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An evaluation of two methods of assessing feral cat and dingo abundance in central Australia

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Abstract. We evaluated the efficacy of spotlight surveys and passive track surveys conducted along roads for assessing the relative abundance of feral cats and dingoes in a semi-arid rangeland environment in central Australia. Track surveys were more time-efficient than spotlight surveys and offered higher precision. We cover a range of issues that need to be considered when using track-based surveys to assess population change. We also discuss the merits of other techniques used to monitor the abundance of mammalian carnivores.

Introduction

Monitoring populations of small to medium-sized mammalian carnivores (body mass <50 kg) presents many challenges. The animals are often shy, cryptic, solitary and usually occur at low densities.

According to Caughley (1977), population abundance can be measured in three ways: as the number of individuals in the population, as the number of individuals per unit area (absolute density), and as the density of a population at a point in time relative to that at some other time or to that of another population (relative abundance). Estimates of population size or absolute density are difficult to obtain, and in the majority of ecological problems are unnecessary (Caughley 1977). Indices of density that are correlates of absolute density are a useful alternative and are used to assess relative abundance (Caughley 1977).

Methods used to estimate the relative abundance of mammalian carnivores include spotlight surveys (Jones and Coman 1982; Kinnear *et al.* 1988; Newsome *et al.* 1989; Stahl and Migot 1990; Ralls and Eberhardt 1997; Short *et al.* 1997), active track counts using attractants (Linhart and Knowlton 1975; Roughton and Sweeney 1982; Conner *et al.* 1983; Smith *et al.* 1994; Thompson and Fleming 1994), passive track counts (Newsome *et al.* 1972; Thompson *et al.* 1989; Allen *et al.* 1996; Mahon *et al.* 1998), scat counts (Henke and Knowlton 1995) and live capture (Jones and Smith 1979; Ross and Jalkotzy 1992). Live capture techniques can provide an estimate of actual density or population size. However, often the assumptions needed to estimate population size are not met, relegating the estimates to indices of abundance (Caughley 1977). Spotlight surveys can also provide an estimate of actual density or population size. However, spotlight surveys are often negatively biased (Jones and Coman 1982) and, unless this bias is known, spotlight data are best used as an index of abundance.

Feral cats (*Felis catus*) and dingoes (*Canis lupus dingo*) are common over much of mainland Australia. Feral cats also occur in Tasmania. Dingoes have been present in Australia for 3500–4000 years (Corbett 1995) whereas feral cats probably arrived only about 200 years ago (Dickman 1996). Across mainland Australia there is interest in monitoring populations of both species, feral cats because they are believed to have a significant detrimental impact on native fauna (Dickman 1996) and dingoes because of potential impacts on native fauna and on the pastoral industry (Corbett 1995). The aim of this study was to evaluate the suitability of spotlight surveys and passive track surveys for assessing the relative abundance of feral cats and dingoes in central Australia.

Methods

The relative abundance of free-living feral cats and dingoes was assessed using spotlight surveys and passive track surveys at a semi-arid rangeland site in central Australia, 150 km north-west of Alice Springs from July 1995 to October 1997.

Study area

The study site was about 550 km^2 in area (Fig. 1) and situated on the western part of Hamilton Downs station (23°31′S, 132°58′E) and the bordering areas of two adjacent stations – Milton Park and Narwietooma. The study site is dominated by a flat plain elevated 650 m above sea level. The southern edge of the plain abuts a range of low hills (elevation 900–1000 m above sea level) that make up the northern fringe of the MacDonnell Range. In the north-east of the study site is Mt Hay (elevation 1250 m above sea level) and to the north-west is Redbank Hill (elevation 1000 m above sea level).

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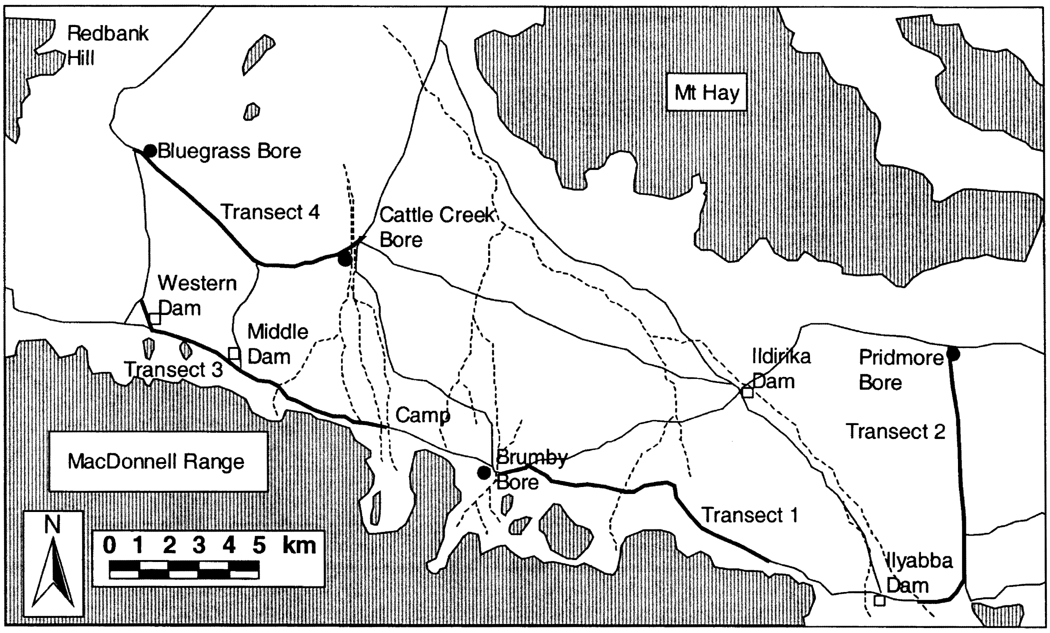

Fig. 1. Map of the study site showing water points (□, ●), station roads (—), areas of relief (shaded) and ephemeral creeks (----). The bold line shows the track survey transects.

The temperature regime at the study site is typical of central Australia. Summers are hot and in January, the hottest month, the mean maximum and minimum temperatures are 36.1°C and 21.2°C respectively (Bureau of Meteorology 1991). Daily temperatures greater than 40°C are common during summer. During the coldest month, July, the mean maximum and minimum temperatures are 19.3°C and 4.0°C respectively (Bureau of Meteorology 1991). During 1984–96 the mean annual rainfall recorded 30 km to the east of the study site at Hamilton Downs Homestead was 256 mm with a coefficient of variation of 54%.

The study site comprises three land systems: (a) Harts land system – crystalline ranges with stony shallow soils; (b) Hamilton land system – plains with texture contrast soils, some red earths and red clay soils flanking crystalline ranges; and (c) Bushy Park land system – plains with red earths flanked by the Hamilton land system (Perry *et al.* 1962). These land systems are characterised by particular vegetation associations. The rocky ranges (Harts land system) support sparse shrubs and grasses. The Hamilton land system supports a mosaic of open grassland dominated by wiregrass (*Aristida contorta*) and open mixed woodland with scattered mulga (*Acacia aneura*), ironwood (*A. estrophiolata*), witchetty bush (*A. kempeana*), whitewood (*Atalaya hemiglauca*) and bloodwood (*Corymbia opaca*). The Bushy Park landsystem supports open to dense stands of mulga woodland with scattered ironwood, witchetty bush and bloodwood over grassland. A series of ephemeral creeks supporting stands of river red gum (*Eucalyptus camaldulensis*) are scattered throughout the study site (Fig. 1). These creeks rise in the ranges to the south and drain onto the plains.

The study site has a history of cattle (*Bos taurus*) grazing and a number of permanent water points exist (bores and dams in Fig. 1). However, Milton Park paddock, which comprises much of the study site, contained very few cattle until January 1995. Two species of large kangaroo also inhabit the study site. The red kangaroo (*Macropus rufus*) typically reaches densities of 5 km^{-2} on the plains (Edwards and de Preu, unpublished) and the euro (*M. robustus*) is common in the rocky ranges. Rabbits (*Oryctolagus cuniculus*) are uncommon (Edwards and de Preu, unpublished). Dingoes and feral cats occur throughout the study site but red foxes (*Vulpes vulpes*) are rare or absent (Edwards and de Preu, unpublished).

Spotlight surveys

Spotlight surveys were conducted at night along station roads throughout the study site (Fig. 1) at approximately 3-monthly intervals from July 1995 to October 1997. Typically, a spotlight survey comprised 5–7 individual counts that were conducted over a 14-day period along different sections of road (i.e. spatially replicated transects). An attempt was made to cover the entire study site over the course of the survey period. Mean transect length per survey was 23.5–32 km. The total distance covered during a survey was 107–178 km. In some surveys one of the transects was counted twice. Observations were made by an individual standing in the back of, or sitting on the roof of, a 4-wheel drive vehicle moving at 10–15 km h^{-1}. Spotlight counts commenced 1–2 h after dark and took 3–4 h to complete, giving a time investment of approximately 42 person-hours per survey (2 persons × 6 counts × 3.5 h per count). A 100-W spotlight was used to detect animals, and data were recorded directly onto a microcassette recorder. Once a carnivore was detected, the vehicle was stopped and an attempt was made to verify the species with binoculars. The perpendicular distance of the observed animal from the road was estimated by eye or determined using a range-finder.

Track surveys

In August 1995, four permanent track-survey transects (i.e. spatial replicates) 10 km long were established along dirt roads in the eastern and western sections of the study site (Fig. 1). The transects were separated by a distance of at least 5 km. The roads that we used were chosen because they had a sandy surface substrate that was ideal for recording tracks and other signs of animals, and because they provided broad coverage of the study site. Transects were surveyed at 3-monthly intervals either before or after the spotlight surveys until October 1997. During each survey period tracks were counted along each transect every day for 2–3 days, usually on consecutive days. Twenty-four hours before the first count of a survey period, each transect was cleared by towing a heavy drag (steel bar 1.8 m long) behind a 4-wheel drive vehicle to erase all previous tracks and signs. Track counts were conducted during the following morning by two observers driving All-Terrain-Vehicles (ATVs) at 5–8 km h^{-1}, each observer counting along two transects. Track counts commenced about 0.5 h after sunrise and were completed within 2.5–3 h, giving a time investment of approximately 18 person-hours per survey (2 persons × 3 counts × 3 h per count). Each time that new carnivore tracks were detected during a count (i.e. fresh tracks on the newly prepared surface), the ATV was stopped and the tracks were closely inspected and identified to species. The following data were also recorded for each new set of tracks: the distance of the tracks from the start point of the transect, the distance that the tracks remained on the transect, the behaviour of the animal (whether it walked along the transect or simply walked across the transect), whether the prints were made by a small or large animal, whether one or more animals had made the tracks and the habitat in which the tracks were observed. The transect was prepared for the next day's count by towing a drag (1.2-m length of steel) behind the ATV.

Data analysis

Too few feral cats and dingoes (maximum $n = 7$) were observed during the individual spotlight surveys to allow a density estimate to be calculated by the distance method (Buckland *et al.* 1993). Instead, spotlight data were used to calculate an index of abundance for each species expressed as the mean number of feral cats or dingoes observed per kilometre averaged over transects. Coefficients of variation were calculated using these means.

Two methods were used to calculate population indices for feral cats and dingoes on the basis of the track surveys. Track Index 1 (TI1) used the total number of sets of tracks recorded along a transect, thereby providing an estimate of the maximum number of individuals that could have made the tracks. The total number of tracks was summed over sampling days for each transect. TI1 was the transect mean expressed as the number of tracks per kilometre. For feral cats it was relatively easy to count individual sets of tracks each day because feral cats are largely solitary (Corbett 1979) and do not follow roads (Mahon *et al.* 1998; Edwards and de Preu, unpublished). For dingoes, assessing the number of individual tracks was more difficult because dingoes often travel in packs (Thomson 1992), are known to follow roads (Mahon *et al.* 1998) and often double back and make small detours off the roadway (Mahon *et al.* 1998). Consequently, the number of individual sets of dingo tracks each day was calculated by adding together the number of sets of tracks entering and leaving the transect over a short distance (<20 m) measured along the axis of the transect and the number of sets of tracks travelling along the axis of the transect for distances greater than 20 m. Tracks travelling along the axis of the transect were classed as 'new tracks' if they were separated by a minimum distance of 1 km from previous tracks travelling along the axis of the transect. This distance was chosen arbitrarily.

Track Index 2 (TI2) was calculated in a similar manner to TI1 but used an estimate of the minimum number of 'individual' animals responsible for tracks on the transect (*sensu* Mahon *et al.* 1998). Feral cat tracks separated by 500 m were deemed to have been made by different animals; multiple sets of tracks less than 500 m apart were attributed to one individual. For feral cats, the 500-m classification criterion was based on knowledge of the size of the daily home range of feral cats in the study area (core area of approximately 20 ha: Edwards and de Preu, unpublished). A distance of 500 m was also used by Mahon *et al.* (1998) to classify the tracks of individual feral cats in a similar study. The minimum number of dingoes responsible for observed tracks on a transect was estimated as zero if no tracks were recorded, one if there were no incidences of two or more dingoes apparently travelling together (i.e. parallel tracks <1.5 m apart) or as the maximum number of tracks travelling side by side in situations where two or more dingoes appeared to be travelling together.

Our main objective was to compare the precision of the various population indices. Replicate spotlight counts and track counts were conducted over *c*. 14 days to assess sampling error. In conducting our analyses we assumed that variation between methods and replicate counts was due to sampling error alone and not actual population change.

Pearson correlation analysis was used to examine the relationships between population indices. These analyses were performed with Systat version 8.03 (SPSS Inc., USA).

Results

Feral cat population surveys

Spotlight survey data for feral cats are shown in Table 1. The mean number of feral cats seen per kilometre was generally low. Coefficients of variation were 68–224%, indicating a large degree of imprecision.

Table 1. Values of the spotlight index (animals per kilometre), track index 1 (TI1: maximum animals per kilometre summed over 3 days) and track index 2 (TI2: minimum animals per kilometre summed over 3 days) for feral cats

Coefficients of variation are shown in parentheses

Survey	Spotlight index	Track index 1	Track index 2
1	0.010	1.025	0.550
	(223.6)	(79.0)	(67.2)
2	0.023	1.575	0.800
	(112.9)	(35.3)	(22.8)
3	0.041	1.150	0.525
	(67.7)	(60.5)	(42.2)
4	0.029	0.450	0.425
	(223.6)	(46.3)	(42.2)
5	0.043	0.350	0.225
	(130.0)	(28.6)	(42.6)
6	0.034	0.400	0.275
	(94.4)	(91.3)	(80.6)
7	0.007	0.225	0.175
	(223.6)	(75.9)	(71.9)
8	0.023	2.100	0.825
	(93.2)	(60.4)	(45.8)
9[A]	0.016	1.825	0.325
	(177.0)	(117.3)	(68.2)

[A]Cat tracks summed over two days.

Track survey data for feral cats are shown in Table 1. Values of TI1 (maximum number of feral cats per kilometre) were 1–6 times higher than values of TI2 (minimum number of feral cats per kilometre) but there was a positive relationship between the two indices ($r = 0.75$, $P < 0.05$). The minimum number of feral cats per kilometre encountered during track surveys was 5–55 times higher than that for the corresponding spotlight surveys and the maximum number of feral cats per kilometre was 8–114 times higher than that for the corresponding spotlight surveys. Spotlight survey data and track survey data were uncorrelated. Coefficients of variation for both track indices were much lower than those for the corresponding spotlight data in seven of the nine surveys.

Dingo population surveys

Spotlight survey data for dingoes are shown in Table 2. The number of dingoes seen per kilometre was very low and often no dingoes were seen over 5–6 nights of spotlight counting. Coefficients of variation were 137–265%, indicating a large degree of imprecision.

Track survey data for dingoes are shown in Table 2. Values of TI1 (maximum number of dingoes per kilometre) were 1–3 times higher than values of TI2 (minimum number of dingoes per kilometre). TI1 and TI2 were highly correlated ($r = 0.90$, $P < 0.001$). The minimum number of dingoes per kilometre encountered during track surveys was 8–50 times higher than that for the corresponding spotlight surveys and the maximum number of dingoes per km was 10–125 times higher than that for the corresponding spotlight surveys. Spotlight survey data and track survey data were uncorrelated. In all cases where comparisons could be made, coefficients of variation for both track indices were much lower than those for the corresponding spotlight data.

Table 2. Values of the spotlight index (animals per kilometre), track index 1 (TI1: maximum animals per kilometre summed over 3 days) and track index 2 (TI2: minimum animals per kilometre summed over 3 days) for dingoes

Coefficients of variation are shown in parentheses. n.a. = not applicable

Survey	Spotlight index	Track index 1	Track index 2
1	0.007	0.875	0.325
	(223.6)	(81.0)	(68.2)
2	0	0.625	0.325
	(n.a.)	(61.8)	(46.2)
3	0.016	0.575	0.250
	(137.3)	(50.0)	(51.6)
4	0	0.575	0.250
	(n.a.)	(120.8)	(69.3)
5	0.030	0.750	0.375
	(183.9)	(25.5)	(33.6)
6	0	0.400	0.250
	(n.a.)	(89.0)	(51.6)
7	0.006	0.650	0.300
	(223.6)	(44.4)	(27.2)
8	0.015	0.450	0.175
	(138.9)	(75.9)	(54.7)
9[A]	0.012	0.125	0.100
	(264.6)	(151.4)	(141.4)

[A]Dingo tracks summed over two days.

Discussion

Spotlight counts appear to be generally unsuitable for monitoring populations of mammalian carnivores. Spotlight surveys consistently failed to detect the presence of dingoes that were shown by track surveys to be present at our study site (i.e. surveys 2, 4 and 6). Mahon *et al.* (1998) similarly found that spotlight counts often failed to detect feral cats and red foxes. These types of problems arise because some animals do not look at the spotlight (Jones and Coman 1982) and because spotlight counts do not sample the entire time that carnivores are active (Mahon *et al.* 1998). The latter issue can be important if there are marked temporal variations in activity patterns of the species under investigation. Changes in habitat structure through space or time can have major impacts on the detectability of animals during spotlight counts.

Another problem is that spotlight counts of mammalian carnivores often have low precision. Coefficients of variation greater than 100% were common in this study. Similarly, Stahl and Migot (1990) found coefficients of variation of 0–173% for spatially and temporally replicated counts (n = 3) of red foxes. However, Short *et al.* (1997) recorded somewhat lower coefficients of variation of 40–50% for temporally replicated counts (n = 4) of feral cats. Stahl and Migot (1990) found that higher precision tended to be associated with higher density for red foxes.

Our results indicate that, provided suitable substrates are available, passive track surveys offer several advantages over spotlight surveys for assessing the relative abundance of mammalian carnivores. Track-based surveys are simple to implement, more time-efficient and sample the entire active period of the target species, thereby providing larger sample sizes resulting in higher precision. Another important attribute of track-based methods is that they are likely to detect the presence of other species that can similarly be indexed (Allen and Engeman 1995).

Passive track-based methods of estimating mammalian carnivore abundance are increasing in popularity and a variety of approaches have been used. Newsome *et al.* (1972) estimated the number of dingoes at two sites in central Australia by counting tracks around bores over four consecutive nights. Approximate coefficients of variation of 20 and 43% were recorded. Thompson *et al.* (1989) counted tracks in snow to assess the relative occurrence of marten (*Martes americana*), lynx (*Felis lynx*) and red foxes in Canadian forest habitats. They reported high variances with the technique for red foxes and lynx and attributed this to the short transects (1 km) that they used. Allen *et al.* (1996) used tracking stations consisting of 1-m-wide strips of sand placed across roads at 1-km intervals along 50-km transects to assess dingo abundance. Their method reliably detected dingoes (Allen *et al.* 1996) and can offer high precision (Engeman *et al.* 1998). The tracking-station technique used by Allen *et al.* (1996) offers an advantage over the continuous transects used in this study as it requires less time. However, unless a very large number of tracking stations is used, the technique is not suited to species like the feral cat, which do not follow roads (Mahon *et al.* 1998).

The studies described above (and others) have highlighted several important issues that need to be considered when using track-based surveys, whether passive or active, for estimating population abundance. Perhaps the most important is that of serial correlations, which often occur in track-based survey data. Repeated counts of the same sampling unit are not independent because environmental conditions are not unrelated across days (Engeman *et al.* 1998) and because some animals may follow the same pattern of movement from day to day. Similarly, sampling units that are close together may not be spatially independent in respect of animal movements (Engeman *et al.* 1998). The effects of such serial correlations need to be considered from two perspectives. First, while it is possible to estimate variances for population measures based on serially correlated data (Engeman *et al.* 1998), non-independence invalidates the use of many parametric statistical analyses, including analysis of variance (Fortin and Gurevitch 1993; Underwood 1997), which is usually the most powerful method employed for testing whether there are significant treatment effects in ecological field experiments (Fortin and Gurevitch 1993; Underwood 1997). In this study we avoided the problem of temporal non-independence in our track-based population indices by summing our counts over sampling days. Also, because our sampling units were separated by at least 5 km, they were spatially independent in respect of feral cat movements because feral cats have long-term home ranges of approximately 20 km^2 at our study site (Edwards and de Preu, unpublished). However, our sampling units may not have been wholly independent in respect of dingo movements because dingoes have larger home ranges than feral cats (Thomson 1992). The second point to consider in respect of serial correlations is that where counts are conducted along roads, serial correlations across sampling units cause positive bias in population indices for species like red foxes and dingoes, which are known to follow roads (Mahon *et al.* 1998). The population index proposed by Allen *et al.* (1996) is particularly prone to this problem (Engeman *et al.* 1998). Such potential biases need to be taken into consideration when comparing population indices between species (Mahon *et al.* 1998).

Some track-based and other population indices estimated from frequency data are unlikely to have the desired monotonic linear relationship with abundance (see Caughley 1977). Our measure TI2 for feral cats fits into this category. As density increases, TI2 will plateau and then start to decrease as the number of feral cat tracks separated by 500 m declines due to track-saturation effects. However, although population irruptions of feral cats have been documented in some areas of Australia in response to *Rattus villosissimus* plagues (Pettigrew 1993; McRae 1993), such events are uncommon and the key point is that at the low densities typical of carnivores over much of arid Australia, TI2 should vary more or less monotonically with absolute density for feral cats. Population indices such as TI1 and the tracking station technique used by Allen *et al.* (1996), which estimate the maximum number of individuals, are likely to have a monotonic linear relationship with carnivore abundance over a much greater range of densities. It is worth noting that at high densities the precision of spotlight surveys might improve due to larger sample sizes, making them a more viable alternative.

The number of tracks of a particular species encountered during a track-based survey is likely to be a function of activity levels of individuals in the population as well as population density. Activity levels may be influenced by seasonal changes in behaviour (especially those related to breeding) (Phillips and Catling 1991; Thomson 1992; Allen and Engeman 1995; Fleming *et al.* 1996; Edwards *et al.* 1997), the availability of resources including food (Marlow 1992) and shelter, and the presence of competing species. In territorial animals such as dingoes (Corbett 1995) and red foxes (Ables 1969; Phillips and Catling 1991) activity levels may also be dependent on the actual density of individuals in the population. For such species a reduction in population density may result in an elevation of activity levels in the remaining members of the population. Under these circumstances track-based population indices may remain static or even increase in magnitude (Allen *et al.* 1996). This phenomenon could lead to spurious results in studies in which track-based methods are used to assess the efficacy of control programmes. In some instances wildlife ecologists can account for activity-related impacts on population indices through careful survey design (e.g. surveying outside of the breeding season) or by using appropriate statistical analyses (e.g. using covariates to account for differences in resource availability and/or the abundance of another species). However, this is not so easy with density-dependent activity changes.

In addition to the problems outlined above, active track methods may be influenced by bait shyness (Thompson and Fleming 1994), learned responses (Allen and Engeman 1995) and territorial responses (Allen *et al.* 1996).

Scat counts have some elements in common with track-based methods of population assessment: they sample the entire activity period of the target animal, and are likely to have a monotonic linear relationship with abundance. They also offer some advantages: they are less affected by serial correlations and are less prone to problems associated with changes in activity than track-based methods. However, scat counts may be affected by changes in scat deposition rate related to diet composition (Andelt and Andelt 1984) and by differences in scat decomposition rate associated with seasonal conditions and microenvironmental factors (Simonetti 1989). The fact that some felids bury their scats (Bradshaw 1992; Edwards and de Preu, unpublished) would limit the utility of the scat-count technique for this group.

Although they offer the opportunity to estimate population size, live-capture techniques tend to be more labour-intensive than other techniques used to estimate population abundance (Caughley 1977), are based on numerous assumptions and are unsuitable for assessment of low-density populations typical of mammalian carnivores in many areas.

The accuracy of the methods used to assess the relative abundance of mammalian carnivores in this study was not assessed. In theory, this could be done using the index–manipulation–index method (Caughley and Sinclair 1994). This could be achieved with dingoes but possible density-dependent changes in behaviour would need to be taken into account. Problems associated with the control of feral cats (Short *et al.* 1997) would make population manipulation a difficult proposition for that species.

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References

Ables, E. D. (1969). Home range studies of red foxes (*Vulpes vulpes*). *Journal of Mammalogy* **50**, 108–120.

Allen, L., and Engeman, R. M. (1995). Assessing the impact of dingo predation on wildlife populations. In ‘Proceedings of the Tenth Australian Vertebrate Pest Control Conference’. (Ed. M. Strahan.) pp. 72–79. (Tasmanian Department of Primary Industries and Fisheries: Hobart.)

Allen, L., Engeman, R., and Krupa, H. (1996). Evaluation of three relative abundance indices for assessing dingo populations. *Wildlife Research* **23**, 197–206.

Andelt, W. F., and Andelt, S. H. (1984). Diet bias in scat deposition rate surveys of coyote density. *Wildlife Society Bulletin* **12**, 74–77.

Bradshaw, W. S. (1992). ‘The Behaviour of the Domestic Cat.’ (CAB International: Wallingford.)

Buckland, S. T., Anderson, D. R., Burnham, K. P., and Laake, J. L. (1993). ‘Distance Sampling. Estimating Abundance of Biological Populations.’ (Chapman and Hall: London.)

Bureau of Meteorology (1991). ‘Climatic Averages, May 1991.’

Caughley, G. (1977). ‘Analysis of Vertebrate Populations.’ (John Wiley and Sons: Chichester.)

Caughley, G., and Sinclair, A. R. E. (1994). ‘Wildlife Ecology and Management.’ (Blackwell Science: Cambridge.)

Conner, M. C., Labisky, R. F., and Progulske, D. R. (1983). Scent-station indices as measures of population abundance for bobcats, raccoons, gray foxes, and opossums. *Wildlife Society Bulletin* **11**, 146–152.

Corbett, L. K. (1979). Feeding ecology and social organisation of wildcats (*Felis sylvestris*) and domestic cats (*Felis catus*) in Scotland. Ph.D. Thesis, University of Aberdeen, Scotland.

Corbett, L. (1995). ‘The Dingo in Australia and Asia.’ (University of New South Wales Press: Sydney.)

Dickman, C. R. (1996). ‘Overview of the Impact of Feral Cats on Australian Native Fauna.’ (Australian Nature Conservation Agency: Canberra.)

Edwards, G. P., Piddington, K. C., and Paltridge, R. M. (1997). Field evaluation of olfactory lures for feral cats (*Felis catus* L.) in central Australia. *Wildlife Research* **24**, 173–183.

Engeman, R. M., Allen, L., and Zerbe, G. O. (1998). Variance estimate for the Allen activity index. *Wildlife Research* **25**, 643–648.

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.

Fortin, M., and Gurevitch, J. (1993). Mantel tests: spatial structure in field experiments. In ‘Design and Analysis of Ecological Field Experiments’. (Eds S. M. Scheiner and J. Gurevitch.) pp. 342–359. (Chapman and Hall: New York.)

Henke, S. E., and Knowlton, F. F. (1995). Techniques for estimating coyote abundance. In ‘Proceedings of the Symposium: Coyotes in the Southwest’. pp. 71–78. (Parks and Wildlife Department: Austin, Texas.)

Jones, E., and Coman, B. J. (1982). Ecology of the feral cat, *Felis catus* (L.), in south-eastern Australia. III. Home ranges and population ecology in semiarid north-west Victoria. *Australian Wildlife Research* **9**, 409–420.

Jones, J. H., and Smith, N. S. (1979). Bobcat density and prey selection in central Arizona. *Journal of Wildlife Management* **43**, 666–672.

Kinnear, J. E., Onus, M. L., and Bromilow, R. N. (1988). Fox control and rock-wallaby population dynamics. *Australian Wildlife Research* **15**, 435–450.

Linhart, S. B., and Knowlton, F. F. (1975). Determining the relative abundance of coyotes by scent station lines. *Wildlife Society Bulletin* **3**, 119–124.

Mahon, P. S., Banks, P. B., and Dickman, C. R. (1998). Population indices for feral carnivores: a critical study in sand-dune habitat, south-western Queensland. *Wildlife Research* **25**, 11–22.

Marlow, N. J. (1992). The ecology of the introduced red fox (*Vulpes vulpes*) in the arid zone. Ph.D. Thesis, University of New South Wales.

McRae, P. (1993). Feline enterlikesit: a case for feline enteritis. *Australian Ranger* **25**, 27–28.

Newsome, A. E., Corbett, L. K., and Stephens, D. R. (1972). Assessment of an aerial baiting campaign against dingoes in central Australia. CSIRO Division of Wildlife Research Technical Paper No. 24.

Newsome, A. E., Parer, I., and Catling, P. C. (1989). Prolonged prey suppression by carnivores – predator-removal experiments. *Oecologia* **78**, 458–467.

Perry, R. A., Mabbutt, J. A., Litchfield, W. H., and Quinlan, T. (1962). Part II. Land systems of the Alice Springs area. In ‘Lands of the Alice Springs Area, Northern Territory, 1956–57’. (Ed. R. A. Perry.) pp. 20–128. (CSIRO: Melbourne.)

Pettigrew, J. D. (1993). A burst of feral cats in the Diamantina – a lesson for the management of pest species? In ‘Cat Management Workshop Proceedings – 1993’. (Eds G. Siepen and C. Owens.) pp. 25–32. (Queensland Department of Environment and Heritage: Brisbane.)

Phillips, M., and Catling, P. C. (1991). Home range and activity patterns of red foxes in Nadgee Nature Reserve. *Wildlife Research* **18**, 677–686.

Ralls, K., and Eberhardt, L. L. (1997). Assessment of abundance of San Joaquin kit foxes by spotlight surveys. *Journal of Mammalogy* **78**, 65–73.

Ross, P. I., and Jalkotzy, M. G. (1992). Characteristics of a hunted population of cougars in southwestern Alberta. *Journal of Wildlife Management* **56**, 417–431.

Roughton, R. D., and Sweeny, M. W. (1982). Refinements in scent station methodology for assessing trends in carnivore populations. *Journal of Wildlife Management* **46**, 217–229.

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

Simonetti, J. A. (1989). Microhabitat use by *Oryctolagus cuniculus* in central Chile: a reassessment. *Mammalia* **53**, 363–368.

Smith, W. P., Borden, D. L., and Endres, K. M. (1994). Scent-station visits as an index to abundance of raccoons: an experimental manipulation. *Journal of Mammalogy* **75**, 637–647.

Stahl, P., and Migot, P. (1990). Variabilité et sensibilité d'un indice d'abondance obtenu par comptages nocturnes chez le renard (*Vulpes vulpes*). *Gibier Faune Saugage* **7**, 311–323.

Thompson, I. D., Davidson, I. J., O'Donnell, S., and Brazeau, F. (1989). Use of track transects to measure the relative occurrence of some boreal mammals in uncut forest and regeneration stands. *Canadian Journal of Zoology* **67**, 1816–1823.

Thompson, J. A., and Fleming, P. J. S. (1994). Evaluation of the efficacy of 1080 poisoning on red foxes using visitation to non-toxic baits as an index of fox abundance. *Wildlife Research* **21**, 27–39.

Thomson, P. C. (1992). The behavioural ecology of dingoes in north-western Australia. IV. Social and spatial organisation, and movements. *Wildlife Research* **19**, 543–563.

Underwood, A. J. (1997). 'Experiments in Ecology.' (Cambridge University Press: Cambridge.)

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