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Lethal control reduces the relative abundance of dingoes but not cattle production impacts

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Abstract

Context. Lethal control through the application of 1080 baits is widely used in Australia to manage the negative impacts of wild dogs (dingoes, wild domestic dogs and their hybrids) on cattle production, but its effectiveness in this regard is not well understood.

Aims. To evaluate the efficacy of once yearly 1080 baiting on dingoes and its effects in mitigating predation and sublethal impacts on beef cattle.

Methods. A replicated experiment with two paired treatments (1080 poisoned and non-poisoned) was conducted on each of four cattle stations of 3782–10 850 km^2, over 2.5 years (2000–02) in the southern Northern Territory. The study was undertaken in relatively good rainfall years.

Key results. Track-based surveys indicated that dingo abundance declined on poisoned relative to non-poisoned areas immediately following a single baiting episode. However, there was no detectable difference about 8 months after baiting. No difference was detected in observed levels of calf damage or calf loss between poisoned and non-poisoned areas.

Conclusions. The results add to the growing body of consistent evidence that contemporary dingo control practices yield little benefit to rangeland beef producers most of the time.

Implications. Routine dingo baiting (as currently undertaken) may be largely unnecessary for beef cattle producers in arid and semiarid areas. Alternative strategies and practices to reduce dingo mauling and predation impacts should be investigated using replicated and controlled field studies.

Keywords: calf wastage, dingo control, Northern Territory, predation impacts, poison baiting.

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Introduction

The dingo (*Canis familiaris*) has existed in Australia for the past 4000 years (Corbett 2001). At ~15 kg in mean body mass, aside from humans, dingoes are Australia's largest contemporary terrestrial predator (Fleming *et al.* 2012*a*). Dingoes can and do interbreed with domestic dogs, and in many parts of Australia free-ranging wild dog populations include dingoes, wild domestic dogs and their hybrids (Corbett 2001; Stephens *et al.* 2015).

Dingoes were once widely distributed across mainland Australia but their distribution has declined since European settlement (Fleming *et al.* 2001; Allen and West 2013). In south-eastern Australia and also in the far south-west, 1.6-m high netting 'barrier' fencing was erected in the late 19th century to control the movement of dingoes. For much of the 20th Century, sustained coordinated control programs in combination with the barrier fences all but eliminated dingoes from the enclosed sheep production areas (Fleming *et al.* 2001). However, the effectiveness of barrier fencing in preventing the ingress of dingoes has declined in recent years. Some areas within the barrier fences that used to be relatively free of dingoes now have established populations (Allen and West 2013). Outside the barrier fences, beef production is the predominant agricultural enterprise (NLWRA 2001; Allen 2011; Fleming *et al.* 2012*b*). In these areas, dingoes have remained widespread and common (West 2008), and may even have increased due to the provision of stock watering points, the spread of rabbits and buildups in some macropod populations Corbett (2001).

Although the presence of dingoes has not hindered the development of the beef industry in the extensive rangelands outside the barrier fences (Allen 2011), many cattle graziers regard dingo predation on stock as a serious impediment to production (Eldridge and Bryan 1995; Allen and Sparkes 2001; Hewitt 2009; McGowan *et al.* 2014). Dingoes are known to prey on cattle of all ages (Fleming *et al.* 2001), but normally attack calves and weaners (Fleming and Korn 1989; Hewitt 2009). Producer estimates for annual calf losses attributable to dingo attack typically range from 1 to 7% (Eldridge and Bryan 1995; Hewitt 2009; McGowan *et al.* 2014; Binks *et al.* 2015). In addition to direct predation, reported impacts include mauling injuries (which can reduce sale value and may reduce weight gain) and transmission of production-affecting disease (Hewitt 2009; Burns *et al.* 2010; King *et al.* 2011; Binks *et al.* 2015). In order to mitigate these negative impacts, dingo populations in cattle production systems are commonly subjected to lethal control programs (Allen 2014; Fleming *et al.* 2014; Campbell *et al.* 2019). The most widely used control method is the broadscale application of 1080-poisoned fresh or manufactured meat baits (Thomson 1986; Fleming *et al.* 2001; APVMA 2008; Fleming *et al.* 2014; Allen *et al.* 2015). Other methods such as trapping and opportunistic shooting are also applied, but these are not considered to be cost effective for reducing dingo populations and associated impacts on large holdings (Fleming *et al.* 2014).

In the Northern Territory (NT), most free-ranging wild dogs are considered to be pure or near-pure dingoes (Eldridge *et al.* 2002; Newsome *et al.* 2013; Stephens *et al.* 2015), and are thus classified as indigenous wildlife and are protected (*Territory Parks and Wildlife Conservation Act* 1976). However, free-ranging dingo populations are managed under a government-approved management program to mitigate their negative impacts in beef producing areas.

In the most recent review of cattle predation in northern Australia, Fleming *et al.* (2012*b*) identified substantial knowledge gaps that impede our ability to strategically mitigate the damage caused by dingoes to cattle. These included, but were not limited to, a lack of understanding of: (1) the factors that cause dingoes to prey on cattle; (2) the degree to which predation, as opposed to other factors, influences calf survival; and, (3) the efficacy of dingo control measures in reducing target dingo populations and their effectiveness in mitigating impacts on cattle.

We conducted a study over ~2.5 years in the southern NT that investigated the effects of 1080 baiting of dingoes in a cattle production system. Our specific aims were to ascertain the efficacy of contemporary baiting practice in reducing dingo abundance, and whether baiting dingoes has an effect on cattle production. Our specific hypotheses were: (1) 1080 baiting reduces dingo abundance; (2) 1080 baiting reduces calf damage; and (3) 1080 baiting reduces calf loss.

Methods

Study locality and climate

The research described here is a part of a broader study described more fully in the comprehensive unpublished report of Eldridge *et al.* (2002). The present study commenced in May 2000 at four study sites (Fig. 1): Andado station (25°25′S, 135°17′E; area 10 850 km^2), Umbearra station (25°45′S, 133°41′E; area 4045 km^2), Lyndavale station (25°36′S, 132°53′E; area 3782 km^2) and Henbury station (24°33′S, 133°15′E; area 5273 km^2). The study ceased between May and September 2002 on Andado, Umbearra and Lyndavale stations. However, Henbury station ceased involvement in the study in August 2001. All of the stations were commercially operated cattle enterprises at the time of the study. Cattle on each station were continually mated (bulls remained with cows year round), as was typical of cattle management in central Australia at the time. Herds were mustered at least once (usually twice) per year, and calves were ear tagged and branded at each muster. Calves in most herds were left to self wean, although Lyndavale calves were separated from their mothers at 6–12 months of age. Cattle breed varied considerably among stations, with *Bos taurus* breeds (Red Angus, Shorthorn, Charolais) and *Bos indicus* × *taurus* composite breeds (Charbray, Santa Gertrudis, Droughtmaster) represented.

The climate in the southern NT is classified as semiarid and the rainfall is highly variable from year to year (Slatyer 1962). The mean annual rainfall for Kulgera, the nearest rainfall recording location to all four stations (Fig. 1), for the period 1983–2001 was 260.5 mm (s.d. 123.1; Commonwealth Bureau of Meteorology, Canberra, ACT, Australia; http://www.bom.gov.au/jsp/ncc/cdio/wData/wdata?p_nccObsCode=139&p_display_type=dataFile&p_stn_num=015603, accessed 14 April 2021). The temperature regime in the southern NT is highly seasonal, with average daytime maxima ranging from ~19°C in July to ~37°C in January (Commonwealth Bureau of Meteorology).

The years 2000 and 2001 were the wettest years recorded at Kulgera since records commenced in 1969 ($n = 28$ years of complete data). During 2000, a total of 467 mm of rain was recorded, with the bulk of this (206 mm) falling during February. Rainfall during 2001 totalled 652 mm, with falls greater than 100 mm recorded in March, June and December, and a further 94.2 mm of rain in February 2002. Thus, seasonal conditions in the study locality could be described as 'above average' in 2000 leading into 2001, tending to 'well above average' over the second half of 2001 extending into 2002.

Experimental treatments

We established paired treatment areas (experimental units) on each station to minimise differences in rainfall, habitat, cattle type and management. Thus, the experiment was a randomised block design with 'station' as the blocking factor. The treatment areas were 650–1500 km^2 in size and encompassed several fenced paddocks. The treatment areas were relatively large compared with the size of most recorded home ranges (mean 25–77 km^2: Corbett 2001, ~24 km^2: Allen 2012) of dingoes in pastoral landscapes. However, dingoes in non-pastoral settings living away from habitation may have considerably larger home ranges (Newsome *et al.* 2013; mean 757 km^2). Paired treatment areas were separated by a distance of at least 40 km, which helped to ensure their independence in respect to dingo movements during each survey (see below). Each treatment area contained a herd of 250–1200 breeder cows, a small number of bulls and a variable number of calves.

We applied fresh meat baits containing 1080 poison in one of the paired treatment areas on each station once per year in August 2000 and July 2001 (Table 1), which is the onset of the dingo breeding season in central Australia (Corbett 2001). Baits

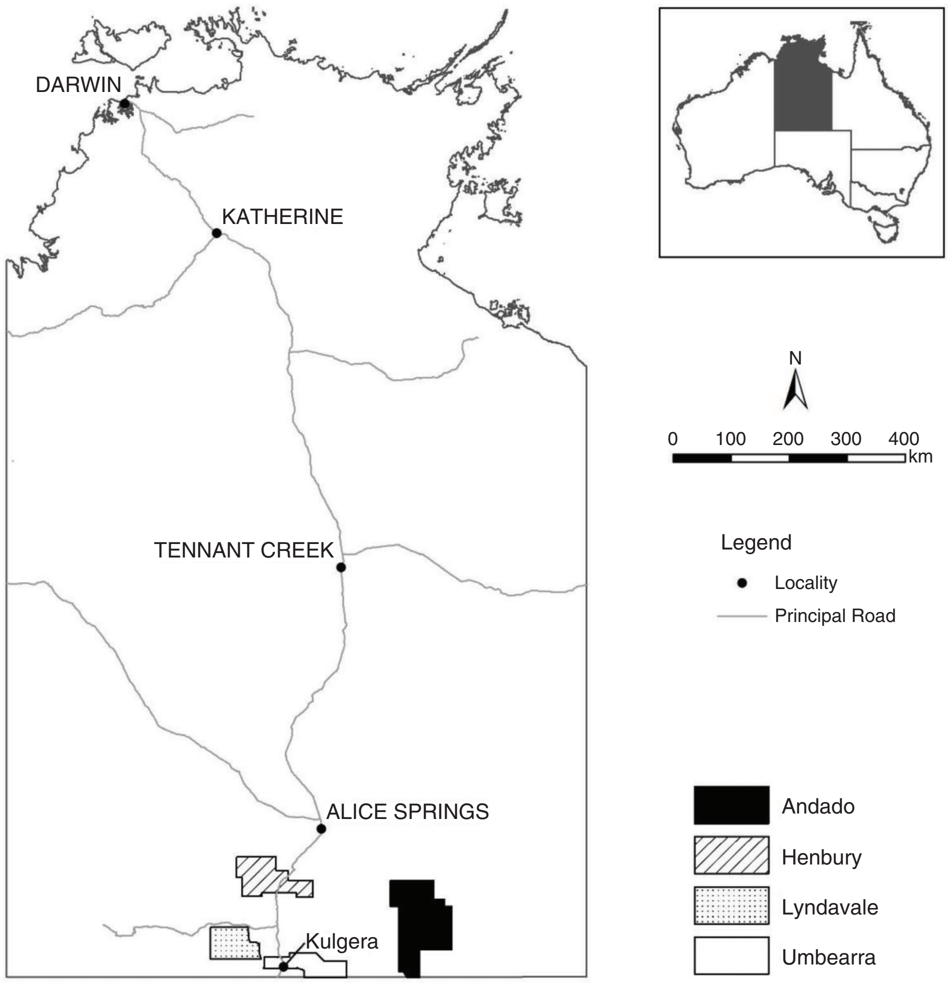

Fig. 1. Map of the Northern Territory showing the location of the four study sites (Andado, Henbury, Lyndavale and Umbearra stations). The location of Kulgera where rainfall data were collected is also shown.

Table 1. Schedule showing when baiting (purple) and track surveys for dingoes (blue) were undertaken on each station

Calendar years in top row, calendar months in second row. Bottom two rows show how years were defined (orange Year 1, green Year 2) and how surveys were categorised for ANOVA analysis

	2000								2001												2002			
	M	J	J	A	S	O	N	D	J	F	M	A	M	J	J	A	S	O	N	D	J	F	M	A
Andado																								
Umbearra																								
Lyndavale																								
Henbury																								
	Pre-baiting 1					<3 months post-baiting 1					>8 months post-baiting 1													
											Pre-baiting 2				<3 months post-baiting 2							>8 months post-baiting 2		

were applied in a fashion typical of that used throughout the NT at the time of the study. We applied non-poisoned fresh meat baits also in July–August in the other paired experimental areas. Thus, the treatments were 'Poisoned' and 'Non-poisoned', with the latter treatment acting as an experimental control. We assigned treatments at random to each paired treatment area on each station. Baits were cubes of beef or camel meat weighing 400–500 g and cured for 12–24 h to form a dry skin. The poisoned baits were then injected with 1.5 mL of 40 mgL^{-1} 1080 solution, delivering a nominal 6 mg 1080 per bait, which is the estimated lethal dose for a large (20 kg) dingo (McIlroy 1981; Twigg *et al.* 2000; Fleming *et al.* 2001). Baits were transported by vehicle and placed on the ground by hand, usually under low vegetation to minimise uptake by non-target avian species. Within the treatment areas, 20–25 baits were placed at artificial (i.e. man-made) water points because these are regularly visited by dingoes (Allen 2012), and single baits were placed at 500-m intervals along unformed station roads and fence lines with obvious dingo tracks. Using this approach, ~175–350 baits were laid on each treatment area – the actual number varying in accordance with the number of water points and extent of the road network within each area. The density of applied baits (calculated on the basis of the size of the treatment areas delineated by their perimeter fences) ranged from 0.12 to 0.44 baits km^{-2}. This density range straddles the mean annual bait density applied across the NT over the period 1999–2008: 0.33 baits km^{-2} (W. Dobbie, unpubl. data, 2020) and was similar to that used in cattle rangelands in northern South Australia, northern Western Australia and parts of western Queensland at the time the present study was conducted (Allen *et al.* 2015). Untaken baits were not recovered. Three of the four stations (Lyndavale, Umbearra and Henbury) did not lay poisoned baits for dingoes in the year preceding the study (1999). Generally speaking, cattle stations in the region (south of Alice Springs) baited on average every second year over the period 1999–2008 (W. Dobbie, unpubl. data, 2020).

Track surveys

Passive track counts conducted along transects are widely used to assess the abundance of small to medium-sized carnivores (and other animals) in Australia (Allen *et al.* 1996; Edwards *et al.* 2000; Twigg *et al.* 2000; Paltridge and Southgate 2001; Kennedy *et al.* 2012; Eldridge *et al.* 2016). However, the measure they provide is actually a composite measure of activity and abundance (Kennedy *et al.* 2012; Eldridge *et al.* 2016).

We established three permanent 10-km track survey transects in each treatment area to monitor the abundance of dingoes. All transects were established at least 5 km apart along existing unformed station roads with a surface substrate sandy enough to record track impressions and other signs of animals. Although spatially separated, the transects were unlikely to be independent with respect to dingo movements. We surveyed transects in accordance with the schedule shown in Table 1. In each treatment area within a station, a trained observer driving an All-Terrain Vehicle (ATV) at 10 km h^{-1} assessed tracks made along each transect for three consecutive days during each survey in most instances, commencing ~0.5 h after sunrise. We cleared old tracks along each transect the day before the first count by driving along the transect in a 4-wheel drive vehicle towing a heavy drag (a steel bar 1.5 m in length with heavy gauge chain attached). It took ~75 min to complete a survey along one 10-km transect. Observers recorded the presence and behaviour of dingoes and other species during the surveys. Observers stopped the ATV each time new carnivore tracks – dingo, fox (*Vulpes vulpes*) or cat (*Felis catus*) – were detected during a count, and closely inspected the tracks to identify them to species (see Triggs 2004). The transects were prepared for the next day's count by towing a lightweight drag (1.2-m steel fence picket with chain attached) behind the ATV during days 1 and 2 of each survey.

We used methods similar to those described in Edwards *et al.* (2000) to derive an index of abundance (*Ab*) for dingoes in each treatment area for each survey period. This involved pooling and averaging data across transects and survey days to account for pseudoreplication (Hurlbert 1984; Crawley 2005). *Ab* was derived as follows:

1. For each individual transect, estimate the number of individual dingoes (DT) responsible for tracks observed on the transect each day. For this purpose, two or more tracks entering and leaving the transect over a short distance (<20 m) were classed as one set of tracks made by the same individual. Two (or more) tracks travelling along the axis of the transect for a distance of more than ~20 m were deemed to have been made by two (or more) individuals (i.e. two or more sets of tracks). Tracks separated by a minimum distance of 5 km measured along the axis of the transect were attributed to a new dingo.
2. Sum the number of individual dingo tracks (DT) across transects each day.
3. Divide this figure by 30 to give the number of dingoes km^{-1}.
4. Sum the number of dingoes km^{-1} across days.
5. Divide this figure by the number of days that surveys were conducted to give dingoes km^{-1} day^{-2}.
6. In summary, *Ab* was calculated according to the following formula:

$$Ab = \left(\sum_{1}^{m}\left(\left(\sum_{1}^{n}DT\right)/30\right)\right)/m$$

where n is the number of transects (three in this instance), DT is an estimate of the total number of dingoes responsible for the observed tracks on a transect (see above) and m is the number of days counts were undertaken (typically three, but two on one occasion). Although there are some untested assumptions inherent in this type of approach (Hayward and Marlow 2014) that may cause some problems in interpretation, we did not consider this to be of major consequence in the present study (see Discussion).

Livestock damage

Livestock losses were evaluated for the treatment areas on three of the four stations (Lyndavale, Umbearra and Andado) during 2001 and 2002 using two methods. The first involved observations of sublethal damage to unbranded calves yarded during mustering operations. Calves were assessed for signs of dog attack (such as scarring on the hind quarters and scarring or missing tissue on the ears and tail) while constrained in a calf cradle before branding. The number of unbranded damaged calves was used to calculate an index of calf damage expressed as a percentage of the total number of unbranded calves yarded in each mustering operation in each treatment area. The second method involved observations of the lactation status of cows

Table 2. Schedule showing when mustering was undertaken on each station (orange) and whether calf damage only (D) or calf damage and lactation status of cows (DC) was assessed

Calendar years in top row, calendar months in second row. Henbury station withdrew from the study in August 2001

	2000								2001												2002								
	M	J	J	A	S	O	N	D	J	F	M	A	M	J	J	A	S	O	N	D	J	F	M	A	M	J	J	A	S
Andado																D							DC	DC	DC				
Umbearra												D					DC	DC				D							D
Lyndavale													D			D		D						DC					

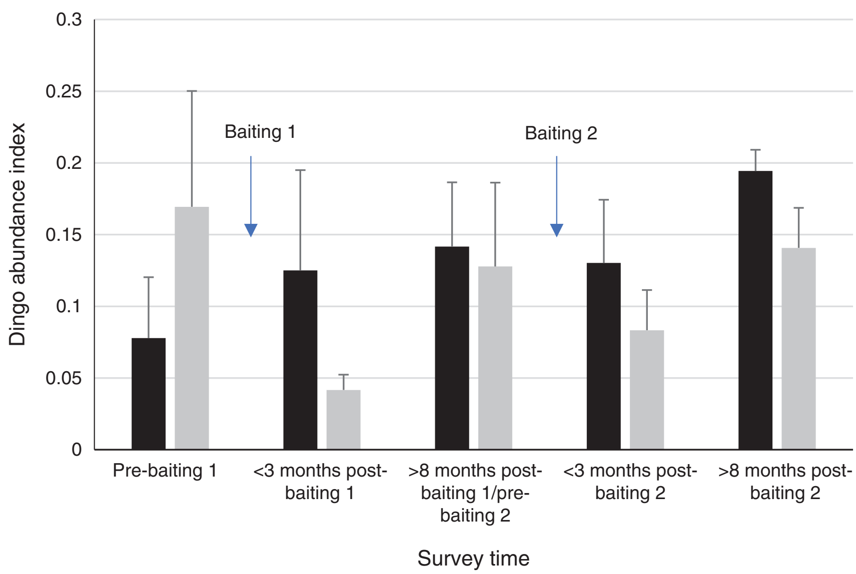

Fig. 2. Bar graph showing changes in dingo abundance index across time. Data are station means for each treatment. Dark bars are non-poisoned, light bars are poisoned. Baiting events are indicated by arrows. Error bars are standard deviations.

yarded during some mustering operations. Udder condition was scored as either 'wet' (lactating) or 'dry' (not lactating). We used these data to determine the percentage of dry cows detected in individual mustering operations in each treatment area and used this as a measure of calf loss. Both measures were compared between treatments to determine the effect of poisoning.

The timing of mustering operations was determined by the manager on each station and varied among years, stations and treatment areas (Table 2). All calves were yarded during each mustering event and any unbranded calves were branded. Thus, the dataset for calf damage contained only one measurement for each individual because we assessed only unbranded calves for damage. Although mustering activities were sometimes spread over several months (Table 2), individual cows were only yarded and inspected once on each of the three stations. Thus, the cow wet–dry dataset also contained only one measurement for each individual.

Statistical analyses

We used analysis of variance to model differences in dingo abundance (*Ab*) among the treatments, surveys and 'years'. We looked for temporal autocorrelation in values of *Ab* over time by examining bivariate plots for the Non-poisoned areas. In the analysis of variance, 'Station' was a fixed blocking factor (categorical) with four levels. 'Year' was a fixed factor with two levels, 'Year 1' and 'Year 2'. Note that the Years were 12-month blocks but were not calendar years (Table 1; Fig. 2). 'Treatment' was a fixed factor (categorical) with two levels, Poisoned and Non-poisoned. 'Survey' was a fixed factor (categorical) with three levels ('Pre-baiting', '<3 months post-baiting' and '>8 months post-baiting'). The >8 months post-baiting data for baiting event 1 were used as the Pre-baiting data for baiting event 2 (Table 1; Fig. 2). We followed model simplification procedures outlined in Crawley (2005) to determine the minimal adequate model.

Because of the large differences in the timing of mustering operations among years, stations and treatment areas, we pooled calf damage and cow lactation data across years and stations for each baiting treatment. We arcsine transformed percentages before analysis and used either the Binomial test or the Mann–Whitney test to test for differences between the treatments.

All statistical analyses were performed using either SYSTAT 8.1 (SPSS Inc.) or the R core software package (R Development

Core Team 2008), including the R package ‘car’ (Fox and Weisberg 2019).

Results

The efficacy of 1080 baiting in reducing dingo abundance

A histogram of the *Ab* values, with a normal curve superimposed, indicated that the data were slightly platykurtic but not strongly skewed. Neither square root nor logarithmic transformation of *Ab* improved the fit. We found no strong indication of consistent temporal autocorrelation in *Ab* values for the non-poisoned areas. The minimal adequate ANOVA model contained just four parameters: Station ($F_{3,37} = 2.96$, $P = 0.01$), Treatment ($F_{1,37} = 1.81$, $P = 1$), Survey ($F_{1,37} = 5.69$, $P < 0.01$) and the Survey-by-Treatment interaction ($F_{2,37} = 5.12$, $P = 0.01$). *Post hoc* testing showed that the difference between the treatments was confined to the <3 months post-baiting level of the survey main effect: in the poisoned treatment, dingo abundance <3 months post-baiting was significantly lower than that at Pre-baiting and >8 months post-baiting (Fig. 2). This effect was not apparent at the non-poisoned areas (Fig. 2). When considered relative to the non-poisoned treatment areas and taking into account initial differences in dingo abundance (following Twigg *et al.* 2000), the apparent decline in dingo abundance was on average 76% in poisoned areas after baiting event 1, and 33% lower after baiting event 2.

The effects of 1080 baiting on cattle damage

We inspected a total of 3593 calves for signs of sublethal dingo attack during 38 mustering events. The mean percentage of calves showing signs of attack was 0.23% in non-poisoned areas and 0.33% in poisoned areas (Fig. 3). However, the data were strongly skewed as we did not detect any sign of attack in 30/38 musters (i.e. the data contained many zero values). This effect was still evident following arcsine transformation so we further pooled data within treatments and used the Binomial test to compare the mean proportion of damaged calves for each treatment. There was no effect of 1080 baiting on calf damage ($\chi^2 = 0.39$, d.f. $= 1$, $P = 0.53$). We inspected a total of 2605 cows for lactation status. The mean percentage of dry cows was 26.3% in non-poisoned areas and 23.3% in poisoned areas (Fig. 4). The data showed a reasonable fit to the normal distribution after arcsine transformation but the variances were clearly unequal. The Mann–Whitney *U*-test indicated there was no effect of 1080 baiting on the percentage of dry cows (Mann–Whitney $U = 40$, $\chi^2 = 0$, d.f. $= 1$, $P = 1.0$).

Discussion

Before interpretation of the results can commence, some clarification of the limitations of the methodology is required. Hayward and Marlow (2014) highlighted several problems with track-based methods for assessing the ‘abundance’ of animals, with particular reference to predator guild studies and the detection of mesopredator release. The first is an almost universal lack of an established relationship between track-based indices of ‘abundance’ and actual abundance. The track-based index used in this study is certainly not validated in this sense. Furthermore, it is unlikely to be linear across all density values and will almost certainly show signs of saturation at very high densities. However, as long as the relationship is not non-linear recurvate (Caughley 1980) – and there is no reason that it should be – an index like the one used here is still useful for answering general questions of the nature posed in the present study (i.e. ‘does baiting reduce the abundance of dingoes?’) as opposed to a more specific question such as ‘what proportional reduction do we get in dingo abundance if we double the bait density?’.

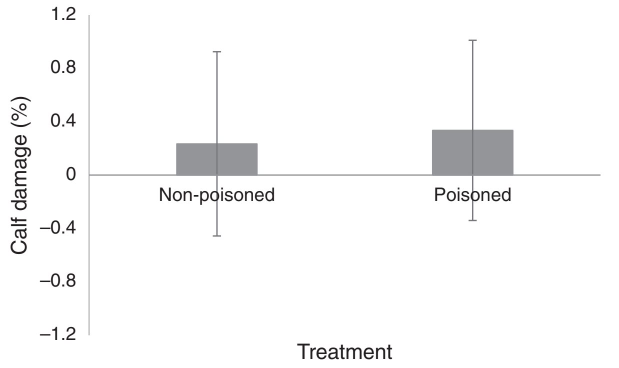

Fig. 3. Bar graph showing the mean percentage of damaged calves in each treatment. Error bars are standard deviations.

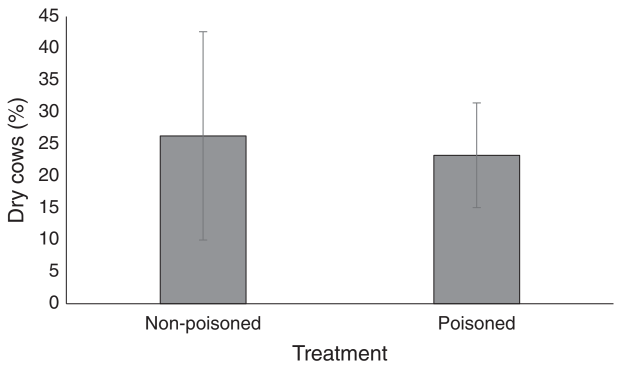

Fig. 4. Bar graph showing the mean percentage of dry cows in each treatment. Error bars are standard deviations.

The second problem raised by Hayward and Marlow (2014) is that changes in detectability are not accounted for in most studies that have used track-based indices. The detectability of some predator species may vary when there are changes in the abundance of other predator species or in different areas or habitats (Hayward and Marlow 2014). It can also vary in response to seasonal shifts in behaviour (e.g. breeding behaviour) or shifts in the availability of resources. Changes in detectability present a problem because they mean that measures taken at different times or under different circumstances may not be directly comparable. Fortunately, thoughtful experimental design and appropriate use of statistical models can account for some shifts in detectability. In the present study, the use of a blocked design and an appropriate ANOVA model largely accounted for any potential differences in the detectability of dingoes among stations due to resource or habitat differences, and any changes in resource abundance over time. However, dingoes are seasonal breeders and shifts in behaviour related to breeding may have affected detectability at different times of the year. Problems due to this could potentially have been compounded in the present study by slight differences in the timing of track surveys among years (Table 1). With the issues in this and the preceding paragraph in mind, we make the following cautious interpretation regarding the efficacy of dingo baiting in this study.

The data support our first hypothesis: 1080 baiting caused a decline in dingo abundance but it was short term. This decline was evident in track surveys conducted 1–3 months following baiting (Fig. 2). It is likely that the reduction in dingo abundance occurred within days to weeks following baiting (Twigg *et al.* 2000), but we were unable to confirm this because of the 1–3 month time lag associated with the conduct of surveys post baiting. Dingo abundance on the poisoned treatment areas could not be distinguished from that on non-poisoned areas at the >8 months post-baiting surveys (Fig. 2).

Several other studies from across cattle rangelands in Australia have investigated the efficacy of dingo baiting using an independent experimental control (non-poisoned treatment). All used track-based indices of abundance and so the same caveats apply. At three cattle stations in central Australia, Twigg *et al.* (2000) found an immediate reduction in dingo abundance at two of three areas poisoned once with 1080 compared with non-poisoned areas. There was no follow-up assessment of dingo abundance to gauge the duration of this effect. The study accounted for initial differences between the paired treatment areas that were separated by a buffer zone. At two cattle stations in Queensland, Allen (2015*a*) reported an immediate reduction in dingo abundance in areas poisoned with 1080 once or twice annually over periods of 4–5 years compared with non-poisoned areas separated by a buffer zone. However, dingo abundance was similar in poisoned and non-poisoned areas after 8 months in the absence of follow-up baiting in most years (Allen 2015*a*). On three additional cattle stations in Queensland, Allen *et al.* (2013) also showed a decline in dingo abundance in areas poisoned continuously every 2–4 months compared with non-poisoned areas over periods of 3–4 years. The study accounted for initial differences between the paired treated and untreated areas. However, as there was no buffer between the paired areas on a station, there were likely to have been baiting effects on the unbaited areas. Campbell *et al.* (2019) showed that dingo abundance was generally lower on areas poisoned with 1080 twice annually compared with non-poisoned areas separated by a buffer zone on four cattle stations in northern South Australia over a 2–4-year period. However, there was high variability and the treatment effect was not consistent over time. It is unclear whether the studies of Allen (2015*a*) or Campbell *et al.* (2019) accounted for any initial differences in dingo abundance between paired poisoned and non-poisoned areas in assessing the efficacy of baiting. The overall picture that emerges from this body of work is that 1080 baiting in rangeland settings generally reduces dingo abundance in the short term. However, the effects of baiting usually cannot be detected after 8–11 months in the absence of further management intervention.

The objective of 1080 baiting programs for dingoes should not be to reduce dingo density *per se* but to reduce their impacts on livestock. Our data did not support our second hypothesis: we did not detect an effect of 1080 baiting on calf damage (Fig. 3). Nor did we find an effect of baiting on the percentage of dry cows (Fig. 4). While the latter finding implied a lack of support for our third hypothesis (1080 baiting reduces calf loss), we would qualify this result. Our measure of calf loss was relatively crude because it could not account for cows that were not mated, or for fertilisation failure, both of which were assumed to be the same on poisoned and non-poisoned areas. Although calf loss estimates based on confirmed pregnancy (Burns *et al.* 2010) offer an advantage in this respect, this approach could not be used in the present study due to opposition from participating station managers. Furthermore, our sampling regime for assessing cow wet–dry status was not consistent across stations, and may have been a suboptimal approach in continuously mated cattle production systems where calving time is not synchronised.

These problems aside, our failure to detect an impact of poison baiting on calf loss accords with the results obtained in similar studies that have used confirmed pregnancy. Campbell *et al.* (2019) found no measurable effect of twice-yearly 1080 baiting on calf loss across both ‘average’ and ‘above average’ seasons at their four study stations in northern South Australia. Furthermore, in an extensive review of regional calf production and 1080 baiting records in northern South Australia, Allen (2015*b*) suggested that repeated, broad scale, long-term poison baiting programs had failed to yield increased calf production over nearly 40 years. Allen (2014) also found no difference in calf loss between poisoned and non-poisoned areas at his two central Queensland stations during ‘above average’ seasonal conditions. However, in ‘below average’ conditions, calf loss was typically higher on poisoned areas. This is a counterintuitive result that does not align with the prediction of Corbett (2001) that there would be an increase in dingo predation of calves during drought.

Despite the issues raised above regarding our methodology, our overall estimate of calf loss (24.6%, averaged across treatments and stations; Fig. 4) is not unduly high for rangeland production systems. It is within the 20–25% range reported by Burns *et al.* (2010) in an extensive review of ~25 years of records of foetal and calf mortalities, from confirmed pregnancy to weaning, in cattle in northern Australia. It is also only marginally higher than the values recorded by Allen (2014), 10–17.1%, and Campbell *et al.* (2019), 18.4%. To keep these types of values in perspective, it must be remembered that calf loss figures represent losses due to all causes, not just predation. In reality, many factors contribute to levels of prenatal and perinatal mortality in cattle, and predation may play only a minor role under most circumstances (Hewitt 2009; Burns *et al.* 2010; Allen 2014). Our overall estimate of calf damage (0.26%) is lower than that reported by Allen and Fleming (2004), 0.4% averaged over 8 years, for a property in Queensland subject to broadscale aerial baiting with 1080, and later annual estimates provided by Queensland producers in a statewide questionnaire survey (1.9%; Hewitt 2009).

The present study showed that a single 1080 baiting episode suppresses dingo abundance in the short term but does not reduce calf damage or losses. These results add to the growing body of consistent evidence that dingo control practices, as currently conducted, yield little benefit to beef producers in arid or semiarid environments most of the time (Wicks and Allen

2012; Allen 2014; Allen 2015*a*; Campbell *et al.* 2019). These findings have important implications for managers and we conclude, as have these others, that routine dingo baiting with 1080 may be largely unnecessary for beef cattle producers in these areas. We suggest that alternative strategies and practices to reduce dingo mauling and predation impacts should be investigated. These may not necessarily involve lethal control of dingoes. For example, benefits may be had through manipulation of cattle husbandry practices to avoid calves being born in the dingo mating season (*sensu* Corbett 2001). However, alternative approaches must be evaluated using replicated and controlled field studies. If lethal control of dingoes is to remain as part of the management mix, greater effort is required to demonstrate some actual benefit to cattle in order to satisfy the ethical and welfare concerns associated with the practice (Allen and Hampton 2020).

Conflicts of interest

The authors declare no conflicts of interest.

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