Aerial baiting with 1080 to control wild dogs does not affect the populations of two common small mammal species

Article *in* Wildlife Research · January 2009

DOI: 10.1071/WR08134

CITATIONS
8

READS
105

3 authors, including:

Karl Vernes
University of New England (Australia)
97 PUBLICATIONS **925** CITATIONS

SEE PROFILE

CSIRO PUBLISHING

Wildlife Research, 2009, **36**, 528–532 www.publish.csiro.au/journals/wr

Aerial baiting with 1080 to control wild dogs does not affect the populations of two common small mammal species

Sally Fenner[A], *Gerhard Körtner*[B,C] *and Karl Vernes*[A]

[A]Ecosystem Management, University of New England, Armidale, NSW 2351, Australia.
[B]Centre for Behavioural and Physiological Ecology, Zoology, University of New England, Armidale, NSW 2351, Australia.
[C]Corresponding author. Email: gkoertne@une.edu.au

Abstract. More than most other animal control techniques, toxic baiting is fraught with the potential impact on non-target species. In the present study, we investigated the effect of aerial baiting with 1080 to control wild dogs in north-eastern New South Wales (NSW), Australia, on populations of southern bush rats (*Rattus fuscipes assimilis*) and brown antechinus (*Antechinus stuartii*), using a controlled experiment. Six populations, three each within widely spaced baited and unbaited trapping grids, were monitored before and after bait laying. To develop capture–mark–recapture indices, separate 4-day trapping surveys were undertaken twice before and twice after meat baits (250 g containing 6 mg sodium fluoroacetate, 1080) were delivered from a helicopter at 40 baits per kilometre. To assess non-fatal bait consumption, all baits contained rhodamine B (RhB), which gets incorporated into the vibrissae of animals that have ingested this marker. Neither mammal population decreased in size after baiting, nor was there any increase in population turnover rates or changes in the movement patterns of either species. Furthermore, no trapped animal tested positive for RhB, suggesting that these small mammals rarely consume meat baits, and that, at the population level, the impact of baiting on them was likely negligible. It is therefore unlikely that the current practise of aerial baiting in NSW, although effective in reducing dog activity, threatens populations of these two common species and perhaps small mammals in general.

Introduction

Large predators are almost universally perceived as a threat to livestock industries and the control programs to mitigate the impacts of predators, especially toxic baiting, potentially place non-target animals at risk (McIlroy 1981, 1982*a*, 1986).

In parts of Australia, populations of wild dogs, including dingoes (*Canis lupus dingo*), are controlled to prevent attacks on livestock, especially sheep (Fleming *et al*. 2001, 2006). Control measures frequently include the maintenance of an extended buffer zone of forested land, including conservation reserves, to reduce the emigration of wild dogs into areas where livestock are grazed. Because of the rugged nature of many of these buffers, aerial baiting with 1080 (sodium fluoroacetate) meat baits is often the only feasible option for wild dog control (Fleming *et al*. 1996). In Australia, this has been an established practice for over 40 years and, in parts of NSW, 1080 meat baits are delivered from a helicopter along pre-determined transects at a rate of up to 40 baits per kilometre of flight path (Fleming *et al*. 2001, 2006). Surface-laid baits potentially expose several native carnivorous and omnivorous species to poisoning (McIlroy 1986). In fact, the current rate of baiting is used not only to maximise the exposure of target animals, but also to compensate for the removal of baits by introduced, e.g. feral pigs, foxes and native non-target animals, e.g. birds, possums, bandicoots, small mammals (McIlroy *et al*. 1986*b*; Allen *et al*. 1989; Fleming *et al*. 1996; Fairbridge *et al*. 2000).

Fortunately, in Australia few native species are as sensitive to 1080 as wild dogs or other placental carnivores (McIlroy 1981, 1986, 1999). However, the often rapid breakdown of 1080 (McIlroy *et al*. 1988; Fleming and Parker 1991) and the large size of wild dogs necessitates a bait loading of at least 4.5 mg and up to 10 mg per bait (this depends on State regulations; 6 mg in NSW). Such levels of 1080 can pose a theoretical risk to some small mammals (McIlroy 1986), although this is mitigated by using large baits (250 g in NSW) that are beyond the meal-size portion of most small animals. Previous work by McIlroy (1982*b*), using small baits (52 g) with ~10.4 mg 1080 in each bait and an excessively high baiting rate (29.4 baits ha^{-1}), showed evidence for a significant reductions in a brown antechinus (*Antechinus stuartii*) population immediately after baiting; however, the population recovered within 3 months because of emigration from surrounding areas (McIlroy 1982*b*). This work was spatially unreplicated and does not represent current aerial-baiting practice. McIlroy *et al*. (1986*a*) also investigated the effects of ground-based, surface baiting for wild dog control on non-target animals, including brown antechinus and *Rattus* spp., and found no change in their numbers; however, this experiment was uncontrolled, unreplicated and the bait placement was mostly in areas that were not favoured by the non-target species investigated. Nevertheless, these and other studies have demonstrated that both surface-laid fresh meat baits and manufactured baits are removed and consumed by some small mammals (McIlroy 1982*a*, 1982*b*; Glen and Dickman 2003; Körtner *et al*. 2003). It remains, however, unclear whether populations of these mammals are adversely affected by the current NSW aerial-baiting practices.

10.1071/WR08134 1035-3712/09/060528

Here we present an assessment of the impact of a control program using NSW standard aerial-baiting procedures on populations of two common small mammal species, the southern bush rat (*Rattus fuscipes assimilis*) and the brown antechinus (Menkhorst and Knight 2001). These species were chosen because they are potentially susceptible to aerial baiting with 1080 meat baits, because of their small size, and because they co-occur with wild dogs in forest areas where aerial baiting is conducted.

Materials and methods

Study area

The study was conducted in Styx River State Forest (30°35′S, 152°14′E), NSW, during autumn and winter of 2005. Topography is rugged and elevation ranges from ~850 to 1250 m. Average annual rainfall is 1514 mm and daily temperatures range from 11–32°C in summer to –3–16°C in winter (Australian Bureau of Meteorology, http://www.bom.gov.au). The study area contains mainly wet and dry sclerophyll forests. Selective logging has occurred in parts of the study area for more than 90 years, although none of the trapping grids used had been logged since 1955 (N. Fuller, NSW State Forests, pers. comm., 2005).

Trapping

Three 80 × 180-m trapping grids, each containing 50 traps spaced at 20-m intervals, were established within the baited (treated) areas, with the long side of the grid orientated parallel to the baiting transect. Three more grids were established at untreated (control) sites. All sites were at least 1 km apart, and the control sites were also at least 0.8 km away from any of the baiting transects. Box aluminium traps (Type A; Elliott Scientific Equipment, Melbourne) that were covered with plastic sleeves and provisioned with sheep wool for insulation and baited with a mixture of rolled oats, peanut butter, honey and vanilla essence, were used for trapping small mammals.

Four trapping surveys were undertaken at each site, with two surveys conducted before baiting, and two surveys after the 1080 meat baits were laid (at 1 and 5 weeks post-baiting). Trapping commenced on the 5 May (22 days before baiting), and finished on the 8 July (42 days after baiting had occurred). The end of the study preceded the annual mating and subsequent die-off of male brown antechinus by more than a month (McAllan and Dickman 1986) and, hence, this event should have had no bearing on the results of the present study. For each survey period, three of the sites were trapped simultaneously for four consecutive nights. Afterwards the traps were collected, washed and transferred to the other three sites. The order in which the six sites were trapped was dictated by the weather-dependent accessibility of the sites, and varied between survey periods.

All trapped animals were identified to species, inspected for any injuries, and their sex and bodyweight were recorded. A numbering system using an ear punch was used to identify individuals. All animals were released at their site of capture after processing and capture locations were recorded with a GPS. If an individual was captured repeatedly at a particular trap station during any one survey, this trap was closed for the remainder of that survey.

Vibrissae analysis

Rhodamine B temporarily stains fur and skin bright red and in addition, when ingested, is incorporated into growing hair and forms distinct bands under fluorescent microscopy (Fisher 1999). To monitor whether animals consumed sublethal quantities of 1080 bait, 50 mg of RhB was mixed with the 1080 solution and injected into all baits. Although the exact RhB detection threshold is unknown for both small mammal species investigated, 50 mg successfully marked spotted-tailed quolls (Körtner 2007) and is also suitable for small mammals including rats and antechinus (which are marked by a dose of at least 20 mg RhB; Fisher 1999; Fairbridge *et al*. 2003).

On capture, eight vibrissae were collected per animal, because the growth rate of individual vibrissae varies and hence not all vibrissae are marked after the ingestion of RhB. Each animal trapped at the three baited treatment sites was sampled once during each survey after 1080 baits were laid (i.e. several individuals were sampled twice). For analysis, vibrissae were cleaned in water followed by 70% ethanol, before being permanently mounted on microscopic slides. Samples were screened for RhB bands under a fluorescent microscope (Fisher 1999).

Baiting

Aerial baiting was conducted on the 27 May 2005, as described by Körtner (2007). Baits were delivered from a helicopter at a nominal rate of 40 baits per kilometre of flight path (see Thompson *et al*. 1990) along four pre-determined transects (total of 20.5 km over ~66 km^2). Prior to baiting, waypoints for the flight path along the baiting transects were entered into the helicopter's GPS. In addition, the centre line through each of the three baited trapping grids was marked with an unbroken strip of white paper (using a roll of laboratory paper towel) to assist the helicopter's pilot and navigator. The track of the flights over the baited sites were also logged on the helicopter's GPS for confirmation of the route travelled.

Following NSW regulations, the 1080 solution was prepared by licenced staff of the Armidale Rural Lands Protection Board following standard practice. In short, pre-packed satchels of 1080 powder were mixed with water. The weighed quantity of RhB powder was dissolved separately in hot water and then allowed to cool, before mixing the two solutions. The injection mixture contained 50 mg of RhB and a nominal quantity of 6 mg 1080 (standard for dog baits in NSW) per 1.2 mL that were injected into fresh meat baits (~250 g of boneless beef, dried overnight). Several baits and the original solution were later assayed for 1080 content for quality assurance (see Körtner 2007).

The efficacy of the baiting program on wild dogs was assessed with 36 track pads (1 m wide sand strips across existing tracks) spaced at 1 km. Track pads were monitored for dog tracks for four nights before and after bait laying.

Data analyses

Abundance of small animals before and after baiting and between sites was compared with a two-way ANOVA followed by a pairwise comparison (Tukey). Data that were not normally distributed (Sharpiro–Wilk normality test) were log-transformed. The distance travelled between captures

provides some estimate of 'residency' during our trial period. Consequently, the maximum distance between capture locations was calculated for *R. fuscipes* captured more than three times and four times for the more abundant *A. stuartii*. From these measures, the minimal distances required for independence between the different sites and of the baited transects were calculated by adding half the mean maximum distance to all sides of the trapping grid. For the baited sites, the maximum distances between capture locations were compared between before and after baiting with a Student's *t*-test. The sample size required to detect the theoretical prevalence of non-fatal bait uptake via RhB was estimated with a power analysis based on confidence level and sample size (Sokal and Rohlf 1981).

The number of track pads positive for dog tracks was compared before and after baiting with a chi-square test. Significance was assumed at the 5% level and averages are presented as means ±1 s.d.

Results

The logged flight pass confirmed that baits were delivered over the three treatment sites. At the approved baiting rate of 40 baits per kilometre, seven or eight 1080 baits should have fallen on each of these trapping grids. All tested baits and the 1080 solution were toxic to wild dogs and both non-target species at the time of baiting. However, the solution contained 4.2 mg of 1080 per dose rather than the nominal 6 mg possible because older powder stock was used that can contain less active ingredient than the nominal 90% (Robert Parker and Martin Hannan-Jones, Alan Fletcher Research Station, Sherwood, Qld, pers. comm.).

Population indices

Individuals (number of individuals in parentheses) of five mammal species were trapped, including *A. stuartii* (91), *A. swainsonii* (1), *R. fuscipes* (22), *R. lutreolus* (8) and *Melomys cervinipes* (1). Only *A. stuartii* and *R. fuscipes* were caught in sufficient numbers to warrant further analyses.

Antechinus stuartii was trapped at all sites and between 3 and 14 individuals were captured per site during surveys (Table 1). There was no difference in the number of animals trapped between sites ($F_{5,23} = 1.87$, $P = 0.15$), nor between surveys conducted before and after the bait delivery at both treatment and untreated control sites ($F_{1,23} = 1.78$, $P = 0.2$). At all sites, the number of new, unmarked individuals trapped declined during the course of the study, indicating that most of the trappable population had been captured and marked. This meant that at the treatment sites there was no pronounced influx of new animals in the 5 weeks following baiting (Table 1) and 74% of the individuals marked before baiting at the treatment sites were recaptured afterwards. The mean maximum distance travelled between capture locations for *A. stuartii* was 96.5 ± 46.9 m ($n = 37$; maximum 189.4 m), which was considerably less than the distance between sites and because no animal was observed to travel between sites, sites were demonstrated as being independent. For the treatment site, the maximum distance between capture locations did not differ between before and after baiting (before: 71.1 ± 29.4 m, $n = 10$; after: 98.2 ± 35.6 m, $n = 12$; $T_{19} = 1.96$, $P = 0.065$).

We encountered several trap mortalities in *A. stuartii* during our surveys (Table 1), presumably because of consistent wet and cold weather (see Lemckert *et al*. 2006). Overall, 12 individuals (8 males, 4 females, 13%) were found dead. Trap mortalities occurred at both treatment and untreated control sites during all four surveys. Mortalities were evidently not 1080 related, because there was no significant difference in mortalities before and after baiting (Fisher exact two-tailed = 1.0000); animals at control sites were also affected and dead animals were not marked with RhB. On average, the body mass of animals that died did not differ from that of surviving individuals from the same site ($T_{2,5} = 1.7$, $P = 0.15$). No significant trends were apparent regarding the number of previous captures or weight loss; however, most of the animals that died did not consume the bait in the trap. Unfortunately, adding bacon and dog biscuits to the bait did not entice bait consumption and hence was ineffective in preventing trap mortalities.

Overall fewer *R. fuscipes* were captured and at one of the untreated control sites, none was captured. Even after excluding this site, there were significant differences in the number of individuals captured between sites ($F_{4,19} = 4.78$, $P = 0.021$). However, there was no difference in population size before and after baiting at either treatment or control sites ($F_{1,19} = 0.03$, $P = 0.86$). Similar to *A. stuartii*, there was no discernable influx of unmarked bush rats at treatment sites following baiting and seven of the eight rats captured and marked initially at the treatment sites were recaptured during the surveys after baiting. Rats were found to travel a mean maximum distance between capture locations of 81.1 ± 41.3 m ($n = 17$; maximum 181 m), and no animal was trapped at more

Table 1. Trap statistics for *Antechinus stuartii*

Total, total number of individuals trapped per survey, followed by the number of unmarked, new animals in parentheses; Mort., number of trap mortalities encountered during that particular survey; Sum, total number of individuals trapped before and after baiting; T1–T3, baited treatment sites; C1–C3, unbaited control sites

Site	Before baiting					After baiting				
	Survey 1		Survey 2		Sum	Survey 3		Survey 4		Sum
	Total	Mort.	Total	Mort.		Total	Mort.	Total	Mort.	
T1	5	0	13 (8)	2	13	11 (2)	0	8 (0)	0	12
T2	4	0	8 (7)	1	11	4 (0)	0	8 (2)	0	8
T3	4	0	5 (3)	0	7	8 (2)	1	6 (0)	1	9
C1	13	1	14 (3)	1	16	11 (2)	1	5 (5)	0	16
C2	14	1	10 (3)	1	17	4 (1)	0	3 (3)	0	7
C3	6	0	4 (0)	0	6	5 (0)	2	3 (3)	0	8

than one site, confirming the independence of sites. For the baited treatment sites, the distance between trap locations did not differ significantly between before and after baiting sessions (before: 100.9 ± 72.8 m, $n = 3$; after: 71.7 ± 33.3 m, $n = 8$; $T_2 = 0.67$, $P = 0.57$). No trap-related deaths were recorded for bush rats.

Dog activity over the whole baited area (~66 km^2) decreased significantly after baiting ($P < 0.01$), although track pads and sightings confirmed that some dogs remained.

Vibrissae analysis

None of the 45 vibrissae samples from 28 individual *A. stuartii* collected at the three baited sites (including those from animals found dead in traps after baiting) exhibited RhB staining. Given the relatively large sample size, a power analysis (based on a power of 80% and an α of 0.05) indicated that non-fatal bait uptake of 4.1% or higher should have been noticed.

Following baiting, 16 vibrissae samples from 10 captured bush rats were collected. None of these showed RhB bands. However, because of the smaller sample size, this result is less reliable and a power analysis suggested that non-fatal bait consumption as high as 22% could have gone undetected.

Discussion

Although most native species are less sensitive to 1080 than are canids (McIlroy 1981, 1986), their smaller body size negates their higher 1080 tolerance to some extent. At 4.2 mg of 1080 injected into each bait during the present study (Körtner 2007), the seven or eight dog baits dropped at each of the baited sites could have killed not only canids, but any of the small mammals that ate a bait. The amount of 1080 in each bait at injection was equivalent to ~60 times the LD_{50} (the amount of toxin required to kill on average 50% of the tested animals) for *A. stuartii* (McIlroy 1981, 1986; Table 5, available as an Accessory Publication on the *Wildlife Research* website) and 35 times that for *R. fuscipes* (McIlroy 1982*b*, 1986; Table 5). Yet, our study showed that a routine aerial 1080-baiting campaign, which significantly reduced dog activity, did not affect the two species of small mammals studied. There was no measurable decrease in the population size, increased population turnover and no discernable changes in movement patterns of marked individual antechinus and rats.

Furthermore, we did not observe evidence of sublethal bait consumption from RhB analysis of vibrissae. It appears therefore that bait density/availability, and perhaps bait size (see McIlroy 1986), effectively prevented mortalities of small mammals. In comparison to wild dogs, the movements of small mammals are restricted to much smaller areas (Lindstedt *et al*. 1986; present data), exposing only a small proportion of the overall population to baits. It is likely that the current maximal bait rate provides insufficient opportunity for brown antechinus and bush rats to find enough baits to affect population size substantially. In our experiment, there was less than one bait potentially availably to each non-target animal known to be present (Tables 1, 2) and this was in the presence of the target animals, wild dogs and other large non-target species such as for example pigs, foxes, cats and spotted-tailed quolls. Alternatively, the absence of any individuals marked with RhB could suggest that, like the marsupial fat-tailed dunnart (*Sminthopsis crassicaudata*) and some other species (Morgan 1982; Sinclair and Bird 1984; Morgan *et al*. 1996; O'Connor *et al*. 2005), brown antechinus and bush rats might detect 1080 and avoid or at least decrease bait consumption to the extent of preventing poisoning and RhB marking.

Table 2. Trap statistics for *Rattus fuscipes*

Total, total number of individuals trapped per survey, followed by the number of new, unmarked animals in parentheses; Sum, total number of individuals captured before and after baiting; T1–T3, baited treatment sites; C1–C3, unbaited experimental control sites

Site	Before baiting			After baiting		
	Survey 1 Total	Survey 2 Total	Sum	Survey 3 Total	Survey 4 Total	Sum
T1	0	2 (2)	2	2 (0)	3 (1)	3
T2	3	4 (1)	4	5 (2)	5 (0)	6
T3	1	1 (1)	2	1 (0)	1 (0)	1
C1	0	0 (0)	0	0 (0)	0 (0)	0
C2	4	5 (1)	5	4 (4)	1 (1)	5
C3	2	4 (3)	5	3 (0)	0 (0)	3

In conclusion, our data and several other studies conducted during the actual baiting campaigns to control canids suggest relatively low mortality rates of the native non-target species assessed and often a negligible impact on population levels (McIlroy 1982*b*; King 1989; Claridge and Mills 2007; Körtner 2007). They also highlight that laboratory (e.g. McIlroy 1981, 1982*a*) and simulation trials (e.g. McIlroy 1982*b*; Murray and Poore 2004) are not sufficient for predicting the impact of 1080-baiting campaigns on populations (King 1989; Claridge and Mills 2007; Körtner 2007). Therefore, determination of theoretical risk should be regarded only as a first step in assessing the actual risk faced by non-target species (King 1989; Körtner *et al*. 2003; Körtner and Watson 2005; Claridge *et al*. 2006; Claridge and Mills 2007; Körtner 2007). Without monitoring the fate of individual animals and populations during the actual baiting campaigns, any risk assessments of baiting remain highly speculative and can lead to erroneous management decisions.

Acknowledgements

We are grateful for the assistance of Stuart Green and Peter Watson during the fieldwork and thankful for the active support from NSW Department of Environment and Climate Change (formerly DEC) staff, the Jeogla Wild Dog Control Association, the Armidale Rural Lands Protection Board, NSW State Forests (now part of the DPI). We also thank Harold and Shirley Heffernan for providing accommodation. Michael Johnston from Victorian Department of Sustainability and Environment and David Backhouse from the University of New England helped with the vibrissae analysis, and Nigel Andrews (University of New England) and Joanne Lello (Cardiff University) assisted with some of the statistical procedures. Peter Fleming and two anonymous referees provided valuable comments on earlier drafts of the manuscript. Baiting was approved by the NSW Department of Primary Industries and by NSW DECC following a Conservation Risk Assessment. Further licences were issued by the UNE Animal Ethics Committee (AEC05/050), DECC Wildlife Licensing (S11537), Special Purpose Permit, Forestry Commission of NSW (Styx River State Forest, No: XX23591), Australian Pesticide and Veterinary Medicine Authority (Ref. No: 8232) and Department of Environment and Heritage under the Commonwealth Environment Protection and Biodiversity Conservation Act 1999 (Ref. 2005/1962).

References

Allen, L. R., Fleming, P. J. S., Thompson, J. A., and Strong, K. (1989). Effects of presentation on the attractiveness and palatability to wild dogs and other wildlife of two unpoisoned wild-dog bait types. *Australian Wildlife Research* **16**, 593–598. doi: 10.1071/WR9890593

Claridge, A. W., and Mills, D. J. (2007). Aerial baiting for wild dogs has no observable impact on spotted-tailed quolls (*Dasyurus maculatus*) in a rainshadow woodland. *Wildlife Research* **34**, 116–124. doi: 10.1071/WR06151

Claridge, A. W., Murray, A. J., Dawson, J., Poore, R., Mifsud, G., and Saxon, M. J. (2006). The propensity of spotted-tailed quolls (*Dasyurus maculatus*) to encounter and consume non-toxic meat baits in a simulated canid-control program. *Wildlife Research* **33**, 85–91. doi: 10.1071/WR05039

Fairbridge, D., Fisher, P., Busana, F., Pontin, K., Edwards, A., Johnston, M., and Shaw, M. (2000). Observations of the behaviour of the free living bush rat *Rattus fuscipes* and the southern brown bandicoot *Isoodon obesulus* at buried bait stations. *Australian Mammalogy* **22**, 125–127.

Fairbridge, D., Anderson, R., Wilkes, T., and Pell, G. (2003). Bait uptake by free living brush-tailed phascogales *Phascogale tapoatafa* and other non-target mammals during simulated fox baiting. *Australian Mammalogy* **25**, 31–40.

Fisher, P. (1999). Review of using rhodamine B as a marker for wildlife studies. *Wildlife Society Bulletin* **27**, 318–329.

Fleming, P. J. S., and Parker, R. W. (1991). Temporal decline of 1080 within meat baits used for the control of wild dogs in New South Wales. *Wildlife Research* **18**, 729–740. doi: 10.1071/WR9910729

Fleming, P. J. S., Thompson, J. A., and Nicol, H. I. (1996). Indices for measuring the efficacy of aerial baiting for wild dog control in north-eastern New South Wales. *Wildlife Research* **23**, 665–674. doi: 10.1071/WR9960665

Fleming, P. J. S., Corbett, L., Harden, R., and Thompson, P. (2001). 'Managing the Impact of Dingoes and Other Wild Dogs.' (Australian Government Publishing: Canberra.)

Fleming, P. J. S., Allen, L. R., Lapidge, S. J., Robley, A., Saunders, G. R., and Thomson, P. C. (2006). A strategic approach to mitigating the impacts of wild canids: proposed activities of the Invasive Animals Cooperative Research Centre. *Australian Journal of Experimental Agriculture* **46**, 753–762. doi: 10.1071/EA06009

Glen, A. S., and Dickman, C. R. (2003). Monitoring bait removal in vertebrate pest control: a comparison using track identification and remote photography. *Wildlife Research* **30**, 29–33. doi: 10.1071/WR01059

King, D. R. (1989). An assessment of the hazard posed to northern quolls (*Dasyurus hallucatus*) by aerial baiting with 1080 to control dingoes. *Australian Wildlife Research* **16**, 569–574. doi: 10.1071/WR9890569

Körtner, G. (2007). 1080 aerial baiting for the control of wild dogs and its impact on spotted-tailed quoll (*Dasyurus maculatus*) populations in eastern Australia. *Wildlife Research* **34**, 48–53. doi: 10.1071/WR06076

Körtner, G., and Watson, P. (2005). The immediate impact of 1080 aerial baiting to control wild dogs on a spotted-tailed quoll population. *Wildlife Research* **32**, 1–8.

Körtner, G., Gresser, S., and Harden, B. (2003). Does fox baiting threaten the spotted-tailed quoll, *Dasyurus maculatus*? *Wildlife Research* **30**, 111–118. doi: 10.1071/WR02107

Lemckert, F., Brassil, R., Kavanagh, R., and Law, B. (2006). Trapping small mammals for research and management: how many die and why? *Australian Mammalogy* **28**, 201–207.

Lindstedt, S. L., Miller, B. J., and Buskirk, S. W. (1986). Home range, and body size in mammals. *Ecology* **67**, 413–418. doi: 10.2307/1938584

McAllan, B. M., and Dickman, C. R. (1986). The role of photoperiod in the timing of reproduction in the dasyurid marsupial *Antechinus stuartii*. *Oecologia* **68**, 259–264. doi: 10.1007/BF00384797

McIlroy, J. C. (1981). The sensitivity of Australian animals to 1080 poison. II. Marsupial and eutherian carnivores. *Australian Wildlife Research* **8**, 385–399. doi: 10.1071/WR9810385

McIlroy, J. C. (1982*a*). The sensitivity of Australian animals to 1080 poison. IV. Native and introduced rodents. *Australian Wildlife Research* **9**, 505–517. doi: 10.1071/WR9820505

McIlroy, J. C. (1982*b*). The sensitivity of Australian carnivorous mammals to 1080 poison. In 'Carnivorous Marsupials'. (Ed. M. Archer.) pp. 267–271. (Royal Zoological Society of New South Wales: Sydney.)

McIlroy, J. C. (1986). The sensitivity of Australian animals to 1080 poison. IX. Comparisons between the major groups of animals, and the potential danger non-target species face from 1080-poisoning campaigns. *Australian Wildlife Research* **13**, 39–48. doi: 10.1071/WR9860039

McIlroy, J. C. (1999). Species Impact Statement: Aerial Baiting with 1080 Poison for Wild Dog Control in New South Wales National Parks and Wildlife Service Reserves. Prepared for the NSW National Parks and Wildlife Service, Hurstville, NSW.

McIlroy, J. C., Gifford, E. J., and Cooper, R. J. (1986*a*). Effects of non-target populations of wild dog trail-baiting campaigns with 1080 poison. *Australian Wildlife Research* **13**, 447–453. doi: 10.1071/WR9860447

McIlroy, J. C., Cooper, R. J., Gifford, E. J., Green, B. F., and Newgrain, K. W. (1986*b*). The effect on wild dogs, *Canis f. familiaris*, of 1080-poisoning campaigns in Kosciusko National Park, NSW. *Australian Wildlife Research* **13**, 535–544. doi: 10.1071/WR9860535

McIlroy, J. C., Gifford, E. J., and Carpenter, S. M. (1988). The effect of rainfall and blowfly larvae on the toxicity of '1080'-treated meat baits used in poisoning campaigns against wild dogs. *Australian Wildlife Research* **15**, 473–483. doi: 10.1071/WR9880473

Menkhorst, P., and Knight, F. (2001). 'A Field Guide to the Mammals of Australia.' (Oxford University Press: Melbourne.)

Morgan, D. R. (1982). Field acceptance of non-toxic and toxic baits by populations of the brushtail possum (*Trichosurus vulpecula*, Kerr). *New Zealand Journal of Ecology* **5**, 36–43.

Morgan, D. R., Morriss, G. A., and Hickling, G. J. (1996). Induced 1080 bait-shyness in captive brushtail possums and implications for management. *Wildlife Research* **23**, 207–211. doi: 10.1071/WR9960207

Murray, A. J., and Poore, R. N. (2004). Potential impact of aerial baiting for wild dogs on a population of spotted-tailed quolls (*Dasyurus maculatus*). *Wildlife Research* **31**, 639–644. doi: 10.1071/WR03067

O'Connor, C., Morriss, G., and Murphy, E. (2005). Toxic bait avoidance by mice. In 'Proceedings of the 13th Australasian Vertebrate Pest Conference'. pp. 102–105. (Landcare Research: Lincoln, New Zealand.)

Sinclair, R. G., and Bird, P. L. (1984). The reaction of *Sminthopsis crassicaudata* to meat baits containing 1080: implications for assessing risk to non-target species. *Australian Wildlife Research* **11**, 501–507. doi: 10.1071/WR9840501

Sokal, R. R., and Rohlf, F. J. (1981). 'Biometry.' (Freeman & Co.: New York.)

Thompson, J. A., Fleming, P. J. S., and Heap, E. W. (1990). The accuracy of aerial baiting for wild dog control in New South Wales. *Australian Wildlife Research* **17**, 209–217. doi: 10.1071/WR9900209

Manuscript received 23 September 2008, accepted 14 July 2009