CENTRE FOR INVASIVE SPECIES SOLUTIONS

ASSESSMENT OF THE BIODIVERSITY, ECONOMIC AND PRODUCTIVITY GAINS FROM EXCLUSION FENCING, WESTERN AUSTRALIA

FINAL REPORT FOR PROJECT P01-L-006

AUTHORS

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OCTOBER 2022

Prepared for the Centre for Invasive Species Solutions

The Centre for Invasive Species Solutions gratefully acknowledges the financial and in-kind contributions made by its members, associate members, and partners.

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We acknowledge the Traditional Custodians of the lands on which we meet and work and pay our respects to Elders — past, present and emerging.

We acknowledge all Aboriginal and Torres Strait Islander peoples and their continuing connection to country, culture and community.

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CITATION
This report should be cited as: Kreplins T, Kennedy M, Fleming T, Dawson S, Miller J, Barwick J, Macleay C, Omogbene M, O'Leary R, Renwick J, and Cowan M (2022). *Assessment of the Biodiversity, Economic and Productivity Gains from Exclusion Fencing, Western Australia: Final Report for Project P01-L-006*. Report for the Centre for Invasive Species Solutions.

invasives.com.au

ISBN e-Book 978-1-925727-94-4

ISBN Print 978-1-925727-95-1

ACKNOWLEDGEMENT OF PROJECT PARTNERS

The *Assessment of Biodiversity, Economic and Productivity Gains from Exclusion Fencing (WA)* project was led by the Western Australia Department of Primary Industries and Regional Development in partnership with Department of Biodiversity, Conservation and Attractions, Murdoch University and Meat and Livestock Australia.

The project was funded by the Australian Government Department of Agriculture, Fisheries and Forestry, Meat & Livestock Australia, Murdoch University with in-kind support from Western Australian Department of Primary Industries and Regional Development, Western Australian Department of Biodiversity, Conservation and Attractions, and Murdoch University.

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Cover image: Deployment of canid pest ejectors and camera traps. Credit Tracey Kreplins Western Australian Department of Primary Industries and Regional Development.

ASSESSMENT OF THE BIODIVERSITY, ECONOMIC AND PRODUCTIVITY GAINS FROM EXCLUSION FENCING (WA)

FINAL PROJECT REPORT FOR P01-L-006

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INTRODUCTION

Rangelands account for over three-quarters of the Australian landmass. Rangelands are areas of land where livestock are grazed on native vegetation or where the rainfall is too low for intensive agriculture. They are biologically diverse and economically important. However, livestock production in the rangelands faces several significant challenges (Safstrom et al. 2013). The current situation in the southern rangelands of Western Australia (WA) is representative of issues in small-stock production rangelands across Australia. In WA the southern rangelands encompass the Gascoyne, Murchison, Goldfields and Nullarbor regions. The presence of wild dogs and artificially enhanced populations of native and introduced herbivores act to limit production and enterprise choice. Reductions in rainfall over time are also affecting the productivity of rangelands.

These factors are of particular concern in the southern rangelands of WA where there are few other enterprise choices other than small stock. Landholders are currently investigating and implementing options to allow them to develop sustainable enterprises. These include: wild dog fencing from paddock- (approximately 200 km^2) to large-cell- (88,000 km^2) scale; manipulation of water availability to direct stock; and implementation of established and new pest control measures at the landscape-scale.

Fencing has occurred for many years for conservation (e.g. Hayward et al. 2009), keeping out invasive species (e.g. Ens et al. 2016), mitigating human–wildlife conflicts (e.g. Pekor et al. 2019), and for farming and pastoralism purposes (e.g. Newsome et al. 2001b; Pickard 2007; Cockfield et al. 2018) around the world. In addition to these purposes, fencing is often used to delineate boundaries, generate financial and social benefits, and reduce conflict. However, fencing can be costly, requires continual maintenance, and cause negative impacts on wildlife (Pekor et al. 2019). In Australia alone there are thousands of fences (and cell fences) used to keep unwanted species out of agricultural enterprises and conservation estate (Smith et al. 2020). The earliest exclusion fence in WA was the State Barrier Fence in the 1860s to halt the spread of rabbits (Coman 1999). In the present-day, cell fences are still being constructed in WA to halt the movement of wild dogs.

This project evaluated cell fencing in Western Australia (WA). Cell fencing entails a group of landholders fencing the outer perimeter of their collective property boundaries using fencing which is impermeable to target species, typically wild dogs (dingoes, free-living domestic dogs and their hybrids, *Canis familiaris* [Jackson et al. 2017, 2019]). Western Australian Regional Biosecurity Groups involved received matching funds under the Western Australian Wild Dog Action Plan 2016–2021 (Department of Agriculture and Food Western Australia 2015) in February 2018 to purchase the building materials for four fences in total. In this report we address biodiversity and productivity impacts of two fences within the Murchison area as well as of the WA State Barrier Fence. The economics of all four WA cell fences will be addressed in another report.

Within this report we aim to investigate relationships between active predator management, cell fencing and water availability; and their influence on native herbivores, introduced herbivores and introduced predators. To address these aims, the project will determine changes in the numbers of introduced predators, native and introduced herbivores in response to fencing, predator densities and water availability. It will also identify how changes in predator and herbivore density can be practically used by landholders to improve small-stock production and native biodiversity.

The landholders in this study are part of the Meekatharra Recognised Biosecurity Group (MRBG), which coordinates control across the landscape to mitigate ongoing losses from wild dogs. Landholders are legally required to control declared pests such as wild dogs under the *Biosecurity and Agriculture Management Act 2007* (State Government of Western Australia 2007). In addition to their legal obligations and pressures from neighbouring landholders and the group, the landholders involved in this project are strongly committed to wild dog control given the ongoing emotional and financial pressures from wild dog attacks on their small stock. This group completed (in November 2021; construction commenced in May 2018) a large area of fencing called the Murchison Region Vermin Cell (MRVC; Figure 1) that incorporates 52 pastoral properties and nine Department of

Biodiversity, Conservation and Attractions properties. It spans from the State Barrier Fence, to as far north as Meekatharra. Since completion, the landholders now have the considerable task of removing all wild dogs from within the 1,400-km fence line. Some landholders within the larger cell successfully sought funding to build a smaller cell, the Murchison Hub Cell (MHC; commenced construction in May 2018 and not yet completed; completion due in early 2023; Figure 1), which lies within the MRVC. Chapter 1 and 2 focus their studies within the smaller MHC. Chapter 3 investigates wild dogs and kangaroos along the Western Australian State Barrier Fence.

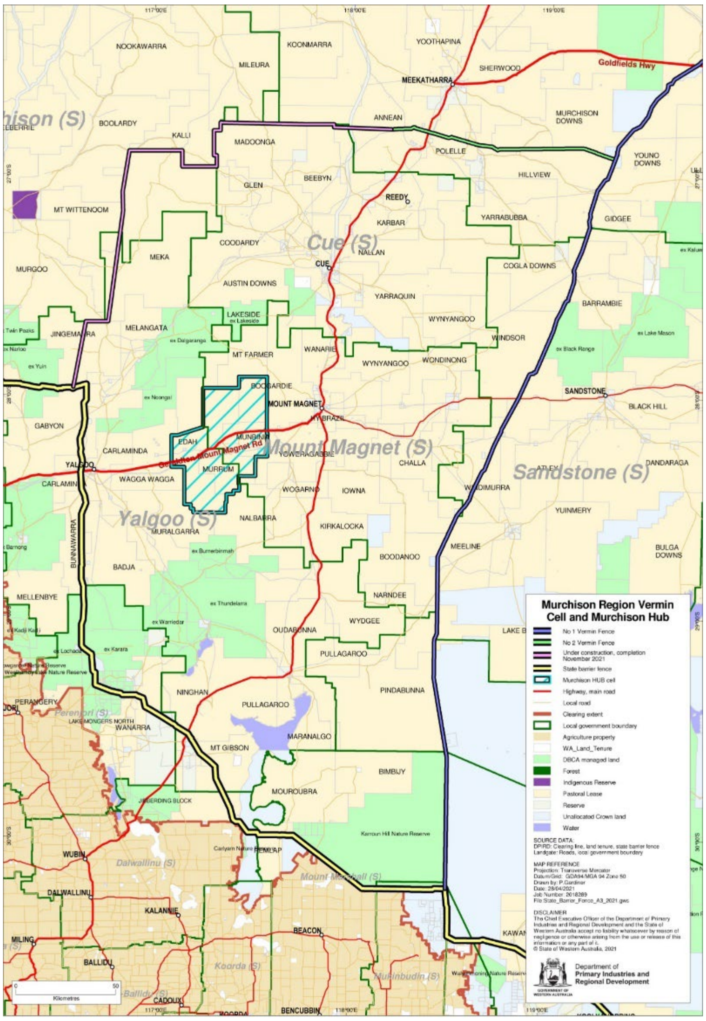


Figure 1. The layout of the Murchison Region Vermin Cell (MRVC) and the Murchison Hub Cell (MHC)

CHAPTER 1. WILD DOG AND SMALL-STOCK SPATIAL RESPONSES TO FENCES IN THE SOUTHERN RANGELANDS OF WESTERN AUSTRALIA

INTRODUCTION

Wild dogs are significant predators of livestock, costing an estimated $89 million per year in control action and lost productivity nationally (McLeod 2016). In addition, wild dog impacts also have significant emotional costs to producers, and have costs associated with their control (Binks et al. 2015; Ecker et al. 2015). Wild dog predation on livestock must be almost non-existent for small-stock enterprises (i.e. sheep and goats) to be financially viable (Thomson 1986b; Allen et al. 2001b; Fleming et al. 2001b; Thomson et al. 2006). While cattle can withstand greater impacts from wild dogs, cattle production systems can suffer significant losses as a result of wild dog impacts (McGowan et al. 2014). Effective wild dog control requires integrated use of multiple tools. In addition to conventional forms of control (i.e. baiting, shooting and trapping), non-lethal forms of control are receiving increasing attention (Eklund et al. 2017).

Fencing is a well-established, worldwide wildlife management tool for protecting livestock from predation. Fencing is used extensively for the conservation of threatened species to reduce or remove predation pressure on populations of native wildlife (Moseby et al. 2009; de Tores et al. 2012; Somers et al. 2012). Fencing is also used to reduce or remove human–wildlife conflicts. For example, in South America fencing on cattle stations is installed to restrict jaguar (*Panthera onca*) attacks on cattle (Cavalcanti et al. 2012). In Australia extensive fencing (e.g. the Dingo Fence in South Australia, New South Wales and Queensland; and the State Barrier Fence in WA) restricts the passage of wild dogs, macropods (*macropod* spp.) and emus (*Dromaius novaehollandiae*) (Pople et al. 2000b; Allen et al. 2001b; Bradby et al. 2014) for protection of agricultural enterprises. There is a dearth of published literature on the effectiveness of the fencing itself for livestock production. However, ‘exclosure’ (or cell) fencing (rather than barrier fencing) involving multiple landholders fencing their outer perimeter, specifically for livestock production, is an emerging issue and largely undocumented in Australian landscapes. Most literature investigates the impact fencing has on native fauna and wild dog population size (Caughley et al. 1980b; Pople et al. 2000b; Newsome et al. 2001b), rather than livestock enterprise viability.

Landholders in southern rangelands regularly experience wild dog predation of livestock. Properties that are part of this study are some of the few left in the southern rangelands that run small stock due to ongoing wild dog predation. Even these producers regularly note lost lambs or kids, bite-marks and attacks, mis-mothering due to wild dog presence and overall reduced productivity due to wild dog activities (Kreplins et al. 2018b; Pacioni et al. 2018a).

MURCHISON HUB CELL – FENCING TO REDUCE WILD DOG PREDATION OF SMALL STOCK

The MHC is a fencing cell under construction surrounding four stations in the southern rangelands of Western Australia (Figure 2). This smaller cell within the large MRVC is aimed at reducing stock losses to wild dogs and ‘reinvigorate the sheep industry in the southern rangelands’ (Jones 2017). All the stations within the cell are the last few to run merino sheep for hundreds of square kilometres, but also run a herd of goats which are included in the study. Once completed (the aim is for late 2022) the MHC will undergo wild dog control to remove all wild dogs from the fenced area, providing a predator-proof area to run small stock. The stations received funds under the Western Australian Wild Dog Action Plan 2016–2021 (Department of Agriculture and Food Western Australia 2015) funding in February 2018 to purchase the building materials.

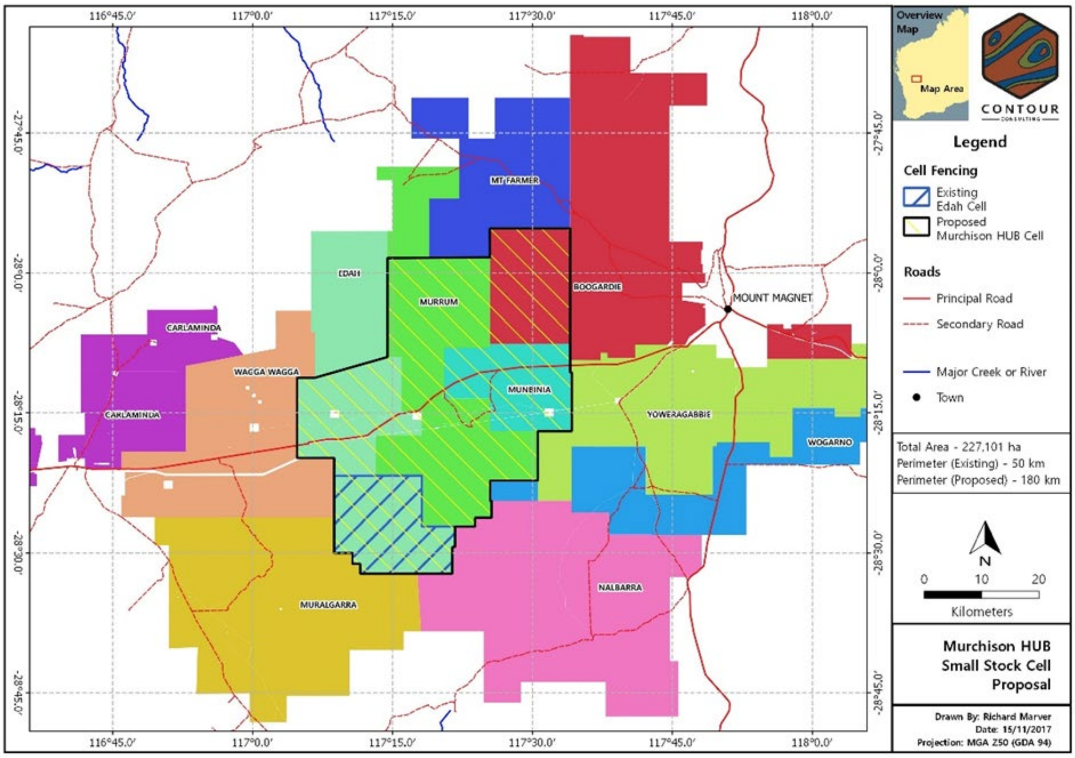


Figure 2. The layout of the MHC

Aims of the project are to determine:

1. If activity of wild dogs decrease as the cell fence is completed?
2. How the activity of the small stock, feral cats, native and non-native herbivores change
    a. as the cell fence is completed?
    b. the wild dog activity events?
3. If the daily spatial movements of small stock increase due to reduced wild dog presence because of the cell fencing?

METHODS

SITE DESCRIPTION

The study area comprises occasional ranges separated by stony slopes and alluvial plains upslope of salt lakes. Sandy soils on sandplains and granitic country predominate; most habitat types are shrublands (typically dominated by the genera *Acacia*, *Atriplex*, *Eremophila*, *Maireana*, *Ptilotus* and *Senna*) with *Triodia* spp. grasslands occurring in some locations (Watson et al. 2007). Average annual rainfall in the southern rangelands ranges from 200 to 250 mm, increasing on the south-western margins.

The dominant land use is pastoral production, although conservation estate and mining leases occur on a significant portion of the land. Historically the dominant livestock enterprise has been merino sheep; however, this has declined in recent decades with increases in cattle, meat sheep and managed goats (Watson et al. 2007).

The MHC, located in the Yalgoo shire, 30–60 km west of Mount Magnet in the southern rangelands of Western Australia, encompasses four stations and is in varying stages of completion over the project. The MHC is bisected by the Geraldton–Mount Magnet Highway. In November 2020, half the cell below the highway was enclosed. This half incorporates two smaller 30-ha cells that will initially house all the small stock until the larger MHC is complete. North of the highway, bisecting the property, only the north-east corner of the cell requires completion at the time of this report.

Throughout the project, landholders undertook wild dog control to remove the wild dog presence from inside the cell fence. Twice-annual baiting occurred by the landholders as part of a coordinated effort by all landholders within the region, as well as licensed pest management technicians conducting trapping, shooting and additional baiting. During the fence construction, landholders implemented intensive wild dog control efforts using canid pest ejectors to remove all wild dogs within the cell fencing area (Chapter 2 describes this work).

DOES THE ACTIVITY OF WILD DOGS DECREASE UPON COMPLETION OF THE FENCE?

HOW DO THE SMALL STOCK, FERAL CATS, NATIVE AND NON-NATIVE HERBIVORES CHANGE AS THE FENCING IS COMPLETE AND REACT TO WILD DOG ACTIVITY?

Thirty-six infrared Reconyx HFX2 professional cameras were placed at water points to record the number of wild dogs, small stock, native and non-native herbivores in the paddock (details on the running time for these cameras are in Appendix 1). Cameras were set up in February 2019 and are still running; data analyses in this report include data up to December 2020. Cameras were deployed on fence droppers 0.5 m off the ground, facing a south-west direction where possible to reduce excess camera triggers. Camera-monitoring work covered north (incomplete cell) and south (complete cell in November 2020) of the highway; the tracking of small stock occurred south of the highway as the cell was completed.

Cameras monitored the activity events of wild dogs, goats, sheep, macropods, emus and feral cats for the project. An independence threshold was set at 10 minutes between activity events recorded on camera as per other published works (Kreplins et al. 2018a; Kreplins et al. 2020). Age structure of livestock flocks was also recorded (adult, sub-adult and juvenile). The number of days each camera ran for was recorded (Appendix 1) and the activity events for each species was averaged over those days and then standardised per month (for 30 days).

Activity indices (number of independent events per month; 30 days) for each species were log-transformed, except wild dog activity events (which were sparse and therefore not significantly different from a Gaussian distribution). To determine the change in wild dogs, goats, sheep, macropod, emu and feral cat activity events per month (30 days; dependent variables in separate analyses), generalised linear models were run with month (1–26; continuous variable), rainfall and delayed rainfall as independent variables. Monthly rainfall data was taken from the Bureau of Meteorology climatic data online (station 007024). Number of wild dog activity events per month was also used as an independent variable for the analyses on goat, sheep, macropod, emu and feral cat activity events per month. Models were run with the entire dataset, and then in two separate parts for north (incomplete cell) and south (cell completed November 2020, 23 months into the study).

The sheep activity events for adults, sub-adults and juveniles per month were analysed with generalised linear models were run; month (1–26), rainfall, delayed rainfall and wild dog activity events were the independent variables (RStudio Team 2018). The same analysis was completed for goat activity events.

Collinearity between predictor models was investigated using the *car* package and *vif* function (Kassambara 2018), and found values of ≤ 1. Distribution of residuals were checked for all dependent variables using the *DHARMa* package in R (Hartif 2022). No significant issues were noted, with the exception of the wild dog and feral cat values when analysed collectively north and south of the highway. When analysed separately for each side of the highway, the data had no significant issues. These values had non-linear relationships.

Wild dogs are individually recognisable and identified. To compare changes in density of wild dogs before and after the closure of the cell fence south of the highway, we used spatially-explicit capture–recapture analyses (*secr* package in R) (Efford 2017) with the assumption that the wild dog populations were closed during the study. A combination of state (animal home range) and observation (probability of detecting an individual at a camera trap in relation to the individual's home range) were used to construct models derived from Cormack-Jolly-Seber or Jolly-Seber models with refinements. 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*; individuals per km^2). Each model included the parameters:

- g0: 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 g0 is constant across animals, occasions and detectors
- Sigma (σ): the spatial scale of detection
- D_j: density at a flat scale (i.e. ignoring any intervening topography) taking into account the spatial distance between traps.

Jointly, σ and g0 define the detection probability as a function of location – interpreting their meaning alone should be done with caution (Efford 2017). Covariates included session (before and after cell fence closure). A mask was constructed with a buffer of 2.7 km from the furthermost camera traps.

DO THE DAILY SPATIAL MOVEMENTS OF SMALL STOCK INCREASE DUE TO REDUCED WILD DOG PRESENCE BECAUSE OF FENCING?

Two types of tracking devices were used on the small stock to retrieve the same animal movement information:

1. 12 UNEtracker (constructed in-house at the University of New England, New South Wales) were deployed on nine sheep and three goats between 13 February 2019 and 28 February 2020.
2. 11 Lotek wildlife tracking collars (3 LiteTrack 420 store onboard collars and eight LiteTrack Iridium 420 collars, Canada) were deployed on nine sheep and two goats between 3 June 2020 and 2 January 2022.

Figure 3. A sheep wearing the GPS tracking collar

Collars were attached to the small stock while they were restrained in a stock crate during regular yard work (Figure 3). In most high-rainfall areas, sheep work occurs regularly (as often as monthly). In the rangelands area, this is not practical and small stock are usually only yarded annually. At the initial yarding event, each individual is given an electronic ear tag. At each yarding event, small stock were weighed (Tru-Test, Prattley Animal Management Systems), given a body condition score (van Burgel et al. 2011) and the tracking devices were refitted/attached.

At present the flock/herd sizes in the southern rangelands are small due to the ongoing predation pressures. Station A previously had 40 sheep and a small herd of free-ranging goats. Station B had 4,000 sheep and goats residing in the cell below the highway. As of February 2021, the sheep at Station B were agisted to rest the land and the lack of rainfall/feed present. Over the course of the project the small stock will be replenished, but at the time of this report only goats remain inside the cell.

The GPS tracking data was analysed using the *adehabitat* package (Calenge 2006) to calculate the kilometres travelled per day for the collared sheep and goats. These values were estimated before and after the cell was complete south of the highway. This method is able to highlight areas of concentrated use by the small stock. Some individuals were only tracked during the 'before' phase and others were tracked before and after the cell south of the highway was complete (Appendix 3). All animals were tracked south of the highway; no individual livestock travelled north of the highway during the study.

LRI (livestock residency index; a visual representation of time spent across the available area) was calculated as a percentage of GPS tracking points from the goats and sheep in each grid cell based on the total of points in the paddocks. The grid size is 100 m × 100 m.

The formula was:

$$LRI = Count\ in\ polygon \div Sum\ of\ count\ in\ all\ polygons \times 100$$

Activity overlap between wild dogs recorded on camera traps and tracked livestock was investigated. Wild dogs recorded on camera trap (with date stamp) were compared to the GPS points (along with date and time) to investigate whether wild dog activity events impacted livestock activity. The kilometres from water points that livestock travelled after a wild dog activity event on camera was calculated using the latitude and longitude of the camera location and the GPS points recorded on the tracking collars.

RESULTS

CAMERA MONITORING

During 26 months of camera monitoring within the MHC, 45 wild dog, 70,592 goat, 50,025 sheep, 28,531 macropod, 13,361 emu and 110 feral cat activity events were recorded. Results for all generalised linear models are found at Appendix 2.

Rainfall (during, $\beta = -0.05$, $P = 0.40$; or prior to the month, $\beta = -0.009$, $P = 0.16$) did not affect wild dog activity. The month of study had a negative relationship with wild dog activity ($\beta = -0.04$, $P = 0.004$; Figure 4). When the wild dog data was examined in two parts, in the south of the highway there was no relationship with month, rainfall, and rainfall the month prior by the wild dog activity events per month. For wild dog activity north of the highway, there was a negative relationship with month only ($\beta = -0.02$, $P = 0.02$).

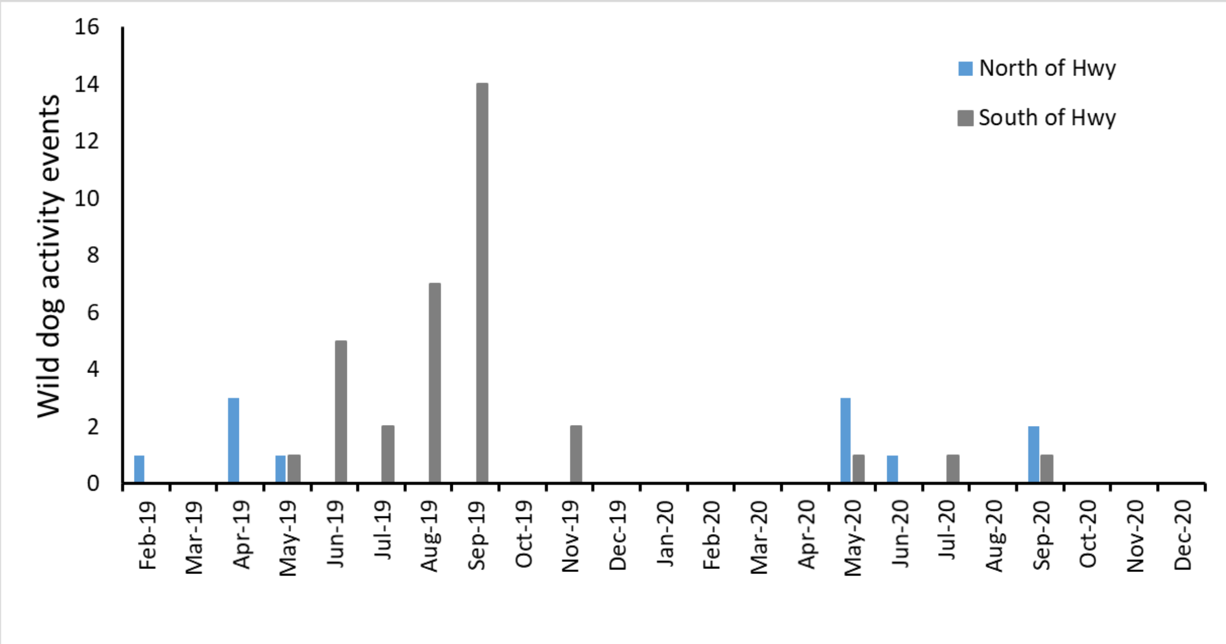


Figure 4. The number of wild dog activity events per month within the MHC, north and south of the highway

There were 25 wild dog individuals during the project. Eighteen individuals were recorded in 2019 (16 below and two above the highway) and only seven in 2022 (three below and four above the highway). None of the wild dogs in 2019 were seen in 2020, and vice versa. Before the cell was fenced in the southern end was enclosed, there were 0.144 wild dogs per km^2, and afterwards there were 0.053 wild dogs per km^2.

Rainfall (during, β = –0.02, P = 0.41), rainfall in the preceding month (β = –0.03, P = 0.28), or month of study (β = –0.05, P = 0.30; Figure 5) did not affect sheep activity overall. Sheep activity outside the complete part of the cell fence, north of the highway had a negative relationship with month (β = –0.07, P = 0.02); sheep south of the highway were unrelated to month (β = –0.02, P = 0.23). The sheep residing outside the cell were all predated upon by the writing of this report, as recorded by the landholders. Sheep activity south or north of the highway did not respond to rainfall, rainfall the month prior or wild dog activity per month.

Breaking down sheep responses by age cohort: adult, sub-adult and juvenile sheep activity events were unrelated to month (Figure 5), rainfall, rainfall the month prior or wild dog activity events. However, sub-adult sheep activity events were negatively related to rainfall (β = –0.01, P = 0.03).

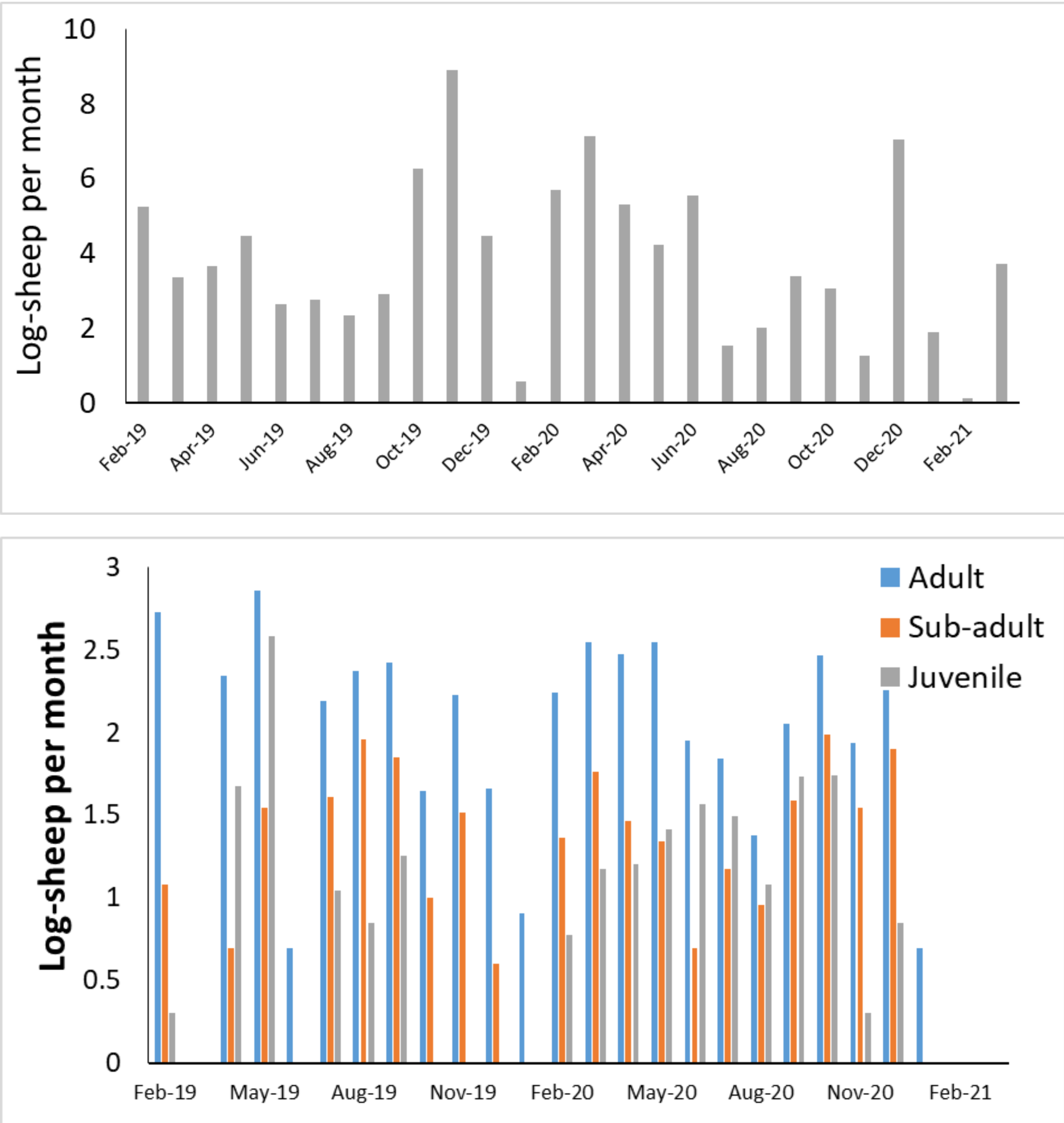
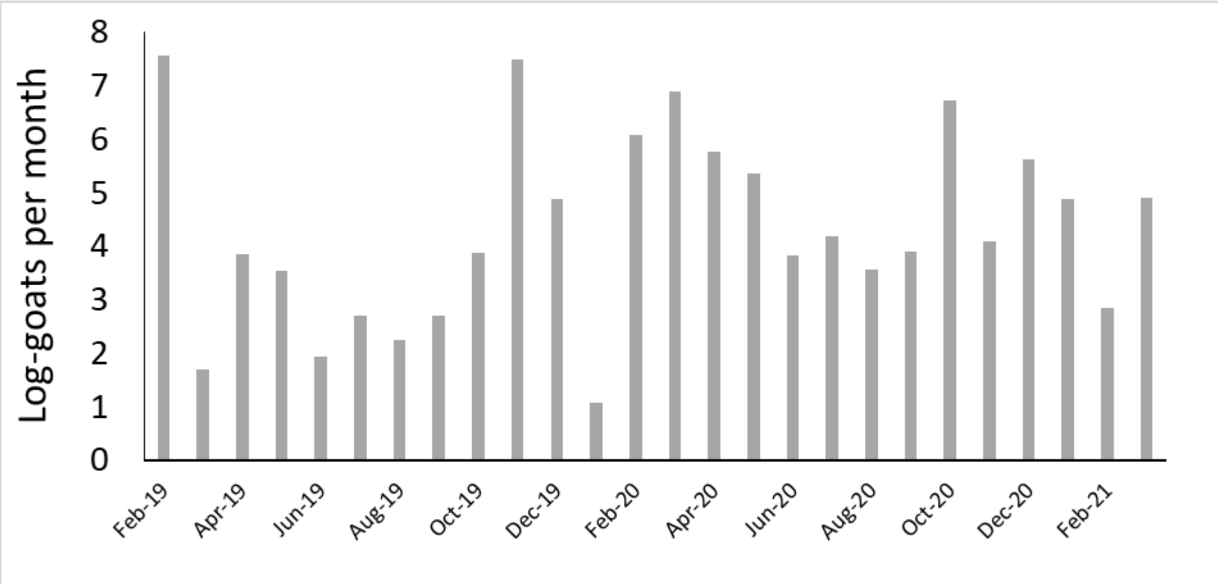


Figure 5. (a) Log-sheep activity events per month within the MHC and (b) log-sheep activity of adults, subadults and juveniles per month at water points inside and outside the MHC at (a) station A and (b) station B

Goat activity per month was not correlated with rainfall (during, β = –0.02, $P = 0.28$) rainfall the month prior (β = –0.04, $P = 0.15$), or month (β = 0.04, $P = 0.30$; Figure 6). Goat activity did not respond to the wild dog activity events in the entire cell (β = –0.36, $P = 0.64$). When the dataset was divided between north and south of the highway, goat activity per month was negatively related to the rainfall the preceding month north of the highway (β = –0.04, $P = 0.02$) but there was no relationship with the goats south of the cell. Goat activity per month either side of the highway was unrelated to rainfall, month and wild dog activity events per month.

Breaking down goat responses by age cohort: adult and sub-adult goat activity events were unrelated to month, rainfall, rainfall the month prior and wild dog activity events. Juvenile goat activity events were positively related to month (β = 0.03, $P = 0.03$; Figure 5b), and unrelated to rainfall (β = –0.01, $P = 0.06$), rainfall the month prior (β = –0.001, $P = 0.89$) and wild dog activity events (β = 0.10, $P = 0.65$).



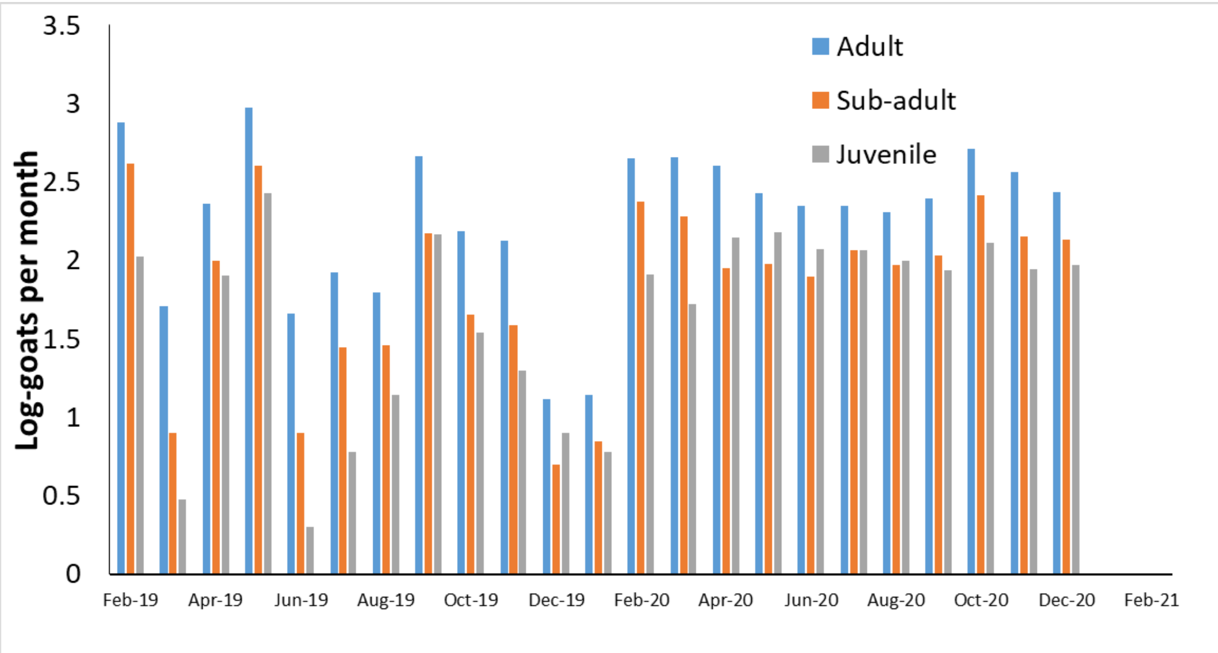


Figure 6. (a) Log-goat activity events per month within the MHC and (b) log-goat activity of adults, subadults and juveniles per month at water points inside and outside the MHC

Macropod activity events per month did not respond to rainfall (β = –0.04, $P = 0.28$), rainfall in the month prior (β = –0.07, $P = 0.09$), or the month (β = –0.07, $P = 0.38$; Figure 7). Macropod activity events did not respond to the wild dog numbers in the entire cell (β = –0.89, $P = 0.45$). This result did not differ when examining the macropod numbers south of and north of the highway (Figure 8).

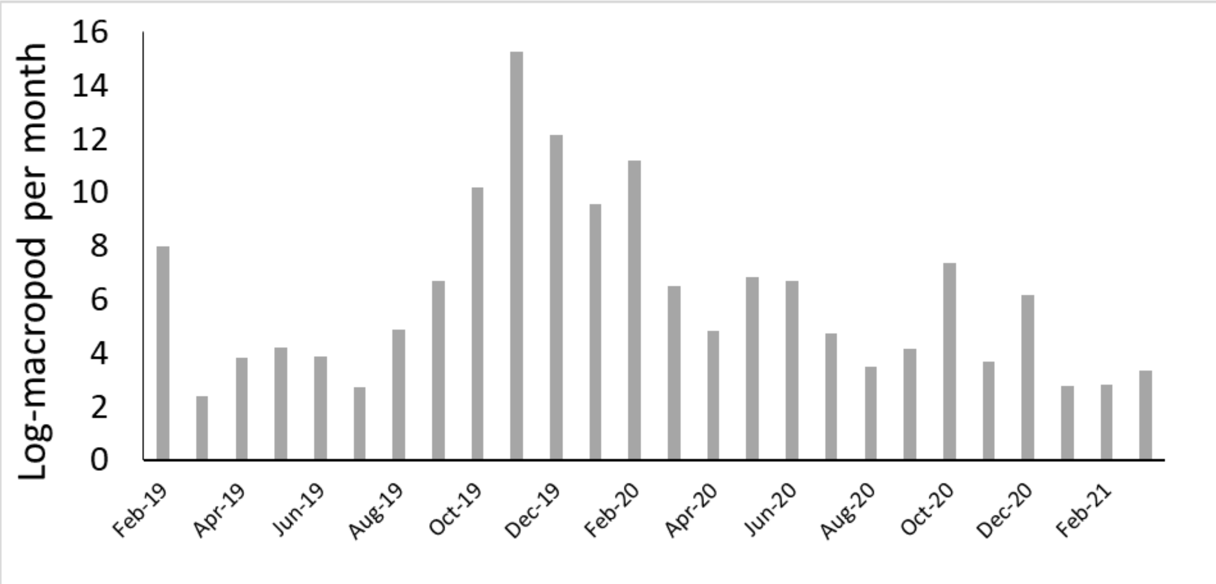


Figure 7. Log-macropod activity events per month at water points within the MHC

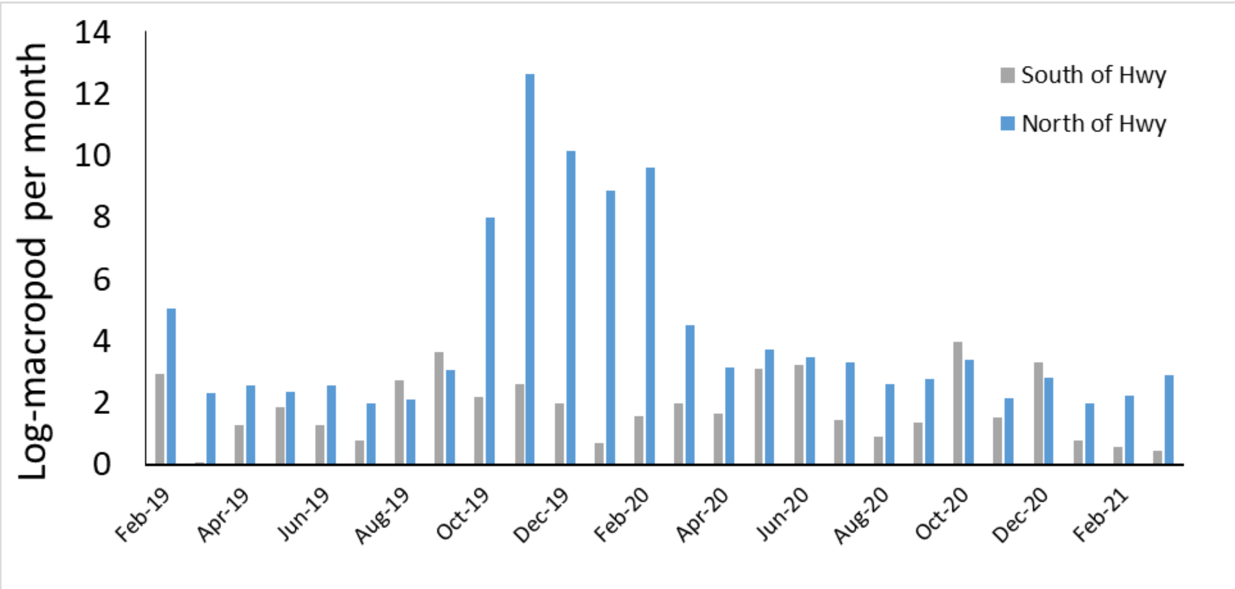


Figure 8. Log-macropod activity per month north (fence incomplete) and south (fence complete 23 months into study) of the highway

Emu activity events per month did not respond to rainfall (β = –0.02, *P* = 0.26), rainfall in the month prior (β = –0.01, *P* = 0.44), or the month (β = 0.009, *P* = 0.81; Figure 9). Emu activity events did not respond to the wild dog numbers in the entire cell (β = –0.38, *P* = 0.51). This result differs for emu activity events north of the highway in the incomplete cell, with a negative relationship with month (β = –0.03, *P* = 0.01) and rainfall (β = –0.02, *P* = 0.004), but the same group of emu activity events had no relationship with rainfall the month prior or with wild dog numbers. Emu activity events south of the highway were not related to any variables.

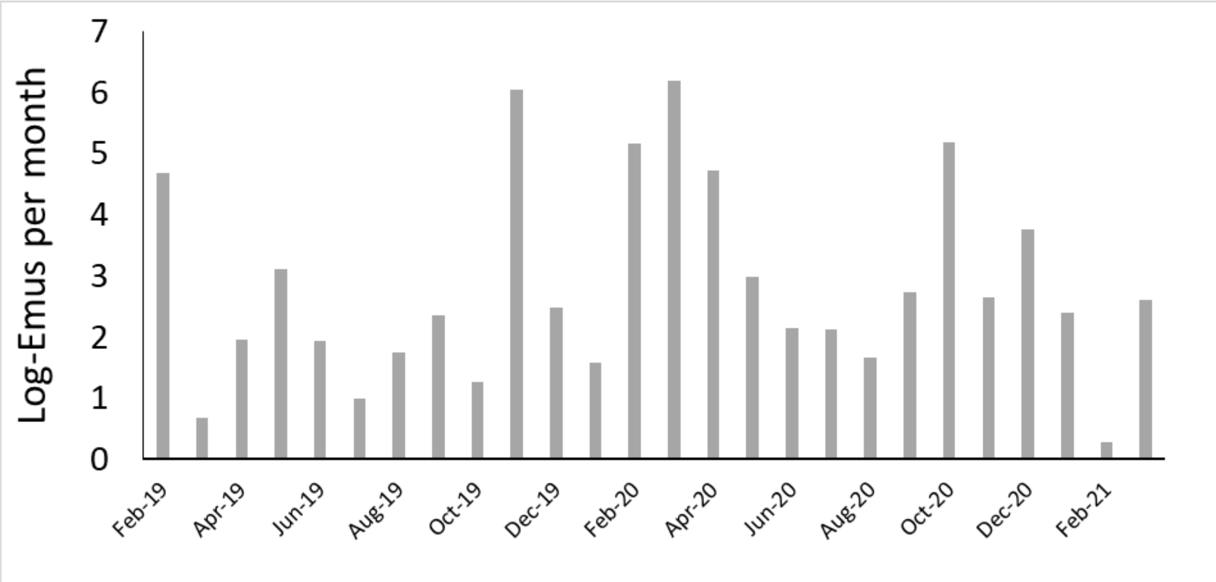


Figure 9. Log-emu activity events per month at water points within the MHC

Feral cat activity events per month were not related to month (β = –0.01, *P* = 0.65; Figure 10), rainfall (β = –0.01, *P* = 0.31) or rainfall in the month prior (β = –0.01, *P* = 0.15). Wild dog activity events were unrelated to feral cat activity events (β = 0.35, *P* = 0.23). These results were unchanged by the separation of the cell either side of the highway.

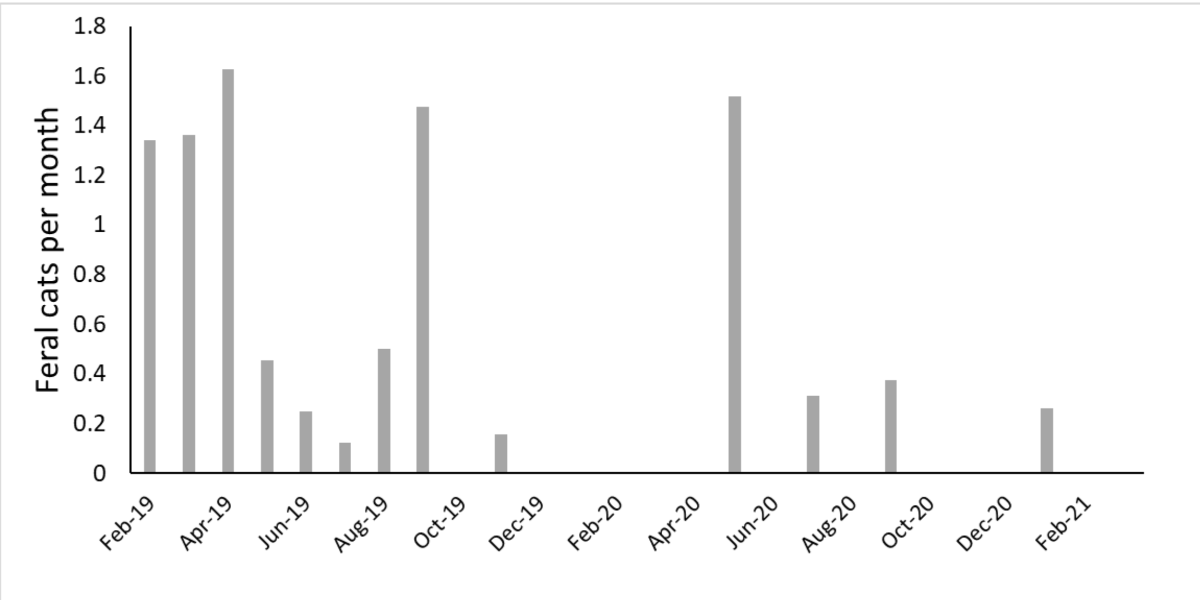


Figure 10. Feral cat activity events per month within the MHC

TRACKING SMALL STOCK

Tracking data for 23 individual sheep (*n* = 17) and goats (*n* = 5) was recorded from February 2019 until the end of 2020. Of these individuals, only eight recorded data after the cell south of the highway was complete (sheep = 7; goats = 1). On average, the sheep and goats moved similar distances per day, 2.4 km (range: 0.57–6.17 km) and 1.80 km (range: 0.49–6.60 km; Table 1) respectively.

Sheep moved further per day after the cell was closed south of the highway (sheep were only tracked south of the highway). On average, before and after distances travelled by sheep were 1.74 km (range: 0.57–6.17 km) and 4.12 km (range: 3.38–4.65 km), respectively. The distance travelled by goats was reduced after the cell closure; before 2.07 km (range: 0.57–6.60 km) and after 0.49 km; however, there was only one goat tracked in the 'after' period (Table 1).

It is evident that during the 'after' period when the cell fence south of the highway was enclosed that the residency index was reduced. There were also fewer animals being tracked, but they moved their core activity from seven to two paddocks (Figure 11).

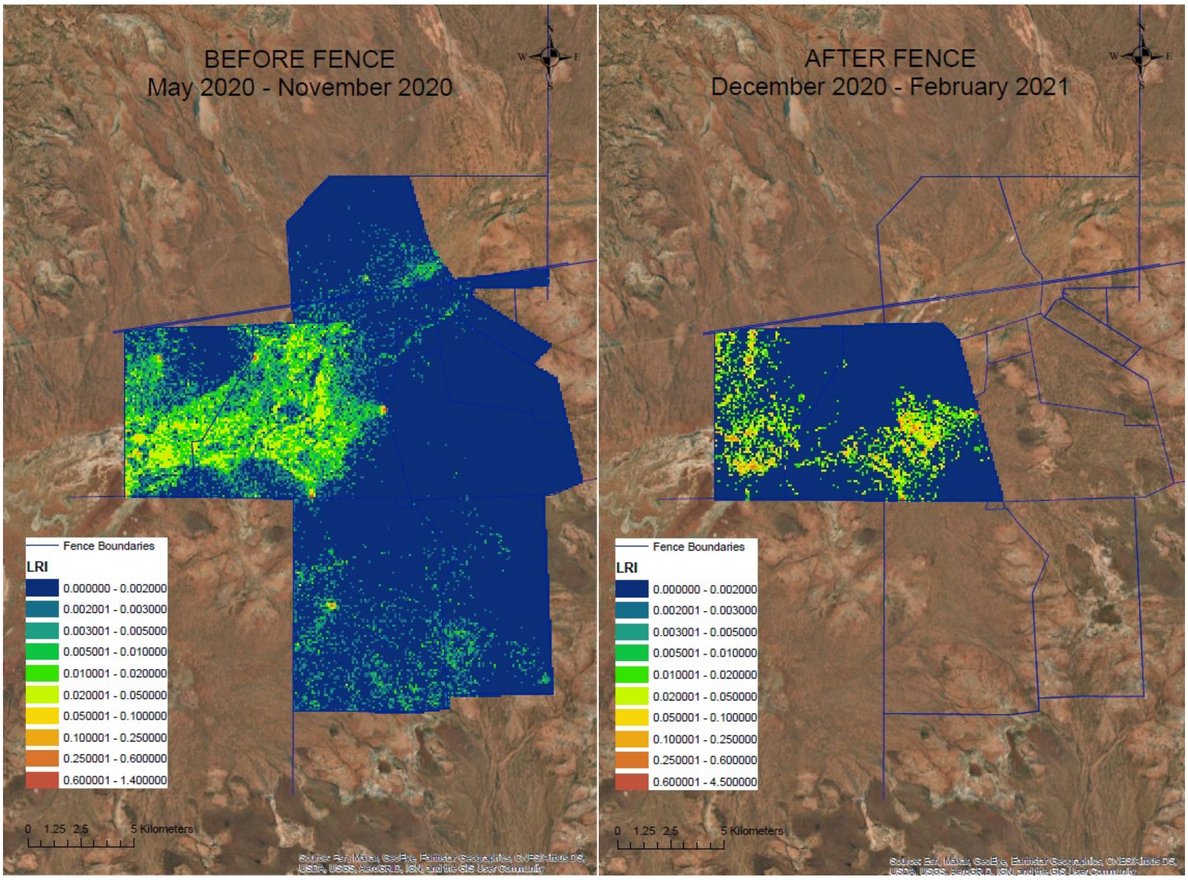
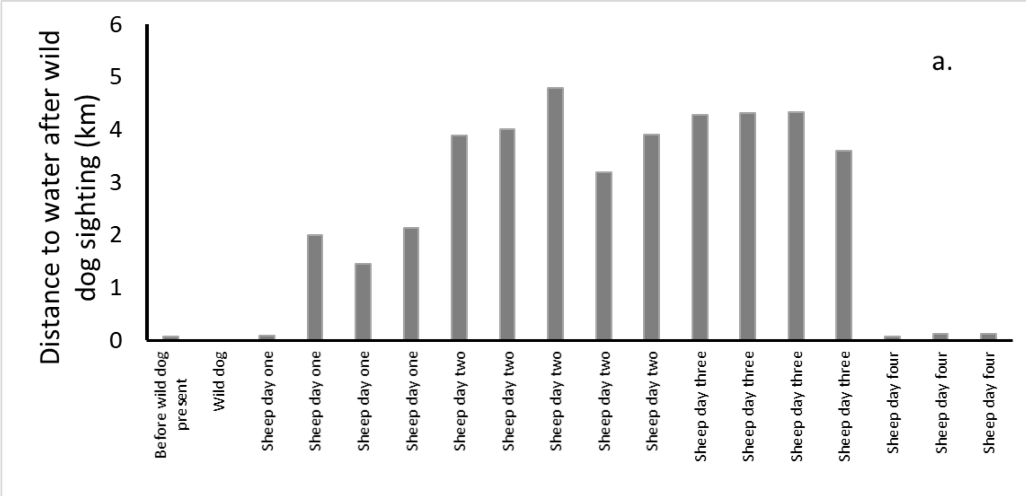


Figure 11. The livestock residency index for the tracked sheep and goats in 2019 and 2020. During 2019 and in 2020; 23 and eight animals were tracked, respectively.

Table 1. The livestock (sheep or goat), sex, days the individual was tracked, distance travelled per day (km) before and after the cell south of the highway was enclosed

			Before		After		
Animal ID	**Species**	**Sex**	**Days**	**Per day**	**Days**	**Per day**	**Time period**
6	Sheep	F	273	0.97			Before
23	Sheep	F	184	1.14			Before
43	Sheep	F	151	1.91			Before
54	Sheep	F	334	0.79			Before
58	Goat	F	365	0.58			Before
60	Sheep	F	365	0.79			Before
72	Sheep	F	151	1.74			Before
74	Sheep	F	365	0.58			Before
76	Goat	F	365	0.79			Before
77	Goat	F	212	1.24			Before
82	Sheep	F	365	0.58			Before
84	Sheep	F	365	0.79			Before
Goat886	Goat	F	183	6.60			Before
Karrot	Sheep	M	122	6.17			Before
Kurly	Sheep	F	153	6.18			Before
Sheep874	Sheep	F	183	1.44	62.00	4.25	Before/After
Sheep875	Sheep	F	183	1.15	62.00	3.39	Before/After
Sheep876	Sheep	F	183	1.58	62.00	4.66	Before/After
Sheep878	Sheep	F	183	1.44	62.00	4.25	Before/After
Sheep882	Sheep	F	183	1.15	62.00	3.39	Before/After
Sheep883	Sheep	F	183	1.58	62.00	4.66	Before/After
Sheep885	Sheep	F	183	1.44	62.00	4.25	Before/After
Goat86400	Goat	F	183	1.15	427.00	0.49	Before/After

Overlaps in wild dogs recorded on camera and the tracking activity of sheep were noted. A wild dog was recorded at a water point on 5 September 2019, one hour after a sheep was recorded from the tracking collars at the same water point. Examining the sheep movement after the wild dog was also recorded at the water point, it took the sheep until 8 September to return to the water point and it was not recorded visiting another water point (Figure 12a). This wild dog was recorded on camera chasing sheep and goats at the time (Figure 12b and c).



b.

c.

Figure 12. (a) Distance the sheep was recorded (from the GPS tracking collar) from the wild dog activity event on camera (at the water point) over four days. The images are the wild dog in question at the water point chasing both (b) sheep and (c) goats.

Another wild dog was recorded on camera on 18 July 2019 at a different water point. A goat (ID 58) was recorded at the same water point four hours after the wild dog was recorded. The goat did not return to the water for another 12 days. The same goat was at the same water point on 5 September 2019 five hours after a wild dog. Following this second potential interaction event, the goat returned to water after four days on 8 September 2019 (Figure 13).

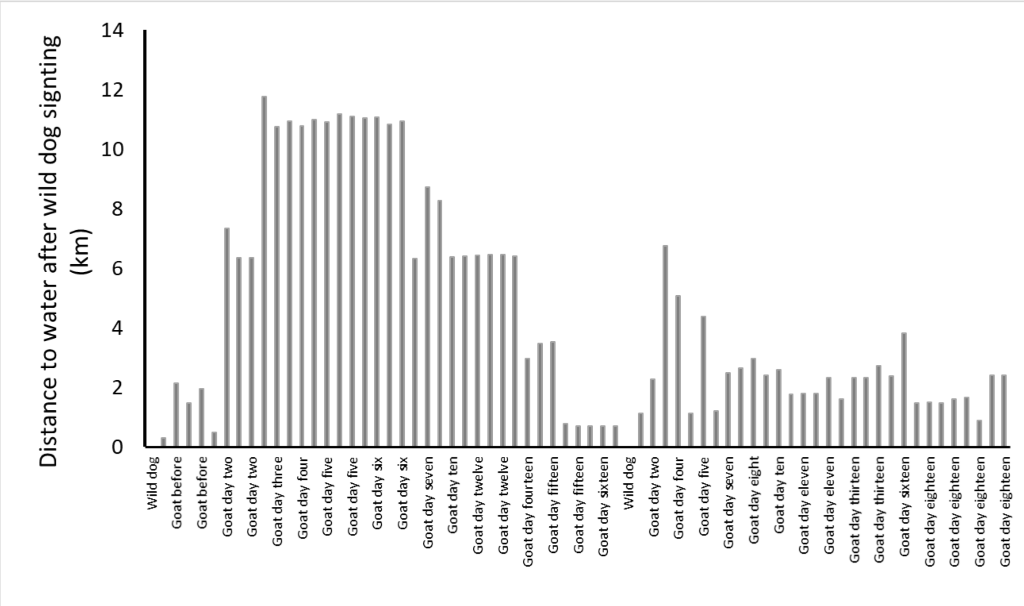


Figure 13. Distance the goat (58) was recorded (from the GPS tracking collar) from the two wild dog activity events on camera (at the water point) over 67 days.

No interactions were recorded between the livestock and wild dogs in 2020 after the cell south of the highway was complete.

DISCUSSION

Fencing for livestock production has been undertaken for many decades in Australia. It demarcates where tools such as traps and baits can be laid to maintain small-stock production through wild dog control. Fencing has been evaluated as a useful tool for keeping out invasive predators (Pacioni et al. 2018a; Pacioni et al. 2021). In this study of changes after cell fencing, we noted that wild dog activity decreased over time, along with increases in emu and juvenile goat activity events recorded on camera. Macropod and sheep activity was stable throughout the project. Feral cat activity did not respond to the change in wild dog activity. However, it should be noted that monitoring water points to estimate activity of these species can be influenced by differential requirements for water by the species being recorded in this arid environment. Sheep travelled further per day after the closure of the cell south of the highway but used a smaller core area based on the observed and calculated livestock residency index. Goat movement per day was reduced after the closure of the cell south of the highway, but the 'after' period only included one goat's movement data. There is evidence that when wild dogs are seen on the camera traps at water points that livestock take many days to return to the water point.

AS THE CELL FENCE IS COMPLETED DO THE ACTIVITY EVENTS OF WILD DOGS REDUCE?

Over the entirety of the project from February 2019 to late 2020, there were 45 wild dog activity events (by 25 individual wild dogs) recorded on camera. The number of individual wild dogs dwindled from 18 to seven during the 26 months of monitoring. The analyses show that, over time, there was a reduction in wild dog activity events on camera for the entire dataset and for wild dog activity events north of the highway. Indeed, of the 45 activity events, 36 occurred in 2019 and only nine in 2020 over the entire cell-fence region. A majority of these wild dog activity events (33/45) were in the completed part of the cell fence (south of the highway) where thousands of sheep also resided. While there was a decrease in wild dog activity events south of the highway (18 in 2019; 7in 2020), this result was not statically significant. Overall, this decrease in wild dog activity events per month demonstrates the strong effort to control wild dogs and remove them from the cell fence through the use of baiting, trapping and shooting by the landholders and by licensed pest management technicians. According to the licensed pest management technicians' records, a total of 48 wild dogs were removed in 2019 and 2020; this does not include the captures by the landholders who are very active in wild dog control. This level of predator removal will likely result in a broadscale extinction of the species within the cell (Lennox et al. 2018).

Pacioni et al. (2021) predicted that within a cell twice the size of the MHC, if one wild dog per 100 km^2 can be maintained (or even achieved in the first place) then the density of wild dogs inside the cell will be 0.03 per 100 km^2. In this study wild dogs commenced at 0.144 km^2 and were reduced to 0.053 km^2 due to the removal effort by landholders and researchers. This final value is slightly higher than modelled by Pacioni et al. (2021) for successful sheep production in the southern rangelands, but it is definitely a good place to start given the incomplete cell fence.

HOW DO THE SMALL STOCK, FERAL CATS, NATIVE AND NON-NATIVE HERBIVORES RESPOND TO THE FENCING AND WILD DOG ACTIVITY EVENTS?

Wild dog predation commonly occurs in sheep paddocks and is a strong driver for lethal control of invasive predators (Allen et al. 2001b; Allen et al. 2013b). Sheep activity events for those residing north of the highway reduced over time where the cell was incomplete and wild dog activity events were common, until the end of this project. Sheep outside the protection of the cell fence did not survive long, most likely due to wild dog predation (Tomlinson 1955; Thomson 1984b). In fact, at the writing of this report the landholders note that no sheep persisted outside the complete part of the cell fence and evidence of wild dog predation events were observed. Sheep residing inside the cell fence had consistent numbers throughout the project (where wild dog activity events were minimal) and then were agisted in early 2021. This is not dissimilar to other studies on predators and livestock; for example, in Montana, Wyoming and Idaho when complete wolf (*Canis lupus*) packs were removed, sheep depredation events were reduced by 79%. Partial pack removal, however, only resulted in a 29% reduction in sheep depredation events (Bradley et al. 2018). This indicates that complete pack removal of predators is required when running small stock, and the success of this hub will only continue if wild dogs are continually excluded from the sheep paddock.

Emu activity events north of the highway decreased over time where the cell was incomplete. Overall, the literature on emus is minimal in Australian research, but emus are commonly recorded in sheep grazing country in Australia (Grice et al. 1985). However, we do know that emus mostly move in groups, but in the cell they were solo or in pairs (as recorded on the camera traps). This would mean that the emus spend more time feeling or detecting predators than if they were in groups and the burden was shared by the individuals (Boland 2003). Emus are commonly preyed on by wild dogs (Caughley et al. 1980a; Pople et al. 2000b), but whether this is population regulation is still unknown.

As the cell fence south of the highway has neared completion, the number of juvenile goat (or kid) activity events have increased. This is most likely a product of a reduction in predation events, as the number of wild dog activity events also decreased over time. An increase in livestock production has been recorded previously where fencing has been installed and predator control undertaken (McKnight 1969; Allen et al. 2001b). The same result cannot be said for juvenile sheep (or lamb) activity events; the lambs were agisted near the end of this dataset, so a change in their activity events could not be observed.

Macropod activity had no relationships with any of the variables. Macropod activity events seemed stable within the cell fence, north of and south of the highway. In late 2019 north of the highway there was an increase in macropod activity events but this did not relate to rainfall. It is difficult to elicit a cause for this increase in activity events from a large landscape, but other potential causes may include a change in food resources or a migration of macropods through an area. Overall, estimating and evaluating macropod number, activity events and density can be a challenging task given the species' ability to migrate long distances (Bayliss 1987) and follow resources over long distances (Newsome 1965; Dawson et al. 2022). This study successfully recorded stable macropod activity from camera traps over a large area, but grazing competition within the cell-fenced area between the livestock and macropods will most likely be an issue in the future (Edwards et al. 1996).

Feral cat activity events remained stable over the project. Wild dog activity events decreased over time and feral cat activity events did not significantly increase or decrease in response. This supports the theory that wild dogs do not suppress feral cat activity or numbers, and the mesopredator release theory (when populations of medium-sized predators rapidly increase in ecosystems after the removal of larger, top carnivores) is not supported in this case (Fancourt et al. 2019; Kreplins et al. 2020), although robust data is required to elicit the threads of this relationship, and requires manipulative data (Hayward et al. 2014).

A limitation to monitoring water points to estimate species' activity in arid rangelands is the boom-and-bust nature of the species populations themselves. Food and water is often provided for livestock and other valuable species for profit, but their use of these resources is not well studied (Armenteros et al. 2021). In the arid southern rangelands of Western Australia all species heavily rely on water for survival (James et al. 1995; James et al. 1999). In the cooler months when rainfall occurs individuals can move further afield for resources and water, compared to the warmer months where a daily visit to the waterpoint is crucial (Dawson et al. 1975). This will impact the ability to monitor species over time. The activity events of goats, sheep, emus and sub-adult sheep (north of the highway only) had a negative relationship with rainfall, most likely a function of being able to move and graze further afield and being recorded on the cameras at each waterpoint. This is more true for livestock than for native species (Leeuw et al. 2001), and potentially the reason behind a lack of a negative relationship between rainfall and macropod and feral cat activity events. Feral cats are known not to rely heavily on water points in different seasons (Paltridge et al. 1997), but to use them for capturing prey all year round (DeStefano et al. 2000). Furthermore, a lack of a relationship with the prey species and wild dog activity events might be a consequence of the difficulty capturing wild dogs and other predators on camera (Kelly et al. 2008). Wild dogs are a cryptic species and, given the incompleteness of the cell fence, wild dog movement might have been more. Camera set-up is very important to evaluating species' activity and numbers (Sun et al. 2014; Newey et al. 2015).

DO THE DAILY SPATIAL MOVEMENTS OF SMALL STOCK INCREASE DUE TO REDUCED WILD DOG PRESENCE BECAUSE OF CELL FENCING?

Sheep movements in this study differ to a similar study in western Queensland investigating sheep and wild dog interactions. Sheep movement per day increased after the cell was complete below the highway. This is contradictory to Evans et al. (2022), who recorded sheep travelled less per day (9.2 km) when wild dogs were absent, and in the presence of wild dogs travel longer distances in a day (11.6 km). Again, the ‘after’ period for the tracking data in this study was during the warm summer months, and it could be assumed that the livestock would travel smaller distances from the water and generally be less active similar to the Evans et al. (2022) study. However, Evans et al. (2022) studied western Queensland where foraging resources and rainfall significantly differ to those in the southern rangelands of Western Australia, indicating that resources are a strong driver for livestock after predation risk. Thomas et al. (2008) investigated the movements of southern rangeland sheep and revealed that sheep adjust their movements and behaviour in warm weather to conserve energy and find more water. It is likely that the sheep in this study travelled further to better grazing areas after the wild dogs were removed from their paddock. Although travelling further each day, the sheep remained in smaller core area – perhaps where shade and water resources were nearby in the warm weather (December 2020 to February 2021) of the tracking months after the cell was complete south of the highway. The reduction in wild dog activity events and threat of predation is likely to increase the animals’ likelihood to travel further afield for better grazing and water points.

Unfortunately, with minimal goat data for after the cell was complete below the highway, it is difficult to determine the changes in goat movements per day (Mayberry et al. 2010).

Visits to water points were also reduced by wild dog presence as seen in Evans et al. (2022) and this study. On several occasions sheep and goats took several days to return to water: four, 16 and 18 days for the livestock to return. Distance of the livestock from the water points increased after a wild dog was recorded on camera traps at the water points. This is a welfare issue resulting from predation pressure on livestock, and indicates again that livestock and wild dogs cannot share a paddock.

CONCLUSION

In Australia there are many vermin, exclusion, cell and predator-proof fences for livestock and biodiversity protection (Pickard 2007). Estimating and evaluating the success of fencing is a difficult task given the wide landscapes of Western Australia and technical difficulties of using camera traps (Meek et al. 2015) and tracking devices. The MHC encompasses 2,600 km^2 of the southern rangelands for livestock production in the absence of wild dogs. This project not only attempted to assess the changes in wild dog activity over time in relationship to sheep and goat activity (i.e. production) but also the changes in other herbivores and predators. Despite this large task, the number of wild dog activity events over the project have decreased.

It is difficult to estimate what impact the reduction in wild dog activity events per month has had on the sheep production of the cell (given the sheep are currently agisted), but where wild dogs could enter the cell fence easily, sheep activity events decreased. Daily movements by the sheep also increased after wild dogs were removed from the complete portion of the cell fence. There is evidence that wild dog activity altered the ability of the sheep to access water. Sheep residing outside the cell did not fare well and likely all were lost to wild dog predation. Juvenile/kid goat activity events increased over time, indicating that there was recruitment in the absence of predation pressure. Going forward, the foremost need is fence maintenance by landholders to ensure wild dog incursions do not occur within the sheep and goat paddock.

Competition for grazing between the livestock and macropods will also become an issue as macropod numbers change with the food on offer and lack of predation pressure (Newsome 1975; Ellis et al. 1977). Macropod numbers remained stable over the project and were unrelated to any of the variables studied, such as rainfall and predation. This is most likely a result of inadequate monitoring methods for macropods (i.e. camera trapping alone), which require aerial surveys for a reliable estimate of numbers.

Alterations to biodiversity within the cell in the absence of the larger predator, the wild dog, seem minimal to date. Feral cat activity per month was unchanged by the reduction in wild dogs, indicating that the mesopredator release hypothesis was not at play here. More positively, the number of feral cats is unlikely to have also risen and negatively impacted the native smaller species present in the cell. Emu activity dropped outside the complete cell fence north of the highway, potentially as a consequence of wild dog activity or other factors.

Overall, this study focused on the early results of the cell fencing enterprise, indicating a positive direction for landholders in wild dog control. Minimising wild dogs initially is a good start; now the maintenance of the low wild dog numbers must continue. As the country is rested, the return of the sheep flock will increase the economic outputs of the cell fencing in the southern rangelands of Western Australia.

APPENDICES

APPENDIX 1. CAMERAS IN THE MURCHISON HUB CELL AND THEIR RUNNING TIME (DAYS) OVER THE DURATION OF THE PROJECT

Month	Feb-19	Mar-19	Apr-19	May-19	Jun-19	Jul-19	Aug-19	Sep-19	Oct-19	Nov-19	Dec-19	Jan-20	Feb-20	Mar-20	Apr-20	May-20	Jun-20	Jul-20	Aug-20	Sep-20	Oct-20	Nov-20	Dec-20	Jan-21	Feb-21
Station A South of Hwy																									
IS130	N/A	N/A	N/A	N/A	N/A	20	31	30	31	30	31	31	29	31	30	31	31	31	31	30	31	30	31	31	28
IS98	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	31	30	0	0	0	13	30	31	31	28
IS152	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	31	30	31	31	31	13	30	31	31	28
North of Hwy																									
IS110	14	31	30	30	30	31	31	4	0	22	0	0	10	31	28	31	30	31	30	30	31	30	31	31	28
IS113	14	31	30	30	30	31	0	30	9	22	0	0	20	31	28	31	30	31	30	30	31	30	31	31	28
IS123	5	0	0	30	30	31	31	30	18	0	0	0	11	9	0	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	28
IS138	14	31	30	30	30	31	31	30	0	11	0	0	29	15	0	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	28
IS118	N/A	N/A	N/A	N/A	N/A	20	31	30	0	0	0	0	0	31	30	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	28
IS102	N/A	N/A	N/A	N/A	N/A	20	31	17	31	30	31	31	29	31	30	31	30	31	31	30	31	30	31	31	28
IS107	N/A	N/A	N/A	N/A	N/A	20	31	17	31	30	31	31	29	31	30	31	30	31	31	30	31	30	31	31	28
IS119	14	31	30	30	30	31	31	30	31	30	31	31	31	31	30	31	30	31	31	30	31	30	31	31	28
IS125	14	31	30	30	30	31	31	30	31	30	31	31	31	31	30	31	30	31	31	30	31	30	31	31	28
IS130	10	0	0	0	0	0	31	30	31	30	31	31	31	31	30	31	30	31	31	30	31	30	31	31	28
IS140	15	0	0	0	0	0	31	30	31	30	31	31	31	31	30	31	30	31	31	30	31	30	31	31	28
Station B South of Hwy																									
IS112	N/A	N/A	N/A	N/A	N/A	21	31	30	31	30	31	31	29	31	30	31	30	31	31	30	31	30	31	31	28
IS104	12	0	0	20	0	31	31	21	0	6	0	0	18	18	30	31	30	31	31	30	31	30	17	31	28
IS139	7	0	30	31	30	31	31	30	31	13	12	0	0	31	30	31	30	31	31	30	31	30	31	31	28
IS135	10	0	0	31	30	31	31	30	31	30	31	31	29	31	30	31	5	31	31	30	31	30	31	31	28
IS96	4	0	0	31	30	31	31	30	21	12	0	0	11	31	30	31	5	31	31	30	31	30	31	31	28
IS155	18	31	30	31	30	31	22	30	31	30	12	31	18	31	30	31	5	31	31	30	31	30	14	31	28
IS133	2	0	0	31	30	31	31	15	0	8	0	0	11	31	18	31	26	31	1	30	10	30	31	31	28
IS132	4	0	0	21	30	31	31	10	0	20	0	0	29	1	0	0	0	31	31	30	7	30	11	0	0
IS129	3	0	30	21	0	31	31	30	1	8	0	0	29	31	28	31	30	15	31	30	31	30	4	0	0
IS127	11	0	0	5	0	31	31	30	12	0	0	0	0	31	30	31	30	31	31	30	31	30	31	31	28
IS111	1	0	0	0	0	0	31	17	0	2	0	0	29	3	30	31	30	31	31	30	31	30	31	31	28

Month	Feb -19	Mar -19	Apr -19	May -19	Jun -19	Jul-19	Aug -19	Sep -19	Oct -19	Nov -19	Dec -19	Jan -20	Feb -20	Mar -20	Apr -20	May -20	Jun -20	Jul-20	Aug -20	Sep -20	Oct -20	Nov -20	Dec -20	Jan -21	Feb -21
IS108	1	31	30	31	30	31	31	20	0	2	0	0	18	3	30	30	30	31	31	30	5	0	2	0	0
IS128	18	31	30	0	0	0	31	10	6	17	12	0	29	31	28	31	3	31	31	30	31	30	19	0	0
IS120	7	31	30	20	0	21	31	9	0	7	0	0	29	30	0	31	3	31	31	30	3	0	19	0	28
IS131	8	0	30	11	0	0	24	3	0	2	0	0	29	31	30	31	0	15	0	0	6	30	31	31	28
IS126	11	0	30	12	0	31	5	0	0	0	0	0	0	31	30	31	30	31	31	30	31	30	3	31	28
IS137	3	0	0	0	0	16	0	0	0	2	0	0	0	0	3	1	5	0	0	0	1	30	21	0	0
North of hwy																									
IS141	18	31	30	0	0	0	0	0	0	13	0	0	13	31	30	31	0	0	0	0	16	30	31	0	0
IS122	1	0	30	31	1	31	22	0	0	4	0	0	0	31	30	31	30	31	31	30	31	30	31	31	28
IS109	1	0	5	0	0	31	22	7	0	0	0	0	0	31	30	31	30	31	31	9	3	0	31	31	28
IS136	N/A	N/A	N/A	N/A	N/A	N/A	N/A	30	31	30	31	31	29	31	30	31	30	16	31	24	30	30	31	31	28
IS149	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	31	30	31	31	29	31	30	31	30	16	31	30	30	30	31	9	0

APPENDIX 2. GENERALISED LINEAR MODELS FOR WILD DOGS, SHEEP, GOATS, MACROPODS, EMUS AND FERAL CATS ACTIVITY EVENTS PER MONTH (COMBINED DATA, NORTH AND SOUTH OF THE HIGHWAY) AND THE AGE CATEGORIES OF SHEEP AND GOATS (ADULTS, SUB-ADULTS AND JUVENILES) WITH RAINFALL, RAINFALL IN THE PRECEDING MONTH, MONTH AND WILD DOG ACTIVITY EVENTS PER MONTH

Independent variables	**Rainfall (mm)**	**Rainfall in the preceding month (mm)**	**Month (1–26)**	**Wild dog activity events per month**
Wild dogs	β = –0.05, p = 0.40	β = –0.009, p = 0.16	**β = –0.04, p = 0.004**	N/A
Sheep	β = –0.02, p = 0.41	β = –0.03, p = 0.28	β = –0.05, p = 0.30	β = –0.31, p = 0.62
Goats	β = –0.02, p = 0.28	β = –0.04, p = 0.15	β = 0.04, p = 0.30	β = –0.36, p = 0.64
Macropods	β = –0.04, p = 0.28	β = –0.07, p = 0.09	β = –0.07, p = 0.38	β = –0.89, p = 0.45
Emus	β = –0.02, p = 0.26	β = –0.01, p = 0.44	β = 0.009, p = 0.81	β = –0.38, p = 0.51
Feral cats	β = –0.01, p = 0.31	β = –0.01, p = 0.15	β = –0.01, p = 0.65	β = 0.35, p = 0.23
North the highway				
Wild dogs	β = –0.005, p =0.31	β =–0.009, p = 0.08	**β =–0.02, p =0.02**	N/A
Sheep	β = –0.01, p = 0.22	β =–0.01, p = 0.40	**β =–0.07, p =0.02**	β = 0.40, p = 0.34
Goats	β = –0.01, p = 0.61	**β =–0.04, p = 0.02**	β =0.04, p =0.21	β = –0.36, p = 0.50
Macropods	β = –0.02, p = 0.64	β =–0.05, p = 0.19	β =–0.05, p =0.53	β = –0.42, p = 0.20
Emus	**β = –0.02, p = 0.004**	β = 0.006, p = 0.43	**β =–0.03, p =0.01**	β = –0.04, p = 0.81
Feral cats	β = 0.003, p = 0.50	β = –0.001, p = 0.83	β = 0.01, p = 0.22	β = –0.03, p = 0.79
South of Highway				
Wild dogs	β = –0.001, p =0.70	β = –0.0006, p = 0.89	β = –0.01, p =0.11	N/A
Sheep	β = –0.01, p =0.80	β = 0.001, p = 0.89	β = –0.02, p =0.23	β = 0.23, p =0.32
Goats	β = –0.01, p =0.07	β = –0.003, p = 0.63	β = 0.004, p =0.76	β = –0.009, p =0.96
Macropods	β = –0.01, p =0.25	β = –0.02, p = 0.09	β = 0.01, p =0.70	β = 0.17 p =0.65

Independent variables	Rainfall (mm)	Rainfall in the preceding month (mm)	Month (1–26)	Wild dog activity events per month
Emus	β = –0.004, p =0.82	β = –0.02, p = 0.12	β = 0.05, p =0.18	β = –0.49 p =0.32
Feral cats	β = –0.01, p = 0.26	β = –0.01, p = 0.23	β = –0.01, p = 0.34	β = 0.38, p = 0.23
Sheep				
Adult	β = –0.01, p = 0.10	β = 0.007, p = 0.45	β = 0.07, p = 0.74	β = 0.08, p = 0.74
Sub-adult	**β = –0.01, p = 0.03**	β = 0.009, p = 0.26	β = 0.03, p = 0.06	β = –0.13, p = 0.58
Juvenile	β = –0.005, p = 0.60	β = 0.01, p = 0.06	β = 0.02, p = 0.37	β = 0.16, p = 0.54
Goats				
Adult	β = –0.01, p = 0.10	β = –0.001, p = 0.88	β = 0.01, p = 0.50	β = 0.13, p = 0.46
Sub-adult	β = –0.01, p = 0.10	β = 0.001, p = 0.92	β = 0.01, p = 0.31	β = 0.10, p = 0.63
Juvenile	β = –0.01, p =0.06	β = –0.001, p =0.89	**β = 0.03, p = 0.03**	β =0.10, p = 0.65

APPENDIX 3: DATA-TRACKING OVERVIEW FOR EACH TRACKING COLLAR DEPLOYED ON SHEEP AND GOATS IN THE MURCHISON HUB CELL

Animal ID	Species	Sex	Time period	Deployed	Removed
6	Sheep	F	Before	13/02/2019	28/02/2020
23	Sheep	F	Before	13/02/2019	28/02/2020
43	Sheep	F	Before	13/02/2019	28/02/2020
54	Sheep	F	Before	13/02/2019	28/02/2020
58	Goat	F	Before	13/02/2019	28/02/2020
60	Sheep	F	Before	13/02/2019	28/02/2020
72	Sheep	F	Before	13/02/2019	28/02/2020
74	Sheep	F	Before	13/02/2019	28/02/2020
76	Goat	F	Before	13/02/2019	28/02/2020
77	Goat	F	Before	13/02/2019	28/02/2020
82	Sheep	F	Before	13/02/2019	28/02/2020
84	Sheep	F	Before	13/02/2019	28/02/2020
Goat886	Goat	F	Before	13/02/2019	18/01/2021
Karrot	Sheep	M	Before	3/06/2020	18/01/2021
Kurly	Sheep	F	Before	3/06/2020	18/01/2021
Sheep874	Sheep	F	Before/after	3/06/2020	18/01/2021
Sheep875	Sheep	F	Before/after	3/06/2020	18/01/2021
Sheep876	Sheep	F	Before/after	3/06/2020	18/01/2021
Sheep878	Sheep	F	Before/after	3/06/2020	18/01/2021
Sheep882	Sheep	F	Before/after	3/06/2020	18/01/2021
Sheep883	Sheep	F	Before/after	3/06/2020	18/01/2021
Sheep885	Sheep	F	Before/after	3/06/2020	18/01/2021
Goat86400	Goat	F	Before/after	3/06/2020	2/02/2022

CHAPTER 2. ARE CANID PEST EJECTORS AN EFFECTIVE CONTROL TOOL FOR WILD DOGS IN AN ARID RANGELAND ENVIRONMENT?

Published in the journal *Wildlife Research*.

INTRODUCTION

Wild dogs (dingoes, free roaming dogs and their hybrids; *Canis familiaris*; Fleming *et al.* 2014; Jackson *et al.* 2017) can have significant negative impacts on livestock and native animals in Australia (Eldridge *et al.* 2002; Allen 2014; Ecker *et al.* 2016). Financial impacts of wild dogs on livestock production exceed $89M per annum nationally (McLeod 2014) and are most pronounced for sheep, which do not have suitable defensive responses to wild dog predation and thus cannot tolerate their impacts (Thomson 1984, Allen and Sparkes 2011, Fleming et al 2014).

Toxic baiting using sodium fluoroacetate (1080), integrated with other control tools such as shooting and trapping is advocated as best practice management of wild dogs (Allen 2017). Toxic baiting is the most commonly used control method for many landholders (Binks et al 2014). It has the advantage of being a relatively cost-effective and time-efficient practice that allows for the control of wild dogs across large areas (Allen 2017). Further, many native Australian species have a tolerance to 1080 (Twigg *et al.* 2010). This means that baiting can be undertaken in most areas with limited risk to non-target native species (Eastman *et al.* 1988; Claridge *et al.* 2007; Buckmaster *et al.* 2014).

While baiting, either via aerial (Ballard *et al.* 2020) or on-ground (Thomson *et al.* 2000; Marlow *et al.* 2015) deployment is a commonly used technique it does have some shortcomings. These include consumption or removal of baits by non-target species before the target species, which can reduce baiting effectiveness (Dundas *et al.* 2014; Kreplins *et al.* 2018a). Leaching, and insect, microbial and fungal action is known to reduce the amount of 1080 in baits over time (Fleming & Parker, 1991; Gentle & Cother, 2014; McIlroy et al., 1988), although this is primarily of concern in mesic environments (Twigg et al 2001).

Canid pest ejectors (CPEs), developed from a device known as M44 ejectors (Robinson 1943; Marks *et al.* 1999; Marks *et al.* 2003) and ‘coyote getters’ (Robinson 1943) are static mechanical devices for toxin deployment. CPEs are comprised of a ground stake in which a spring loaded piston and ejector unit reside. The lure head sits upon the piston and contains a sealed capsule of toxin. CPEs are inserted into the ground stake beneath the dirt, and the lure head (with toxin capsule) protruding at earth level. The toxin can be either 1080 or Para-aminopropiophenone (PAPP). Once the lure head is pulled upwards the CPE is triggered to puncture the capsule and ejects toxin into the mouth of the animal pulling on the device. These devices are target-specific, as only wild dogs and foxes have the required upward-force to trigger them (Animal Control Technologies Australia Pty Ltd 2017; Young *et al.* draft) and they are fixed to the ground, preventing removal by non-target species. CPEs offer the potential of eliminating the problem of non-target inference during baiting programs, and for mesic environments, the rapid degradation of 1080.

Canid pest ejectors can be fitted with a range of lures for the target species. This provides opportunities for land manages to identify and select lure heads with maximum effectiveness. Lures such as canid faeces and urine (Mitchell *et al.* 1992), food lures (Saunders *et al.* 2000), synthetically constructed lures (Fleming 1996; Brawata *et al.* 2011) have been trialled to enhance the response of canids to a variety of tools such as traps, baits camera traps and CPEs. At present there is very little information available on lures types for canid control and monitoring programs.

The southern rangelands of Western Australia has historically been a wool production area, however economic challenges, including the need to control wild dogs has reduced small stock in the region markedly since the 2000's (Foran *et al.* 2019). Producers in the region are dependent on effective wild dog control to return to small stock production. Recent work indicates that uptake of dried meat baits by wild dogs, and baiting effectiveness, in the southern rangelands of WA can be unexpectedly low due to a high level of interference of baits by non-target species (Kreplins *et al.* 2018a; Kennedy *et al.* 2021a). Here we aim to determine if CPEs are an effective tool for controlling wild dogs in the southern rangelands of Western Australia through reduced interference by non-target species. We also seek to investigate the effectiveness of a range of different CPE lure heads in increasing wild dog interactions with CPEs.

METHODS

SITE DESCRIPTION

This work was conducted on three properties in the southern rangelands of Western Australia. All three properties have a history of running small stock (sheep and/or goats) and long-term wild dog control. Across the 320, 926ha there are unmanaged goats and some small stock present. The area is typified by an arid environment, with a mean annual rainfall of 239.1 mm, and mean maximum temperatures in January reaching 38.2°C (Mount Magnet Station, 007057; Bureau of Meteorology 2020). The habitat is composed primarily of *Acacia* spp. woodlands.

CANID PEST EJECTOR DEPLOYMENT

Autumn and Spring are the periods of greatest wild dog activity and when most wild dog control commonly occurs (Thomson 1986b) so bursts of control using CPEs were trialled in these periods. One hundred CPEs (Animal Control Technologies, Somerton, Victoria, Australia) were deployed for periods of two months during autumn 2018, spring and autumn 2019 and spring 2020, equating to four sessions of CPE deployments (sessions 1-4). CPE sessions were split into two sequential month periods, denoted as 'a' and 'b' (dates of deployment and servicing described in Table 1).

The first three CPE sessions used 6mg 1080 capsules and the final session trialled 1000mg Para-aminopropiophenone (PAPP) capsules (Animal Control Technologies Australia). Twenty CPEs were modified to have a 30cm x 3cm auger end; Figure 2). This design allowed for use in soft sand on which wild dog like to travel on, on roads (unpublished data Kennedy and Kreplins). Canid pest ejectors were deployed on property vehicle tracks in transects of 20km with CPEs spaced 1km apart and serviced at monthly intervals. Some of these transects were moved slightly between sessions but remained within a home ranges of a wild dog for Western Australia (22.2 km2; Thomson 1992b; Figure 1). These changes were due to the erection of wild dog-proof fencing for protection of small stock.

Table 1. Dates of canid pest ejector deployment, servicing (CPEs and camera traps) and removal during the four sessions of control at three properties in the southern rangelands of Western Australia in 2018 to 2020.

Session of control	Lures used	Date deployed	Date serviced	Date removed
1	• dried meat bait • fish oil lure (felt soaked)	25-27 September 2018	17-18 October 2018	20-21 November 2018
2	• animal fat (felt soaked) • horse hoof oil (felt soaked)	19-21 March 2019	23-25 April 2019	21-22 May 2019
3	• synthetic fermented egg (felt soaked) • dried liver treat/PVA combination	17-19 September 2019	16-18 October 2019	12-13 November 2019
4	• government call (felt soaked) • vanilla essence (felt soaked)	17-18 March 2020	29-30 April 2020	4-6 June 2020

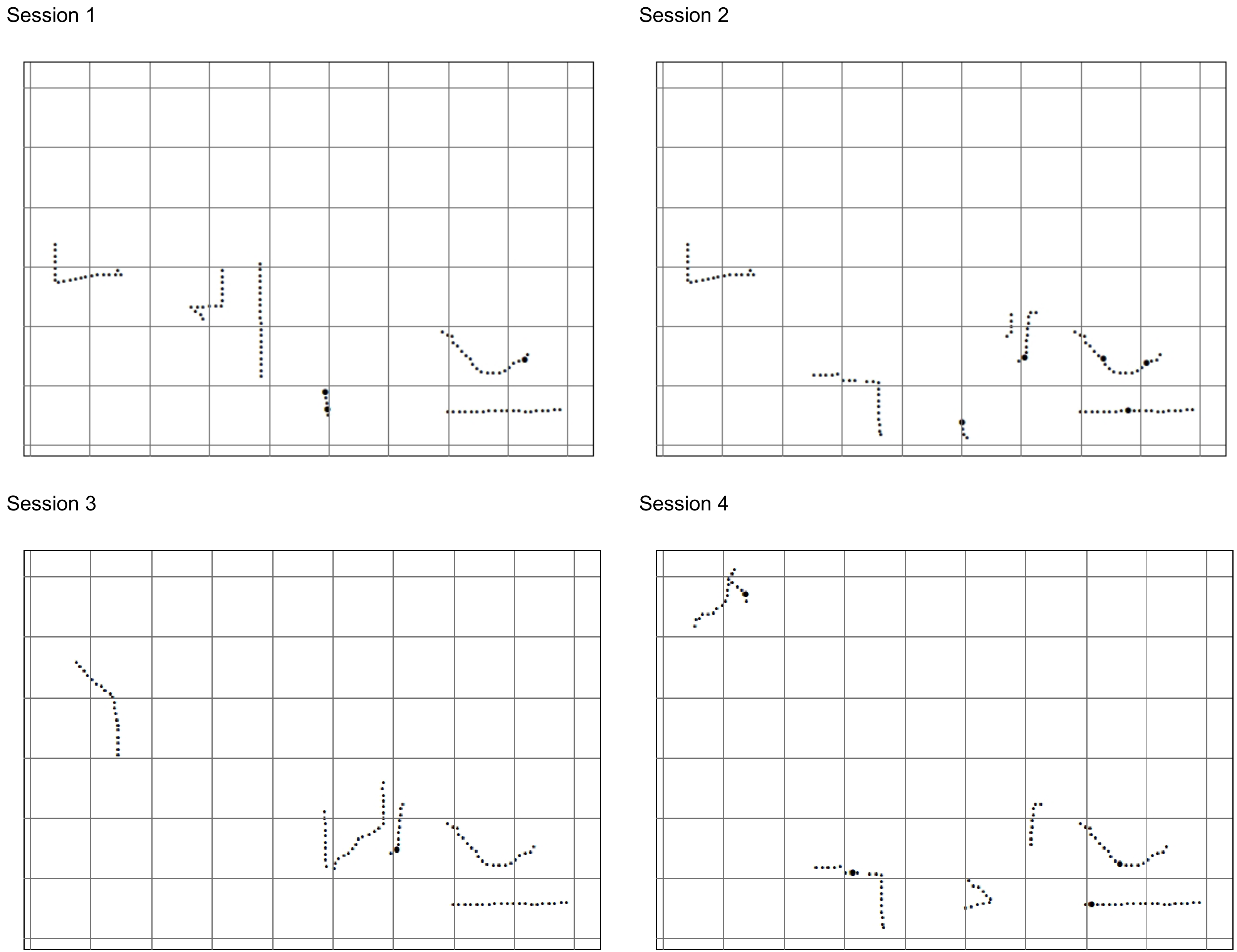


Figure 1. Locations of the canid pest ejectors and camera traps during each of the four sessions in the southern rangelands of Western Australia. Each dot represents a canid pest ejector and camera, spaced 1km along each transect, each square is 10x10km. Highlighted dots are the locations of CPE 'fires'.

A total of eight different lure heads were trialled, with two lures trialled each session (Table 1, Figure 1). During session 1 (Spring 2018) commercially available kangaroo dried meat bait (DMB; Animal Control Technologies Australia; n=50) and felt soaked in fish oil lure (Fish Oil Mate, Wangara, WA; n=50) were used. In session 2 (Autumn 2019) felt soaked in either animal fat (beef and lamb fat from meat scraps; n=50) and horse hoof (hoof trimmings boiled down into a liquid; n=50) were utilised. In session 3 (Spring 2019) a combination of dog dried liver treats (Schmackos Strapz beef flavour; Mars, Raglan, NSW) and PVA glue (Gorilla Glue; Sharonville, Ohio; n=50) and synthetic fermented egg on felt (SFE; Wildlife Control Supplies, East Granby, CT, US; n=50) were trialled. In the last session, (Autumn 2020) O'Gorman's Government Call on felt (synthetic lure; O'Gorman, Broadus, MT, US; n=50) and vanilla essence on felt (Queen, Imitation Vanilla, Alderley, Queensland; n=50) were deployed on the CPE lure heads. All the soaked felt lure heads were made of either red, yellow or blue felt and attached to the CPE head with zip-ties. For each session fifty of each lure head were used. Two different lures heads were deployed for each session, totalling eight different lure heads for the entire trial. Within in each session every second lure head was the same. Each lure head was in the field an entire session (two months) and changed at the CPE servicing event (every month).

a)

b)

c)

d)

Figure 2. Images of a) the modified CPEs for soft ground with an auger end, b) dried meat bait lure heads, c) felt soaked lure head (used for several different lures trialled including, fish oil, animal fat, horse hoof, synthetic fermented egg, Government call and vanilla essence), and d) dried liver treat/PVA glue combination.

MONITORING CPE DEPLOYMENT

Each CPE was monitored with an infrared camera trap Reconyx™ H500 ™ (Reconyx, Holmen, WI) or Scoutguard (Scoutguard, China). Camera traps were deployed approximately 5m from the CPE at a height of 0.5m. 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, cameras took a burst of three images with no time delay between images.

All images were viewed with FastStone image viewer for Windows (FastStone Soft, 2019) program and the station, session of control date, time, temperature (°C), species, behaviour of the individual species, age, sex and lure head type (including the colour of the felt if used) were recorded. Individual wild dogs were identified independently by two researchers (TK and MK) as recommended by Kelly et al. (2008) and Kennedy et al. (2021a).

Activity events of non-target species including sand monitors (*Varanus gouldii*), corvids (*Corvus* spp.), emus (*Dromaius novehollandiae*), kangaroos (*Macropus fuliginosus* and *M. robustus*), small stock (sheep, *Ovis aries*; unmanaged goats, *Capra hircus*), wild horses (*Equus caballus*) and feral cats (*Felis catus*) were also recorded. For individuals of the same species, a separate capture event was recorded if the images were captured at least 10 minutes apart (Kreplins *et al.* 2021). If two or more individuals of the same species were seen on camera at the same time, they were classified as multiple individuals (one capture event per individual).

Interest in a CPE by wild dogs or non-target species was recorded as more than 5 seconds spent sniffing, mouthing, rolling, playing with or licking, but not 'firing' a CPE. A fired CPE was recorded as a pull on the device where the toxin capsule was discharged.

STATISTICAL ANALYSIS

We calculated the densities of wild dogs during each month (four sessions 1-4,a and b) of CPE trials using spatially-explicit capture-recapture analyses using the *secrlinear* package (Efford 2017; Efford 2020) in R (R Core Team 2019). Using the R package *secrlinear* (Efford 2017), a combination of the state (animal home range) and observations (probability of detecting an individual at a detector, i.e. camera, in relation to the individual's home range) are used to construct models, with the assumption that the wild dog population was closed for each month period. *secrlinear* was used rather than simply *secr* given the camera trap deployment along roads only; therefore the estimation of wild dog density along a linear habitat is expressed per km instead of the number of individuals per unit area. Density of wild dogs was estimated across transects on the three properties. All models were derived from the Cormack-Jolly Seber or Jolly Seber models.

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 km). Each model included the parameters:

g0 – 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, g0 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 g0 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.

Data from the three properties was combined for analysis not only as wild dogs can transverse across all three properties, but there would have been insufficient data to analyse the properties individually. A linear mask was constructed with a buffer of 1 km from each camera using poly line shape files 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, indicating that wild dogs use the tracks for moving around but their movement is not solely restricted to the tracks. Akaike Information Criterion adjusted for small sample size (AICc) was used to rank models. Only models with ΔAICc <2 are shown and dealt with further (these models have the greatest likelihood of all the model-set to be the best model fit to the data) (Burnham *et al.* 2002). AICc weights (w_i) were calculated for these top models as a proportion of all models tested.

A general linear model was performed to determine if the density of wild dogs changed over the session of CPE control in R (R Core Team 2019). Predictor variables included the control session (1-4 a and b) and the density of wild dogs.

We used Pearson's Chi-square tests to compare differences between each control session of: number of wild dog observations on camera (expected values were 48.25), the number of wild dog CPE fires (expected values were 3.25), and the number of wild dogs showing interest in a CPE but not firing the CPE (expected values were 10.5).

We used Pearson's Chi-square tests to compare CPE lure head fires and interest shown towards each lure head type by wild dogs; expected values were calculated assuming that lure heads were fired on 1.62 occasions and interest was shown 5.25 time by wild dogs. Pearson's Chi-square tests were used to determine if wild dogs fired CPEs with a particular lure on felt colour (red, blue and yellow) or lure type (fish oil, animal fat, horse hoof, SFE, vanilla and Gov call) more often; expected values were calculated as 4.3 CPE fires.

Pearson's Chi-square tests were used to compare differences between each control session of non-target species interest of CPEs but not firing a CPE (expected values proportional to the number of activity events). Pearson's Chi-square tests were also used to compare interest shown towards each lure head type by non-target species; expected values were calculated assuming that interest was shown 21.62 time by non-targets. All Pearson's Chi-square tests were performed in Excel (Microsoft).

RESULTS

WILD DOG DENSITY

The density of wild dogs during the eight months of canid pest ejector trials varied from –0.38 (± 0.26 SE) to –0.11 (± 0.04 SE) wild dogs per linear km (or 38 to 11 wild dogs per 100km^{-1}; $F_{1,14}$=4.6, P=0.0002; Figure 3). Wild dog density was lower during the second month of each session of CPE control (percentage decrease in density for session; -46%, -5%, 13%, -38%).

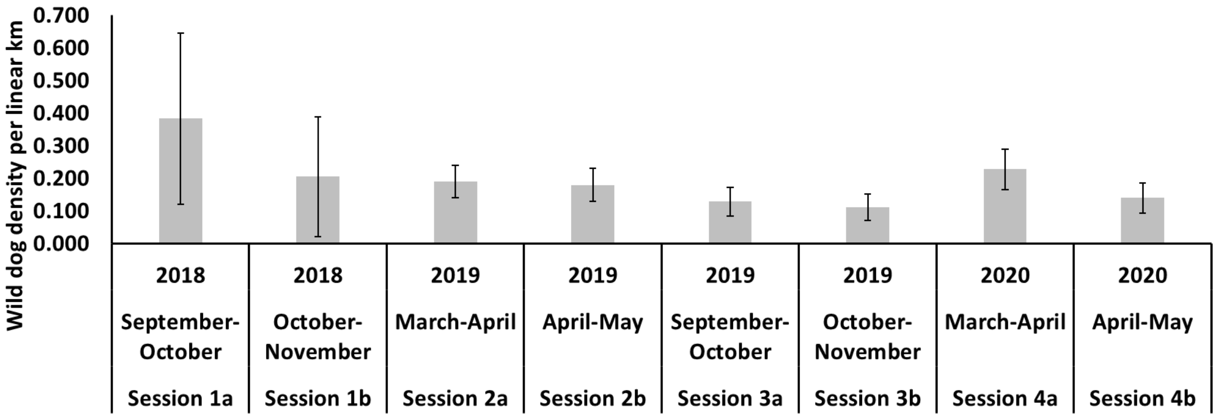


Figure 3. Wild dog density (± SE) on the three properties during the four sessions (two months, each month is split into a and b) of canid pest ejector trials in the Southern Rangelands, Murchison region, Western Australia.

CANID PEST EJECTOR FIRES BY WILD DOGS

There were differences between control sessions in the number of wild dogs captured on camera (χ_1= 44.11, P<0.05; Table 2). During each session the number of wild dog captures on camera varied from 20, 84, 41, to 48 over the four sessions of control consecutively. Session two (March-May 2019) recorded the highest number of wild dogs activity events on camera but a large number of these were the same individuals appearing on several cameras (e.g. one individual was captured in 8 activity events). The peak in activity events in Session 2 corresponded with the highest number of wild dog firings of CPEs for the entire trial (n=5). During session 1, 3 and 4, another 3, 1, and 4 fires by wild dogs were recorded respectively. On average it took 9.3 days (range 1-28 days) from deployment or CPE servicing to a wild dog firing the CPE. The number of CPE fires by wild dogs did not significantly differ between control sessions (χ_1= 2.69, $P=0.44$; Table 2) but overall wild dogs fired more CPEs than expected based on the proportion of wild activity events on camera (χ_1= 417.05, P<0.05; Figure 4). No other species apart from a wild dog fired a CPE. There were a number of occasions where wild dogs showed interest in the CPEs but did not trigger the mechanism. These occasions increased from each session; 1, 12, 13, and 16 (χ_1= 12.29, P<0.05; Table 2).

LURES FOR WILD DOGS

None of the eight lures trialled on the CPE lure heads were significantly better than another in eliciting wild dog triggering of CPEs (χ_1= 7.30, P=0.40). Food-based lures, including the DMB and the ground liver treat and PVA combination) did not result in a CPE fire during the study. All the lures placed on the felt resulted in at least one fire by a wild dog. Fish oil, horse hoof and Gov call all resulted in three fires by wild dogs. Animal fat based lures on the felt resulted in two fires and the SFE and vanilla essence resulted in a single fire by a wild dog (Table 3). It is interesting to note that in contrast to felt lure heads, on four occasions wild dogs showed interest in, but never fired a CPE with a food-based lure head.

Wild dogs demonstrated differences in their level of interest between the lure heads (χ_1= 16.66, P=0.01). Interest by wild dogs decreased in records from vanilla (n=10), SFE (n=9), horse hoof (n=7), Gov call (n=6), animal fat (n=5), dried liver treat (n=4), fish oil (n=1) and DMB (n=0; Table 3). There was a difference in numbers when comparing the interest in food based lures (n=4) and scent based lures (n=38).

Wild dogs did not prefer a particular felt colour (red, yellow and blue) for scented lure heads (fish oil, animal fat, horse hoof, SFE, vanilla or Gov call; χ_1= 58.69, P=0.35)

NON-TARGET SPECIES

The number of individuals of non-target species demonstrating interest in the CPEs during the trial varied for each lure type (χ_1= 31.44, P<0.05) and session (χ_1= 40.50, P<0.05; Table 2 and 3; Figure 4). The species demonstrating this interest (but not firing CPEs) were feral cats (χ_1= 1347.41, P<0.05), emus (χ_1= 157.75, P<0.05), macropods (χ_1= 6.1, $P=0.01$), corvids (χ_1= 1636.37, P<0.05), small stock (sheep and goats; χ_1= 149.72, P<0.05) and varanids (χ_1= 1349.24, P<0.05; Figure 4, Table 3). A single wild horse was observed to be interested in a CPE.

UNAVAILABLE CPES

A number of CPEs were rendered unavailable to wild dogs as a non-target species had pulled the lure head or piston out of the CPE capsule. The lure heads that were present during these few occasions were; DMB (n=46), fish oil (n=6) and animal fat (n=8) (Table 3). Of the forty-six occasions where the DMB lure heads were rendered unavailable, fourteen of these were by varanids, two CPEs were pulled apart by Corvid *spp*. and another two occasions the DMB lure head was consumed by ants. The remainder of the occasions the species interfering with the CPE was unknown. Two of the fish oil lures were chewed by varanids and another four interfered with by an unknown species. Of the seven animal fat lure heads that were rendered unavailable to wild dogs, four were run over by vehicles, one had the felt removed by a corvid and another two had the felt removed by a wild dog but the CPE was not fired.

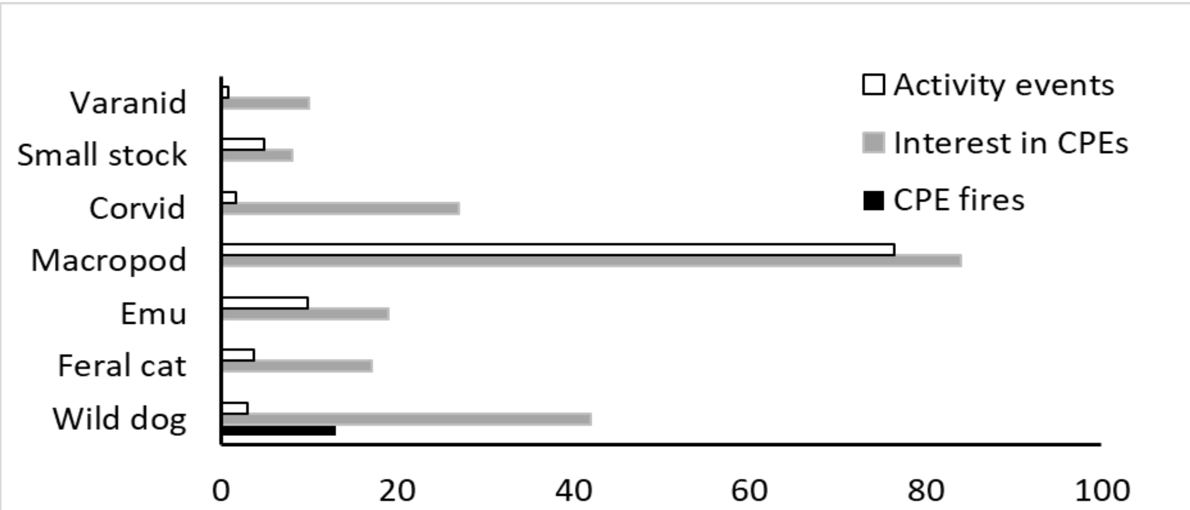


Figure 4. Activity events, interest in canid pest ejectors and fires of CPEs over the entire monitoring period for seven non-target species and wild dogs. Each species' activity is a percentage of the total 6572 activity events (of all species) seen on camera. Interest in CPEs and CPE fires are raw values.

DISCUSSION

Canid pest ejectors are a potentially useful tool for wild dog management in Australia and canid control globally. This paper contributes to the limited amount of literature available on the use of CPEs for canid control. In the harsh conditions of the southern rangelands of Western Australia the CPEs deployed during this study reduced the wild dog population during each session despite high temperatures, multiple rainfall events and livestock trampling. Indeed the use of CPEs led to a more marked reduction in wild dog density per linear km than has been reported for baiting in the same landscape (Kennedy *et al.* 2021a). CPEs allow a range of novel lures to be used and, when serviced monthly, could be integrated with other control tools to improve canid control. Altering the CPEs for soft substrate (i.e. auger CPE capsule into the ground) meant the CPEs could be deployed into a wider range of locations than the original design allows.

For wild dog control to be effective over time it needs to annually reduce approximately 75% of the wild dog population (pacioni; Hone *et al.* 2010). Detecting reductions in wild dog populations as a result of control (i.e. baiting, trapping, fencing and shooting) can be challenging, and where these tools have been evaluated the outcomes can vary considerably (e.g. McIlroy *et al.* 1986a; Thomson 1986b; Fleming *et al.* 1996; Ballard *et al.* 2020). There are a number of factors which can contribute to this variation, including environmental factors such as season, prey availability, behaviour of wild dogs and non-target species (McIlroy *et al.* 1986b; Thomson 1986b; Kreplins *et al.* 2018a). Operational factors such as type of control, method and rate of deployment, and monitoring methods can also contribute to variation in control effectiveness (Saunders *et al.* 2007; Allsop *et al.* 2017; Fancourt *et al.* 2021). During the eight months of CPE canid control in this study a total of thirteen CPEs were triggered by wild dogs. Over the same period we found a modest decline in wild dog density during each session (-5% to -46% changes in wild dog density per linear km). We attribute the decreases in density to mortality due to CPE uptake, although, note the monitoring approach and behavioural responses of wild dogs to human activity (i.e. fence and road construction, vehicle movements, livestock management) may have influenced density estimates.

Factors which may have contributed to the changes in density we detected include seasonal changes in wild dog activity, use of an appropriate spatial scale and limitations of our monitoring methods; namely the ability of camera traps to detect intelligent species (Kreplins *et al.* 2018b; Pacioni *et al.* accepted). To address these challenges we used bursts of control coupled with monitoring targeted to reduce the likelihood of learned aversion to the control and monitoring tools. We targeted the control bursts for short periods during times of peak wild dog activity, and deployed the CPEs along 100km of property tracks (likely to bisect approx. 4-5 wild dog home ranges (Thomson 1992)). Addressing the issues of variable detection, spatial scale and seasonal variation in activity provides confidence that the population reductions are due to CPEs. Deploying cameras throughout the entire study period including in-between bursts of control may have given a more accurate measure wild dog density change after each burst of control finished and allowed calculation of post-control immigration. However, the number of recorded wild dog activity events on camera did not decrease between bursts and the interest in the CPEs increased over the two-year trial indicating this is a valid assessment of the CPEs for wild dog population control.

CPEs have not previously been trialled in arid Western Australia at a landscape-scale. The devices have been successfully trialled for canid control in other parts of Australia (Hunt 2010a; Hunt 2010b; Speed *et al.* Unpublished data). The devices are essentially static or fixed baits so it is informative to compare changes in wild dog population size as a result of CPE control to conventional baiting. Recent evaluation of repeated landscape-scale baiting in the same area noted a low percentage of change in wild dog density per linear km in response to each baiting session (≤7%) (Kennedy *et al.* 2021a), against which the results from CPEs in this study compare favourably. Although, noting that neither individual bursts of CPE use, nor deployment over the eight month period resulted in a 75% population reduction. There are differences in the deployment of baits and CPEs which are also important to recognise. In this environment baits are typically deployed at 10 baits per linear km along extensive lengths of property tracks, but are also rapidly removed by both target and non-target species (Kreplins et al 2018). In comparison, this deployment of 100 CPEs did not cover the landscape as extensively as landscape-scale baiting, although the majority of CPEs were available for the full period of deployment. It is valuable to note that there was some spatial concentration in triggered CPEs. For example, two CPEs located on the same transect were fired during a session more than once.

One of the key factors contributing to the greater effectiveness of CPEs is the lack of interference with non-target species. Non-target species in the area are tolerant to the 1080 poison (King 1990) and are known to consume a large proportion of deployed baits (Kreplins *et al.* 2018a). This results in reduction of the baits being present for the target species, wild dogs, limiting the effectiveness of control efforts. All species recorded on camera (i.e. varanids, small stock, corvid *spp*., macropods, emus, feral cats and wild dogs) demonstrated higher interest in the CPEs than predicted based on their activity events on camera. However, a majority of the CPEs remained intact during the trial. Only target species (i.e. canids) were able to successfully trigger CPEs in this study. Of one hundred CPEs deployed over four sessions (n=800) only 60 CPEs were rendered ineffective by non-targets (7.5%). This contrasts markedly to previous bait uptake work in the same area. Kreplins *et al*., (2018a) recorded that of the 337 baits with a known fate 71% of those were removed by non-target species. Other trials of CPEs around Australia have also noted minimal non-target interference due to the design of CPEs (Allen 2002; Hunt 2010a; Allen 2019; Gil-Fernandez *et al.* 2021; Young *et al.* draft).

The lure head construction material (i.e. food based or felt-scent lure head) affected the effectiveness of the CPEs. The use of the felt lure heads with a scent lure resulted in only thirteen occasions (of the sixty) where the lure head was rendered unavailable to wild dogs. There were many occasions when non-target species showed interest in the felt lure heads (another one hundred and twenty one occasions) but they did not alter the CPE device in any way. Using food-based lures had much higher rates of interference. When using the ‘off-the-shelf’ dried meat bait lure heads, on forty seven occasions non-target species interfered with the CPEs and meat lure heads. The liver treat lure head had no interference from non-targets but also no interest from wild dogs. Using the felt lure heads is recommended where non-target numbers are high, but interestingly no scent on the felt lure heads fared better (i.e. resulted in a CPE fire). The choice of felt colour did not seem to alter the wild dog' choice of lure heads. However, interest by wild dogs in the lures did differ. There has been much research into the use of scent lures of canid species (Jolly *et al.* 1992; Saunders *et al.* 2000; Hunt *et al.* 2007) but to date there a universal scent lure has not been identified.

Canid pest ejectors are an effective tool in the arid environments for wild dog control. Here we demonstrated reductions of wild dog density over 100km of property tracks and that they are robust to interference by non-target species, however the reductions in density were not sufficient on their own to cause a population decline over time. Potentially if a large numbers of CPEs are deployed they could be used to decrease wild dog density over larger areas, although there are significant financial and labour costs to broad-scale deployment of CPEs. Alternatively, landholders could use a smaller number of CPEs in areas of known high wild dog activity to reduce the number of wild dogs in their area. In combination with other tools from the wild dog control toolbox CPEs can assist to achieve greater wild dog population reductions.

CHAPTER 3. LAND USE AND DINGO BAITING EFFECT THE DENSITY OF KANGAROOS IN RANGELAND SYSTEMS

Published by the *Journal of Integrative Zoology*.

INTRODUCTION

Worldwide, the raising of livestock by people is associated with conflict with predators, and in many cases leads to widespread lethal predator control (Berger 2006; Zimmermann *et al.* 2010; Ripple *et al.* 2014). Through predator control to protect livestock, humans have manipulated predator-prey relationships and altered natural limitations on herbivore abundance (Schmitz *et al.* 2000; Berger *et al.* 2008; Estes *et al.* 2011). Additionally, the introduction of free standing water for livestock and altered vegetation to improve grazing can also benefit native herbivores (James *et al.* 1999; Smit *et al.* 2007; von Wehrden *et al.* 2012). In many cases, the consequent overabundance of herbivore species has caused degradation of rangelands (Katona *et al.* 2019; Mills *et al.* 2020).

Kangaroo and common wallaroo or euro (hereafter collectively referred to as 'kangaroos') populations have increased markedly since European colonisation of Australia. Their increase in numbers is likely driven primarily by an increase in permanent water availability (James *et al.* 1999; Dawson *et al.* 2006; Fensham *et al.* 2008), modification of vegetation (Newsome 1975), and broadscale control of wild dogs/dingoes (*Canis familiaris*; Jackson *et al.* 2017) (Caughley *et al.* 1980a; Pople *et al.* 2000a; Letnic *et al.* 2013). There are estimated to be, on average, a combined total 40 million red (*Osphranter rufus*), and grey (*Macropus gigantus*, *M. fuliginosus*), kangaroos in Australia, the vast majority of which are on rangelands in inland regions, used for pastoralism (Wilson *et al.* 2019).

Total grazing pressure (TGP) is the summed pressure applied by all grazers present in a system, which in the Australian southern rangelands (in semi-arid and arid Australia) includes livestock, kangaroos, unmanaged goats, rabbits, feral pigs, equids and dromedary camels (Hacker *et al.* 2020). When combined with grazing by domestic livestock (representing the primary land use of many rangeland areas), populations of feral herbivores and kangaroos have resulted in unsustainably high TGP that can degrade landscapes and lead to negative outcomes for agriculture and biodiversity (Page *et al.* 2000; Mills *et al.* 2020; Fisher *et al.* 2021). There is some uncertainty around the degree to which kangaroos compete with sheep and cattle for fodder because of differences in diet (Pahl 2020b) and in the degree to which individual kangaroos contribute to grazing pressure (eg. Grigg 2002; Pahl 2020a). Nevertheless, kangaroo populations, together with other unmanaged herbivores such as rabbits, pigs and donkeys can contribute significantly to total grazing pressure of an area (Hacker *et al.* 2020).

The grazing pressure applied by unmanaged herbivores, over which pastoralists have little control can limit the effectiveness of management actions to achieve rangeland regeneration (Norbury *et al.* 1993). Examples from the Gascoyne region of Western Australia (WA) indicate that, following removal of sheep, the density of kangaroo dung increased six-fold (Norbury *et al.* 1993), suggesting that kangaroos move into areas with increased fodder availability. In addition to contributing significantly to TGP, kangaroo populations experience heavy mortality during drought with associated, and widely publicised, poor welfare outcomes (Wilson *et al.* 2019).

Managing TGP in pastoral landscapes requires greater understanding of the factors determining population dynamics of each kangaroo species (**Fig. 1**), including land use, the kangaroo commercial harvest effort, the impact of predator abundance, and environmental variables. Here, we briefly explore each of these potential drivers.

LAND USE AND MANAGEMENT

There is some evidence that grazing by livestock has modified the understorey of the rangelands to the benefit of kangaroos (Newsome 1971; Newsome 1975). Grazing by sheep in the Pilbara resulted in an increase in *Triodia pungens*, which, once mature, is avoided by sheep but beneficial for euros (*Osphranter robustus*), contributing to an increase in their abundance (Newsome 1975). Similarly, the creation of subclimax grassland by ruminant livestock in central Australia resulted in greater availability of green pick (i.e. new growth promoted by rainfall), which benefited red kangaroos (Newsome 1971; Newsome 1975). Since the seminal work by Newsome (1971, 1975), the rangelands sheep flock has largely been replaced with cattle, driven by declining demand for wool, increasing price of alternative commodities, and in some areas, dingo predation (Allen *et al.* 2013a; Forsyth *et al.* 2014), however, kangaroo populations remain high. Research published in 1982 showed that in south-eastern Australia, kangaroo densities were greater in pastoral than intensive wheat and sheep farming areas or within ungrazed natural vegetation such as mallee (Short *et al.* 1982). This difference has been attributed to the lack of shelter trees in cleared wheat and sheep holdings as well as the intense control effort by these farmers, while natural vegetation contains few palatable grasses and an absence of water points (Short *et al.* 1982). Despite recognised habitat preferences of the three kangaroo species and euros (**Fig. 1**), some studies have not detected differences in kangaroo habitat use between land tenures, which typically reflects land use (Jonzén *et al.* 2005; Letnic *et al.* 2013).

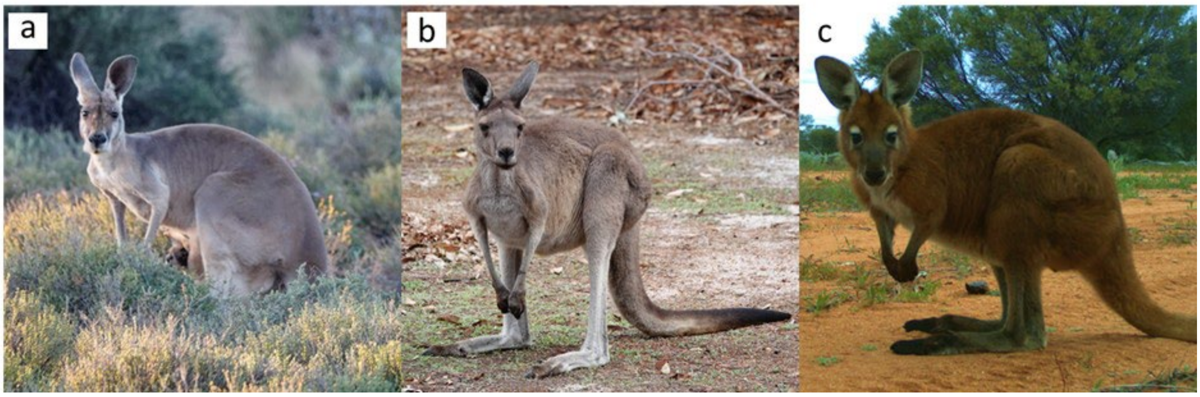


Figure 14. Three large kangaroo species are found in southern Western Australia. Each species shows different habitat preferences. (a) The xeric-adapted red kangaroo (Osphranter rufus) is capable of high mobility (Norbury et al. 1994), and populations are unevenly distributed with respect to vegetation and land use (Johnson et al. 1981). (b) Western grey kangaroos (Macropus fuliginosus) appear to be more dependent on cover than red kangaroos (Caughley 1964) with a preference for habitat heterogeneity (Short et al. 1983) (c) Common wallaroos or euros (O. robustus) are sedentary and localised to rocky landscapes (Ealey 1967).

Production of livestock grazing in Australia has required installation of artificial water points (AWPs) (Ealey 1967; James et al. 1999). Sheep and cattle must drink more frequently than kangaroos (Dawson et al. 1975), and therefore do not move as far from water as kangaroos (Fensham *et al.* 2008). The proliferation of AWPs is believed to be a cause of the increase in kangaroo abundance within the southern rangelands over the past century (James et al. 1999). However, there is some uncertainty about the role of AWPs in influencing kangaroo distribution and abundance, with food availability, landscape features and predation frequently being identified as limiting macropod densities rather than water availability (reviewed in Lavery *et al.* 2018). However, Lavery *et al.* (2018) also identified a lack of experiments assessing the role of AWP on macropod density at appropriate spatial and temporal scales.

Throughout much of the Australian southern rangelands, kangaroos are harvested commercially for meat and leather. The primary goal of the industry is to provide kangaroo products to consumers, but ecologically-sustainable commercial harvest of kangaroos also provides an alternative management approach to reducing the damage caused by over overgrazing (Grigg 1987; Read *et al.* 2021). Conservative harvest quotas have been in place to allay concerns regarding the exploitation of kangaroos. As a result, after 40 years of monitoring, there is no evidence that commercial kangaroo harvest threatens populations of the four harvested species (Ampt *et al.* 2006; Lunney *et al.* 2018; Read *et al.* 2021).

DINGOES

The dingo is Australia's largest terrestrial predator. While sheep production is incompatible with dingoes (Thomson 1984a; Allen *et al.* 2001a; Fleming *et al.* 2001a), the effect of dingoes on cattle enterprises is more complex (Allen 2015; Prowse *et al.* 2015). The widespread control of dingoes has been synonymous with the spread of sheep grazing throughout agricultural and pastoral regions of Australia. There are currently two landscape scale barrier fences intended to reduce dingo impacts on sheep production: the State Barrier Fence in south-western Australia and the Dingo Barrier Fence in eastern Australia. Dingoes are now less common on the sheep/agricultural sides of these fences (Pople *et al.* 2000a; Woolnough *et al.* 2005).

Ecological theory suggests that herbivore numbers are directly linked to primary productivity (bottom-up), as well as being controlled by predators (top-down) (Choquenot *et al.* 2013; Letnic *et al.* 2013). The strength of top-down regulatory processes is expected to be weaker in areas where productivity is unpredictable and stochastic (Morgan *et al.* 2017). Theoretical and field-based studies have concluded that the abundance of kangaroos can be determined by dingo predation. Letnic *et al.* (2013), Caughley *et al.* (1980a), Rees *et al.* (2017) and Pople *et al.* (2000a) describe field-based studies comparing areas either side of the Dingo Barrier Fence. Such natural experiments often have confounding factors, with land capability and productivity likely to have determined the location of the fence in the first place. For example, differences in vegetation structure and complexity (Mills *et al.* 2020) or fractional vegetation cover (Fisher *et al.* 2021) have been attributed directly to kangaroo overgrazing due to lower dingo density on the 'inside' of the Dingo Barrier Fence. However, Newsome *et al.* (2001a) examined one area surveyed by Caughley *et al.* (1980a), but included multiple years of data, and concluded that landscape differences in productivity explained kangaroo numbers, obscuring any potential impacts of dingo predation. Manipulation experiments are likely to have the greatest ability to demonstrate the relationships between kangaroos and dingoes. Two studies monitored kangaroo populations over time following the introduction (Moseby *et al.* 2019; albiet within a fenced reserve) or removal (Thomson 1992a) of dingoes; both studies found that dingoes had some regulatory effect on kangaroo populations.

While these field-based studies show dingoes can play a regulatory role on kangaroo populations, the effect of predation on kangaroo density is not independent of plant biomass, habitat and land management (e.g. Newsome *et al.* 2001a) suggesting the relationship is far from well understood. Nevertheless, some cattle graziers have ceased control of dingoes anticipating regulation of kangaroos and reduced TGP (Pollock 2019; Emmott 2021), providing additional field trials.

ENVIRONMENTAL FACTORS

Population growth rate of various kangaroo species have been strongly linked to rainfall (Cairns *et al.* 1993; Letnic *et al.* 2013; Lunney *et al.* 2018). Increases in kangaroo density in response to rainfall show a lagged response, reflecting increased reproduction rate in response to vegetation growth and standing dry matter (Cairns *et al.* 1993; Lunney *et al.* 2018). However, negative responses can be more immediate. Kangaroo mobility increases in drought conditions, presumably to find resources (Norbury *et al.* 1994), and there can be dramatic reductions in kangaroo density through death during drought (Ealey 1967; Caughley *et al.* 1985; Newsome *et al.* 2001a; Wilson *et al.* 2019; Zanker 2021). Rainfall is therefore used to predict population size in years between aerial surveys, which is then used to set harvest quotas (e.g. Department of Biodiversity Conservation and Attractions 2019).

AIMS OF THIS STUDY

Control of dingoes and provision of AWPs across Australian southern rangelands are likely to have removed important limitations to kangaroo populations that are now only food limited (Bayliss 1987; Cairns *et al.* 1993) and contributing significantly to TGP. In this study, we analysed 22 years of aerial monitoring data from the southern rangelands of WA to test whether the density of three kangaroo species (red kangaroo, western grey kangaroo, euro) is associated with:

1. Environmental and management factors
2. Dingo control
3. Presence of the State Barrier Fence

METHODOLOGY

KANGAROO SURVEYS

From 1994 onwards, aerial surveys for red and western grey kangaroos and euro kangaroos have been flown by the WA Department of Biodiversity, Conservation and Attractions (DBCA) to estimate abundance to determine the annual harvest quota. Survey transects are flown at 0.5-degree latitude intervals (**Fig. 2**). The survey is broken into four monitoring zones: northern, central, south-eastern and south-western. As the present study is focused on the southern rangelands, only the central and south-eastern zones were analysed. The central zone contains the Murchison, Gascoyne, Yalgoo, Avon Wheatbelt, and Geraldton Sandplains IBRA regions, and the southern-eastern zone contains the Murchison, Coolgardie, Nullarbor, Great Victoria Desert, Mallee, and Esperance Sandplains IBRA regions (Thackway *et al.* 1995). The smaller south-western block is generally surveyed annually, while each of the larger monitoring blocks is surveyed every three years on a rotational basis; e.g, the south-eastern zone was surveyed in 1996, the central zone in 1997, and the northern zone in 1998. From 1981 until 1993, a triennial aerial survey of WA was conducted by the then Australian Nature Conservation Agency; however the raw data were not available and we have therefore not included these data in the present study.

The aerial survey technique is described in detail in Pople *et al.* (1999). Broadly, fixed-wing aircraft are flown along transects at 100 knots, at 76 m (250 feet) north of ground level (AGL), with a 200 m-wide strip searched for the three species of kangaroo (Department of Biodiversity Conservation and Attractions 2018). A 'cell' represents 5 km of flown transect, which is equivalent to surveying 1 km^2. Standard correction factors are applied for temperature and vegetation type (Supplementary Information Table S2).

LAND USE AND MANAGEMENT

Land use category is likely to incorporate elements of other factors such as availability of grasses for foraging or availability of drinking water. For example, much of the variation in the availability of AWPs is likely to be tied to land use; pastoral land generally has abundant AWPs for livestock, agricultural land may also have many AWPs (unless the operation is predominantly grain growing), while reserve and government land generally have no AWP or they have been turned off. Land use and management type were held constant at the individual property level. Across the entire survey, 61% of survey cells were on pastoral land, 7% on agricultural land, and 31% of government land and reserves, the majority of which was conservation estate, unallocated crown land, and miscellaneous reserves. Livestock type and livestock numbers were extracted from Australian Bureau of Statistics census data at the level of Local Government Area (LGA). Goats are also recorded during aerial surveys. To investigate the potential for a competitive effect on kangaroos by goats, the density of goats, averaged at the level of LGA, was tested.

DINGO DENSITY ESTIMATE

To assess the impact of the dingo predation on the density of kangaroos, we created three spatial data layers.

Restricted Chemical Permits as a surrogate of permitted dingo control. We used the Department of Primary Industries and Regional Development (DPIRD) Restricted Chemical Permit (RCP) database, a record of all granted permits and the property they are associated with, to calculate the percentage of each LGA in which poison baiting, or trapping with strychnine, was permitted for 2010–2020 (this was the only period for which electronic data was available) at the level of LGA. This does not necessarily imply that control was undertaken in these areas, only that a permit existed for control. We extrapolated from these available data back to the beginning of the study period. LGAs where dingo control had only recently commenced (areas 'inside' the State Barrier Fence where broadscale control only commenced since 2013) were assumed to have no dingo control prior to 2010. LGAs with consistently high dingo control were assumed to have consistent dingo control since 1996; average percentage RCP-permitted area for 2010–2020 was therefore extrapolated back for 1996–2009. For LGAs with a consistent increase in the percentage of the RCP-permitted area, the average value over a shorter period (2010–2012) was extrapolated back for 1996–2009.

Dingo density. We estimated dingo density across the study area in the years 1996–2018 using the approach of Woolnough *et al.* (2005). To create our estimate of dingo density, we conducted interviews with four DPIRD staff directly working on dingo management in WA for ~20 years each, who also consult broadly with land managers over then time to monitor vertebrate pests. For each LGA, participants were asked to estimate the density of dingoes as absent (0), rare (1), medium (2), or common (3) for each year between 2003–2018. An average value was then calculated for each LGA for each year. Each transect point was assigned a value for dingo density for the matching time point.

State Barrier Fence. We recorded which side of the State Barrier Fence data were collected for each 5 km-length of transect.

ENVIRONMENTAL FACTORS

We recorded terrain ruggedness and vegetation cover for each 5-km length of aerial transect, and rainfall (in the previous 12 and 12-24 months) and Total Standing Dry Matter (TSDM) (in the previous 12 and 12-24 months) at the level of LGA (details in **Table 1**). The coordinates of the start of each 5 km-length of transect were used as a survey point for extracting environmental information from Geographic Information System (GIS) input layers for each year of the study (Table 1). The previous year's kangaroo density in the relevant Kangaroo Management Zone was included as a covariate.

Table 2. Layers used to extract environmental and anthropogenic data as covariates to model the density of red kangaroos (Osphranter rufus), western grey kangaroos (Macropus fuliginosus), and euros (O. robustus). Details relating to sources and scale of data in Supplementary information

Variable name	Description
Kangaroos (dependent variables)	
Separate density estimates for red kangaroo (RK), western grey kangaroo (GK) and euro (E)	The number of individuals recorded in a 5km-length of transect (~1 km^2), after correction for temperature and vegetation.
Land use and management	
Livestock type	Proportion of total DSE that are sheep
Livestock density	Dry sheep equivalent per km^2 in LGA
Goat density	Average goats per km^2 in LGA
Land use	The type of land tenure under which the land is held, aligning broadly with the type of management.
Previous kangaroo harvest	Kangaroos harvested per km^2 in that management zone in the previous calendar year
Dingoes	
Percentage of LGA covered by Restricted Chemical Permits (RCP)	RCP-permitted area as % of the total LGA area
Density of dingoes	Average of four expert rankings for each year
State Barrier Fence (SBF)	Inside (southwest of fence) Outside (northeast of fence)
Environmental	
Terrain ruggedness	The standard deviation of elevation within a 5-km radius.
Vegetation cover	Mean tree cover within a 5 km radius.
Rainfall	Rainfall decile in the 12 months prior to survey
Lagged-rainfall	Rainfall decile lagged 12 months (12-24 months prior to survey)
Total Standing Dry Matter (TSDM)	Average TSDM in previous 12 months
Lagged-TSDM	Average TSDM in lagged previous 12 months (12 to 24 months prior to survey)
Previous kangaroo density	Individuals per km^2 in the previous year

STATISTICAL ANALYSIS

Two datasets were analysed for each kangaroo species (six datasets in total):

(1) Full survey dataset. As red kangaroos are found throughout the survey area, the entire survey area was analysed for this species. The analysis included a smaller area for western grey kangaroos and euros, including only those locations where these species are likely to occur (**Fig. 2**).

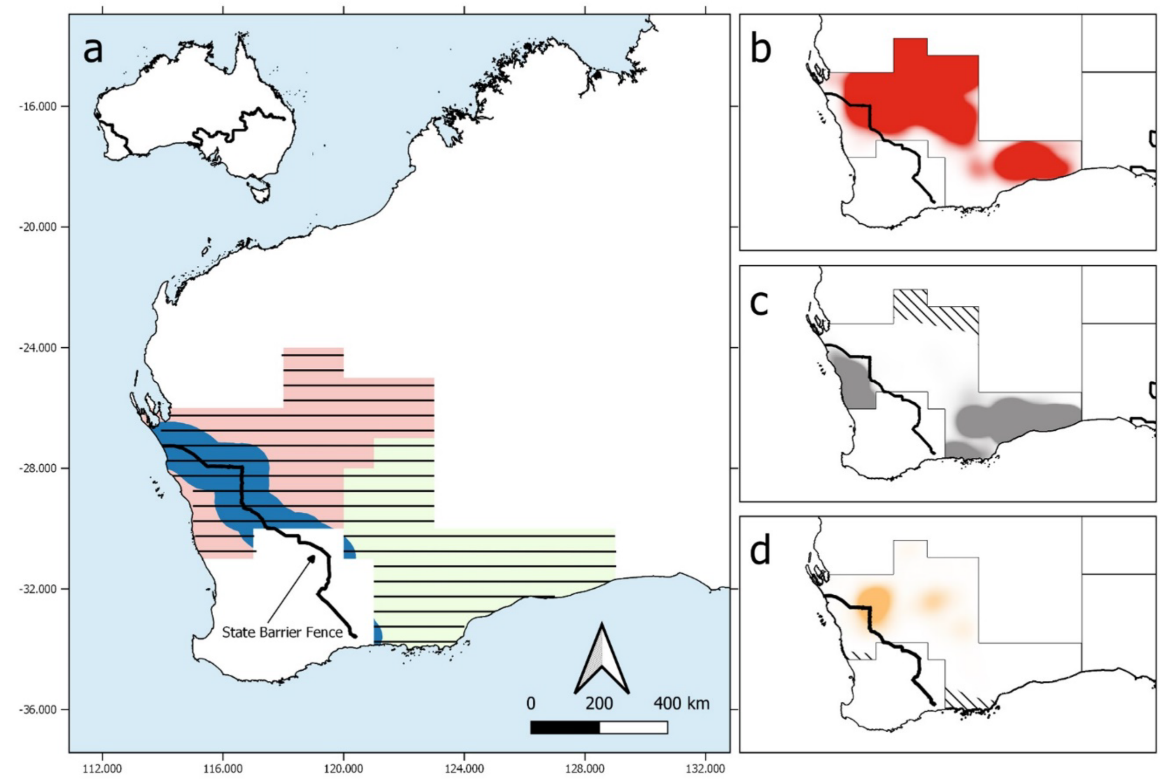


Figure. 15. (a) Location of the kangaroo density survey transects (horizontal black lines), in the central (pink shading), and south-eastern (green shading) zones, flown on a triennial basis across Western Australia (inset the two dingo barrier fences: the Western Australian State Barrier Fence and the Dingo Barrier Fence running across four states in eastern Australia). Right hand panel: heat maps indicating the areas of greatest density of (b) red kangaroos (Osphranter rufus) (red), (c) western grey kangaroos (Macropus fuliginosus) (grey), and (d) euros (O. robustus) (orange), over the period 1996 to 2018. All three species occur outside of the study area but shaded areas show locations with greatest density. Areas to the southwest of the State Barrier Fence are referred to as 'inside', while areas to the northwest are 'outside'. Hatched areas indicate parts of the study area that were outside the distribution of western grey kangaroos and euros (c, d). State Barrier Fence dataset is shown by the blue area in plot a.

(2) State Barrier Fence-dataset. A separate analysis was conducted on datapoints from within 100 km of the State Barrier Fence (**Fig. 2**), allowing a finer scale comparison of the effect of the fence between relatively similar areas.

Prior to inclusion, each dataset was filtered to exclude missing values, which were generally due to data gaps in layers. All input variables were scaled (each value has the mean subtracted and then divided by the standard deviation) before being included in analysis.

Kangaroo data are collected as count data, but integers were converted to a continuous scale after correction for temperature and vegetation. The data were highly zero-inflated (>80% zeros). Given these characteristics, a GLM was fitted for each species with a Tweedie link, using the 'tweedie' package (Dunn 2017) in R (R Core Team 2018). The Tweedie distribution allows for highly zero-inflated continuous data, with true zeros, to be used without the need for dramatic data filtering or pooling to fit the assumptions of alternative distributions (Dunn *et al.* 2005). When fitting the model, kangaroo density data was log-transformed (+1) to achieve best fit. The variable power of the Tweedie link can be specified for each individual model in order to optimise fit. We compared alternative models using the derived Tweedie-AIC value (Dunn 2017).

A single interaction term, *land use x 25 month lagged-rainfall*, was included in all models as the availability of AWPs on pastoral land make it likely that rainfall will have minimal effects on drinking water for kangaroos on pastoral land (compared with relative absence from reserves and public estate). A global model was fitted for each of the three species, using both datasets (full and SBF subset), containing all 15 independent variables plus the land use x lagged rainfall interaction term (Table 1). All combinations of the variables within the global model were fitted using the *dredge* function in the 'MuMIn' package (ver. 1.43.17) (Barton *et al.* 2020) in R. Variance Inflation Factors (VIF) were calculated for each variable in the fitted global model. Variable pairs that resulted in a VIF>3 were specifically excluded from *dredge* analysis, so that models containing collinear variables were not fitted (all excluded combinations specified in Supplementary Information Table S3). Model averaging was performed on the top models (all models within Δt-AIC <2 of the best model).

Forest plots of the model-averaged beta estimates (±95% confidence intervals) of each predictor variable were made using 'ggplot2' (Wickham 2016) in R. For visualisation, graphs were also made using a fitted model that contained all significant predictors from model averaging, with significant predictors were those for which the 95% confidence interval did not overlap zero. Individual relationships were also plotted using the *ggeffect* function in the 'ggeffects' package (Lüdecke 2018) in R. This method is useful when displaying fitted models with multiple explanatory variables, as it holds other variables constant (at an average value) while displaying the effect of the variable in question (Lüdecke *et al.* 2020). All plots are back-transformed to be plotted on the original scale of the variables displayed.

All analysis was performed in the R statistical environment (version 3.5.2; R Core Team 2018).

RESULTS

OVERALL

Red kangaroos were observed over a total of 13,440 individual 5 km-length cells (i.e. 13,440 km^2), western grey kangaroos over 11,639 km^2, and euros over 12,909 km^2. Abundance of red kangaroos was highly variable between years in the central monitoring zone (**Fig. 3**) and western grey kangaroo numbers were highly variable between years in the south-east monitoring zone (**Fig. 3**). Taking red kangaroos as an example, as they are the most evenly distributed across the survey area, there was a maximum population size of approximately 1,189,886 (95% CI: 1,072,777 – 1,306,996) red kangaroos in 2000, and a minimum of 139,270 (95% CI: 105,044 – 173,496) in 2012.

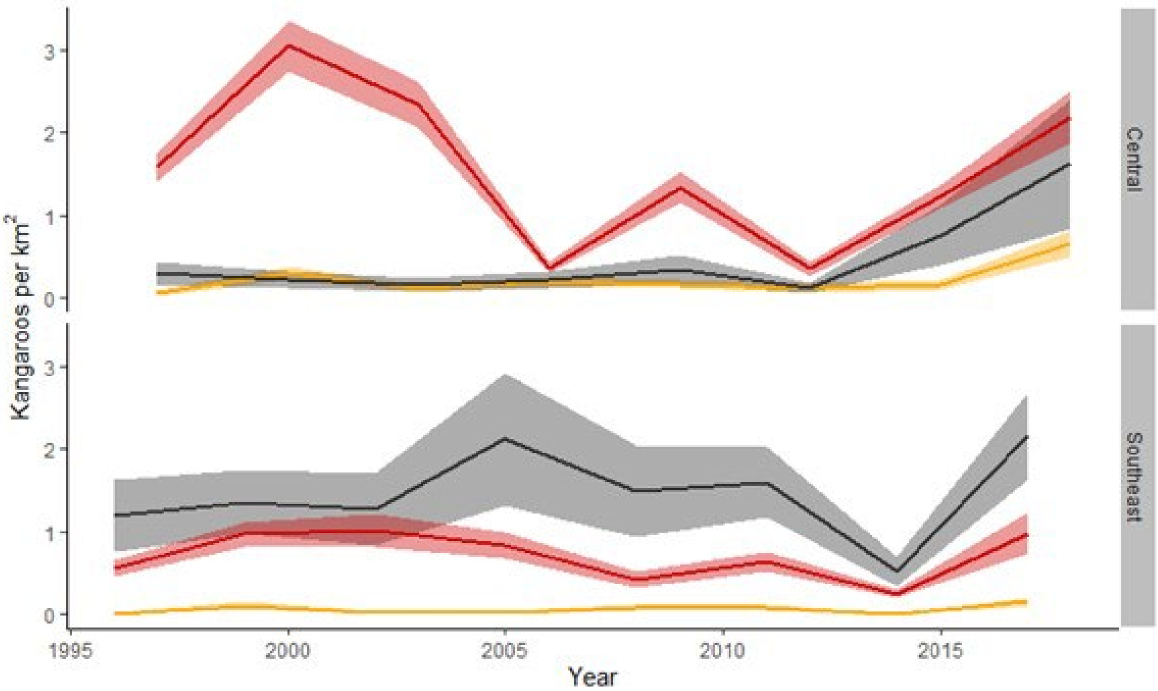


Figure 16. The mean density of red kangaroos (Osphranter rufus, red line), western grey kangaroos (Macropus fuliginosus, grey line) and euros (O. robustus, orange line) in the central and southeast monitoring zones of Western Australia. Shaded areas indicate 95% confidence intervals.

FULL DATASET

The density of red kangaroos was greatest on pastoral land and lowest on agricultural land (**Figs. 4, 5**). The density of red kangaroos was positively associated with the number of red kangaroos harvested in the previous year. Red kangaroos were positively associated with the RCP-permitted area (dingo density was not retained in the top models and there was no significant effect of the presence of the State Barrier Fence). There was some evidence of preference for flat ground (negative relationship with terrain ruggedness) and red kangaroos were more common in areas with less vegetation cover (**Figs. 4, 5**). Red kangaroo density was positively correlated with 12 month lagged-rainfall (**Figs. 4, 5**).

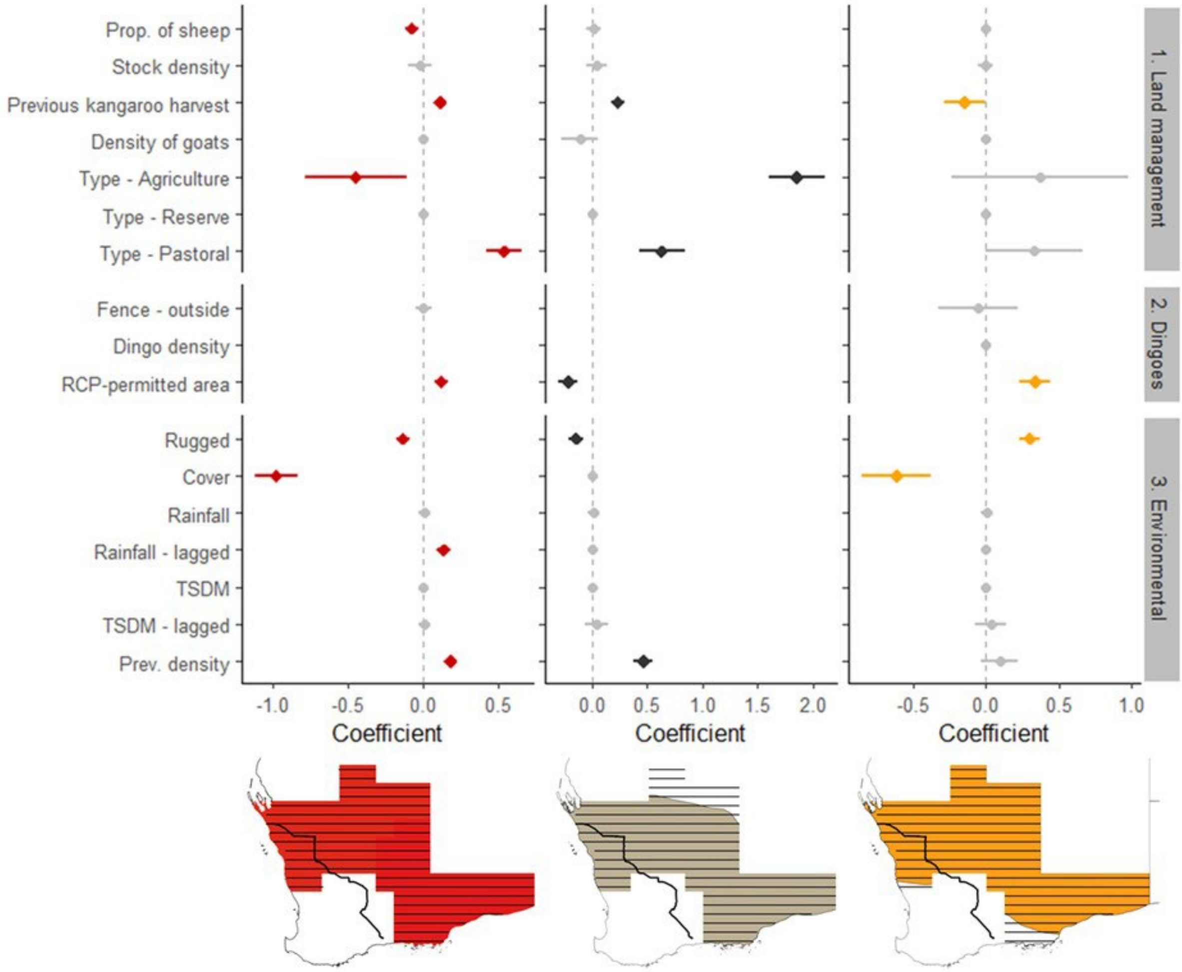


Figure 17. Model-averaged beta values of explanatory variables on the density of red kangaroos (Osphranter rufus), western grey kangaroos (Macropus fuliginosus), and euros (O. robustus). Bars indicated the 95% confidence intervals, and bars that overlap zero are considered to be non-significant predictors (grey points and bars). The maps at the bottom indicated the area over which data was included.

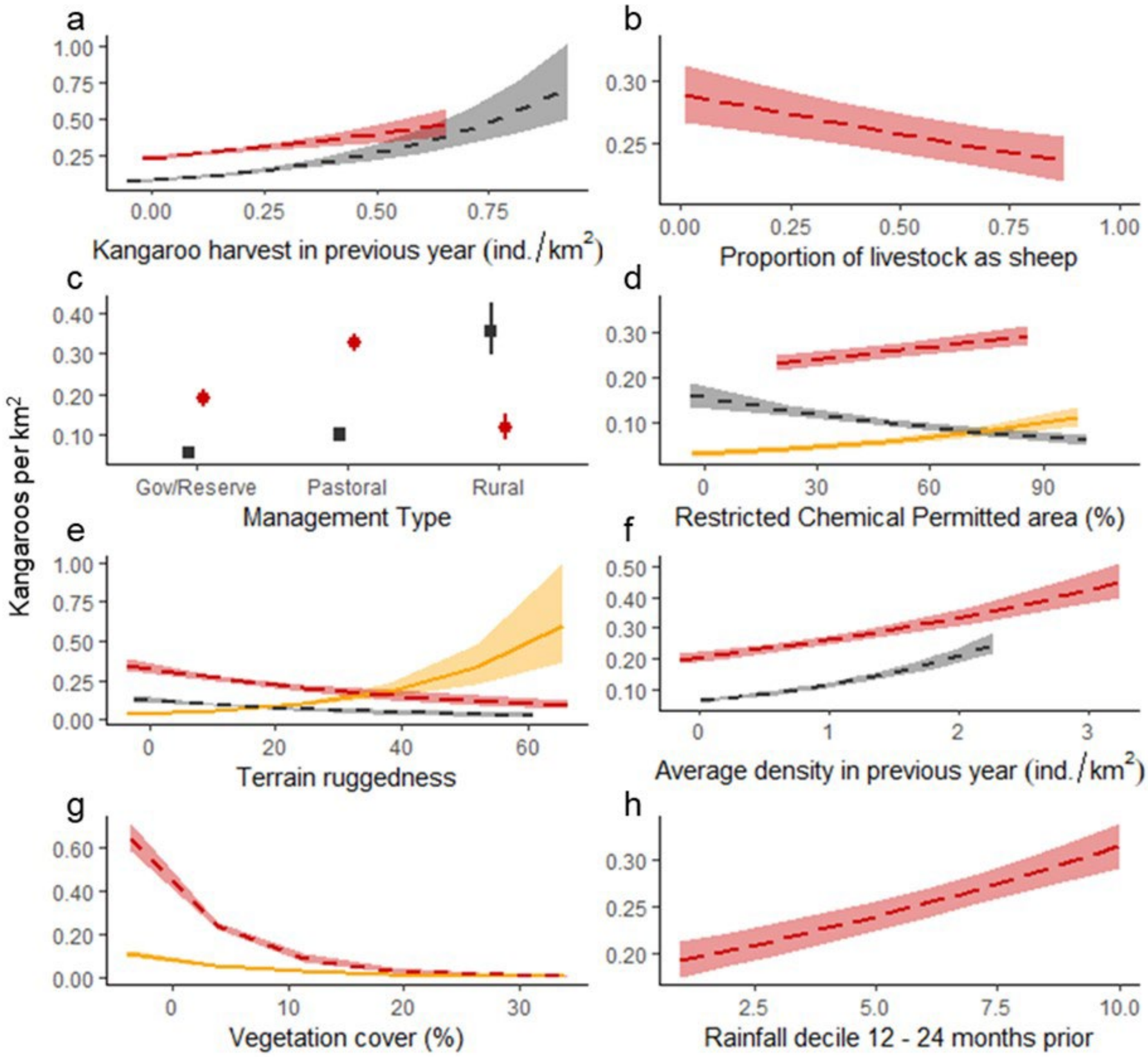


Figure 18. The effect of land use variables (a, b, c), dingo management (d), and environmental variables (e, f, g, h) on the density of red kangaroos (Osphranter rufus), western grey kangaroos (Macropus fuliginosus), and euros (O. robustus). Only those variables that were significant after model averaging are included here, and only in the species for which they were relevant.

Western grey kangaroos were most abundant on agricultural land, and least abundant on reserve and government land (**Figs. 4, 5**). The density of western grey kangaroos was positively associated with the number of western grey kangaroo harvested in the previous year. Western grey kangaroos were also less abundant in areas with a greater RCP-permitted area (**Figs. 4, 5**) (dingo density and presence of the State Barrier Fence were not retained in the top models). Western grey kangaroos preferred flat ground (indicated by the negative relationship with terrain ruggedness). Western grey kangaroos were also negatively associated with lagged-rainfall (**Figs. 4, 5**).

Euro density was not significantly different between the three land use types (**Figs. 4, 5**) and euro density was not significantly associated with number of euros harvested in the previous year. Euros were more common in areas with greater RCP-permitted area (dingo density and presence of the State Barrier Fence were retained in the top models but were not significant factors). Euros showed preference for rugged terrain (**Figs. 4, 5**) and were more common in areas with less vegetation cover (**Figs. 4, 5**).

It was notable that the density of the three species was not correlated with the density of goats, livestock density, the presence of the State Barrier Fence, estimated dingo density, rainfall in the immediate year, or the current or 12 month-lagged TSDM.

STATE BARRIER FENCE-DATASET

Data from the State Barrier Fence subset was collected between 1996 to 2018, representing 2,870 km2 of sampling for all three species. As this scale, there was no difference in density across the fence for any of the three species (**Fig. 6**).

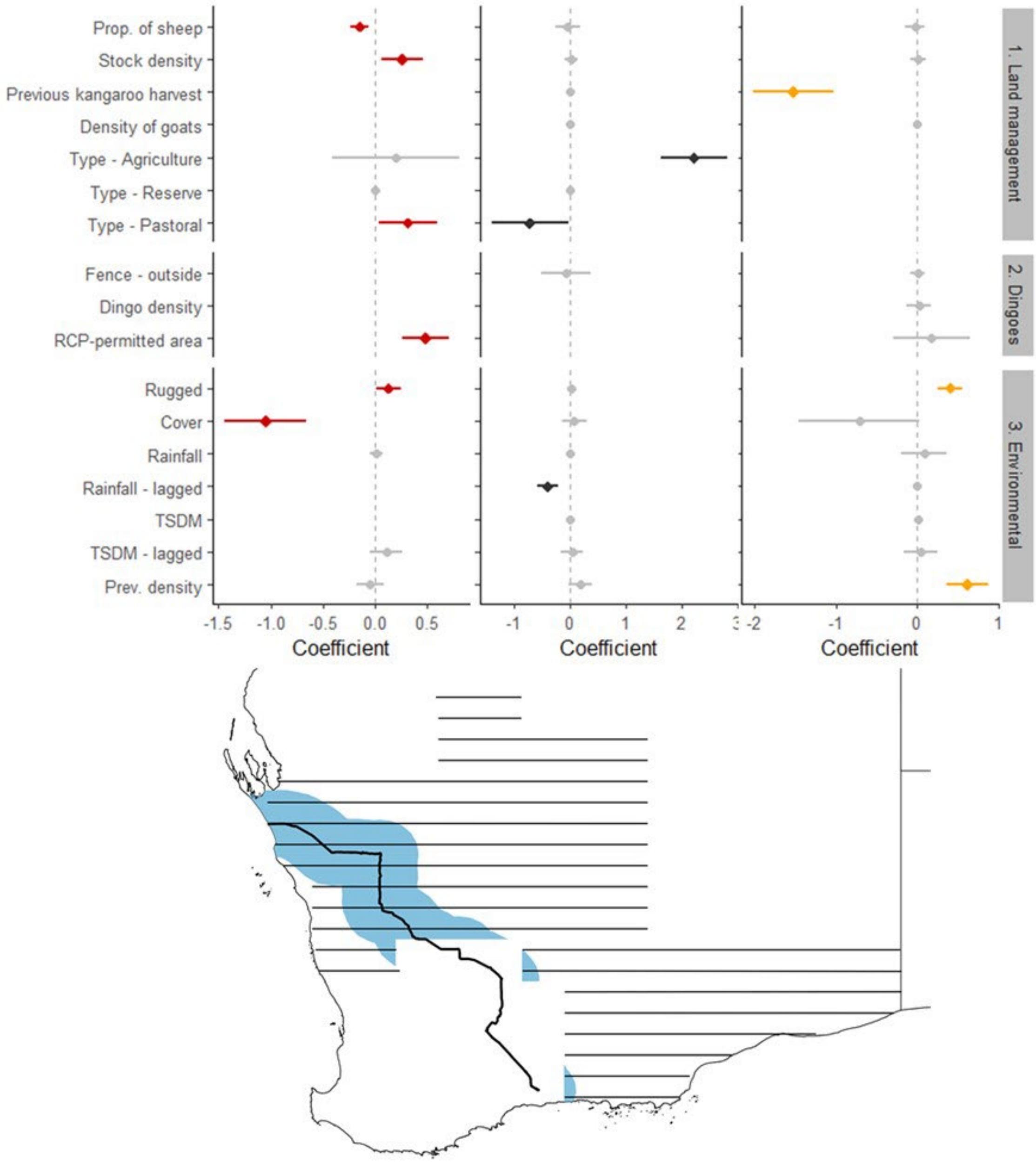


Figure 19. Model-averaged beta values of explanatory variables retained in the top model set (delta AIC <2) on the density of red kangaroos (red), western grey kangaroos (grey), euros (orange) at points within 50 km of the state barrier fence. Bars indicated the 95% confidence intervals, and bars that overlap zero are considered to be non-significant predictors (light grey points and bars). Variables are sorted from most negative estimates on the bottom, to positive estimates at the top.

Within 100 km of the State Barrier Fence, red kangaroos were negatively associated with increased proportion of sheep grazing (red kangaroos were more common in areas with cattle), but positively associated with average livestock density. Similar to the full analysis, red kangaroos were most common on pastoral land and more common with greater RCP-permitted area (**Fig. 6**). In addition, red kangaroos were negatively associated with vegetation cover and the density of red kangaroos in the previous year, and positively associated with terrain ruggedness (**Fig. 6**).

Western grey kangaroos were most abundant on agricultural properties and least abundant on pastoral properties within 100 km of the State Barrier Fence. They were also negatively associated with 12 month lagged-rainfall (**Fig. 6**).

Within 100 km of the State Barrier Fence, euros were negatively associated with the previous euro harvest, but positively associated with the previous euro density (**Fig. 6**). Again, euros were more common in rugged-terrain areas.

DISCUSSION

The long-term degradation of rangelands around the world present a significant threat to biodiversity and livelihoods of livestock producers (Harris 2010; Bedunah *et al.* 2012). In Australia, there is a renewed focus on understanding the contribution of kangaroo populations to TGP, and the impact that this has on rangeland health and livestock production (Mills *et al.* 2020; Emmott 2021; Fisher *et al.* 2021). Broadly, the habitat associations identified in this study for the southern rangelands of Western Australia were similar to those recorded previously for red kangaroos (Newsome 1975; Short *et al.* 1983), western grey kangaroos (Short *et al.* 1983), and euros (Ealey 1962) (**Fig. 1**). However, density of these kangaroo species also varied with land management practices and potential dingo control (estimated as the proportion of land covered by Restricted Chemical Permits for dingo control). Here we discuss the basic habitat requirements of each of the three species investigated in this study, followed by an exploration of the potential impact of land use, commercial kangaroo harvest, and dingo control on their abundance.

While red kangaroos are the most broadly distributed macropod species in WA, their density is lower than recorded in eastern Australia (Short *et al.* 1983). Red kangaroos are primarily grazers, and are therefore less common in woodlands and shrublands in favour of open plains where grasses predominate (Short *et al.* 1983). This habitat preference is reflected in the greater density in areas with low vegetation cover and low terrain ruggedness. The rate of increase of red kangaroo populations is known to be driven by primary productivity, which is in turn driven by antecedent rainfall (Bayliss 1987; Cairns *et al.* 1993). Here, we were unable to directly calculate the annual rate of increase, as each area was only surveyed every three years. To account for this temporal non-independence, we included the modelled previous density, and the 12-month lagged rainfall, both of which were positively associated with red kangaroo density.

The distribution and density of western grey kangaroos is driven largely by climatic factors, such as the seasonality of rainfall, preferring areas with winter rain (May-October) (Short *et al.* 1983). As the winter-rainfall regions of WA are less subject to stochastic rainfall driven resource pulses than the arid zone, it is likely that temporal changes in rainfall and pasture availability were insufficient to be a significant driver of western grey kangaroo density. Euros are colloquially known as ‘hill kangaroos’, and displayed their well-documented preference for rugged-terrain country, including rocky outcrops and breakaways (Ealey 1967) in the present study. Euros are highly selective for grasses (Ellis *et al.* 1977), which may account for the preference for grazing in more open areas, as opposed to the often shrub-dominated woodlands.

DINGOES

Numerous studies have concluded that predation by dingoes regulates the abundance of kangaroo populations, and that widespread dingo control in food production systems has alleviated this kangaroo population control through predation (Caughley *et al.* 1980a; Pople *et al.* 2000a; Letnic *et al.* 2013). While kangaroo populations are significantly influenced by bottom-up processes, when macropod density is high enough to support dingo populations, kangaroos may be regulated by a top-down predator-herbivore feedback loop (Choquenot *et al.* 2013). As such, suppression of dingo populations is hypothesised to result in increased kangaroo populations that are limited only by pasture availability (Choquenot *et al.* 2013). We assessed the effect of dingoes on kangaroo populations using three variables: the RCP-permitted area, an estimate of dingo density, and presence of the State Barrier Fence. Red kangaroos and euros were positively associated with the RCP-permitted area (note that this is not actual control effort, but a surrogate measure of likely dingo control effort). By contrast, western grey kangaroos were negatively associated with RCP-permitted area. In the State Barrier Fence-subset analysis, only red kangaroos were influenced by RCP-permitted area.

This result suggests that dingo control is associated with an increase in red kangaroo and euro density, as revealed by numerous previous studies (Caughley *et al.* 1980a; Pople *et al.* 2000a; Choquenot *et al.* 2013; Letnic *et al.* 2013). However, as a correlation it does not demonstrate causation. An alternative explanation posits that this relationship, as with the positive relationship between western grey kangaroos and RCP-permitted area, reflects habitat features and land use other than dingo abundance where there is different levels of reliance on dingo control.

There are other important caveats with these dingo datasets. The relationship between the RCP-permitted area and dingo density can be weak and highly variable. For example, in recent years, baiting in the southern rangelands of WA has been demonstrated to be particularly ineffective (Kennedy *et al.* 2021b) compared to previous studies (Thomson 1986a). Consequently, greater investment in dingo control does not necessarily result in fewer dingoes. Furthermore, obtaining an RCP does not necessarily mean dingo control was actually carried out on the ground. Well-organised community biosecurity groups may encourage and enable landholders to gain RCPs to the maximum extent allowed, but landholders may never actually carry out dingo control (T. Kreplins, pers. obs.). Collecting data from pest control professionals who undertake the control on behalf of landholders is likely to be an informative addition to the data collected in the present study.

To attempt to account for the lack of direct relationship between RCP-permitted area and dingo density, we included a qualitative, expert elicitation estimate of dingo density in our analysis. Density of dingoes is extremely difficult to quantify at the site-scale, let alone gaining an understanding across multiple years at the regional scale. Our approach of interviewing employees of DPIRD was based on that used by Woolnough *et al.* (2005), and was undertaken because there are no regional estimates of dingo density in Australia. Such an approach is naturally limited by the knowledge and memory of interviewees, the scale at which density is estimated, and subject to individual biases. Nevertheless, we believe that observed trends in dingo density over the period of the study are likely to representative and course trends, even if ultimately these broadscale temporal changes in estimated dingo density proved uninformative as predictors of kangaroo density.

When limiting our analysis to areas within 100 km of the WA State Barrier Fence, we found no evidence of a difference in density on either side of the fence, and an effect of RCP-permitted area only on red kangaroos. Most studies that have investigated the effect of dingo predation on kangaroo populations have relied on comparison across the Dingo Barrier Fence in Queensland, New South Wales, and South Australia, inferring that differences in kangaroo populations are driven by dingo density (Caughley *et al.* 1980a; Pople *et al.* 2000a; Letnic *et al.* 2013; Rees *et al.* 2017). In contrast with recent studies on the Dingo Barrier Fence in eastern Australia (Mills *et al.* 2020; Fisher *et al.* 2021), we found no significant differences in density on either side of the WA State Barrier Fence for three kangaroo species. It is possible that the differences in dingo density within 100 km of the fence was not great enough to result in a difference in control applied to kangaroos by dingoes. Dingo control effort is high on both sides of the WA State Barrier Fence and the fence has some known locations where it is permeable to dingoes (e.g. road crossings). The few remaining pastoral enterprises running sheep on the ‘outside’ are generally within 100km of fence, and properties ‘inside’ the fence running small livestock (i.e. sheep) have seen incursions by dingoes through the fence for over 10 years (Pacioni *et al.* 2018b). As such, dingoes are present, but subject to relatively high control effort, on both sides of the WA State Barrier Fence.

LAND USE AND WATER

In the present study, the density of red kangaroos was greatest on pastoral land, and lowest on agricultural land. It is likely that rather than avoiding agricultural land, the natural distribution of red kangaroos tends to end at the western boundary of the pastoral zone, which is approximately the 250 mm isohyet (Short *et al.* 1983). Conversely, western grey kangaroos are most abundant on agricultural land, and least abundant on reserves. Western grey kangaroos generally occur in areas dominated by winter rainfall, which largely corresponds to the ‘wheatbelt’, the grain growing region of southwest WA (Short *et al.* 1983), with the notable exception of high western grey kangaroo densities across the Nullarbor region.

Both red and western grey kangaroos occur in greater density in food production landscapes (pastoral for reds, and agricultural for western greys) than in reserves. This difference may reflect a number of variables including the density of AWPs which are likely to be correlated with land use, with more water available on pastoral and agricultural land, and less on reserves where AWPs are generally absent or removed (Short *et al.* 1982). Alternatively, differences in land productivity may be driving the difference between production areas and reserves.

COMMERCIAL HARVEST OF KANGAROOS

There was no evidence that commercial harvesting had a negative effect on the density of red and western grey kangaroos in the southern rangelands of WA. In fact, there was a significant positive relationship between kangaroo harvesting in the previous calendar year and the density of red and western grey kangaroos. This result is likely an artefact of commercial shooters targeting areas of high density, which remain at high density the following year, and may be considered a likely result given the spatial approach used.

Grigg (1987) postulated that kangaroo harvesting would reduce grazing pressure, resulting in lower TGP and better long-term grazing practices. However, there was also a lack of relationship between kangaroo density and kangaroo harvesting, which is unsurprising given that the conservative harvest quota of 15–17% has been met in only 2 years since 1972 for red kangaroos, and 3 years since 1983 for western grey kangaroos (Department of Biodiversity Conservation and Attractions, Conservation and Attractions 2018). At current harvest rates, the commercial kangaroo harvest appears to provide no regulatory effect on kangaroo populations, let alone presenting a threat, as suggested by Ben-Ami *et al.* (2010)

In contrast to red and western grey kangaroos, there was a significant negative relationship between the harvest of euros and their density. There has been no commercial harvest of euros in WA since 2009 (Department of Biodiversity Conservation and Attractions 2019), largely due to the small number of euros taken annually and the significant cost of the monitoring required to continue to support a commercial harvest under a Wildlife Trade Management Plan. The negative relationship is likely an artefact of no harvest in the second half of the time series corresponding with an increase in euros due to some factor unrelated to the harvest.

CONCLUSIONS

In the current study, we analyse and present results from one of the largest annual, broadscale surveys of a group of native species in Australia, comparable with the Eastern Australian Waterbird Survey (Kingsford *et al.* 2020). Results of macropod monitoring surveys across Australia are rarely interrogated or reported in the scientific literature, which is concerning given the important ecological role of large macropods, and impacts of their overabundance. Red kangaroos, western grey kangaroos, and euros select habitat according to environmental factors such as terrain ruggedness and vegetation cover. In addition, all three species were significantly impacted by anthropogenic factors, including livestock grazing, abundant water, and potential dingo control (RCP-permitted area). Red and western grey kangaroos were more abundant in food production landscapes (pastoral and agricultural land, respectively), than reserves. Red kangaroos and euros were more abundant in areas with greater RCP-permitted area, while western grey kangaroos showed the opposite pattern, although the confounding effects of these species' natural distribution make further interrogation of this relationship problematic. Given most jurisdictions in Australia conduct regular macropod monitoring of some scale, we implore researchers to use these existing datasets build our understanding long-term landscape-level change in Australian ecosystems.

SUPPLEMENTARY INFORMATION

TABLE S1

Layers used to extract environmental and anthropogenic data as covariates to model the density of red kangaroos (*Osphranter rufus*), western grey kangaroos (*Macropus fuliginosus*), and euros (*O. robustus*), including sources and scale of data.

Variable name	Description	Range and units	Scale of extracted data†	Source
Kangaroos (dependent variables)				
Separate density estimates for red kangaroo (RK), western grey kangaroo (GK) and euro (E)	The number of individuals recorded in a 5km-length of transect (~1 km^2), after correction for temperature and vegetation.	Continuous: RK (0–111.78 ind./km^2) GK (0 –360.58 ind./km^2) E (0–42.42 ind./km^2)	Point	Data provided by DBCA in 2020
Land use and management				
Livestock type	Proportion of total DSE that are sheep	Continuous: 0–1	LGA	Australian Bureau of Statistics census, provided by DPIRD in 2020
Livestock density	Dry sheep equivalent per km^2 in LGA	Continuous: 0.13–216 DSE/km^2	LGA	ABS census, sourced from DPIRD
Goat density	Average goats per km^2 in LGA	Continuous: 0–30 ind./km^2	LGA	Data provided by DBCA in 2020
Land use	The type of land tenure under which the land is held, aligning broadly with the type of management.	Categorical: (1) Pastoral (2) Agricultural (3) Government / Reserve (other uses, e.g. roads, were excluded from analysis).	Property	DPIRD (2021)
Previous kangaroo harvest	Kangaroos harvested per km^2 in that management zone in the previous calendar year	Continuous: RK (0–0.63 ind./km^2) GK (0 –0.85 ind./km^2) E (0–0.03 ind./km^2)	Kangaroo Management Zone	DBCA (2018)
Dingoes				
Percentage of LGA covered by Restricted Chemical Permits (RCP)	RCP-permitted area as % of the total LGA area	Continuous: 0–100%	LGA	Data provided by DPIRD in 2021

Variable name	Description	Range and units	Scale of extracted data†	Source
Density of dingoes	Average of four expert rankings for each year	Continuous: 0–3	LGA	
State Barrier Fence (SBF)	Inside (southwest of fence) Outside (northeast of fence)	Categorical: Inside or outside	Point	DPIRD (2018)
Environmental				
Terrain ruggedness	The standard deviation of elevation within a 5-km radius.	Continuous: 1.31–65.4	Point	(Jarvis *et al.* 2008)
Vegetation cover	Mean tree cover within a 5 km radius.	Continuous: 0–33.7%	Point	(Hansen et al. 2003)
Rainfall	Rainfall decile in the 12 months prior to survey	Continuous calculated as a decile value‡: 1–10	LGA	DSITI (2021)
Lagged-rainfall	Rainfall decile lagged 12 months (12-24 months prior to survey)	Continuous calculated as a decile value‡: 1–10	LGA	DSITI (2021)
Total Standing Dry Matter (TSDM)	Average TSDM in previous 12 months	Continuous: 23.6–2479 kg DM/ha	LGA	DSITI (2021)
Lagged-TSDM	Average TSDM in lagged previous 12 months (12 to 24 months prior to survey)	Continuous: 23.6–2533 kg DM/ha	LGA	DSITI (2021)
Previous kangaroo density	Individuals per km^2 in the previous year	Continuous: RK (0.18–3.05 ind./km^2) GK (0.04–2.21 ind./km^2) E (0.03–0.19 ind./km^2)	Kangaroo Monitoring Zone (Central or Southeast)	DBCA (2018)

Abbreviations: Local Government Authority (LGA), Western Australian Department of Primary Industries and Regional Development DPIRD, Western Australian Department of Biodiversity, Conservation and Attractions (DBCA), Queensland Department of Science, Information Technology and Innovation (DSITI).

† LGA: Data extracted for the whole Local Government Authority, 33 LGAs within the study site; point: data extracted for each 5-km length of transect, Kangaroo monitoring zone: Data extracted for each of the two monitoring zones analysed.

‡ to correct for differences in average rainfall between regions. As surveys were generally conducted during June or July, the total rainfall between July and June the previous year was calculated for every year since 1900 based on data from AussieGRASS (available from https://www.longpaddock.qld.gov.au/aussiegrass/). The AussieGRASS environmental calculator

(Carter *et al.* 2000) is a continental scale spatial implementation of the GRASP daily time-step pasture production and water balance model (Rickert *et al.* 2000) using daily climate data from SILO (Jeffrey *et al.* 2001). Then, for each year in the current survey, the rainfall in the 12 months leading up to the survey (July to June) was ranked according to the distribution of rainfall total in the same period since 1900 (e.g. a value of 1 indicated the total rainfall was in the 1st decile, and a value of 10 indicated rainfall was in the 10th decile).

The density of kangaroos in the previous calendar year was calculated based on the population estimate of the DBCA in the previous year

TABLE S2.

Standardised correction values, used during surveys of Western Australia to correct for vegetation and temperature (eg. Department of Biodiversity Conservation and Attractions 2018).

Vegetation correction factors					
Species	Open vegetation	Light vegetation	Medium vegetation	Dense vegetation	Reference
Red	2.29	2.36	2.43	2.57	(Caughley *et al.* 1976)
Western grey	4.8	4.8	4.8	4.8	(Pople *et al.* 1999)
Euro	4.8	4.8	4.8	4.8	(Pople *et al.* 1999)

Temperature correction factors			
Temperature	**Correction factor**	**Temperature**	**Correction factor**
1 °C – 15 °C	1	26 °C	1.53
16 °C	1.03	27 °C	1.61
17 °C	1.07	28 °C	1.70
18 °C	1.10	29 °C	1.79
19 °C	1.14	30 °C	1.90
20 °C	1.19	31 °C	2.02
21 °C	1.23	32 °C	2.16
22 °C	1.28	33 °C	2.32
23 °C	1.34	34 °C	2.50
24 °C	1.40	35 °C	2.72
25 °C	1.46	36 °C	2.97

TABLE S3.

Combinations of independent variables that were excluded from dredge analysis due to collinearity (VIF > 3).

Red **Full dataset**	**Grey**	**Euro**
RCP & Density of dingoes	Density of dingoes & RCP	Density of dingoes & Livestock type
TSDM & Lagged-TSDM	Density of dingoes & State Barrier Fence Density of dingoes & Type Density of dingoes & Livestock density	TSDM & Lagged-TSDM

Red	Grey	Euro
Full dataset		
	State Barrier Fence & Type	
	State Barrier Fence & Prev.dense1	
	State Barrier Fence & Livestock density	
	TSDM & Lagged-TSDM	
State Barrier Fence subset		
Density of dingoes & RCP	Previous kangaroo density & RCP	Density of dingoes & RCP
TSDM & Lagged-TSDM	Previous kangaroo density & Lagged-TSDM	TSDM & Lagged-TSDM
TSDM & Livestock density	Previous kangaroo density & Density of dingoes	
	RCP & Lagged-TSDM	
	RCP & Density of dingoes	
	Density of dingoes & State Barrier Fence	
	Density of dingoes & Lagged-TSDM	
	TSDM & Lagged-TSDM	
	TSDM & Previous kangaroo harvest	

CONCLUSION

In the southern rangelands of Western Australia, the ability to run small stock (i.e. sheep and goats) has been lost due to high levels of predation by wild dogs, along with the collapse in the wool price (Forsyth et al. 2014). Around Australia, wild dog predation costs the livestock industry millions of dollars per annum through direct predation events, mis-mothering, bite-marks, stress and other welfare issues for livestock. Many methods are used to mitigate predation impacts on the livestock industry including baiting, trapping and shooting; but in Western Australia, large investment has gone into cell-fencing pastoral country for producing livestock without predation pressure. Historically, the southern rangelands was sheep country – but wild dog impacts have made this untenable. Many producers have tried changing enterprises or increasing their off-station income. However, the desire to and possibility of continuing to run sheep in the southern rangelands is an idea strongly held by many producers in the area.

The aim of this project was to assess the effectiveness of cell fencing to enable producers to again run small stock in the southern rangelands. Many factors affect the likelihood of this such as the ability to remove wild dogs from within the cell-fenced area, and competition between the native and non-native herbivores grazing. Biodiversity of the cell-fenced area is also very important to maintain and improve. Lastly, the bioeconomics of cell fencing in Western Australia is highly relevant to this study; landholders and government departments need to ensure that profits are made from the large fencing projects.

Camera traps and GPS trackers were employed to monitor the numbers of wild dogs, small stock, macropods and other native species inside and outside the cell-fenced area. GPS trackers were worn by the small stock to investigate changes in their movement as wild dogs were removed from the landscape. Not only did landholders and licensed pest management technicians undertake landscape baiting, trapping and shooting to reduce wild dog numbers, but researchers also undertook a landscape-scale assessment of canid pest ejectors to remove wild dogs from the area. Camera traps were also employed to investigate the biodiversity of the cell-fenced area inside and outside of the large MRVC and smaller MHC.

Before their removal from the paddock, wild dogs had a significant impact on the movement and survival of sheep – despite the sheep being agisted at the study’s end due to a lack of rainfall. Sheep took longer to return to water points in the presence of wild dogs. Sheep also changed their movement after the southern end of the cell was completed. In the portion of the cell yet to be completed, sheep were likely all predated by wild dogs. Goats are slightly more robust to wild dog predation; however, in the absence of wild dogs, goat recruitment increased.

Competition for grazing is not only difficult to estimate, but to manage, as macropod numbers/events were not related to any variables studied in the project, nor did macropod numbers vary much within the cell. Emus were recorded as being prone to wild dog predation. Feral cats were unchanged despite the reduction in wild dog numbers, which again indicates that mesopredator release is not an ecosystem function in Western Australia (similar to other studies in Western Australia such as Kreplins et al. 2021). Other native species are continually being evaluated and will be the topic of a PhD thesis.

The bioeconomic evaluation of the cell fences for the entirety of Western Australia demonstrated that the magnitude of increase in livestock weaning rate following eradication of wild dogs is the most important factor determining the return on investment in cell fencing. While previous studies have predicted the increase in livestock weaning rate following wild dog eradication using speculative and optimistic estimates, the bioeconomic analyses showed that the projected the livestock weaning rate using historical livestock records from 1985 to 1995, the period prior to reports of significant wild dog impacts in the region. As many previous estimates of return on investment assumed a greater increase in weaning rate following cell fencing than was used in the current study, we argue that estimates of expected benefits of cell fencing in the southern rangelands may have been overstated. Unsurprisingly, small stock, particularly wool sheep, reap the most benefit from cell fencing, as wild dog impacts small stock more dramatically than cattle. In all beef options, the investment in cell fence construction, maintenance and dingo eradication is not outweighed by the increase in revenue. The bioeconomic assessment data was collected from landholder interviews and stations records as well as Pastoral Lands boards' annual returns data.

Station owners, Recognised Biosecurity Groups, other agencies (e.g. WA Department of Biodiversity, Attractions and Conservation), and pest management technicians were incorporated into many stages of the project, its results and ongoing outcomes from the project. For example, the canid pest ejector–control of wild dogs was published, presented at national conferences, and is a flyer now used by landholders and other agencies to inform control work.

Despite the cell fence being incomplete, its future success looks bright. If the fence can be maintained and wild dog numbers keep decreasing, there is no reason that sheep production cannot return to the MHC. MRVC has a 'long road ahead' to reduce wild dog numbers over its large expanse.

ACKNOWLEDGMENTS

CHAPTER 1

Many thanks to the Nichols and Jones families for allowing the research to be conducted on their properties. Significant amounts of time were spent by Glen Coupar, Jim Miller, Ben Kreplins, Peter Adams and John-Michael Stuart to conduct this work. Thanks to Claire Macleay for the use of the weight crate. This study was completed with the Department of Primary Industries and Regional Development Animal Ethics Committee approval (18/06/17).

CHAPTER 2

Many thanks go to all the field assistances, Glen Coupar, Simon Merewether, Madgalena Zabek, Paul Pitaro, Stuart Dawson, John-Michael Stuart and Moses Omogbeme. Thanks to Greg Mifsud, National Wild Dog Action Plan facilitator, for the advice on CPE lure head construction. A special thanks to Ben Kreplins for altering the CPEs to function in all terrain types with the auger addition to the CPEs. This study was completed with the Department of Primary Industries and Regional Development Animal Ethics Committee approval (18/04/08).

CHAPTER 3

The authors would like to thank Sam Giles, Kate Pritchett, Jim Miller, Michael Jones, Craig Robins and Mark Lethbridge for their help. We would like to acknowledge the valuable comments of the two reviewers.

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*Note: bold references indicate project outputs. Other CISS and IA CRC publications are marked with an **

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