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Responses of dingo (*Canis familiaris*) populations to landscape-scale baiting

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ABSTRACT

Carnivores are important drivers of ecological processes around the world. However, medium-large carnivores are often the focus of human-wildlife conflicts and are subject to control efforts. Determining the effectiveness of predator control efforts in reducing predator abundance or impact is critical to ensuring control is achieving its intended aims. This information is also vital to understanding any impacts of control efforts on the ecological functions of carnivores. In this study, we deployed camera traps on two properties in the southern rangelands of Western Australia to examine the effectiveness of repeated rounds of landscape-scale toxicant baiting in reducing dingo populations. Biannual baiting at each property was temporally offset from the other to provide a comparison of short-term changes in dingo activity and density over 16 months' monitoring. While there were significant differences in dingo density between properties, there was no significant differences between months categorised as 'month of baiting', 'month immediately post-baiting' or 'between baiting'. Further, there was no overall decline in dingo density on either property over the duration of the study and survival of dingoes exceeded 84% on each site. Neither individual nor sequential rounds of baiting therefore resulted in a reduction in dingo density approaching 75%, which is necessary for dingo population control in this environment. Several factors are likely to have contributed to the limited effect of baiting, including bait uptake by non-target species, low encounter rate with baits and aversive responses to baits. Consideration of baiting practices, including bait rate, frequency and attractiveness, as well as evaluation of the net returns of baiting should be addressed in future work to ensure dingo control achieves its intended aims to reduce dingo abundance and/or impacts.

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1. Introduction

Carnivores are important drivers of ecological processes around the world (Krebs et al., 2001; Terborgh and Estes, 2013). However, medium-large carnivores are also often the focus of human-wildlife conflicts where they impact humans or agriculture (Treves and Karanth, 2003). Disputes can arise where predator control for protection of humans and their agricultural assets lacks sound evidence (Van Eeden et al., 2018) or is at odds with attempts to preserve or reinstate the ecological function of carnivores (Treves and Karanth, 2003). For example, grey wolves (*Canis lupus*) can affect livestock across their range (Treves and Karanth, 2003), coyotes (*Canis latrans*) are significant predators of sheep in North America (Knowlton et al., 1999; Saitone and Bruno, 2020), black-backed jackals (*Canis mesomelas*) and caracals

(*Caracal caracal*) are the primary predators of livestock in South Africa (Bergman et al., 2013; Kerley et al., 2018). These species are the focus of control programs to mitigate their impacts on livestock, however the effectiveness and justification of these programs are contested (Berger, 2006; Eklund et al., 2017; Natrass et al., 2020). Understanding the outcomes of control programs is central to determining their effectiveness in mitigating negative effects of carnivores, and their impacts on ecological functions of carnivores (Fleming et al., 2014).

Dingoes (*Canis familiaris*, Jackson et al., 2017, 2019, which include dingoes, free-living domestic dogs and dingo-domestic dog hybrids) are the largest terrestrial predator in Australia. Dingoes are capable of regulating native herbivores (Allen, 2015; Choquenot and Forsyth, 2013; Pople et al., 2000) and possibly introduced predators (Letnic et al., 2012; Ritchie and Johnson, 2009, although see Allen et al., 2013; Fancourt et al., 2019; Hayward and Marlow, 2014; Kreplins et al., 2020; Stobo-Wilson et al., 2020). Dingoes can also have significant impacts on livestock (Fleming et al., 2014) and threatened native species (Allen and Fleming, 2012). For example, sheep cannot persist in the presence of dingoes (Fleming et al., 2001; Thomson, 1984), and as such, dingoes have influenced the distribution of sheep production

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across Australian rangelands (Allen and West, 2013, although see also Forsyth et al., 2014 and Allen and West, 2015).

To mitigate dingo impacts on livestock, coordinated landscape-scale control using a suite of tools is advocated as best practice management (Fleming et al., 2001; Allen and Harris, 2020). The tools available to producers include shooting, trapping, guardian animals, fencing and toxicant baiting (hereafter 'baiting'). Baiting is widely used in the control of vertebrate pests for conservation and agricultural production (e.g. Eason et al., 1993; Gentle et al., 2007; Olivera et al., 2010; Short et al., 1997; Snow et al., 2017). The use of sodium fluoroacetate (1080) baits for dingo control is relatively low cost and can be deployed at scale (Allen, 2017); consequently, 1080 baiting is the most commonly used technique for dingo control across Australia (Binks et al., 2015).

The impact of 1080 baiting on dingo populations is variable, ranging between 22 and 100% (Ballard et al., 2020; Fleming, 1996; McIlroy et al., 1986a; Thomson, 1986). To sustain a reduction in dingo density, 75% or greater of the population needs to be removed annually (Ballard et al., 2020; Hone et al., 2010; Pacioni et al., 2020a). The effectiveness of baiting can be influenced by environmental factors such as season, prey availability, activity of target and non-target species, and the age and social status of target species (Kreplins et al., 2018; McIlroy et al., 1986b; Thomson, 1986). Additionally, variables in baiting programs such as bait type (Eldridge et al., 2000; Thomson, 1986), rate of bait lay (Ballard et al., 2020) and bait placement (Kennedy et al., 2014) can influence the availability, attractiveness, palatability and lethality of baits (Allsop et al., 2017; Fancourt et al., 2020; Saunders and McLeod, 2007).

Exclusion of sheep by dingoes is particularly problematic when there are no economically viable enterprise alternatives for producers. The southern rangelands of Western Australia were traditionally a wool sheep production area (Government of Western Australia, 2009; Foran et al., 2019). Since the 1990s, livestock production in this area has moved away from sheep to cattle production or non-grazing enterprises as a result of economic challenges associated with wool production, including difficulties in controlling dingoes (Government of Western Australia, 2009; Foran et al., 2019). Due to a lack of suitable feed for cattle and greater profitability of sheep production, producers in the southern rangelands seek to regain the ability to run sheep enterprises, in part through dingo control. Expenditure on dingo control accounts for over 75% of the annual pest plant and animal control budget for biosecurity groups in the area, much of which is spent on landscape-scale baiting (DPIRD unpublished data). Landscape-scale baiting of dingoes is likely to have implications, not only for livestock producers, but the native and introduced prey of dingoes. Here we seek to determine the responses of dingoes to landscape-scale baiting in the southern rangelands.

There are limited methods to accurately estimate abundance or density of medium-sized carnivores such as dingoes (Hayward and Marlow, 2014). Dingoes, are cryptic, occupy large home ranges, can be difficult to individually identify, and usually occur at low densities. Reliable detection of individuals at a scale suitable for mark-recapture population estimates is therefore challenging. Further, accurate assessment of dingo populations and the effects of control actions can be confounded by variation in dingo activity due to age, sex (Allen et al., 2014b; Thomson, 1992a) and season (Choquenot and Forsyth, 2013). To address these challenges, we deployed camera traps at a large-scale on two properties, allowing monitoring over sequential rounds of baiting (16 months total). We compared short-term changes in dingo responses to landscape-scale baiting using an index of dingo activity, density (calculated using spatially explicit methods), and survival estimates.

2. Methods

2.1. Site description

This study was conducted on two pastoral properties (Property 'A' and Property 'B') in the southern rangelands, Murchison region,

Western Australia. The southern rangelands are typified by an arid environment with annual rainfall of 239 mm and mean maximum temperatures in January of 38.2 °C (Mount Magnet Station, 007057; Bureau of Meteorology, 2017). The vegetation is composed of *Acacia* spp. woodlands. The two properties of 75,713 ha and 23,096 ha respectively were separated by approximately 70 km. Both have a history of sheep farming, but only unmanaged cattle and goats grazed the properties during the study. Dingo control has occurred at varying intensities on the properties for a long period (~40 y).

2.2. Baiting events and camera trap monitoring

Within the southern rangelands biannual coordinated baiting for dingo control is generally undertaken on livestock production properties by regional biosecurity groups. A round of baiting consists of deployment of dried meat baits containing 6 mg of 1080 prepared as per Thomson and Rose (2006). Between March 2016 and July 2017, three rounds of coordinated baiting occurred in the study area. In each round of coordinated baiting, one of the study properties was included in the coordinated program. At the other property, bait deployment was delayed for approximately 6 weeks (Table 1). The property baited as part of the coordinated baiting program was alternated between the two properties for successive rounds of baiting (Table 1). This schedule of baiting allowed us to test the short-term response of dingo populations to baiting without unduly impacting on the landholders' baiting programs. For each baiting round approximately 2000 baits were deployed at 100 m intervals (10 baits per linear km, APVMA, 2008) along property vehicle tracks.

Rapid uptake of dried meat baits in arid areas means that bait availability is greatest in the month immediately following baiting (Kreplins et al., 2018; Twigg et al., 2000). While baiting occurred at both properties during the study, we categorised each property as 'baited' (where baits had been deployed that month), while the other property was categorised as 'unbaited' for that same time point (sensu Kreplins et al., 2020). For each property, we classified months as 'month of baiting', 'month immediately post-baiting' or 'between baiting'.

Ninety-two camera traps were deployed at Property A on 3 March 2016 (77 Reconyx® Hyperfire H500™, 15 Scoutguard® SG560), and 90 camera traps at Property B on 5 March 2016 (79 Reconyx® Hyperfire H500™, 11 Scoutguard® SG560). Camera traps were removed on 9 June 2017 (Property A) and 22 July 2017 (Property B). On each property, camera traps were placed at 1 km intervals on the side of property vehicle tracks along approximately parallel 20–30 km transects. Transects were separated by 3–16 km and covered a total of 90–92 km of vehicle tracks on each property. Vehicle tracks were graded, well-travelled pathways generally wide enough for one vehicle to travel along. Scoutguard® cameras were interspersed evenly between the Reconyx® cameras to account for potential differences in detection.

Camera traps were mounted on posts or trees 0.3–0.5 m above the ground, 1–2 m from the roadside, directed horizontally at an angle of approximately 22° along property tracks (Meek et al., 2012) and were serviced every 2 months. All cameras were equipped with an infrared flash for nocturnal images and were set to operate continuously. When the passive infra-red motion detector of a camera was triggered, it took a burst of three images with no time delay between images. Time stamps on all camera trap images enabled identification of when animals were detected. During rounds of baiting, baits were deployed on property tracks comprising and immediately surrounding the camera trap arrays. As part of each baiting round, a single bait was laid on the track in front of each camera.

2.3. Dingo activity index

Images were viewed as jpeg files in an image viewer program (Windows Live Photo Gallery, Windows 7 Edition, Microsoft Corporation, Redmond, WA, USA). Images of dingoes at greater than 10 min intervals

Table 1

The density (per linear km [km^{-1}] \pm SE) of dingoes, g_0 , σ (km^{-1}) and percent change in density during each month of monitoring at properties A and B.

Month	Property A						Property B						Post-baiting difference between sites
	Bait	Density (km^{-1})	SE	g_0	σ (km^{-1})	Monthly Δ density	Bait	Density (km^{-1})	SE	g_0	σ (km^{-1})	Monthly Δ density	
1 February		0.082	0.029	0.010	3.494		0.081	0.032	0.006	1.394			
2 March		0.152	0.035	0.011	7.090	85%	0.098	0.034	0.083	4.007		21%	
3 April	*	0.144	0.034	0.014	0.843	-5%	0.176	0.048	0.004	4.830		80%	
4 May		0.065	0.024	0.008	6.688	-55%	0.087	0.032	0.008	3.604		-51%	-4%
5 June		0.107	0.030	0.007	6.058	65%	0.207	0.062	0.005	2.933		138%	
6 July		0.048	0.021	0.006	9.978	-55%	0.078	0.031	0.010	2.914		-62%	-7%
7 August		0.171	0.038	0.005	9.353	256%	0.089	0.053	0.008	2.029		14%	
8 September		0.175	0.040	0.008	4.431	2%	* 0.211	0.052	0.005	4.239		137%	
9 October		0.132	0.049	0.002	9.067	-25%	0.255	0.066	0.002	1.085		21%	46%
10 November		0.075	0.026	0.005	7.711	-43%	0.430	0.155	0.002	2.306		69%	
11 December		0.172	0.039	0.004	6.859	129%	0.106	0.035	0.029	0.194		-75%	204%
12 January		0.142	0.039	0.004	5.626	-17%	0.283	0.061	0.001	7.263		167%	
13 February		0.196	0.042	0.004	7.139	38%	0.263	0.054	0.006	4.431		-7%	
14 March		0.141	0.038	0.002	1.085	-28%	0.349	0.060	0.001	2.471		33%	
15 April	*	0.107	0.030	0.005	8.110	-24%	0.290	0.057	0.001	1.978		-17%	
16 May		0.127	0.146	0.001	1.460	19%	0.172	0.047	0.004	4.929		-41%	60%

Dark shading indicates the months baiting occurred, asterisks (*) indicate baiting occurred as part of the regional biosecurity group co-ordinated baiting. Months with light shading are the months immediately after baiting. Post-baiting difference between sites indicates the difference between pre- and post-baiting on the baited site relative to the change on the unbaited sites. Negative figures indicate a decrease in density on the baited site.

were recorded as separate photo capture events. We recorded bait presence in each image and used these data to determine bait uptake (data on bait-take are reported in Kreplins et al., 2018).

A two-way analysis of variance (ANOVA) conducted in R (R Development Core Team, 2017) was used to determine the effect of (1) Property (A or B) and (2) month ('month of baiting', 'month immediately post-baiting' or 'between baiting') on average daily photo capture rate (hereafter 'activity') over the 16-month monitoring period. While analyses that better partition these data are possible, we considered they were not necessary in the context of the limited changes in response to baiting.

2.4. Dingo density

To determine the density of dingoes over the 16 months of monitoring, spatially explicit capture-recapture analyses were performed using the package *secrlinear* (Efford, 2017) in R. Unique identification is required to analyse density estimates of dingoes using this approach. We therefore identified individuals based on physical traits including sex, size, shape and pelage (Forsyth et al., 2019; Gabriele-Rivet et al., 2020). As with any operator-reliant method, there are errors in this approach. To combat this, as recommended by Kelly et al. (2008), two researchers (TK and MK) identified individuals independently. Unidentifiable photographs were excluded from density estimate analyses.

A combination of the state (animal home range) and observations (probability of detecting an individual at a detector in relation to the individual's home range) were used to construct models with the assumption that the dingo populations were closed during each month of monitoring. The detection function used was hazard rate and the detector type was identified as count. Models were fitted numerically, maximising the log likelihood over the capture histories with spatial information to determine animal density (D ; animals per linear km). Each model included the parameters:

g_0 – detectability or the probability of capture when the distance between the animal's activity centre and the camera trap is zero. In a null model, g_0 is constant across animals, occasions and detectors;

σ – the spatial scale of detection. More specifically defined as the spatial scale parameter of detection function or an index of home range. σ and g_0 jointly define the detection probability as a function of location and interpreting their meaning alone should be done with caution (Efford, 2017); and.

D_j – density at a flat scale taking into account the spatial distance between traps, but ignoring any intervening topography.

Secrlinear was used rather than simply *secr* given the camera trap deployment along roads only; estimation of dingo density along a linear habitat is therefore presented in linear km instead of the number of dingoes per unit of area. A linear mask was constructed with a buffer of 1 km from each camera using a poly line shape file of the track transects (as camera detections would be well inside a 1 km buffer). Models were run assuming a linear habitat map and the default Euclidean distance model, which would calculate estimates assuming that dingoes use the tracks for moving around but their movement is not solely restricted to the tracks.

All models were derived from the Cormack-Jolly Seber or Jolly Seber models with refinements. Akaike Information Criterion adjusted for small sample size (AICc) were used to rank models and AICc weights (w_i) were calculated. Only models with $\Delta\text{AICc} < 2$ are reported and dealt with further (these models have the greatest likelihood of all the model-set to be the best model fit to the data).

Linear regression using *lm* function in R was used to compare monthly dingo density estimates between the two properties. Visual scrutiny of temporal patterns in the data also recommended comparison between the properties with a 1-month lag in density estimates for Property B.

Changes in dingo density were calculated as the differences in estimates for consecutive months on each property. To determine potential changes in density that could be attributable to baiting, change in

density in the month immediately post-baiting on the baited property was then subtracted from the change in density in the same month on the unbaited site.

As conducted for dingo activity, a two-way analysis of variance (ANOVA), conducted in R, was used to determine the effect of (1) Property (A or B) and (2) month (classified into 'month of baiting', 'month immediately post-baiting' or 'between baiting') on dingo density over the monitoring period.

2.5. Dingo survival, residential status and immigration

Survival analysis of dingo individuals was estimated using the *wiqid* package (Meredith, 2020) in R. Individuals were identified as present (1) or absent (0) for each of the 16 months. If a dingo was not seen during a month, but was recorded during the previous and proceeding months, it was also assumed present. A simple model was constructed including the month (1–16) to estimate survival (φ) over time. The proportion of individual dingoes detected on camera while a bait was present was calculated for each property.

Resident status of individually-identifiable dingoes was categorised as: resident, partially using the site or transient. Individuals present for three or more months consecutively, were considered resident. Animals recorded for three or more months, but <3 months consecutively, were considered to be partially using the site. Individuals were considered transient if they were detected for 1 or 2 months.

Residency status was compared between properties using a Chi-squared test with expected values estimated assuming an equal proportion of each category between the two properties.

Immigration following the first two rounds of baiting on each property was assessed by calculating the number of individually-identifiable dingoes detected in the 2 months after baiting at the baited and unbaited properties as a proportion of the number of individual known to be alive (KTBA) during that period. The proportion of new

immigrants at the four rounds of baiting was compared between baited and unbaited sites using a two sample *t*-test.

3. Results

Dingoes were continually active on both properties over the 16-month study period, with an average of 57 ± 47 SD dingo photo captures per month on Property A and 46 ± 29 SD on Property B. Individual dingoes could be identified for the majority of capture events (Property A: 81.5% of 1116 capture events, Property B: 81.6% of 852 capture events). A total of 50 individually-identifiable dingoes were observed at Property A (average 16.0 ± 5.3 SD [range 6–21] individuals per month) and 81 on Property B (average 20.0 ± 10.9 SD [range 7–35] individuals per month). There was no evidence of individuals moving between properties.

3.1. Dingo activity index

Dingo activity peaked in April 2016 at Property A (6.7 ± 5.3 SE, Fig. 1), with a more pronounced peak (11.3 ± 9.1 SE) occurring simultaneously at Property B. Subsequent daily activity ranged between 1.1 ± 1.6 SE and 2.6 ± 3.1 SE at Property A and 0.8 ± 1.2 SE and 2.5 ± 3.2 SE at Property B. ANOVA showed no significant effect of property on dingo activity ($F_{3,31} = 1.16, p = 0.291$) or months categorised as pre-baiting, post-baiting or between-baiting ($F_{2,31} = 4.01, p = 0.060$).

3.2. Dingo density

Detectability (g_0) and sigma (σ), or index of home range, varied between properties and over time (Table 1). At Property A, g_0 ranged between 0.001 and 0.014, peaking in April 2016, and at Property B ranged between 0.001 and 0.083, peaking in March 2016. Sigma estimates at

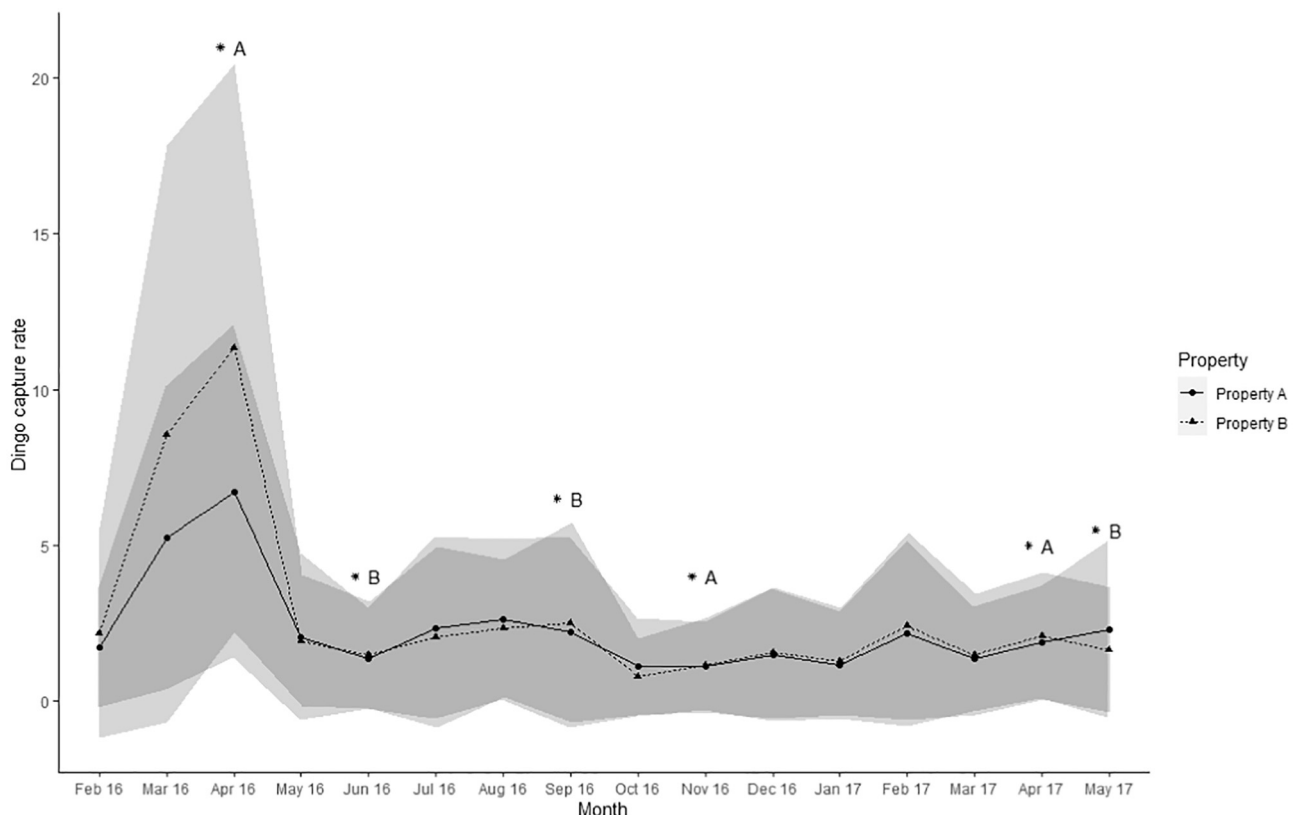


Fig. 1. Daily dingo activity at properties A and B over the 16 months of monitoring. A baiting event is indicated by marked by * and A or B for each property.

Property A ranged from 0.84 to 9.98 dingoes per linear km and at Property B ranged from 0.19 to 7.26 dingoes per linear km.

Dingo densities on Property A ranged from 0.05 to 0.20 individuals per linear km, peaking in winter though to early spring (July–September 2016) and again in summer 2017 (November–February; Table 1, Fig. 2). Densities were generally higher on Property B, ranging from 0.08 to 0.43 individuals per linear km, peaking a month after the peak on Property A. There was no significant relationship between dingo densities estimates at Properties A and B ($R^2_{14} = 0.004, p = 0.110$), but there was a positive relationship between properties when estimates for Property B were lagged by 1 month ($R^2_{13} = 0.341, p = 0.022$, Fig. 3). Regressions of the change in dingo density over the duration of the study showed no significant change in density at Property A ($R^2_{14} = 0.082, p = 0.281$, Fig. 4a). Dingo density estimates showed a significant increase for Property B, equivalent to a trebling of dingo density over the 16-month monitoring period (increase by 0.013 dingoes per linear km per month, $R^2_{14} = 0.353, p = 0.015$, Fig. 4b).

Changes in dingo density in response to baiting (comparing between baited and unbaited properties) were inconsistent. Following the first monitored round of baiting, there was a 4% greater decline in dingoes on Property A (baited) than occurred on the unbaited Property B (Table 1). Following the second round of baiting (Property B) there was similar decline (7%). However, the remaining rounds of baiting showed increases in dingo density in comparison to the unbaited site, ranging from 46% to 204% (Table 1). ANOVA showed a significant effect of property on dingo density ($F_{2,31} = 5.74, p = 0.023$), but no significant difference for months categorised as baiting, post-baiting or between-baiting ($F_{2,31} = 0.41, p = 0.665$).

3.3. Dingo survival, residential status and immigration

Despite sequential rounds of baiting over the 16 month-study, survival of dingoes was high for both properties (Property A: average 88.8% [95% CI 85.1–91.8%]; Property B average 84.4% [95% CI

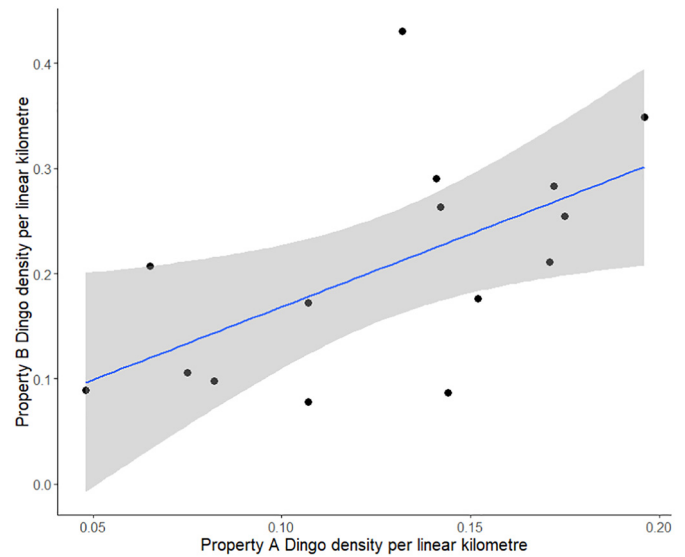


Fig. 3. The relationship between dingo density on Properties A and B, with densities at Property B lagged 1 month. The line indicates the fitted regression, and shaded areas indicate the 95% confidence intervals.

80.1–88.1%]). Approximately half (Property A: 45.9%; Property B: 51.0%) of individually-identifiable dingoes were detected on camera while a bait was present.

There was no significant difference in the residency status of dingoes between the two properties ($\chi^2_3 = 4.34, p = 0.114$, Property A: 28.4% residents, 23.5% individuals partially using the site, 48.1% transients; Property B: 46.0% residents, 20.0% individuals partially using the site, 34.0% transients).

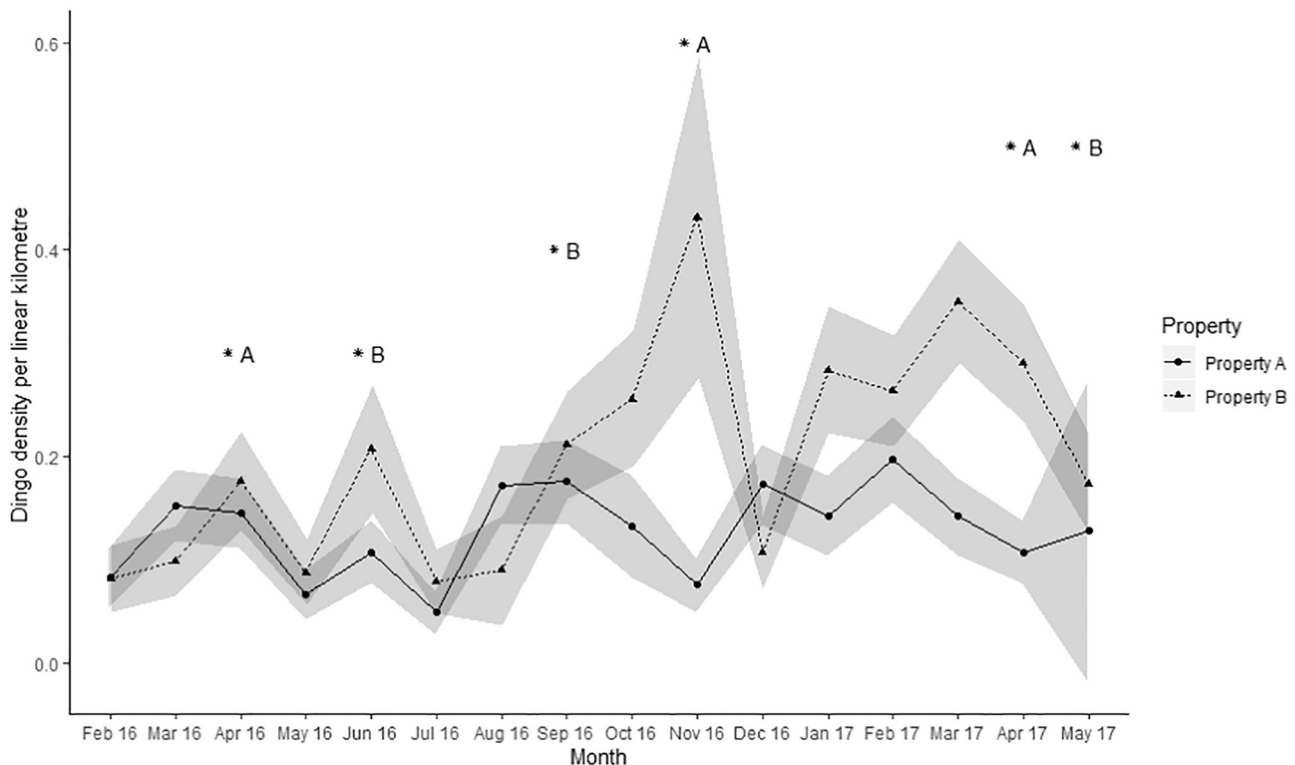


Fig. 2. Densities of dingoes at properties A and B over the 16 months of monitoring as estimated by *seclinear* analysis. A baiting event is indicated by marked by * and A or B for each property.

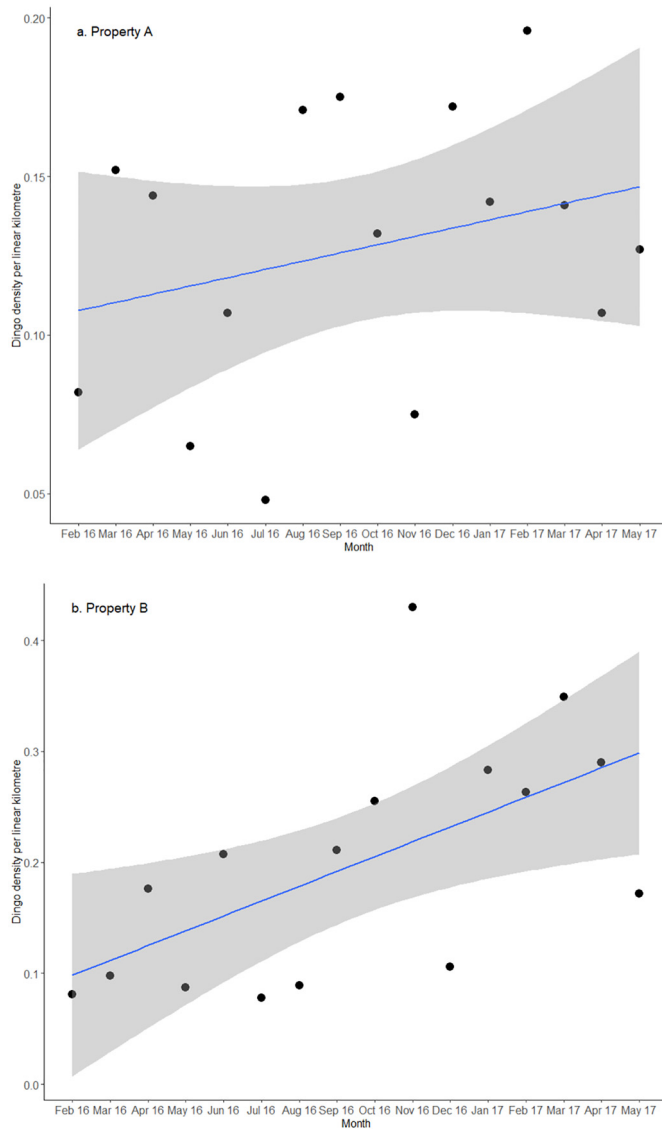


Fig. 4. Relationship between dingo density per linear km and 16 months of monitoring for a) Property A and b) Property B. Line indicates the fitted regression and shaded areas represent 95% confidence intervals.

There was no significant difference in the proportion of immigrants (new animals recorded on a property in relation to the total number of animals KTBA) between baited and unbaited sites following the first

Table 2

The number of immigrants, number known to be alive and the proportion of immigrant dingoes in the 2 months after the first two baiting rounds at properties A and B.

Month	Property A			Property B		
	No. immigrants	KTBA	Proportion of immigrants	No. immigrants	KTBA	Proportion of immigrants
April-16	5	21	0.24	10	18	0.56
June-16	2	17	0.12	3	18	0.17
Sept-16	7	28	0.25	19	31	0.59
Nov-16	2	26	0.08	9	32	0.28

Shading indicates the properties and months at which baiting occurred.

two rounds of baiting on each property (two-sample *t*-test: $t_6 = 0.18$, $p = 0.860$; Table 2).

4. Discussion

Considerable funding and effort is expended on predator control for agricultural and biodiversity outcomes. Determining the effectiveness of control efforts is therefore paramount to understand if these efforts have their intended effect on predator density or impact. For this study in the southern rangelands of Western Australia, we recorded no effect of baiting or post-baiting month on dingo activity or density, and high (>84%) survival of baiting. Neither individual rounds of baiting, nor sequential rounds of baiting over the 16-month monitoring period, resulted in the 75% annual reduction in dingo density that has been identified as necessary for a dingo population control in this environment (Pacioni et al., 2020a).

Dingo densities differed significantly between properties. Despite these differences, there was a positive correlation in dingo density between properties when the data from Property B was lagged by a month. Peaks in density occurred at periods of expected seasonal variation such as preceding mating (February to April), and following emergence of pups from natal dens around October (Thomson, 1992a, 1992b). These seasonal changes, together with the lack of detectable responses to rounds of baiting, indicate that baiting had less impact on density than seasonal variation.

The lack of a response in density to baiting was paralleled in dingo activity. There was limited change in average activity over the duration of the study at both properties, with the exception of high activity in March and April 2016, which rapidly decreased and remained at a lower level for the rest of the study. While the decline coincided with baiting on Property A, a more marked decline occurred simultaneously on Property B, which was not baited in April 2016. We therefore do not attribute the decline from this peak to baiting. As limited human activity occurs on these properties, these temporary peaks in activity may reflect either a rapid, short-term activity increase from February to April 2016, or a decline in dingo activity (after April 2016), in response to increased human activity or camera traps at the sites. We note that dingo activity on cameras was significantly lower when a bait was present (Kreplins et al., 2018; Pacioni et al., 2020b) suggesting the response is more likely to be a decline in activity in response to human activity.

4.1. Factors affecting this baiting program

The effectiveness of baiting programs is influenced by the target species locating (availability), being attracted to (attractiveness), consuming (palatability) and dying as a result of ingestion (lethality) of the bait (Allsop et al., 2017; Fancourt et al., 2020; Saunders and McLeod, 2007). The process for production and deployment of baits in this study was similar to that described by Twigg et al. (2000), who found variable, but lethal, doses of 1080 in field-prepared baits for at least 8 months after baiting. Additionally, the median length of time these baits were on the ground before being taken by any species was <10 days (Kreplins et al., 2018), meaning significant degradation of 1080 was unlikely to have occurred. Consequently, we do not anticipate that bait lethality contributed to the low effectiveness of baiting. Ineffectiveness of the baiting is therefore likely to reflect bait availability, attractiveness and palatability.

Bait availability can contribute to low baiting effectiveness and can be influenced by the rate of baits deployed (Ballard et al., 2020; Fleming, 1996), bait placement (Kennedy et al., 2014), interference by non-target species (Dundas et al., 2014; Fleming, 1996; Kreplins et al., 2018) and density of the target species (Berry et al., 2012). In the current study, reduced availability due to poor deployment or placement in unsuitable habitat or inaccessible locations was minimised through ground, rather than aerial, deployment of baits. However, reduced bait availability due to low baiting rate and/or non-target interference may

have affected baiting effectiveness. Uptake of baits by non-target species can significantly reduce bait availability (Dundas et al., 2014; Kreplins et al., 2018). Uptake of baits by non-target species (including, ants, corvids and varanids), along the same baiting runs as those examined here, was very high (Kreplins et al., 2018). This had the effect of reducing the proportion of baits available to dingoes to <75% of that deployed within 3 days. Low bait availability could therefore have contributed to the limited effectiveness of baiting in this study.

The rate of bait deployment can influence bait availability. The rate of bait deployment in this study was 10 baits per km, in accordance with the Australian Pesticides and Veterinary Medicines Authority (2008), which may have been inadequate for sufficient dingoes to encounter baits to result in a decrease in dingo density. Aerial deployment of baits at 10 baits per km in the New England area of New South Wales has been demonstrated to result in a 55.3% reduction of telemetered dingoes, compared to reduction of 90.6% resulting from a rate of 40 baits per km (Ballard et al., 2020; Fleming and Ballard, 2014). Based on this result, Fleming and Ballard (2014) considered that baiting at a rate of 10 baits per km in the New England environment could not be regarded as control. The probability of a bait being encountered by a dingo is influenced by dingo density, with a baiting regime in high-density populations likely to be more effective than it is in low-density populations (Pacioni et al., 2020b). Dingo densities are likely to be lower in the southern rangelands than the more mesic New England environment studied by Ballard et al. (2020), thus we could anticipate an increase in baiting effectiveness in response to an increased baiting rate in the southern rangelands.

Dried meat baits have been used extensively in the rangelands of Western Australia for several decades (Thomson, 1986). Unrefined meat baits have been found to be preferred over refined meat baits in rangeland environments (Eldridge et al., 2000; Thomson, 1986). Bait attractiveness can potentially be constrained by learned aversive responses by the target species, where an individual has consumed a sub-lethal dose, or through teaching by, or observation of, a conspecific (Allsop et al., 2017). This effect may be more likely to occur with extended reliance on a single control technique (Allsop et al., 2017). In a concurrent study (Kreplins et al., 2018) using the same rounds of baiting on the study sites, bait uptake by dingoes was exclusively by young dingoes ($n = 4$ of 337 baits of confirmed fate) and dingoes were frequently recorded moving past baits without inspecting them (279 of 325 recorded dingo activity events with baits present), or even demonstrated aversive behaviours towards the baits, such as urinating or defecating on them (2 of 325 dingo activity events where bait was present). These results, coupled with lower dingo activity on camera traps on these sites when a bait was present (Kreplins et al., 2018; Pacioni et al., 2020b), suggest low attractiveness and/or bait resistance (sensu Allsop et al., 2017) may be playing a role in baiting effectiveness in this environment.

In rangeland environments, dingoes typically have large home ranges (Allen et al., 2014b; Thomson, 1992b; Wyson et al., 2020) and are capable of large movements in a short time (Thomson, 1992b). Individual dingoes have been recorded periodically undertaking long forays (Allen, 2009; Robley et al., 2010; Thomson et al., 1992). This high level of movement and dispersal has been identified as a constraint to the effectiveness of baiting programs, facilitating rapid immigration back into sites where animals have been removed (Allen, 2017). However, in the present study, the high survival of individuals and the lack of a significant increase in immigration after baiting on each site indicates that low mortality, rather than rapid immigration, was likely to be the key driver in the low effectiveness of the baiting program. The results from this study indicate that there is considerable work to be done to improve bait availability, attractiveness, and palatability for baiting in

this environment to reach levels of effectiveness required to achieve its intended management aims.

4.2. Limitations of this study

There are caveats to our density estimates associated with camera trap type and deployment of baits. Camera traps from different manufacturers have varying abilities and functions to capture the target species (Meek et al., 2015). As a logistical and financial reality, two types of camera trap were used in this study (predominantly Reconyx® camera traps with a small number of Scoutguard® camera traps on each property). The use of a small number of Scoutguard® cameras which have a lower detection rate than Reconyx® cameras (Kennedy and Kreplins unpublished data) may have contributed negative bias to dingo density estimates. Similarly, placing baits in front of cameras may have affected dingo detection. Generally placing a lure in front of a camera should increase detection (e.g. Gerber et al., 2012); however Kreplins et al. (2018) and Pacioni et al. (2020b) found that the detection of dingoes on these camera traps was lower while a bait was present in front of the camera, thus also potentially contributing to a negative bias in density estimates. While cognisant of these caveats, we consider the negligible changes in dingo density and activity in response to baiting over the 16 months of monitoring coupled with high dingo survival, as firm evidence of low baiting effectiveness, and that the result was not due to an artefact of differences in camera trap type or detection function.

4.3. Management implications of this study

Dingo control for production of livestock should include a variety of control tools for greatest effectiveness (Fleming et al., 2014). Baiting is the most commonly used control technique in remote and rugged areas and at large scale, and so it is critical that its effectiveness is evaluated. Under the baiting regime used within our study area, there was negligible impact on dingo densities, and therefore the baiting program was unlikely to benefit livestock production. While, it is important to stress that this is a context-specific finding that differs from other studies on baiting effects on dingoes (e.g. Allen et al., 2013; Ballard et al., 2020; Fleming, 1996; Thomson, 1986), there is a clear need for improved effectiveness to warrant continued use of baiting as the primary dingo control method in the southern rangelands. From a conservation perspective, our findings are consistent with those of Allen et al. (2014a), that the limited effects of the baiting program on dingo density mean that impacts on the ecological functions of dingoes from baiting in this environment are likely to be minor.

4.4. Conclusions

Control programs come at a substantial financial and labour cost to livestock producers, but in our Western Australian study, this effort has not resulted in the desired decrease in dingo density over time, nor has it created the conditions necessary to restore sheep grazing to the region. Thus improvements to baiting practices are required for managers to have confidence that dingo control is both effective and justified. Increasing availability of baits to dingoes (e.g. by changes to bait rate, baiting frequency and targeting periods when non-target species are less active) and improving bait attractiveness (e.g. by varying bait matrices used and inclusion of lures) may improve effectiveness of baiting. It is possible that changes to baiting practices to improve availability and attractiveness of baits may not produce increases in the effectiveness of baiting in this context. Hence, the relative costs and benefits of baiting should also be examined against the implementation of other approaches such as fencing, trapping, canid pest ejectors, enterprise change and no control.

Declaration of competing interest

There are no conflicts of interest concerning our article.

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