Section at DPaW. Finally we would like to thank the various caretakers at Lorna Glen over the years, in particular Bruce and Kay Withnell, for their hospitality and helpfulness. The DPaW Animal Ethics Committee approved protocols 17/2001, 06/2006, and 35/2009 which describe activities undertaken in this project. Prior to the baiting program a DPaW 1080 Risk Assessment was undertaken, and approval from the Australian Pesticides and Veterinary Medicine Authority granted (Permit Numbers PER6333, PER9883 and PER10634).

REFERENCES

Algar D, Angus GJ (2008) Feasibility study for the eradication of feral cats from Faure Island, Shark Bay, Western Australia. *Records of the Western Australian Museum* Supplement **75**, 71–75.

Algar D, Burrows ND (2004) A review of Western Shield: feral cat control research. *Conservation Science Western Australia* **5**, 131–163.

Algar D, Angus GJ, Williams MR, Mellican AE (2007) Influence of bait type, weather and prey abundance on bait uptake by feral cats (*Felis catus*) on Peron Peninsula, Western Australia. *Conservation Science Western Australia* **6**, 109–149.

Algar D, Angus GJ, Brazell RI, Gilbert C, Withnell GB (2010) Eradication of feral cats on Faure Island, Western Australia. *Journal of the Royal Society of Western Australia* **93**, 133–140.

Algar D, Burbidge AA, Angus GJ (2002) Cat eradication on the Montebello Islands. In *Turning the Tide: The Eradication of Invasive Species* (eds CR Veitch, MN Clout), pp. 14–18. IUCN, SSC, Invasive Species Specialist Group, IUCN, Gland, Switzerland and Cambridge, U.K.

Algar D, Hilmer S, Onus M, Hamilton N, Moore J (2011a) New national park to be cat-free. *LANDSCOPE* **26**, 39–45.

Algar D, Johnston M, Hilmer SS (2011b) A pilot study for the proposed eradication of feral cats on Dirk Hartog Island, Western Australia. In *Island Invasives: Eradication and Management* (eds CR Veitch, MN Clout), pp 10–16. IUCN, (International Union for Conservation of Nature), Gland, Switzerland.

Algar D, Richards J (2010) Controlling introduced predators in the rangelands—the conclusion. *LANDSCOPE* **25**, 17–23.

Allen P, Bennett K (2012) *SPSS Statistics a Practical Guide* (Ver. 20). RR Donnelley Asia Printing Solutions Ltd., China.

Beard J (1974) *Vegetation Survey of Western Australia. Murchison. 1:1 000 000 Vegetation Series*. University of Western Australia Press, Perth.

Beier P, Cunningham SC (1996) Power of track surveys to detect changes in cougar populations. *Wildlife Society Bulletin* **24**, 540–546.

Berry O, Algar D, Angus J, Hamilton N, Hilmer S, Sutherland D (2012) Genetic tagging reveals a significant impact of poison baiting on an invasive species. *Journal of Wildlife Management* **76**, 729–739.

Bradshaw JWS (1992) *The Behaviour of the Domestic Cat*. CAB International, Wallingford.

Burbidge AA, McKenzie NL (1989) Patterns in the modern decline of Western Australia's vertebrate fauna: causes and conservation implications. *Biological Conservation* **50**, 143–198.

Burrows ND, Algar D, Robinson AD, Sinagra J, Ward B, Liddelow G (2003) Controlling introduced predators in the Gibson Desert of Western Australia. *Journal of Arid Environments* **55**, 691–713.

Christensen PES, Burrows ND (1995) Project Desert Dreaming: the reintroduction of mammals to the Gibson Desert. In *Reintroduction Biology of Australian and New Zealand Fauna* (ed M Serena), pp. 199–208. Surrey Beatty and Sons, Chipping Norton, NSW.

Dickman CR (1996a) Impact of exotic generalist predators on the native fauna of Australia. *Wildlife Biology* **2**, 185–195.

Dickman CR (1996b). *Overview of the Impact of Feral Cats on Australian Native Fauna*. Report to Australian Nature Conservation Agency. Institute of Wildlife Research and School of Biological Sciences, University of Sydney.

Dickman CR, Mahon PS, Masters P, Gibson DF (1999) Long-term dynamics of rodent populations in arid Australia: the influence of rainfall. *Wildlife Research* **26**, 389–403.

Dunlop J, Morris K (2009) Into the wild: restoring rangelands fauna. *LANDSCOPE* **24**, 52–58.

Environment Australia (1999) *Threat Abatement Plan for Predation by Feral Cats*. Environment Australia, Biodiversity Group, Canberra.

Edwards GP, de Prue ND, Shakeshaft BJ, Crealy IV (2000) An evaluation of two methods of assessing feral cat and dingo abundance in central Australia. *Wildlife Research* **27**, 143–149.

Elzinga CL, Salzer DW, Willougby JW, Gibbs JP (2001) *Monitoring Plant and Animal Populations*. Blackwell Science Inc. Malden, Massachusetts.

Engeman RM (2005) Indexing principles and a widely applicable paradigm for indexing animal populations. *Wildlife Research* **32,** 203–210.

Engeman RM, Allen L, Zerbe GO (1998) Variance estimate for the Allen activity index. *Wildlife Research* **25**, 643–648.

Engeman RM, Pipas MJ, Gruver KS, Bourassa J, Allen L (2002) Plot placement when using a passive tracking index to simultaneously monitor multiple species of animals. *Wildlife Research* **29**, 85–90.

Fischer J, Lindenmayer DB (2000) An assessment of the published results of animal relocations. *Biological Conservation* **96**, 1–11.

Fleming PJS, Corbett L, Harden B, Thomson P (2001) *Managing the Impacts of Dingoes and Other Wild Dogs*. Bureau of Rural Sciences, Australian Government Publishing Service, Canberra.

Gibson DF, Johnson KA, Langford DG, Cole JR, Clarke DE, Willowra Community (1995) The Rufous Hare-wallaby *Lagorchestes hirsutus*: a history of experimental reintroduction in the Tanami Desert, Northern Territory. In *Reintroduction Biology of Australian and New Zealand Fauna* (ed M Serena), pp 171–176. Surrey Beatty and Sons, Chipping Norton, NSW.

Harrison RL, Barr DJ, Dragoo JW (2002) A comparison of population techniques for swift foxes (*Vulpes velox*) in New Mexico. *American Midland Naturalist* **148**, 320–337.

Hemmer H (1979) Gestation period and postnatal development in felids. *Carnivore* **2**, 90–100.

Johnson KA, Burbidge AA, McKenzie NL (1989) Australian Macropodoidea: status, causes of decline and future research and management. In *Kangaroos, Wallabies and Rat Kangaroos* (eds G Grigg, P Jarman, I Hume), pp. 641–657. Surrey Beatty and Sons, New South Wales, Australia.

Jones E, Coman BJ (1981) Ecology of the feral cat, *Felis catus* (L.), in south-eastern Australia. I. Diet. *Australian Wildlife Research* **8**, 537–547.

Jones E, Coman BJ (1982) Ecology of the feral cat, *Felis catus* (L.), in south-eastern Australia. II. Reproduction. *Australian Wildlife Research* **9**, 111–119.

Kendall KC, Metzgar LH, Patterson DA, Steele BM (1992) Power of sign surveys to monitor population trends. *Ecological Applications* **2**, 422–430.

Liberg O (1981) 'Predation and social behaviour in a population of domestic cats: an evolutionary perspective'. PhD thesis, University of Lund, Sweden.

Liberg O, Sandell M (1988) Spatial organisation and reproductive tactics in the domestic cat and other felids. In *The Domestic Cat—The Biology of its Behaviour* (eds DC Turner, P Bateson), pp. 83–89. Cambridge University Press, Cambridge.

Long RA, Donovan TM, Mackay P, Zielinski WJ, Buzas JF (2007) Comparing scat detection dogs, cameras, and hair snares for surveying carnivores. *Journal of Wildlife Management* **71**, 2018–2025.

Mabbut JA, Litchfield WH, Speck NH, Sofoulis J, Wilcox DG, Arnold JM, Brookfield M, Wright RL (1963) *General Report on Lands of the Wiluna-Meekatharra Area, Western Australia.* Land Research Series No. 7. CSIRO, Melbourne.

Marks CA, Gigliotti F, McPhee S, Piggott MP, Taylor A, Glen AS (2009) DNA genotypes reveal red fox (*Vulpes vulpes*) abundance, response to lethal control and limitations of contemporary survey techniques. *Wildlife Research* **36**, 647–658.

Martin GR, Twigg LE, Robinson DJ (1996) Comparison of the diet of feral cats from rural and pastoral Western Australia. *Wildlife Research* **23**, 475–484.

McKenzie NL, Burbidge AA, Baynes A, Brereton RN, Dickman CR, Gordon G, Gibson LA, Menkhorst PW, Robinson AC, Williams MR, Woinarski JCZ (2007) Analysis of factors implicated in the recent decline of Australia's mammal fauna. *Journal of Biogeography* **34**, 597–611.

Morris K, Sims C, Himbeck K, Christensen P, Sercombe N, Ward B, Noakes N (2004) Project Eden—fauna recovery on Peron Peninsula, Shark Bay: *Conservation Science Western Australia* **5**, 202–234.

Morton SR (1990) The impact of European settlement on the vertebrate animals of arid Australia: a conceptual model. *Proceedings of the Ecological Society of Australia* **16**, 201–213.

Moseby KE, Stott J, Crisp H (2009) Improving the effectiveness of poison baiting for feral cat and European fox in northern South Australia: the influence of movement, habitat use and activity. *Wildlife Research* **36**, 1–14.

Ray JC, Zeilinski WJ (2008) Track stations. In *Noninvasive Survey Methods for Carnivores* (eds RA Long, P MacKay, WJ Zielinski, JC Ray), pp. 75–109. Island Press, Washington.

Risbey DA, Calver MC, Short J (1999) The impact of cats and foxes on the small vertebrate fauna of Herisson Prong, Western Australia. I. Exploring potential impact using diet analysis. *Wildlife Research* **26**, 621–630.

Risbey DA, Calver M, Short J, Bradley JS, Wright IW (2000) The impact of cats and foxes on the small vertebrate fauna of Heirisson Prong, Western Australia. II. A field experiment. *Wildlife Research* **27**, 223–235.

Sargeant GA, Johnson DH, Berg WE (1998) Interpreting carnivore scent-station surveys. *Journal of Wildlife Management* **62**, 1235–1245.

Saunders G, Coman B, Kinnear JE, Braysher M (1995) *Managing Vertebrate Pests: Foxes*. Australian Government Publishing Service, Canberra.

Short J, Turner B, Risbey DA, Carnamah R (1997) Control of feral cats for nature conservation. II. Population reduction by poisoning. *Wildlife Research* **24**, 703–714.

Smith AP, Quin DG (1996) Patterns and causes of extinction and decline in Australian conilurine rodents. *Biological Conservation* **77**, 243–267.

Smith EP (2002). BACI design. In *Encyclopedia of Environmetrics* (eds AH El-Shaarawi, WW Piegorsch), pp. 141–148. John Wiley & Sons Ltd, Chichester.

Thackway R, Creswell ID (1995) *An Interim Bioregionalisation for Australia*. Australian Nature Conservation Agency, Canberra.

Wilson GJ, Delahay RJ (2001) A review of methods to estimate the abundance of terrestrial carnivores using field signs and observation. *Wildlife Research* **28**, 151–164.

Witmer GW (2005) Wildlife population monitoring: some practical considerations. *Wildlife Research* **32**, 259–263.

Zieilinski WJ, Stauffer HB (1996) Monitoring *Martes* populations in California: survey design and power analysis. *Ecological Applications* **6**, 1254–1267.