

Two alternate states: shrub, bird and mammal assemblages differ on either side of the Dingo Barrier Fence

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ABSTRACT

The 5500 km long dingo barrier fence (DBF) is a boundary at which the goal of dingo control programs shifts from management to elimination. Since 1980 ecologists have used the discrepancies in dingo densities across the DBF to study the ecological role of Australia's largest terrestrial predator.

We used drone imagery, ground based shrub and tree counts, and camera trap footage to test our hypothesis that there are alternate states in plant, bird and mammal assemblages on either side of the DBF. We found that shrubs and trees were twice as dense where dingoes were rare, and 28 % of shrub and tree species, 78 % of mammal species, and 14 % of bird species recorded were significantly more likely to occur on one side of the DBF than the other.

We provide the first comprehensive snapshot of how flora and fauna assemblages differ across the DBF. This study adds to literature demonstrating that the removal of the dingo has led to profound shifts in the shrub, mammal and bird assemblages in arid Australia. Any expansion of dingo control in arid Australia must be considered against the far-reaching consequences for ecosystem assembly associated with the removal of a top predator.

Key words: Trophic cascades; camera trap; apex predator; human-wildlife conflict; functional extinction; mesopredator release; ecosystem restructuring; Canis dingo; cluster fencing.

DOI: <https://doi.org/10.7882/AZ.2021.005>

Introduction

Canis dingo (hereafter dingo) was introduced to Australia approximately 3500-5000 years ago (Savolainen *et al.* 2004) and is mainland Australia's largest terrestrial predator. Debate surrounds the status of the dingo in Australia: dingoes prey on sheep and cattle and therefore are considered a pest by livestock farmers (Allen and Fleming 2004); however, dingoes also play a crucial part in Australia's ecosystem function (Letnic *et al.* 2012) and culture (Trigger *et al.* 2008). Although there remains debate about the dingo's role and rights in Australia, current legislation promotes the persecution of the dingo in south-eastern Australia (Fleming *et al.* 2006). In Queensland, New South Wales, Victoria and South Australia dingoes are consistently controlled using poison, shooting and barrier fencing. In contrast, other jurisdictions, such as the Northern Territory, list dingoes as a protected species (Northern Territory Government of Australia 2020).

The dingo barrier fence (hereafter DBF) is a 5500 km fence built in the first half of the 20th century. The

DBF was built along the border between New South Wales (NSW) and South Australia (SA) to prevent the immigration of dingoes from the rest of the continent into sheep-grazing land in south-eastern Australia (McKnight 1969) where lethal control of dingoes is common practice (Fleming *et al.* 2006). The NSW-SA border was defined arbitrarily along the meridian 141° E by decree of King William Henry IV, and therefore the location of the DBF in NSW is an arbitrary boundary and does not represent any landscape features.

Barrier fences and other large fences fragment landscapes and impact non-target species by obstructing their migration and movement (Bradby *et al.* 2014; McInturff *et al.* 2020; Smith *et al.* 2020a). For ecologists, the erection of the DBF created a large manipulative experiment where on one side the apex predator is almost entirely absent, and on the other side the apex predator is abundant although livestock farmers have sporadically practiced lethal control throughout these areas.

Since the 1970s, ecologists have exploited the differences in dingo densities created by the DBF to understand the functional role of the dingo in the landscape (Caughley *et al.* 1980; Pople *et al.* 2000; Newsome *et al.* 2001; Letnic and Koch 2010). Caughley *et al.* (1980) was among the first to quantify significant discrepancies in dingo, emu *Dromaius novaehollandiae* and kangaroo *Macropus* spp. and *Osphranter rufus* population densities across the DBF and hypothesised that the high densities of emus and kangaroos observed in NSW were directly attributable to the absence of the top-down control of dingo predation. This hypothesis has since been widely supported in empirical studies (Caughley *et al.* 1980; Pople *et al.* 2000; Letnic and Koch 2010).

More recently, research has focussed on the indirect effects that removal of dingoes has on ecosystems (Letnic *et al.* 2012). The high abundances of kangaroos associated with dingo removal (Robertshaw and Harden 1986; Newsome 1990) have been shown to result in reduced grass cover (Rees *et al.* 2017) and depleted soil nutrients (Morris and Letnic 2017) through trophic cascades. Increased kangaroo abundances in the absence of predators also leads to higher abundances of carcasses, favouring scavenging bird species (Fillios *et al.* 2010; Rees *et al.* 2019; Rees *et al.* 2020). Other species of large invasive herbivores such as donkeys *Equus asinus*, goats *Capra aegagrus*, and pigs *Sus scrofa* have been demonstrated to persist in higher abundances in areas of intensive lethal dingo control (Corbett 1995; Newsome *et al.* 2001; Wallach *et al.* 2010).

The indirect effects of dingo removal on community assembly also arise through the release of mesopredators such as cats *Felis catus* and red foxes *Vulpes vulpes* from top-down control of the apex predator (Johnson and VanDerWal 2009; Brook *et al.* 2012; Feit *et al.* 2019). This mesopredator release and subsequent increase in predation pressure has been correlated with a reduction in the diversity and abundance of small (< 2 kg) and medium-sized (2-5 kg) mammals (Johnson *et al.* 2007; Letnic *et al.* 2009) and reduced abundances of ground-

nesting and low-nesting birds (Gordon *et al.* 2017b; Rees *et al.* 2019). Although the diets of dingoes, foxes and cats all include small and medium-sized mammals such as rodents and rabbits, dingoes will readily prey on large mammals (Paltridge 2002; Cupples *et al.* 2011; Woinarski *et al.* 2018; Doherty *et al.* 2019). An observed increase in shrubs in south-eastern Australia in the past 100 years has also been attributed to an absence of small and medium-sized granivorous mammals resulting from mesopredator release (Gordon and Letnic 2016; Gordon *et al.* 2017a; Mills *et al.* 2018).

In this study we used drone imagery, ground-based shrub and tree counts and camera trap photos to test our hypothesis that alternate states in plant, bird and mammal communities have developed on either side of the DBF. Our study also aims to provide the first assessment of woody shrub community assembly across the DBF and the first comprehensive snapshot of how flora and fauna assemblages differ across the DBF. Our specific predictions are presented in Table 1.

Methods

Study Region

We conducted drone surveys, shrub community surveys, and camera trapping in sites located across the DBF, in sites where dingoes were common and where dingoes were rare (Figure 1). The assignment of DBF treatments to differences in dingo abundance (i.e. “dingoes rare” (NSW, Victoria and southern SA) and “dingoes common” (western Queensland, central and northern SA)) is based on long-term research conducted in the study region (Letnic and Crowther 2013; Letnic *et al.* 2016; Gordon *et al.* 2017b; Morris and Letnic 2017). The exact location of study sites differed for each survey and varied in land-use type and history, and included cattle and sheep grazing properties and conservation reserves. Details of study sites are available in Supplementary Material Table S1. Sites in each dataset were paired for

Table 1 Hypothesised differences in fauna and flora communities across the DBF. Increases and decreases are in areas where dingoes were rare relative to areas where dingoes were common.

Increase where dingoes rare	Decrease where dingoes rare
<ul style="list-style-type: none"> Density of woody shrubs and trees overall (Gordon <i>et al.</i> 2017a). Densities of all species of tree and shrub measured (Mills <i>et al.</i> 2018). Probability of detecting mesopredators on camera traps (Letnic <i>et al.</i> 2011). Probability of detecting scavenging bird species on camera traps (Rees <i>et al.</i> 2019). Probability of detecting large mammals and emus on camera traps (Caughley <i>et al.</i> 1980, Wallach <i>et al.</i> 2010). 	<ul style="list-style-type: none"> Probability of detecting dingoes on camera traps (Letnic and Koch 2010). Probability of detecting small and medium-sized mammals on camera traps (Letnic <i>et al.</i> 2009). Probability of detecting ground-nesting birds on camera traps (Gordon <i>et al.</i> 2017b, Rees <i>et al.</i> 2019).

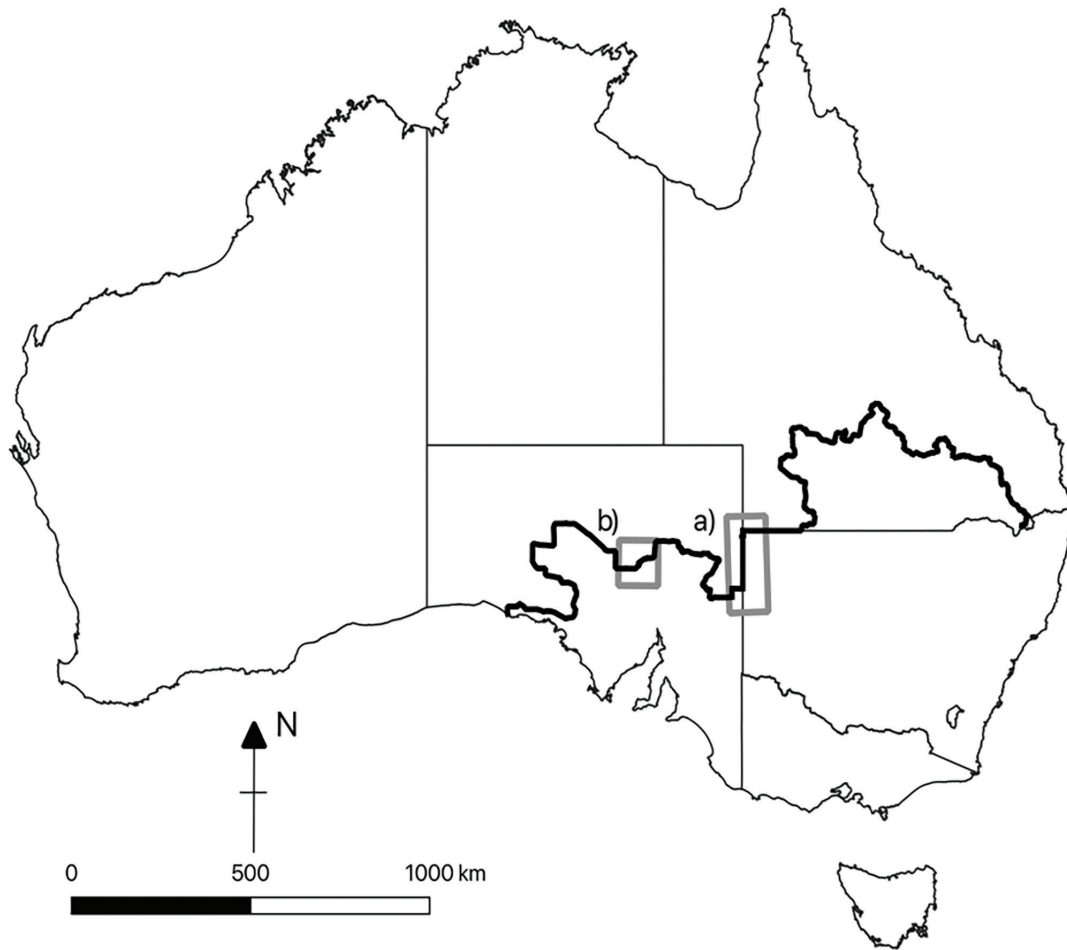


Figure 1 Study region locations (grey squares): a) Strzelecki Desert and b) Moondiepitchnie Dunefield. The DBF is indicated by the bold line.

land-use type. Although the land-use history differed between different paired sites, all of the survey sites were within the Strzelecki Desert and the Moondiepitchnie Dunefield in the arid zone of Australia (annual rainfall < 250 mm) and are characterised by similar longitudinal sand dunes interspersed with clay swales. Dominant vegetation in the study site includes perennial shrubs and ephemeral forbs and grasses (Keith 2006).

Drone Imagery- Shrubs and Trees

To compare the densities of shrubs and trees on sand dunes either side of the DBF we flew a drone (DJI Phantom 3 Pro quadcopter) along 12 dunes which were intersected by the DBF in March 2017. We sampled dunes 2 km apart and used the drone to capture imagery for 300 - 400 m along the same dune either side of the DBF using a paired design. Using the software Pix4D we generated an image orthomosaic with a pixel size of 3 cm from the drone imagery. We calculated the density of trees and shrubs per m² on each of the 12 surveyed dunes by placing 20 parallel transect lines covering equal areas on either side (dingoes common; dingoes rare) of the DBF and manually counting all shrubs and trees in the area encompassed by the transects. We only counted adult shrubs and trees with a crown diameter of at least 1 m.

To compare the density of trees and shrubs on dunes across DBF treatments we used a one-way ANOVA. Further detail on drone image collection and processing methodology and specific locations of dunes is available in Lyons *et al.* (2018).

Ground Surveys – Shrubs and Trees

We used a Bitterlich gauge (Friedel and Chewings 1988) to determine the tree and shrub assemblages on eight sand dunes at four sites where dingoes were rare ($n = 32$) and four sites where dingoes were common ($n = 32$) between 2006 - 2008. Shrub and tree cover estimates produced using the Bitterlich gauge are proportional to shrub abundance (Friedel and Chewings 1988). A Bitterlich gauge (75 cm long rod with 7.5 cm crossbar) was used to estimate cover for all live species of shrub or tree at three points on the top of dunes and three points at the base of dunes, with points separated by a 50 m interval (Gordon & Letnic, 2015). We calculated the cover estimate for each species on each dune using the mean of the six samples and used this cover estimate in our community analyses.

Camera Trapping – Mammals and Birds

To detect the presence of mammal and bird species on either side of the DBF at the intersection of NSW, SA

and Queensland, we used trail cameras at four sites where dingoes were rare for 773 trap nights and four sites where dingoes were common for 1073 trap nights between 2009 and 2015. The trail cameras were set to trigger when movement was detected capturing any mammals or birds on camera. The cameras were placed on water points (troughs or dams), under shade cloth, with foraging trays or in the open. The number of trap nights with shade cloth and foraging trays were equal on either side of the DBF. We removed species which were only recorded for one trap night. Each species was recorded as present or absent for each trap night per camera trap and the resulting presence-absence data was used for the community analysis.

Community Data Statistical Analysis

We ran separate multivariate generalized linear models for each dataset to test for differences in shrub and tree, mammal, and bird community assemblages across the DBF treatment. The multivariate generalized linear models used the model-based approach from the 'mvabund' package in R v3.6.2 (Wang *et al.* 2012). We included an interaction effect of DBF treatment with site as fixed factors to account for different land use histories and fitted a negative binomial distribution to the shrub and tree data as it was the best fit for count data, and a binomial distribution for the presence-absence data of our mammal and bird camera data. We used an ANOVA with likelihood-ratio tests and Monte Carlo resampling with 999 samples to look for significant effects of DBF treatment, site, and the interaction of DBF treatment on the community data. For the mammal and bird camera trap data, we also included presence of water (if the camera was placed on a dam or trough) and its

interaction with DBF treatment as explanatory variables because of the varying original objectives of the camera trap data. To run univariate tests for individual species against the fixed factors we used adjusted resampling within the above ANOVA and reported alpha level = 0.05 as our significance threshold.

We determined the effect size of DBF treatment for each species by subtracting the mean of each species (shrub and tree cover; mammal and bird probability of occurrence) where dingoes were rare from the mean of each species where dingoes were common. To calculate the mean probability of occurrence for each bird and mammal species at each site we divided the total nights the species was observed by the trapping effort for that DBF treatment. We used package 'ggplot2' v3.2.1 in R v3.6.2 (Wickham 2011) to generate figures.

Results

Drone Imagery- Shrubs and Trees

There were approximately twice as many shrubs and trees on dunes where dingoes were rare compared to dunes where dingoes were common (one-way ANOVA; $F_{1,22} = 10.36$, $p < 0.005$; Figure 2a).

Ground Surveys – Shrubs and Trees

We encountered four species of tree and three species of shrub on dunes during Bitterlich gauge surveys. There was a significant difference between shrub and tree cover across the DBF ($LRT_{61,1} = 39.0$, $P = 0.002$, Figure 2b) and across sites ($LRT_{54,7} = 229.5$, $P = 0.001$; Figure S1)

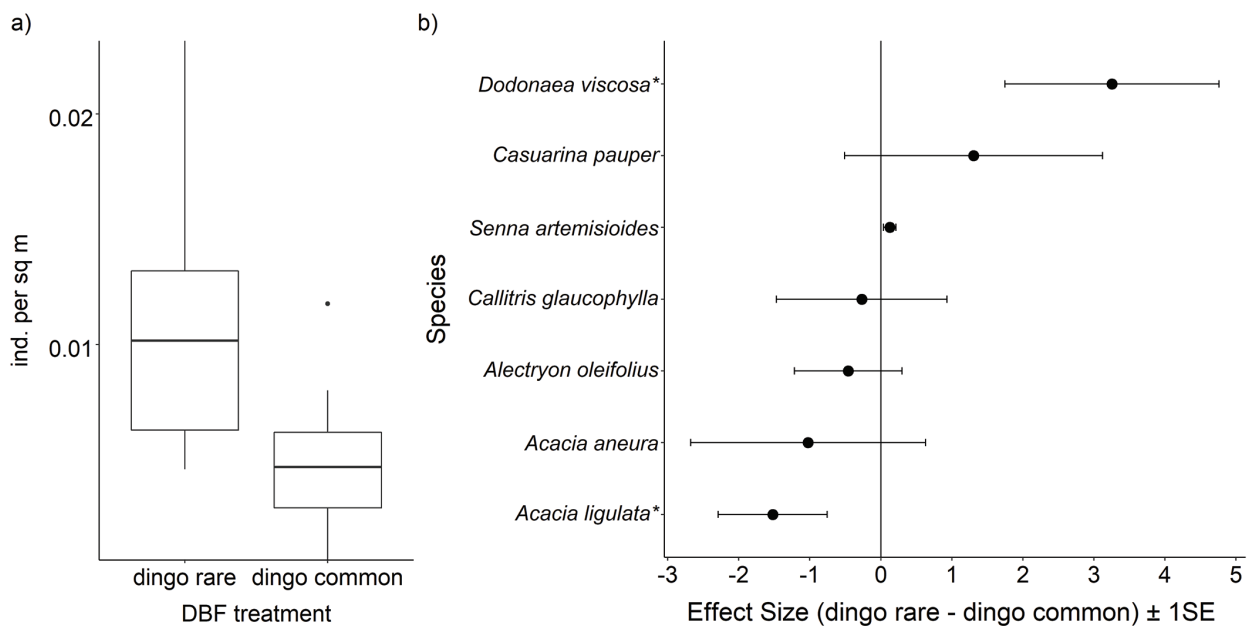


Figure 2 Shrub and tree indices across the Dingo Barrier Fence: a) boxplot of shrub density where dingoes were rare and dingoes common and b) differences in effect size for different shrub species across the DBF. Asterisks indicate species that had significant differences in cover across the DBF according to the generalised linear models. Species with positive effect sizes are more likely to occur where dingoes were rare and species with negative effect sizes are more likely to occur where dingoes were common.

and there was no interaction between DBF treatment and site ($LRT_{55,1} = 0$, $P = 0.205$). *Dodonaea viscosa* subsp. *angustissima* cover was greater where dingoes were rare ($LRT = 14.5$, $P = 0.004$) and *Acacia ligulata* cover was greater where dingoes were common ($LRT = 13.3$, $P = 0.006$), while cover of all other species did not differ with DBF treatment. Cover of all species was different for site ($P < 0.05$; more details in Table S2).

Camera Trapping – Mammals and Birds

Nine species of mammals were detected on the cameras. Mammal assemblages differed significantly on either side of the DBF ($LRT_{1,752} = 797.1$, $P = 0.001$), between sites ($LRT_{7,745} = 210.3$, $P = 0.001$; Figure S2) and with the presence of water ($LRT_{1,744} = 46.2$, $P = 0.001$). There was no interaction between DBF treatment and site ($LRT_{1,744} = 0.4$, $P = 0.07$) nor DBF treatment and water ($LRT_{1,743} = 3.9$, $P = 0.16$). Of the nine species, seven had a significantly higher chance of occurring on one side of the DBF than the other (Figure 3, Table S3). More dingoes were recorded on the western Queensland, central and northern SA side of the DBF ($LRT = 8.9$, $P = 0.009$), consistent with our assignment of these sites as areas where dingoes are common. Kangaroos *Macropus spp.* and *Osphranter rufus* ($LRT = 315.7$, $P = 0.001$), red foxes *Vulpes vulpes* ($LRT = 49.4$, $P = 0.001$) and pigs *Sus scrofa* ($LRT = 6.1$, $P = 0.03$) had higher chances of occurring where dingoes were rare, while Long-haired Rats *Rattus villosissimus* ($LRT = 11.6$, $P = 0.004$), European rabbits *Oryctolagus cuniculus* ($LRT = 34.2$, $P = 0.001$) and Dusky Hopping-mice

Notomys fuscus ($LRT = 368.6$, $P = 0.001$) had higher chances of occurring where dingoes were common. Cats *Felis catus* ($LRT = 0.03$, $P = 0.866$) and Crest-tailed Mulgaras *Dasyercus cristicauda* ($LRT = 2.3$, $P = 0.302$) did not have higher chances of occurring on either side of the fence. Kangaroos were the only mammal that differed with presence of water, and had a higher chance of occurring when water was present ($LRT = 17.4$, $P = 0.001$). Full model output is available in Table S3.

Twenty-one species of birds were detected on the cameras (Table S4). Bird assemblages differed significantly on either side of the DBF ($LRT_{1,752} = 161.3$, $P = 0.001$), between sites ($LRT_{7,745} = 359$, $P = 0.001$; Figure S3) and with the presence of water ($LRT_{1,744} = 241.5$, $P = 0.001$). There was no significant interaction between DBF treatment and site ($LRT_{1,743} = 0$, $P = 0.8$) although there was an interaction between DBF treatment and water ($LRT_{1,744} = 23.8$, $P = 0.005$). Of the 21 species, three had a significantly higher chance of occurring on one side of the DBF compared to the other (Table S4). Emus *Dromaius novaehollandi* ($LRT = 46.7$, $P = 0.001$) and Australian magpies *Gymnorhina tibicen* ($LRT = 18.4$, $P = 0.001$) had higher chances of occurring where dingoes were rare while Cinnamon quail-thrush *Cincoloma cinnamomeum* ($LRT = 37.9$, $P = 0.001$) had higher chances of occurring where dingoes were common. Emus ($LRT = 38.3$, $P = 0.001$), magpie-larks *Grallina cyanoleuca* ($LRT = 26.09$, $P = 0.001$), Little corellas *Cacatua sanguinea* ($LRT = 17.63$, $P = 0.001$), and ducks (Family *Anatidae*; $LRT = 8.15$, $P = 0.05$) were more likely to occur in the presence

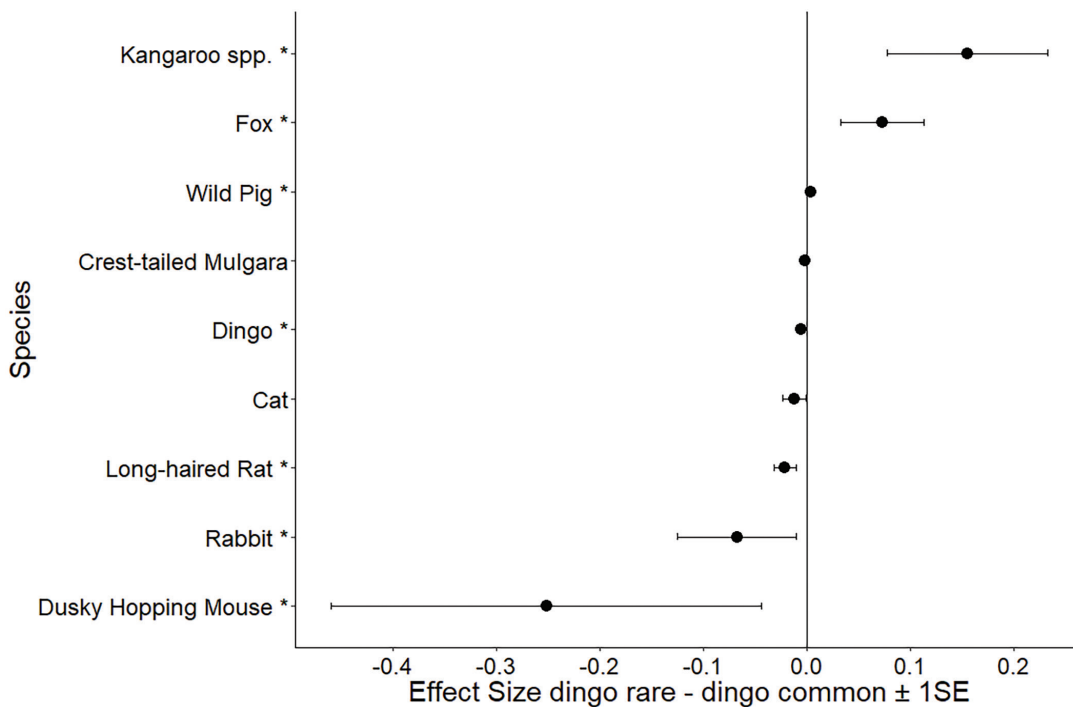


Figure 3 Effect size of the probability of occurrence of mammal species where dingoes were rare compared to where dingoes were common. Asterisks indicate species that are significantly more likely to occur in on one side of the DBF than the other according to the generalized linear models. Species with positive effect sizes are more likely to occur where dingoes were rare and species with negative effect sizes are more likely to occur where dingoes were common.

of a water body while Cinnamon quail-thrush (LRT = 10.7, P = 0.02) were less likely to occur in the presence of water. There was no significant interaction between water and DBF treatment for any individual species. Full model output is available in Table S4.

Discussion

The picture that emerges from our results is that plant, mammal, and bird assemblages are abruptly different across the DBF (see Figure 4 for artist’s interpretation of the resulting alternate states). The density of shrubs and trees on sand dunes was two times greater in areas where dingoes were rare and of the species recorded in our surveys, 28 % of shrub and tree species, 78 % of mammal species, and 14 % of bird species were significantly more likely to occur on one side of the DBF than the other. Our findings provide evidence that widespread efforts to extirpate the dingo from south-eastern Australia have, directly or indirectly, led to dramatic shifts and the creation of alternate states in the flora and fauna assemblages of desert dunefields on either side of the DBF (Caughley *et al.* 1980; Newsome *et al.* 2001; Letnic and Koch 2010).

Although our results support the prediction that where dingoes were rare the density of shrub and tree species would be greater (Gordon *et al.* 2017a; Lyons *et al.* 2018), we found two opposing responses in the species that differed in cover across the DBF. The opposing responses observed in *Acacia ligulata*, which had greater cover where dingoes were common, and *Dodonaea viscosa* subsp. *angustissima*, which had greater cover where dingoes were rare, warrant further investigation. *Dodonaea viscosa* subsp. *angustissima* occurs in dense stands, and is considered a native “woody weed” (Noble *et al.* 2007) which potentially outcompetes *A. ligulata* shrubs. As the species demonstrating the highest discrepancy in density across the DBF, *D. viscosa* was the main contributor to the overall higher density of woody vegetation in the absence of dingoes. High densities of *D. viscosa* have been linked to the functional extinction of granivorous mammals, particularly where dingoes are rare (Gordon and Letnic 2016). We therefore suggest that avenues for further research on this topic include the possibility that other shrub and tree species outcompete *Acacia ligulata* where dingoes are rare, or that *A. ligulata* bears some advantage over other species in areas with high rates of granivory such as has been observed in our sites where dingoes are common (Gordon and Letnic 2016).

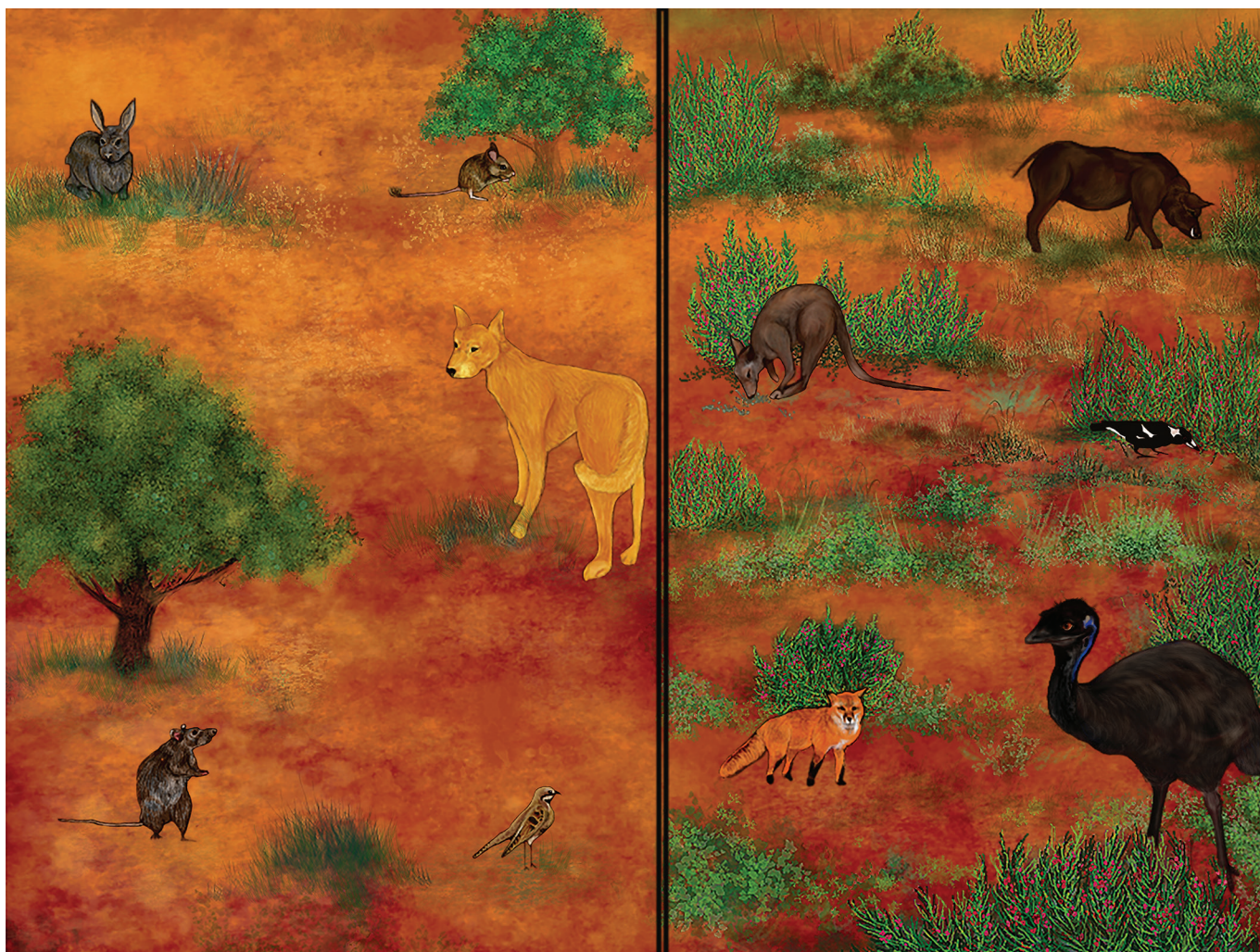


Figure 4 Artist’s interpretation highlighting the alternate states described in this paper. The line in the centre represents the DBF (Artist: Corrine Edwards).

Overall, the differences we observed in mammal and bird assemblages are generally consistent with previous work conducted within the study region (Caughley *et al.* 1980; Newsome *et al.* 2001; Letnic *et al.* 2012; Gordon *et al.* 2017b; Rees *et al.* 2019) and with our hypotheses. Dingoes were more likely to be detected at sites we defined as dingo common indicating that our assignment of sites was appropriate. As predicted and consistent with previous studies large mammals and emus were more likely to occur in areas where dingoes were rare (Caughley *et al.* 1980; Newsome 1990; Corbett 1995; Letnic *et al.* 2009). Our hypothesis that mesopredators would be higher in the absence of dingoes was only partially supported, as red fox numbers differed but cats did not. Although there is consistent evidence that dingoes control red foxes (Letnic *et al.* 2011), the effect of dingoes on cat populations is less clear (Kennedy *et al.* 2012; Hunter *et al.* 2018). In our study region, a long-term study conducted during periods of both wet and dry climatic conditions found that cats were more abundant where dingoes were rare and that cat abundance increased with the availability of small mammal prey, particularly at sites where dingoes were rare (Feit *et al.* 2019). In contrast, a study conducted during a period of wetter climatic conditions and an introduction of dingoes (Moseby *et al.* 2019) found that dingoes had no effect on indices of cat abundance. Taken together, these contrasting findings suggest that cat abundances in our study region may be influenced by both the top-down effect of dingoes and the availability of prey, particularly rodents and rabbits (Feit *et al.* 2019).

Small mammals, medium-sized mammals and a ground-nesting bird were all more likely to occur in areas where dingoes were common, which is consistent with our hypotheses and previous literature (Johnson *et al.* 2007; Gordon *et al.* 2017b; Rees *et al.* 2019). We recorded no differences in scavenging bird species across the DBF, which is at odds with several other studies finding that increased carcass availability resulting from high herbivore abundances favours scavengers (Fillios *et al.* 2010; Rees *et al.* 2019). We suspect that differences between our results and those cited are due to the non-targeted approach to our camera trap surveys.

Our study spanned a large spatial scale (sites were up to 400 km apart) and therefore was subject to considerable natural environmental variation. This variation is reflected by differences in assemblage composition between the study sites for all three taxa. As the DBF was originally built to reduce the risk of dingoes killing sheep, differences in the density and types of livestock present on either side of the DBF are the intentional result of the DBF and therefore impossible to disentangle from the presence or absence of the dingo. To account for differences in land-use our sites were paired and included conservation areas and areas exclusively grazed by cattle on both sides of the DBF. Our results show that the differences we attribute

to the DBF were consistent across sites and water treatments. The absence of interaction effects suggests that our paired study design effectively controlled for confounding factors which may influence the mammal, bird, and shrub assemblages, such as the presence of artificial water bodies and grazing management (Caughley *et al.* 1980; Rees *et al.* 2019).

As an observational study, it is beyond the scope of our data to determine the mechanisms behind the results we report. However, our findings are consistent with an experimental manipulation of dingo abundance which showed that dingoes suppressed kangaroos and foxes and benefited small mammals and rabbits (Moseby *et al.* 2019). Also consistent with previous studies, our results show that the DBF is an effective boundary to dingo populations (Caughley *et al.* 1980; Pople *et al.* 2000; Newsome *et al.* 2001; Letnic and Koch 2010) and shifts in ecosystem assembly similar to those which we report have been connected to dingo suppression in forest and tropical savannah ecosystems in Australia (Colman *et al.* 2014; Leo *et al.* 2019). Indeed, there is evidence that the alternate states across the DBF extend to invertebrate communities (Contos and Letnic 2019), reptiles (Tong 2017) and dune geomorphology (Lyons *et al.* 2018). When our results are considered alongside studies that investigated the mechanisms by which dingo removal restructures ecosystems (Johnson *et al.* 2007; Greenville *et al.* 2014; Gordon *et al.* 2017a; Leo *et al.* 2019) they support the hypothesis that the alternate states observed across the DBF are a consequence of the presence or absence of the dingo on either side of the DBF.

At the time of writing, there are programs in south-eastern Australia and Western Australia which aim to increase dingo control through: more frequent use of poison (South Australian Wild Dog Advisory Group 2016); extending barrier fences (Anonymous 2020; Stadler 2020); and the construction of cluster fencing (Clark *et al.* 2018) where livestock farmers collaborate on large scale fencing projects which exclude large wild herbivores and dingoes. Most of these programs are situated in areas “inside” the existing dingo fence where dingo populations are already subject to intensive population control (Anonymous 2020; Smith *et al.* 2020b). For instance, the extension of the DBF in NSW aims to more than double the DBF, extending it 742 km along the currently unfenced sections of the NSW border and dividing areas which are already within the dingo exclusion area of the existing DBF (Anonymous 2020). The extension of the DBF in NSW and cluster fencing projects in northern NSW and Queensland will fragment the landscape and prevent the movement and migration of non-target wildlife that cannot traverse the fences (Bradby *et al.* 2014; Smith *et al.* 2020a). As large areas of land where herbivores and predators are controlled, cluster fences may confer benefits to any threatened wildlife living within them (Smith *et al.* 2020a). However, cluster fencing projects

are in their infancy and long-term costs and benefits are yet to be seen.

Our study adds to a body of work demonstrating that efforts to control the dingo and the presence of the DBF are associated with unintended and profound shifts in the shrub, mammal and bird assemblages in Australia (Wallach *et al.* 2010; Gordon *et al.* 2017a; Rees *et al.*

2019). The DBF has provided unique insights into the ecological role of the dingo in arid Australia and more broadly the fundamental role that apex predators play in shaping ecosystems. Any expansion of dingo control in arid Australia must be considered against the far-reaching consequences for ecosystem assembly associated with the removal of a top predator.

Acknowledgements

Our thanks go to the station owners and managers at Lindon, Quinyambie, Winnathee, Lake Stewart, Omicron, and Mullyungarie, and to NPWS at Sturt National Park for providing access to the study sites. We also thank Sarah

Allison and James Rees for data collection assistance and Corrine Edwards for the smashing illustration in Figure 4. We are grateful to Thomas Newsome and two anonymous reviewers for constructive comments on the manuscript.

Contributions

CEG, M Letnic, CHM, AF and M Lyons collected data, BW, CEG, M Letnic, AF and AW processed data, CHM and BW analysed data and led the writing of the manuscript. All authors gave final approval for publication.

Funding

M Letnic was funded by the Margaret Middleton Fund of the Australian Academy of Science and Australian Research Council (ADP0666574). M Lyons was funded by the Australian Research Council (LP150100972).

References

- Allen, L. and Fleming, P. 2004. Review of canid management in Australia for the protection of livestock and wildlife—potential application to coyote management. *Sheep & Goat Research Journal* **19**: 97-104.
- Anonymous 2020. Construction begins on World's longest wild dog fence. The National Tribune. Accessed 27 October 2020. <https://www.nationaltribune.com.au/construction-begins-on-world-s-longest-wild-dog-fence/>
- Bradby, K., Fitzsimons, J. A., Del Marco, A., Driscoll, D. A., Ritchie, E. G., Lau, J., Bradshaw, C. J. A. and Hobbs, R. J. 2014. Ecological connectivity or Barrier Fence? Critical choices on the agricultural margins of Western Australia. *Ecological Management & Restoration* **15**: 180-190. <https://doi.org/10.1111/emr.12130>
- Brook, L. A., Johnson, C. N. and Ritchie, E. G. 2012. Effects of predator control on behaviour of an apex predator and indirect consequences for mesopredator suppression. *Journal of Applied Ecology* **49**: 1278-1286. <https://doi.org/10.1111/j.1365-2664.2012.02207.x>
- Caughley, G., Grigg, G. C., Caughley, J. and Hill, G. J. E. 1980. Does dingo predation control the densities of kangaroos and emus. *Australian Wildlife Research* **7**: 1-12. <https://doi.org/10.1071/wr9800001>
- Clark, P., Clark, E. and Allen, B. L. 2018. Sheep, dingoes and kangaroos: new challenges and a change of direction 20 years on. University of Queensland, Gatton, Australia. 173-178
- Colman, N. J., Gordon, C. E., Crowther, M. S. and Letnic, M. 2014. Lethal control of an apex predator has unintended cascading effects on forest mammal assemblages. *Proceedings of the Royal Society B: Biological Sciences* **281**: 20133094. <https://doi.org/10.1098/rspb.2013.3094>
- Contos, P. and Letnic, M. 2019. Top-down effects of a large mammalian carnivore in arid Australia extend to epigeic arthropod assemblages. *Journal of Arid Environments* **165**: 16-27. <https://doi.org/10.1016/j.jaridenv.2019.03.002>
- Corbett, L. 1995. Does Dingo Predation or Buffalo Competition Regulate Feral Pig Populations in the Australian Wet-Dry Tropics? An Experimental Study. *Wildlife Research* **22**: 65-74. <https://doi.org/10.1071/WR9950065>
- Cupples, J. B., Crowther, M. S., Story, G. and Letnic, M. 2011. Dietary overlap and prey selectivity among sympatric carnivores: Could dingoes suppress foxes through competition for prey? *Journal of Mammalogy* **92**: 590-600. <https://doi.org/10.1644/10-mamm-a-164.1>
- Doherty, T. S., Davis, N. E., Dickman, C. R., Forsyth, D. M., Letnic, M., Nimmo, D. G., Palmer, R., Ritchie, E. G., Benshemesh, J., Edwards, G., Lawrence, J., Lumsden, L., Pascoe, C., Sharp, A., Stokeld, D., Myers, C., Story, G., Story, P., Triggs, B., Venosta, M., Wysong, M. and Newsome, T. M. 2019. Continental patterns in the diet of a top predator: Australia's dingo. *Mammal Review* **49**: 31-44. <https://doi.org/10.1111/mam.12139>

- Feit, B., Feit, A. and Letnic, M. 2019. Apex Predators Decouple Population Dynamics Between Mesopredators and Their Prey. *Ecosystems* 22: 1606-1617. <https://doi.org/10.1007/s10021-019-00360-2>
- Fillios, M., Gordon, C., Koch, F. and Letnic, M. 2010. The effect of a top predator on kangaroo abundance in arid Australia and its implications for archaeological faunal assemblages. *Journal of Archaeological Science* 37: 986-993. <https://doi.org/10.1016/j.jas.2009.11.031>
- Fleming, P. J. S., Allen, L. R., Lapidge, S. J., Robley, A., Saunders, G. R. and Thomson, P. C. 2006. A strategic approach to mitigating the impacts of wild canids: proposed activities of the Invasive Animals Cooperative Research Centre. *Australian Journal of Experimental Agriculture* 46: 753-762. <https://doi.org/10.1071/EA06009>
- Friedel, M. H. and Chewings, V. H. 1988. Comparison of crown cover estimates for woody vegetation in arid rangelands. *Australian Journal of Ecology* 13: 463-468. <https://doi.org/10.1111/j.1442-9993.1988.tb00994.x>
- Gordon, C. E., Eldridge, D. J., Ripple, W. J., Crowther, M. S., Moore, B. D. and Letnic, M. 2017a. Shrub encroachment is linked to extirpation of an apex predator. *Journal of Animal Ecology* 86: 147-157. <https://doi.org/10.1111/1365-2656.12607>
- Gordon, C. E. and Letnic, M. 2016. Functional extinction of a desert rodent: implications for seed fate and vegetation dynamics. *Ecography* 39: 815-824. <https://doi.org/10.1111/ecog.01648>
- Gordon, C. E., Moore, B. D. and Letnic, M. 2017b. Temporal and spatial trends in the abundances of an apex predator, introduced mesopredator and ground-nesting bird are consistent with the mesopredator release hypothesis. *Biodiversity and Conservation* 26: 1445-1462. <https://doi.org/10.1007/s10531-017-1309-9>
- Greenville, A. C., Wardle, G. M., Tamayo, B. and Dickman, C. R. 2014. Bottom-up and top-down processes interact to modify intraguild interactions in resource-pulse environments. *Oecologia* 175: 1349-1358. <https://doi.org/10.1007/s00442-014-2977-8>
- Hunter, D. O., Lagisz, M., Leo, V., Nakagawa, S. and Letnic, M. 2018. Not all predators are equal: a continent-scale analysis of the effects of predator control on Australian mammals. *Mammal Review* 48: 108-122. <https://doi.org/10.1111/mam.12115>
- Johnson, C. N., Isaac, J. L. and Fisher, D. O. 2007. Rarity of a top predator triggers continent-wide collapse of mammal prey: dingoes and marsupials in Australia. *Proceedings of the Royal Society B-Biological Sciences* 274: 341-346. <https://doi.org/10.1098/rspb.2006.3711>
- Johnson, C. N. and VanDerWal, J. 2009. Evidence that dingoes limit abundance of a mesopredator in eastern Australian forests. *Journal of Applied Ecology* 46: 641-646. <https://doi.org/10.1111/j.1365-2664.2009.01650.x>
- Keith, D. A. 2006. Ocean Shores to Desert Dunes: The native vegetation of New South Wales and the ACT. Department of Environment and Conservation, Hurstville.
- Kennedy, M., Phillips, B. L., Legge, S., Murphy, S. A. and Faulkner, R. A. 2012. Do dingoes suppress the activity of feral cats in northern Australia? *Austral Ecology* 37: 134-139. <https://doi.org/10.1111/j.1442-9993.2011.02256.x>
- Leo, V., Reading, R. P., Gordon, C. and Letnic, M. 2019. Apex predator suppression is linked to restructuring of ecosystems via multiple ecological pathways. *Oikos* 128: 630-639. <https://doi.org/10.1111/oik.05546>
- Letnic, M. and Crowther, M. S. 2013. Patterns in the abundance of kangaroo populations in arid Australia are consistent with the exploitation ecosystems hypothesis. *Oikos* 122: 761-769. <https://doi.org/10.1111/j.1600-0706.2012.20425.x>
- Letnic, M., Crowther, M. S. and Koch, F. 2009. Does a top-predator provide an endangered rodent with refuge from an invasive mesopredator? *Animal Conservation* 12: 302-312. <https://doi.org/10.1111/j.1469-1795.2009.00250.x>
- Letnic, M., Feit, A., Mills, C. and Feit, B. 2016. The crest-tailed mulgara *Dasymercus cristicauda* in the south-eastern Strzelecki Desert. *Australian Mammalogy* 38: 241-245. <https://doi.org/10.1071/AM15027>
- Letnic, M., Greenville, A., Denny, E., Dickman, C. R., Tischler, M., Gordon, C. and Koch, F. 2011. Does a top predator suppress the abundance of an invasive mesopredator at a continental scale? *Global Ecology and Biogeography* 20: 343-353. <https://doi.org/10.1111/j.1466-8238.2010.00600.x>
- Letnic, M. and Koch, F. 2010. Are dingoes a trophic regulator in arid Australia? A comparison of mammal communities on either side of the dingo fence. *Austral Ecology* 35: 167-175. <https://doi.org/10.1111/j.1442-9993.2009.02022.x>
- Letnic, M., Ritchie, E. G. and Dickman, C. R. 2012. Top predators as biodiversity regulators: the dingo *Canis lupus* dingo as a case study. *Biological Reviews* 87: 390-413. <https://doi.org/10.1111/j.1469-185X.2011.00203.x>
- Lyons, M. B., Mills, C. H., Gordon, C. E. and Letnic, M. 2018. Linking trophic cascades to changes in desert dune geomorphology using high-resolution drone data. *Journal of The Royal Society Interface* 15: 20180327. <https://doi.org/10.1098/rsif.2018.0327>

- McInturff, A., Xu, W., Wilkinson, C. E., Dejid, N. and Brashares, J. S. 2020. Fence Ecology: Frameworks for Understanding the Ecological Effects of Fences. *BioScience*. <https://doi.org/10.1093/biosci/biaa103>
- McKnight, T. L. 1969. Barrier fencing for vermin control in Australia. *Geographical Review* 59: 330-347. <https://doi.org/10.2307/213480>
- Mills, C. H., Gordon, C. E. and Letnic, M. 2018. Rewilded mammal assemblages reveal the missing ecological functions of granivores. *Functional Ecology* 32: 475-485. <https://doi.org/10.1111/1365-2435.12950>
- Morris, T. and Letnic, M. 2017. Removal of an apex predator initiates a trophic cascade that extends from herbivores to vegetation and the soil nutrient pool. *Proceedings of the Royal Society B-Biological Sciences* 284: 9. <https://doi.org/10.1098/rspb.2017.0111>
- Moseby, K. E., Crowther, M. S. and Letnic, M. 2019. Ecological Role of an Apex Predator Revealed by a Reintroduction Experiment and Bayesian Statistics. *Ecosystems* 22: 283-295. <https://doi.org/10.1007/s10021-018-0269-6>
- Newsome, A. 1990. The control of vertebrate pests by vertebrate predators. *Trends in Ecology & Evolution* 5: 187-191. [https://doi.org/10.1016/0169-5347\(90\)90208-U](https://doi.org/10.1016/0169-5347(90)90208-U)
- Newsome, A. E., Catling, P. C., Cooke, B. D. and Smyth, R. 2001. Two ecological universes separated by the Dingo Barrier fence in semi-arid Australia: Interactions between landscapes, herbivory and carnivory, with and without dingoes. *The Rangeland Journal* 23: 71-98. <https://doi.org/10.1071/rj01015>
- Noble, J. C., Hik, D. S. and Sinclair, A. R. E. 2007. Landscape ecology of the burrowing bettong: fire and marsupial biocontrol of shrubs in semi-arid Australia. *The Rangeland Journal* 29: 107-119. <https://doi.org/10.1071/RJ06041>
- Northern Territory Government of Australia 2020. Dingo. Accessed 27 October 2020. <https://nt.gov.au/environment/animals/wildlife-in-nt/dingo>
- Paltridge, R. 2002. The diets of cats, foxes and dingoes in relation to prey availability in the Tanami Desert, Northern Territory. *Wildlife Research* 29: 389-403. <https://doi.org/10.1071/WR00010>
- Pople, A. R., Grigg, G. C., Cairns, S. C., Beard, L. A. and Alexander, P. 2000. Trends in the numbers of red kangaroos and emus on either side of the South Australian dingo fence: evidence for predator regulation? *Wildlife Research* 27: 269-276. <https://doi.org/10.1071/wr99030>
- Rees, J. D., Crowther, M. S., Kingsford, R. T. and Letnic, M. 2020. Direct and indirect effects of carrion subsidies in an arid rangeland: Carrion has positive effects on facultative scavengers and negative effects on a small songbird. *Journal of Arid Environments* 179: 104174. <https://doi.org/10.1016/j.jaridenv.2020.104174>
- Rees, J. D., Kingsford, R. T. and Letnic, M. 2017. In the absence of an apex predator, irruptive herbivores suppress grass seed production: implications for small granivores. *Biological Conservation* 213: 13-18. <https://doi.org/10.1016/j.biocon.2017.06.037>
- Rees, J. D., Kingsford, R. T. and Letnic, M. 2019. Changes in desert avifauna associated with the functional extinction of a terrestrial top predator. *Ecography* 42: 67-76. <https://doi.org/10.1111/ecog.03661>
- Robertshaw, J. and Harden, R. 1986. The Ecology of the Dingo in Northeastern New-South-Wales .4. Prey Selection by Dingoes, and Its Effect on the Major Prey Species, the Swamp Wallaby, *Wallabia-Bicolor* (Desmarest). *Wildlife Research* 13: 141-163. <https://doi.org/10.1071/WR9860141>
- Savolainen, P., Leitner, T., Wilton, A. N., Matisoo-Smith, E. and Lundeberg, J. 2004. A detailed picture of the origin of the Australian dingo, obtained from the study of mitochondrial DNA. *Proceedings of the National Academy of Sciences of the United States of America* 101: 12387-12390. <https://doi.org/10.1073/pnas.0401814101>
- Smith, D., King, R. and Allen, B. L. 2020a. Impacts of exclusion fencing on target and non-target fauna: a global review. *Biological Reviews* Early view. <https://doi.org/10.1111/brv.12631>
- Smith, D., Waddell, K. and Allen, B. L. 2020b. Expansion of Vertebrate Pest Exclusion Fencing and Its Potential Benefits for Threatened Fauna Recovery in Australia. *Animals* 10: 1550. <https://doi.org/10.3390/ani10091550>
- South Australian Wild Dog Advisory Group 2016. South Australian Wild Dog Strategic Plan: A plan to protect the livestock industries and public safety whilst maintaining the integrity of the dingo as a wildlife species. Department of Primary Industries and Regions South Australia
- Stadler, M. 2020. State Barrier Fence Esperance extension update. Accessed 27 October 2020. <https://www.agric.wa.gov.au/invasive-species/state-barrier-fence-esperance-extension-update>
- Trigger, D., Mulcock, J., Gaynor, A. and Toussaint, Y. 2008. Ecological restoration, cultural preferences and the negotiation of 'nativeness' in Australia. *Geoforum* 39: 1273-1283. <https://doi.org/10.1016/j.geoforum.2007.05.010>

Wallach, A. D., Johnson, C. N., Ritchie, E. G. and O'Neill, A. J. 2010. Predator control promotes invasive dominated ecological states. *Ecology Letters* 13: 1008-1018. <https://doi.org/10.1111/j.1461-0248.2010.01492.x>

Wang, Y., Naumann, U., Wright, S. T. and Warton, D. I. 2012. mvabund— an R package for model-based analysis of multivariate abundance data. *Methods in Ecology and Evolution* 3: 471-474. <https://doi.org/10.1111/j.2041-210X.2012.00190.x>

Wickham, H. 2011. ggplot2. *WIREs Computational Statistics* 3: 180-185. <https://doi.org/10.1002/wics.147>

Woinarski, J. C. Z., South, S. L., Drummond, P., Johnston, G. R. and Nankivell, A. 2018. The diet of the feral cat (*Felis catus*), red fox (*Vulpes vulpes*) and dog (*Canis familiaris*) over a three-year period at Witchelina Reserve, in arid South Australia. *Australian Mammalogy* 40: 204-213. <https://doi.org/10.1071/AM17033>

AP Supplementary Material

Table S1 - Details of study sites and which surveys were conducted on which properties.

Site Name	DBF Treatment	Land Use	State	Latitude	Longitude	Drone Surveys (Shrubs and Trees)	Bitterlich Surveys (Shrubs and Trees)	Camera Trapping
Sturt National Park	dingoes rare	Conservation	NSW	-29.1500	141.0330	x		x
Waka Station	dingoes rare	Sheep/Cattle	NSW	-29.2070	141.3445	x		x
Lake Stewart	dingoes rare	Sheep	NSW	-29.2570	141.1400		x	x
Winnathee Station	dingoes rare	Sheep/Cattle	NSW	-29.7500	141.1000		x	x
Andamooka Station	dingoes rare	Cattle	SA	-30.3965	137.0482		x	
Mulyungarie Station	dingoes rare	Sheep/Cattle	SA	-31.3524	140.8096		x	
Omicron	dingoes common	Cattle	SA	-28.9570	141.2484	x	x	x
Bollards Lagoon	dingoes common	Cattle	SA	-29.0630	140.6640			x
Strzelecki Regional Reserve	dingoes common	Conservation	SA	-29.4000	140.5500	x		x
Quinyambie Station (North)	dingoes common	Cattle	SA	-29.8330	140.7667	x	x	
Quinyambie Station (South)	dingoes common	Cattle	SA	-30.8891	140.8364		x	x
Billa Kalina Station	dingoes common	Cattle	SA	-30.0844	136.5037		x	

APPENDIX I

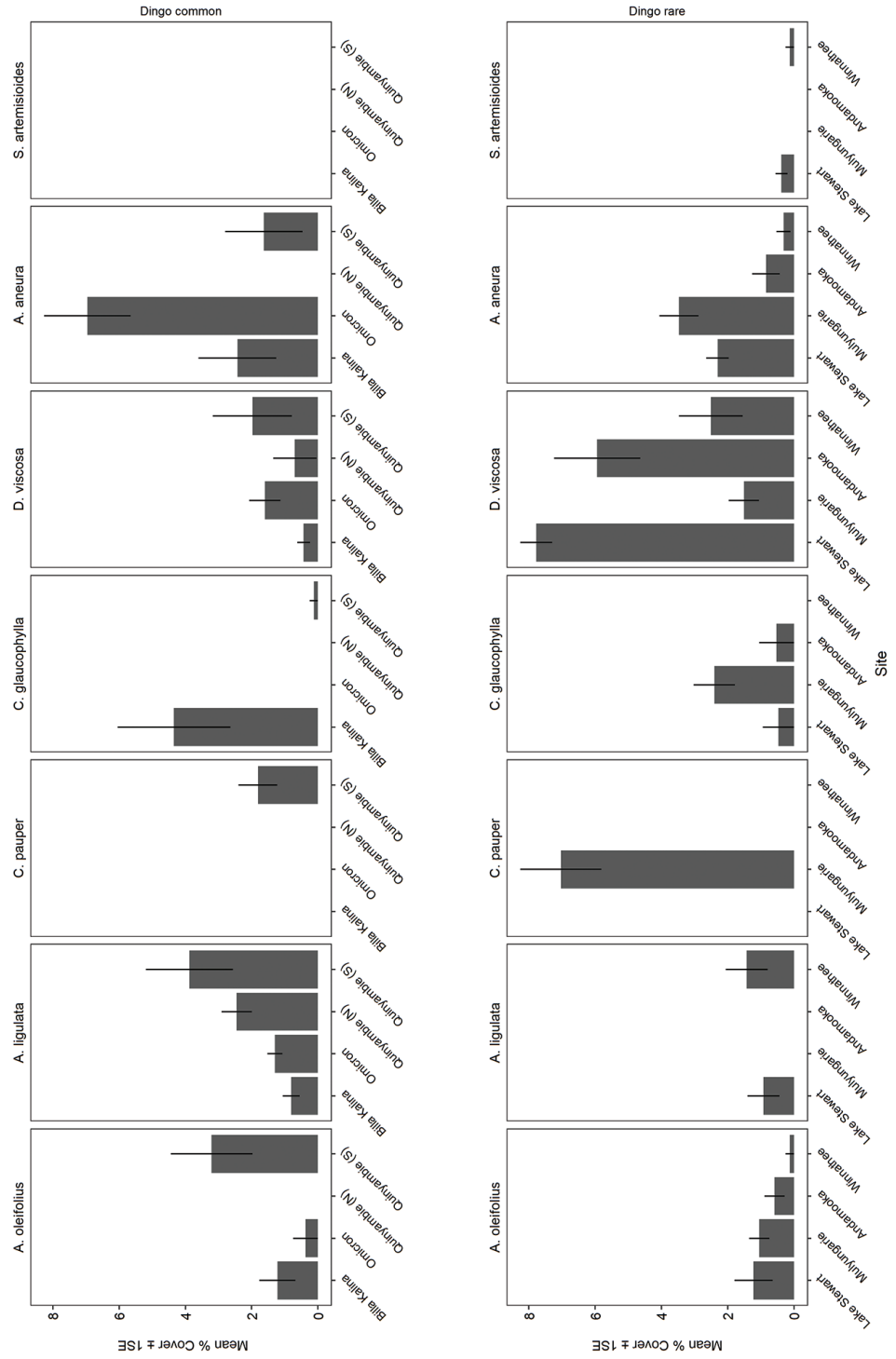


Figure S1 - Mean cover of shrub and tree species at each site. According to the multivariate model output, all species of shrubs and trees were significantly different across sites.

APPENDIX I

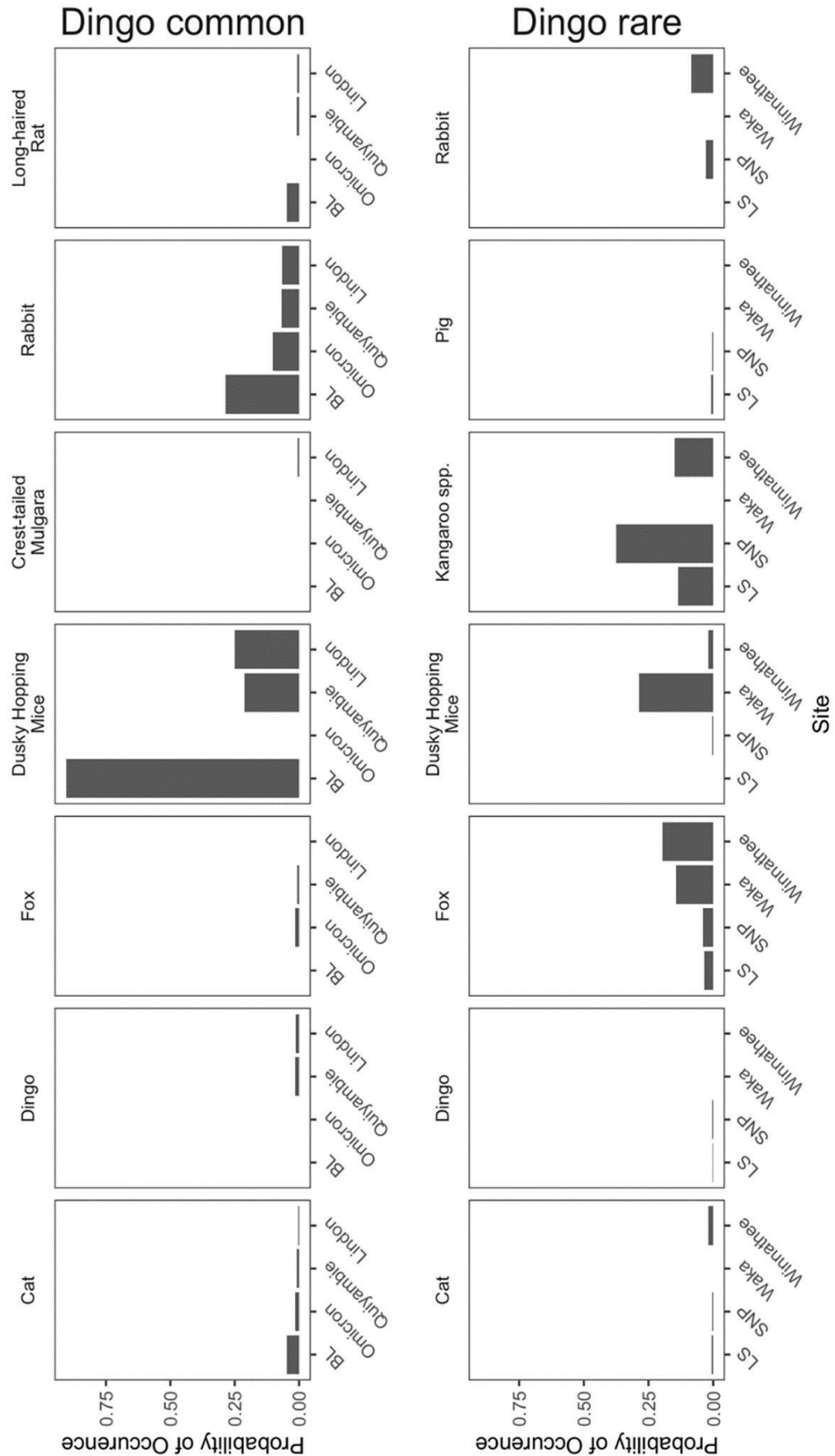


Figure S2 – Probability of occurrence of mammal species on camera traps for each site. According to the multivariate model output, kangaroos, rabbits and Dusky Hopping-mouse were significantly different across sites. Note the species listed for each DBF treatment are different because some species were not detected on both sides of the DBF, for example, there were no Long-haired Rats recorded where dingoes are rare. The BL = Bollards Lagoon, LS = Lake Stewart, SNP = Sturt National Park.

APPENDIX I

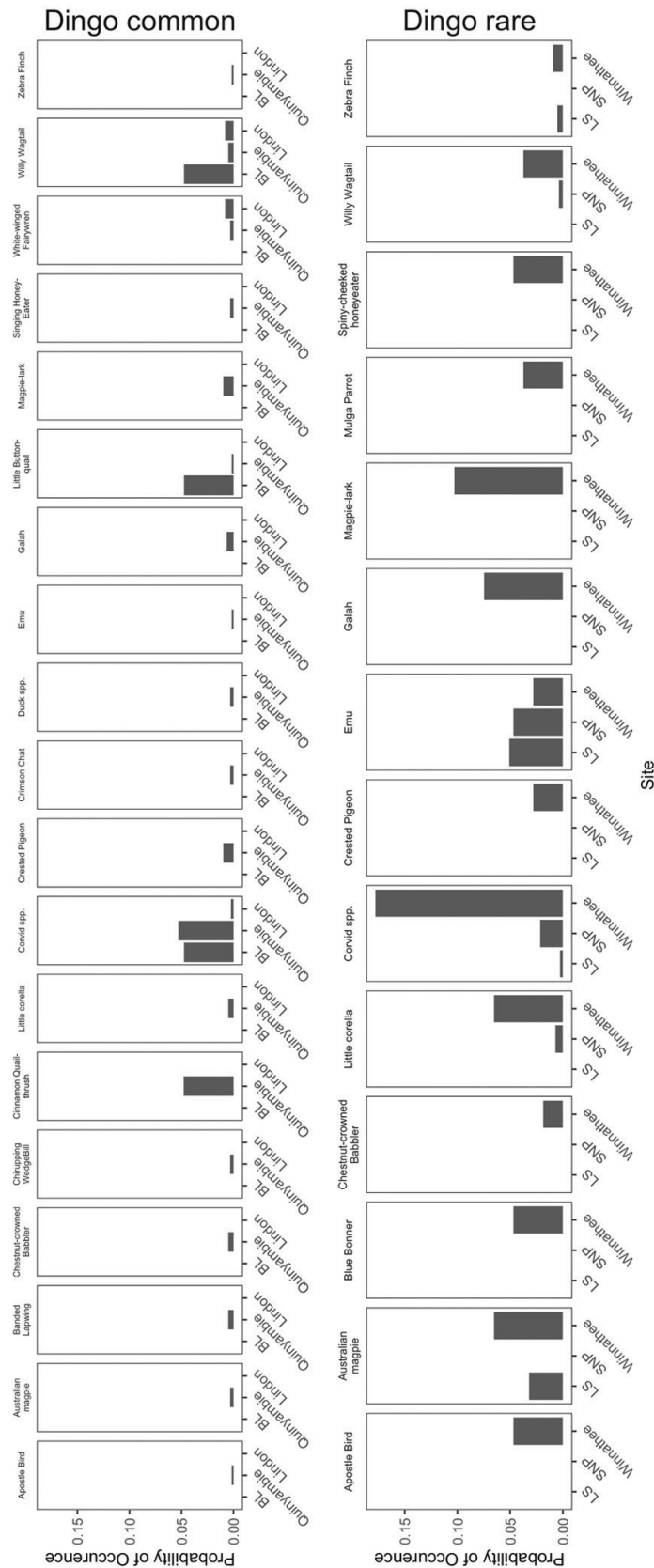


Figure S3 – Probability of occurrence of bird species on camera traps for each site. According to the multivariate model output, corvids, Australian magpies, magpie-larks, galas, spiny-cheeked honeyeaters, blue bonnets, apostle birds, mulga parrots, crested pigeons, chestnut crowned babbler, willy wagtails and cinnamon quail thrush were significantly different across sites. Note the species listed for each DBF treatment are different because some species were not detected on both sides of the DBF. BL = Bollards Lagoon, LS = Lake Stewart, SNP = Sturt National Park.

Table S2 - Univariate test statistics (LRT, P value) output from the multivariate generalized linear model for shrubs and trees. Asterisks mark significant outputs.

Species	DBF Treatment	Site	DBF:Site
<i>Dodonaea viscosa</i>	14.51, 0.003*	18.67, 0.014*	0, 0.62
<i>Callitris glaucophylla</i>	0.12, 0.67	30.86, 0.001*	0, 0.44
<i>Acacia aneura</i>	1.64, 0.46	47.86, 0.001*	0, 0.62
<i>Alectryon oleifolius</i>	1.13, 0.46	26.00, 0.004*	0, 0.88
<i>Senna artemisioides</i>	5.67, 0.09	6.34, 0.04*	0, 0.88
<i>Casuarina pauper</i>	2.62, 0.31	66.40, 0.001*	0, 0.81
<i>Acacia ligulata</i>	13.28, 0.007*	33.40, 0.001*	0, 0.36

Table S3 – Univariate test statistics (LRT, P value) output from the multivariate generalized linear model for mammal species. Each column corresponds to a variable and each row a species following species names in the first column. Asterisks mark significant outputs.

Species	DBF Treatment	Site	Water	DBF:Site	DBF:Water
Kangaroo	395.5, 0.001*	36.46, 0.001*	49.52, 0.001*	0, 0.97	0.004, 0.44
Fox	57.22, 0.001*	0, 1	2.43, 0.48	0, 0.99	0.07, 0.37
Rabbit	29.04, 0.001*	35.26, 0.001*	0.28, 0.90	0, 0.99	15.85, 0.001*
Dingo	7.66, 0.03*	4.17, 0.87	1.12, 0.66	0, 0.81	0.63, 0.37
Dusky Hopping-mouse	401.69, 0.001*	51.49, 0.001*	0.05, 0.90	0, 0.81	0, 0.67
Long-haired Rat	10.53, 0.008*	0.83, 0.99	0.27, 0.90	0, 0.99	0, 0.48
Cat	0.03, 0.87	6.48, 0.63	2.40, 0.48	0, 0.99	0, 0.75
Pig	6.61, 0.04*	4.20, 0.87	0.46, 0.87	0, 0.99	0, 0.67
Crest-tailed Mulgara	2.32, 0.30	3.76, 0.87	0, 0.95	0, 0.99	0, 0.75

APPENDIX I

Table S4 – Univariate test statistics (LRT, P value) output from the multivariate generalized linear model for bird species. Each column corresponds to a variable and each row a species following species names in the first column. Asterisks mark significant outputs.

Species	DBF:Treatment	Site	Water	DBF:Site	DBF:Water
Little buttonquail	2.21, 0.80	0, 1	0.30, 0.98	0.001, 0.97	0, 0.94
Cinnamon quail-thrush	35.79, 0.001*	7.03, 0.09	10.65, 0.015*	0, 0.99	0.001, 0.85
Willie wagtail	0.001, 1.00	19.86, 0.001*	2.58, 0.70	0, 0.98	0.79, 0.76
White-winged fairywren	5.58, 0.17	0, 1	0.61, 0.98	0, 1.00	0, 0.94
Banded lapwing	3.32, 0.70	0.41, 1.00	0.91, 0.97	0, 1.00	0, 0.91
Chirruping wedgebill	2.21, 0.80	0.41, 1.00	0.54, 0.98	0.001, 0.97	0, 0.94
Crimson chat	2.21, 0.80	0.36, 1.00	0.61, 0.98	0, 0.98	0, 0.94
Duck spp.	2.21, 0.80	0.41, 1.00	8.15, 0.05*	0, 0.98	0.001, 0.85
Singing honeyeater	2.21, 0.80	0.41, 1.00	0.61, 0.98	0, 0.98	0, 0.94
Zebra finch	1.82, 0.80	2.37, 0.64	0.27, 0.98	0, 1.00	0.61, 0.76
Chestnut-crowned babbler	0.01, 1.00	6.06, 0.09	0.01, 0.99	0, 1.00	2.13, 0.64
Crested pigeon	0.31, 0.96	9.55, 0.03*	0.68, 0.98	0, 0.99	1.25, 0.76
Mulga parrot	7.04, 0.10	11.22, 0.012*	2.62, 0.70	0, 1.00	0, 0.94
Apostle bird	4.45, 0.38	14.45, 0.006*	7.40, 0.10	0, 0.99	0, 0.94
Blue bonnet	8.84, 0.06	14.25, 0.007*	3.40, 0.60	0, 1.00	0, 0.94
Spiny-cheeked honeyeater	8.84, 0.06	14.25, 0.007*	3.40, 0.60	0, 1.00	0, 0.94
Little corella	5.73, 0.17	16.36, 0.004*	17.63, 0.001*	0, 1.00	0, 0.94
Galah	3.14, 0.70	24.84, 0.001*	4.13, 0.45	0, 1.00	2.42, 0.63
Magpie-lark	3.88, 0.44	36.27, 0.001*	26.09, 0.001*	0, 0.98	0.45, 0.76
Australian magpie	24.56, 0.001*	16.15, 0.004*	0.85, 0.97	0, 0.98	2.68, 0.57
Corvid spp.	0.05, 0.99	55.48, 0.001*	0.001, 0.99	0, 0.99	1.52, 0.76
Emu	64.35, 0.001*	12.18, 0.008*	13.66, 0.004*	0, 0.98	4.69, 0.18