Towards Monitoring the Effects of Artificial Water Provision on Mammalian Species Occupancy in Semi-Arid North-Western Botswana

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Abstract

Water is often the most limiting resource in semi-arid environments, but many such ecosystems are managed through artificial water provision that can alter the distribution and composition of the mammalian community. In addition, tourist lodges in semi-arid environments often establish artificial water points to attract wildlife for visitors without fully understanding the repercussions for sympatric mammals. One such ecosystem exists in north-western Botswana, where we deployed 24 camera traps across a 100 km² grid centred on a tourism lodge to assess occupancy levels of seven mammal species during the hot dry seasons before and after provision of artificial water was initiated by that lodge in November 2015. Occupancy levels of aardvark, brown hyena and duiker declined following artificial water provision, whereas occupancy levels of gemsbok, kudu, porcupine and steenbok increased. For some species, occupancy and/or detection varied with habitat type, but none showed varying occupancy in relation to distance to water. No clear patterns predicting species response were detected, but most herbivore species showed increasing occupancy, in keeping with previous studies that found an initial population increase following artificial water provision. Overall results were inconclusive, most likely because of the short study period that could only encompass one dry season before and after initiation of artificial water provision. Long-term studies of how artificial water provision affects mammalian communities in semi-arid environments would be beneficial considering predictions of extreme rainfall patterns and high temperatures linked to climate change, such as those anticipated in Botswana. This study provides a baseline and recommendations to benefit future research into the effects of artificial water provision in semi-arid environments.

Introduction

Water availability is a primary driver of tropical ecosystems, especially in arid and semi-arid environments (Andrew 1988) where seasonal rainfall promotes vegetation growth and fills ephemeral waterholes (Zhang *et al.* 2005). Herbivores alter their distribution seasonally in relation to cyclical changes in water availability (Western 1975), occupying different habitats and home ranges to take advantage of spatio-temporal heterogeneity in resource availability (Owen-Smith *et al.* 2010). In semi-arid ecosystems such as north-western Botswana, the end of the rainy season leads to a decrease in surface water availability, which can become focused in a small number of large waterholes or disappear altogether as the dry season progresses. By the end of the dry season, most waterholes dry up and any that are left contain water of very low quality (Wolanski and Gereta 2001). Under natural conditions, only arid-adapted animals that can extract sufficient water from their food, including tubers, can survive year-round in such areas, whereas more water-dependent species can only be seasonal visitors to semi-arid environments (Chamaille-Jammes *et al.* 2007).

Waterholes in semi-arid environments are central nodes in wildlife movements, attracting animals that will move across many kilometres to drink (Chamaille-Jammes *et al.* 2016). Heavy foot traffic can create substantial game trails leading to a water source and animals frequently forage around waterholes, affecting vegetation growth patterns (Fernandez-Gimenez and Allen-Diaz 2001). In combination, these

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activities can cause the development of a disturbance zone around a waterhole, known as a piosphere (Andrew 1988), where vegetation is stunted and more sparse than in the surrounding areas, an effect that can be observed for several kilometres around a water source (Epaphras *et al.* 2008). Piosphere effects on vegetation and habitats can lead to cascading ecological effects on many trophic levels, altering species composition and potentially affecting ecosystem functioning (James *et al.* 1999). In Kalahari, or Kgalakgadi, to use traditional terminology, regions such as southern Botswana, piospheres can extend their reach for several hundred metres (Perkins 2018) and their effects can persist for decades beyond the closure of the artificial waterpoint (Jeltsch *et al.* 1997). Natural waterholes dry up seasonally, allowing a period of relief from the pressure associated with herbivore trampling and grazing, but artificial water pumped by humans is provided year-round in many semi-arid ecosystems as a source of drinking water for livestock and wildlife, potentially causing more permanent effects (Rosenstock *et al.* 1999). Artificial water provision can also disrupt ecosystem functioning, for example by disturbing migration patterns (Perkins 2019).

Landscape management in many arid and semi-arid areas in Australia, North America and Africa involves the provision of artificial water with the aim of increasing local wildlife populations and biodiversity (Harris *et al.* 2015 and James *et al.* 1999) by supplementing access to water, altering wildlife distributions (Epaphras *et al.* 2008) and increasing temporal availability of otherwise arid areas for water-dependent species (Rosenstock *et al.* 1999). Such management can greatly increase the value of an otherwise sparse landscape for photographic wildlife tourism, thereby aiding conservation efforts, but the distribution and output of water should be carefully considered (Mwakiwa *et al.* 2013). In Botswana, efforts to diversify tourism beyond the well-known protected areas such as the Okavango Delta and Chobe have led to the establishment of tourist lodges in more remote locations, such as the semi-arid ecosystem of the NG3 (Ngamiland 3) concession, north-western Botswana. Some of these lodges have started, in the last five years, to pump water into artificial water points designed specifically to attract wildlife. Areas such as NG3 have very little naturally-occurring standing water for most of the year and are designated as mixed use, meaning that livestock exist alongside wildlife. Boreholes have, therefore, been used to provide artificial water in the area for decades, but these are often protected from use by wildlife that have limited experience of artificial water provision until the establishment of tourist lodges with associated pumped water points.

In North America and Australia, research has assessed the indirect effects of artificial water provision on birds (Tanner *et al.* 2017), carnivores (Hall *et al.* 2013), invasive species (Letnic *et al.* 2015) and rodents (Kluever *et al.* 2016), but most studies worldwide have focused on mammalian herbivores; in Africa most research has considered African elephants (*Loxodonta africana)* (Shannon *et al.* 2009). Unfortunately, there is a dearth of research on the effects of artificial water provision in north-western Botswana, although some research has been carried out in the southern Kgalakgadi regions of the country (Makhabu *et al.* 2002; Perkins 2018 and Perkins 2019). There is a substantial body of evidence in scientific literature that artificial water provision can lead to negative indirect effects on herbivore species such as smaller home range sizes, higher predation levels, lower biodiversity levels, skewed sex ratios, decreases in rare species and increases in water-dependent herbivores that can have negative consequences for vegetation and lead to trophic cascades, which are the repercussions of disturbances that can alter ecosystem functioning at multiple levels (Harris *et al.* 2015; McKee *et al.* 2015; Owen-Smith 1996; Redfern *et al.* 2003; Ritter and Bednekoff 2005).

The distribution and movements of many wildlife species are restricted by access to water, so providing artificial water can increase the range and abundance of water-dependent species (Chamaille-Jammes *et al.* 2007). Artificial water provision, therefore, removes the limitation of water access for

wildlife, but the associated higher populations of herbivores can lead to depletion of food resources that can subsequently become limiting, particularly during drought years (James *et al.* 1999), when herbivores migrate from drier areas towards waterholes (Chamaille-Jammes *et al.* 2016). In addition, grazing patterns centred around artificial water can homogenise the landscape (James *et al.* 1999), removing key levels of heterogeneity vital for biodiversity conservation (Cromsigt *et al.* 2009, Owen-Smith 2004). These effects are likely to be exacerbated during drought years, which are predicted to become more extreme in Botswana as a result of climate change. Rainfall onset is predicted to be delayed by several months (Byakatonda *et al.* 2019, Mberego 2017) and temperatures are expected to rise (Midgley and Thuiller 2011), creating hotter, drier conditions overall, interspersed with extreme patterns of high and low rainfall (Byakatonda 2018 and Perkins 2019).

Semi-arid environments that have not previously been pumped with artificial water should have an established community of desert-adapted wildlife existing in relative equilibrium. The introduction of artificial water into such an environment could affect the balance of species composition within the community by increasing populations of some species and decreasing populations of others (Owen-Smith 1996 and Razgour *et al.* 2018). North-western Botswana consists primarily of Kgalakgadi Desert habitats with relatively low densities of wildlife. Using a study site in NG3, we assessed mammalian community composition in a semi-arid environment before and after artificial water provision was initiated through camera trapping, which provided images of a variety of species. We used occupancy analysis, which takes probability of detection into consideration, to quantify changes in occupancy for several species within the mammalian community following provision of artificial water. We hypothesised that i) overall species richness (number of species) would increase, ii) grazers would increase with proximity to water, and iii) browsers would not be affected.

Materials and Methods

Study site
The study was conducted in the NG3 concession of north-western Botswana, part of the Kgalagadi desert landscape. Two habitats were identified within the study area, based on woody and herbaceous vegetation: open grassland and scrub. Wildlife density in the study area is relatively low, based on personal observations during fieldwork and the low number of established game trails. The study area receives approximately 500 mm of rainfall annually (Parida and Moalafhi 2008), between November and April, and there are no natural permanent water sources in the area. During the study years, rainfall totalled 422.2 and 349.2 mm in 2015 and 2016, respectively and neither year was considered a drought year. The study was based around 'The Lodge', a tourist lodge operated by the Feline Fields Trust that opened for business in early 2016. At the beginning of November 2015, three artificial waterholes at The Lodge were filled and pumped with water by the operators of The Lodge throughout the rest of the study period.

The Kgalagadi landscape is inhabited by a range of desert-adapted resident mammals. In the NG3 study site, camera traps detected multiple mammal species (Table 1).

Table 1: List of mammal species detected by camera traps deployed in NG3, north-western Botswana

| Common name	Latin name	Diet or taxonomic group	Residence status in study area
Gemsbok	*Oryx gazella*	Grazer	Resident
Greater kudu	*Tragelaphus strepsiceros*	Browser	Resident
Eland	*Taurotragus oryx*	Browser	Resident
Steenbok	*Raphicerus campestris*	Mixed feeder	Resident
Common duiker	*Slyvicapra grimmia*	Mixed feeder	Resident
Warthog	*Phacochoerus africanus*	Omnivore	Resident
Aardvark	*Orycteropus afer*	Insectivore	Resident
Aardwolf	*Proteles cristata*	Insectivore	Resident
Porcupine	*Hystrix cristata*	Rodent	Resident
Spring hare	*Pedetes capensis*	Rodent	Resident
Scrub hare	*Lepus saxatilis*	Lagomorph	Resident
Brown hyaena	*Parahyaena brunnea*	Carnivore	Resident
African lion	*Panthera leo*	Carnivore	Resident
Leopard	*Panthera pardus*	Carnivore	Resident
African wild dog	*Lycaon pictus*	Carnivore	Resident
Black-backed jackal	*Canis mesomelas*	Carnivore	Resident
Caracal	*Caracal caracal*	Carnivore	Resident
African wildcat	*Felis sylvestris lybica*	Carnivore	Resident
Slender mongoose	*Galerella sanguinea*	Carnivore	Resident
Honey badger	*Mellivora capensis*	Carnivore	Resident
Meerkat	*Suricata suricata*	Insectivore	Resident
Zorilla	*Ictonyx striatus*	Carnivore	Resident
African elephant	*Loxodonta africana*	Mixed feeder	Seasonal
Blue wildebeest	*Connochaetes taurinus*	Grazer	Seasonal
Plains zebra	*Equus quagga*	Grazer	Seasonal
Cattle	*Bos taurus*	Grazer	Resident, domestic

Source: The Author

Camera traps

Thirteen M1100i and eleven A-5 Moultrie trail cameras (Birmingham, Alabama, USA) were fitted with 16GB (Gigabyte) SanDisk (Milpitas, California, USA) memory cards and deployed in 4 km² grid cells within a 100 km² grid with The Lodge at its centre (Figure 1). Each camera trap was identified as one sampling unit. This camera trap distribution was selected to maximise the chances of capturing images of the wide variety of species occurring in the area with varying home range sizes. Global Positioning System (GPS) coordinates were used to identify the centre of each grid square and the camera traps were deployed as close to those coordinates as possible, given variations in vegetation density and accessibility.

Camera traps were positioned towards the most open space within the vicinity of the GPS coordinate; for two camera traps the most open space was a road. Three camera traps were attached to trees, but many of the areas did not have large enough trees, so 21 camera traps were attached to mopane (*Colophospermum mopane*) droppers dug into the ground by the researcher (Table 2).

Figure 1: Map of study area, showing configuration of camera traps around The Lodge, a tourist lodge in the NG3 concession of Ngamiland
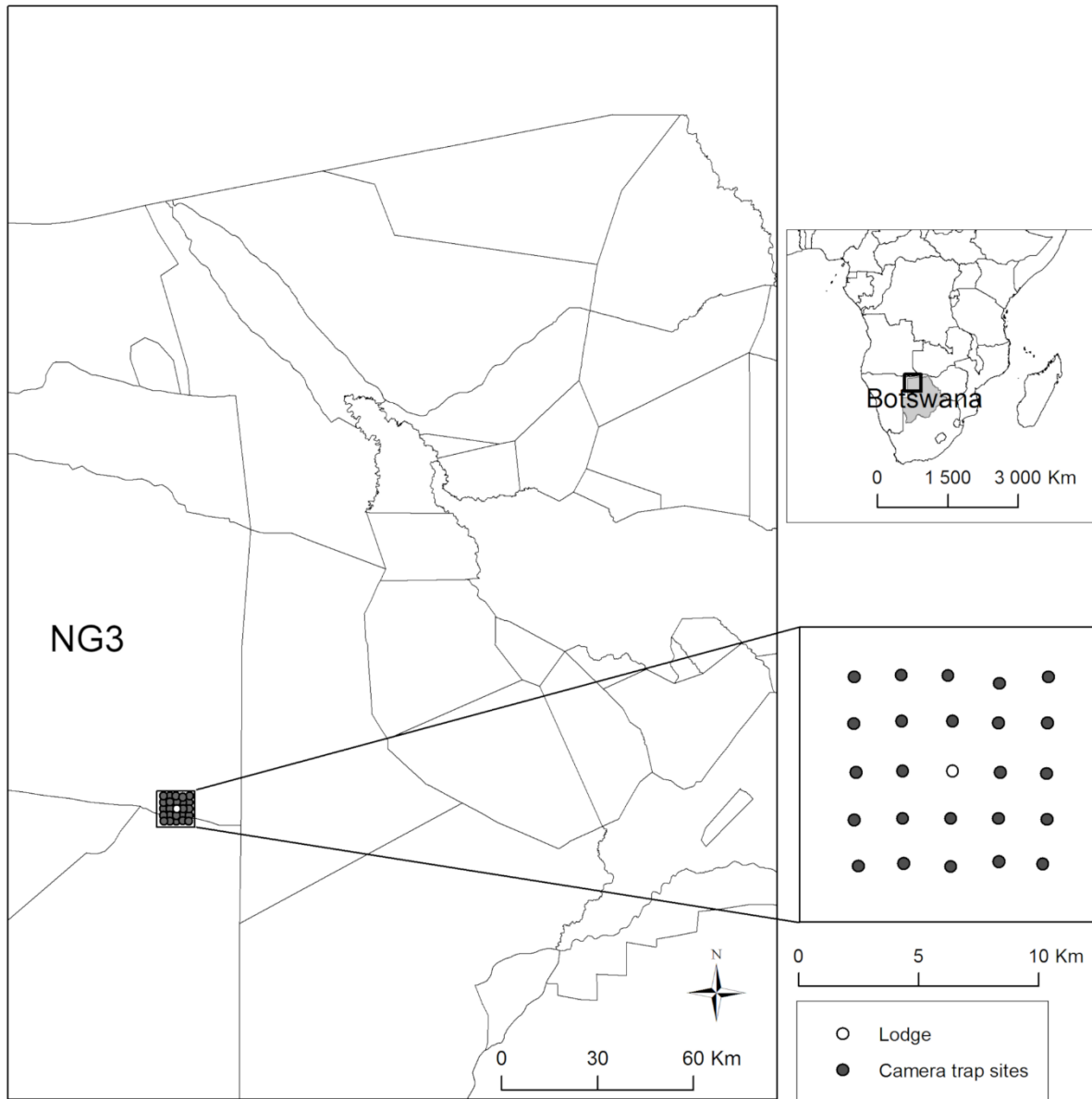
Source: The Author

The camera traps were initially deployed on 1 or 2 July 2015 and were visited every 115.6 ± 18.6 days to replace batteries and download images. This was a relatively long period between visits, but most camera traps had not used up their batteries and still had space on their memory cards. The date of the last recorded image was noted and any gaps in data collection were accounted for during analysis. New batteries and memory cards were inserted into every camera trap during each visit, which coincided with a change of season, as defined by changing rainfall and temperatures: the rainy season (November-February), when most rainfall occurred; the cold dry season (March-June) and the hot dry season (July-October). All camera traps were recovered by 27 January 2017. To assess the effects of artificial water provision on mammalian species occupancy for this study, we focused on the hot dry season months, when natural surface water availability was at its lowest. We compared data from the 2015 and 2016 hot

dry seasons, representing pre- and post- artificial water provision, which began in November 2015. [1]By focusing on the same season in each year, we avoided the potential influence of seasonal migrants and we reduced the likelihood of season-related changes in occupancy (MacKenzie *et al.* 2006). We, therefore, assumed that camera trapping sites were occupied or unoccupied for the duration of the survey in each year. Camera Base was used to sort and store images from camera traps. We recorded information on species, number of individuals and, where possible, animal gender, although only presence/absence data were used in analyses.

Table 2: Camera trap model and deployment information for camera trapping survey in NG3, north-western Botswana

Camera trap ID	X	Y	Location	Height/m	Bearing/°
M1100-04	21.82644	-19.99057	Tree	1.4	298
M1100-05	21.84568	-19.99079	Dropper	1.4	102
M1100-08	21.80732	-19.99066	Dropper	1.4	86
M1100-09	21.82642	-19.93957	Dropper	1.7	27
M1100-10	21.76867	-19.93740	Dropper	1.3	107
M1100-17	21.80602	-19.93673	Dropper	1.4	39
M1100-18	21.78812	-19.97267	Dropper	1.5	141
M1100-19	21.78776	-19.95394	Dropper	1.4	270
M1100-21	21.80742	-20.00873	Dropper	1.4	51
M1100-22	21.78821	-19.99074	Tree	1.4	101
M1100-23	21.84590	-19.93713	Dropper	1.2	325
M1100-24	21.78936	-20.07610	Tree	1.5	305
M1100-28	21.82707	-19.97313	Dropper	1.4	354
A5-01	21.82625	-19.95443	Dropper	1.5	356
A5-04	21.82663	-20.00675	Dropper	1.4	347
A5-06	21.84423	-20.00752	Dropper	1.5	79
A5-07	21.84534	-19.97352	Dropper	1.3	341
A5-10	21.76944	-19.97334	Dropper	1.4	102
A5-11	21.78748	-19.93660	Dropper	1.3	355
A5-12	21.76844	-19.95502	Dropper	1.5	58
A5-13	21.76890	-19.99127	Dropper	1.5	69
A5-14	21.77064	-20.00870	Dropper	1.5	231
A5-16	21.84564	-19.95434	Dropper	1.4	30
A5-20	21.80797	-19.95391	Dropper	1.6	69

Source: The Author

Species richness
We counted the number of species (species richness) captured by each camera during the 2015 and 2016 hot dry seasons and ran a generalised linear mixed model with a Poisson distribution, using the 'lme4' package in R (Bates 2010), to determine whether species richness varied between the two years.

The Number of species was the dependent variable, year was the fixed effect and camera ID (Identity) was the random effect. The most parsimonious model was identified from AIC (Akaike Information Criterion) values (Akaike 1974).

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1 http://www.atrium-biodiversity.org/tools/camerabase/ Accessed 10 February 2019

Variables
We recorded habitat type for each camera trap. During the 2015 hot dry season, no water was provided by The Lodge, so we only included the effect of habitat on occupancy and detection in the occupancy models, but for the 2016 hot dry season we also included the effect of distance to water. We re-projected decimal degrees into the Universal TransMercator coordinate system to allow accurate Euclidean distance from The Lodge, and therefore from permanent water, to be calculated for every camera trap using the 'pointDistance' function from the 'raster' package in R v. 3.4.1 (R Development Core Team 2017).

We included these variables in occupancy models to determine species-specific effects of habitat and distance to water on seasonal occupancy and detection.

Occupancy models
We divided each season into eight occasions, each representing two weeks of data, and recorded presence/ absence data for every species within each occasion. Most camera trapping studies use shorter occasions (Burton *et al.* 2015), but the study area contained relatively low densities of wildlife, so we used long sampling occasions to ensure that we had sufficient occasions with detection to run the analyses. We used Presence v 2.12.10[2] to run two sets of single season occupancy analyses for each species and obtain maximum likelihood estimates for occupancy and detection. Occupancy (Ψ) was defined as the probability that a given site was occupied at the start of the study. Detection (*p*) was defined as the probability that a species would be detected if it was present in a given site. Including detection in models reduces potential bias in estimates for occupancy (MacKenzie *et al.* 2003).

We did not use multi-season occupancy models because the effect of distance to water was not present in 2015, so could not be included in the models for that year. To quantify covariate effects on Ψ or *p* parameters, we ran models that included all covariates for the other parameter, which allowed the models maximum flexibility (Forsyth *et al.* 2016 and MacKenzie *et al.* 2006). We used AIC values to identify the most parsimonious model for Ψ and *p* for each season and species, then we used model averaging to obtain estimates for Ψ and *p*. We summed the weights of all models containing specific covariates to determine whether the effect of those covariates was significant: covariates with summed model weights close to or below 0.5 did not have significant effects on parameters (Forsyth *et al.* 2016 and MacKenzie *et al.* 2006).

The Results
Camera trap functioning varied annually because of software malfunctions and external effects such as disturbance by animals (Table 3), so data were collected from 23 and 19 camera traps in 2015 and 2016, respectively.

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2 https://www.usgs.gov/software/presence, accessed February 2019

Table 3: Camera trap functioning during seasonal occasions, where (*) represents full functioning and (-) represents impaired functioning

Camera trap ID	2015 occasions								2016 occasions							
	1	2	3	4	5	6	7	8	1	2	3	4	5	6	7	8
M1100-04	*	*	*	*	*	*	*	*	*	*	*	*	*	*	*	*
M1100-05	*	*	-	-	-	-	-	-	*	*	*	*	*	-	-	-
M1100-08	*	*	-	-	-	-	-	-	*	*	*	*	*	*	*	*
M1100-09	*	*	*	-	-	-	-	-	-	-	-	-	-	-	-	-
M1100-10	*	*	*	*	*	*	*	*	*	*	*	*	*	*	*	*
M1100-17	*	*	-	-	-	-	-	-	*	*	-	-	-	-	-	-
M1100-18	*	*	*	*	*	*	*	*	*	*	*	*	*	*	*	*
M1100-19	*	*	*	*	*	*	*	*	*	*	*	*	*	*	*	*
M1100-21	*	*	*	*	*	*	-	-	*	*	*	*	*	*	*	*
M1100-22	*	*	*	*	*	*	*	*	*	*	*	*	*	*	*	*
M1100-23	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
M1100-24	*	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
M1100-28	*	*	*	*	*	*	*	*	*	*	*	*	*	*	*	*
A5-01	*	*	-	-	-	-	-	-	*	*	*	*	*	*	*	*
A5-04	*	*	-	-	-	-	-	-	*	*	*	*	*	*	*	*
A5-06	*	*	*	*	*	*	*	*	*	*	*	*	*	*	*	*
A5-07	*	*	*	*	*	*	*	*	*	*	*	*	*	*	*	*
A5-10	*	*	*	*	*	*	*	*	*	*	*	*	*	*	*	*
A5-11	*	*	*	*	*	*	*	*	-	-	-	-	-	-	-	-
A5-12	*	*	*	*	*	*	*	*	*	*	*	*	*	*	*	*
A5-13	*	*	*	*	*	*	*	*	*	*	*	*	*	*	*	*
A5-14	*	*	*	-	-	-	-	-	*	*	*	-	-	-	-	-
A5-16	*	*	*	*	*	*	*	*	*	*	*	*	*	*	*	*
A5-20	*	*	*	*	*	*	*	*	*	*	*	*	*	*	*	*

Source: The Author

Species richness
Images of 15 and 17 different mammalian species were recorded during 2015 and 2016, respectively, with a range of 1-9 species detected by individual cameras. In 2015 and 2016, mean ± S.D. species richness per camera was 3.55 ± 2.11 and 4.44 ± 2.48 species, respectively.

The most parsimonious model was the null model (AIC = 168.73), indicating that there was no difference in species richness between 2015 and 2016.

Occupancy modelling
Most species detected by the camera traps were detected very infrequently, so could not be included in the

occupancy analyses. We identified seven species that each represented at least 5% of the overall species composition: aardvark, brown hyaena, common duiker, gemsbok, kudu, porcupine and steenbok.

We ran single season occupancy analyses for the 2015 and 2016 hot dry seasons for each of these species separately to obtain maximum likelihood estimates for occupancy and detection, and to determine whether those parameters varied with habitat and distance to artificial water.

Aardvark

In 2015, the most parsimonious model for occupancy did not include any effect of habitat (AIC = 74.76, AICω = 0.69; Supplementary Table 1), nor did the most parsimonious model for detection (AIC = 74.78, AICω = 0.69; Supplementary Table 1). Summed occupancy and detection model weights for models containing the effect of habitat were both <0.5 (Table 4), indicating that habitat did not have an effect on occupancy or detection. In 2016, the most parsimonious model for occupancy included the effect of distance to water (AIC = 53.02, AICω = 0.38; Supplementary Table 1), but summed model weights for models containing the effect of habitat and distance to water were <0.5 and 0.52, respectively (Table 4), suggesting that neither habitat nor distance to water had an effect on occupancy.

The most parsimonious model for detection included the effect of distance to water (AIC = 53.02, AICω = 0.37; Supplementary Table 1), but summed model weights for models containing the effect of habitat and distance to water were <0.5 and 0.51, respectively (Table 4), demonstrating that neither habitat nor distance to water had an effect on detection. Overall, aardvark occupancy rates decreased from 2015 to 2016 (Figure 2), while detection rates increased (Figure 3).

Table 4: Summed model weights for covariates affecting occupancy and detection of mammal species in NG3, north-western Botswana

Species	Year	Parameter	Estimate	S.D.	Covariate	Summed model weights
Aardvark	2015	Occupancy	0.82	0.07	Habitat	0.31
		Detection	0.07	0.01	Habitat	0.31
	2016	Occupancy	0.61	0.27	Habitat	0.27
					Water	0.52
		Detection	0.09	0.03	Habitat	0.27
					Water	0.51
Brown hyena	2015	Occupancy	0.37	0.11	Habitat	0.30
		Detection	0.19	0.25	Habitat	0.90
	2016	Occupancy	0.28	0.04	Habitat	0.27
					Water	0.27
		Detection	0.33	0.06	Habitat	0.27
					Water	0.44
Common duiker	2015	Occupancy	0.69	0.00	Habitat	0.27
		Detection	0.29	0.04	Habitat	0.54
	2016	Occupancy	0.59	0.04	Habitat	0.31
					Water	0.33
		Detection	0.38	0.03	Habitat	0.34

					Water	0.34
Gemsbok	2015	Occupancy	0.79	0.02	Habitat	0.31
		Detection	0.32	0.01	Habitat	0.31
	2016	Occupancy	0.91	0.07	Habitat	0.33
					Water	0.45
		Detection	0.34	0.06	Habitat	0.63
					Water	0.56
Kudu	2015	Occupancy	0.32	0.04	Habitat	0.27
		Detection	0.19	0.02	Habitat	0.27
	2016	Occupancy	0.58	0.07	Habitat	0.33
					Water	0.31
		Detection	0.13	0.03	Habitat	0.44
					Water	0.17
Porcupine	2015	Occupancy	0.42	0.26	Habitat	0.74
		Detection	0.19	0.06	Habitat	0.70
	2016	Occupancy	0.44	0.21	Habitat	0.38
					Water	0.38
		Detection	0.09	0.03	Habitat	0.35
					Water	0.29
Steenbok	2015	Occupancy	0.51	0.00	Habitat	0.27
		Detection	0.25	0.03	Habitat	0.44
	2016	Occupancy	0.64	0.03	Habitat	0.42
					Water	0.30
		Detection	0.28	0.01	Habitat	0.27
					Water	0.29

Source: The Author

Brown hyena

In 2015, the most parsimonious model for occupancy did not include any effect of habitat (AIC = 33.71, AICω = 0.70; Supplementary Table 1) and summed model weights for models containing the effect of habitat were <0.5 (Table 4), indicating that habitat did not have an effect on occupancy. The most parsimonious model for detection contained the effect of habitat (AIC = 35.36, AICω = 0.90; Supplementary Table 1) and summed model weights for models including the effect of habitat were 0.9 (Table 4), suggesting that habitat had an effect on detection: brown hyaena were more detectable in grassland than in scrub (Figure 4).
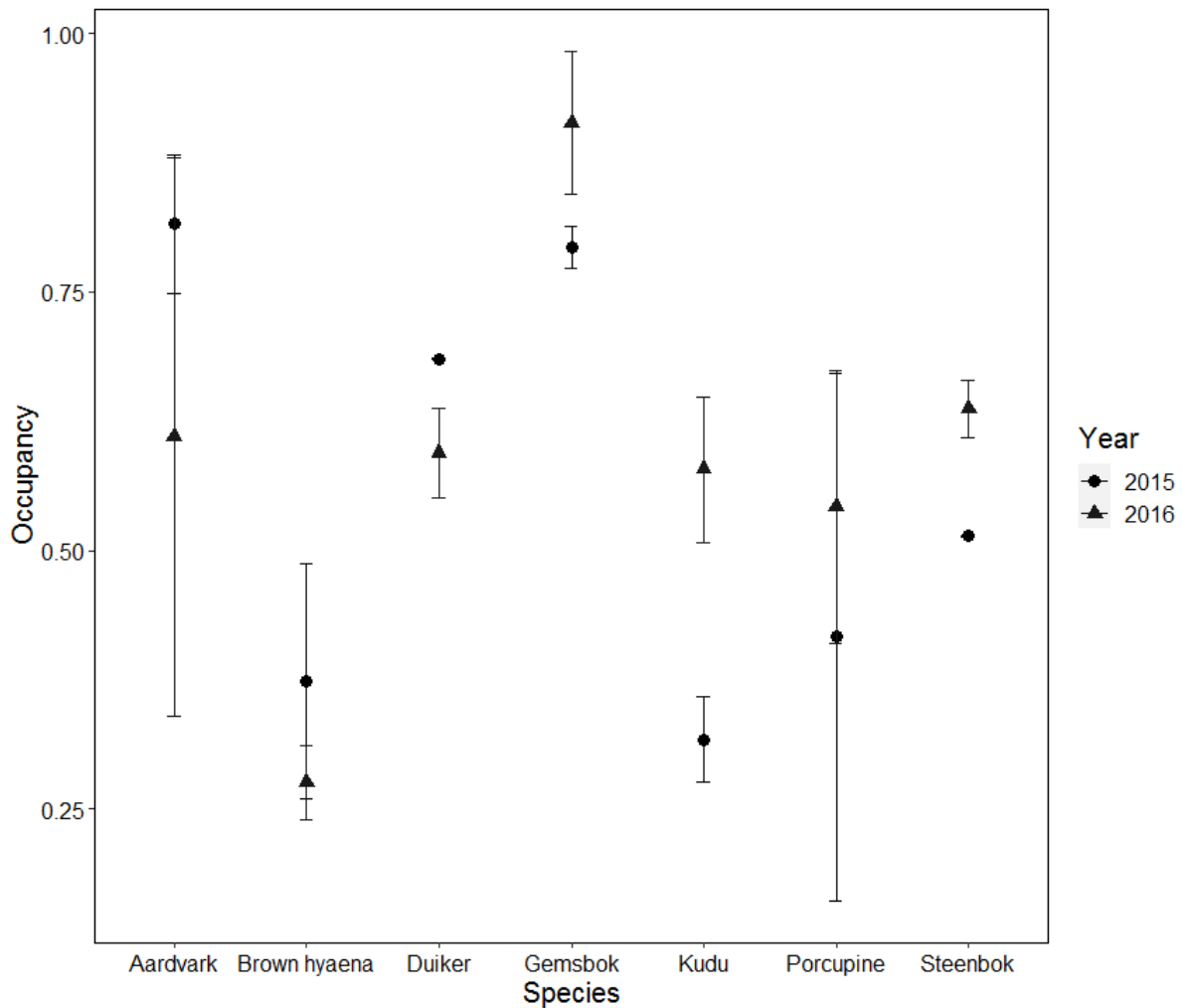
In 2016, the most parsimonious model for occupancy did not contain any effect of habitat or distance to water (AIC = 65.70, AICω = 0.53; Supplementary Table 1); neither did the most parsimonious model for detection (AIC = 67.18, AICω = 0.41; Supplementary Table 1). Summed occupancy and detection model weights for models including the effect of habitat and distance to water were <0.5 (Table 4), implying that neither habitat nor distance to water affected occupancy or detection. Overall, brown hyaena occupancy rates decreased from 2015 to 2016 (Figure 2), while detection rates increased (Figure 3).

Common duiker
In 2015, the most parsimonious model for occupancy did not contain any effect of habitat (AIC = 144.79, AICω = 0.73; Supplementary Table 1) and summed model weights for models including the effect of habitat were <0.5 (Table 4), showing that habitat did not have an effect on occupancy. The most parsimonious model for detection contained the effect of habitat (AIC = 146.79, AICω = 0.54; Supplementary Table 1), but summed model weights for models including the effect of habitat were 0.54 (Table 4), suggesting that habitat did not have an effect on detection.

In 2016, neither the most parsimonious model for occupancy (AIC = 150.84, AICω = 0.45; Supplementary Table 1) nor the most parsimonious model for detection (AIC = 151.35, AICω = 0.44; Supplementary Table 1) contained any effect of habitat or distance to water. Summed occupancy and detection model weights for models including the effect of habitat and distance to water were <0.5 (Table 4), demonstrating that neither habitat nor distance to water had an effect on occupancy or detection. Overall, common duiker occupancy rates decreased from 2015 to 2016 (Figure 2), while detection rates increased (Figure 3).

Figure 2: Occupancy of seven mammal species in NG3, north-western Botswana, in the 2015 and 2016 dry seasons
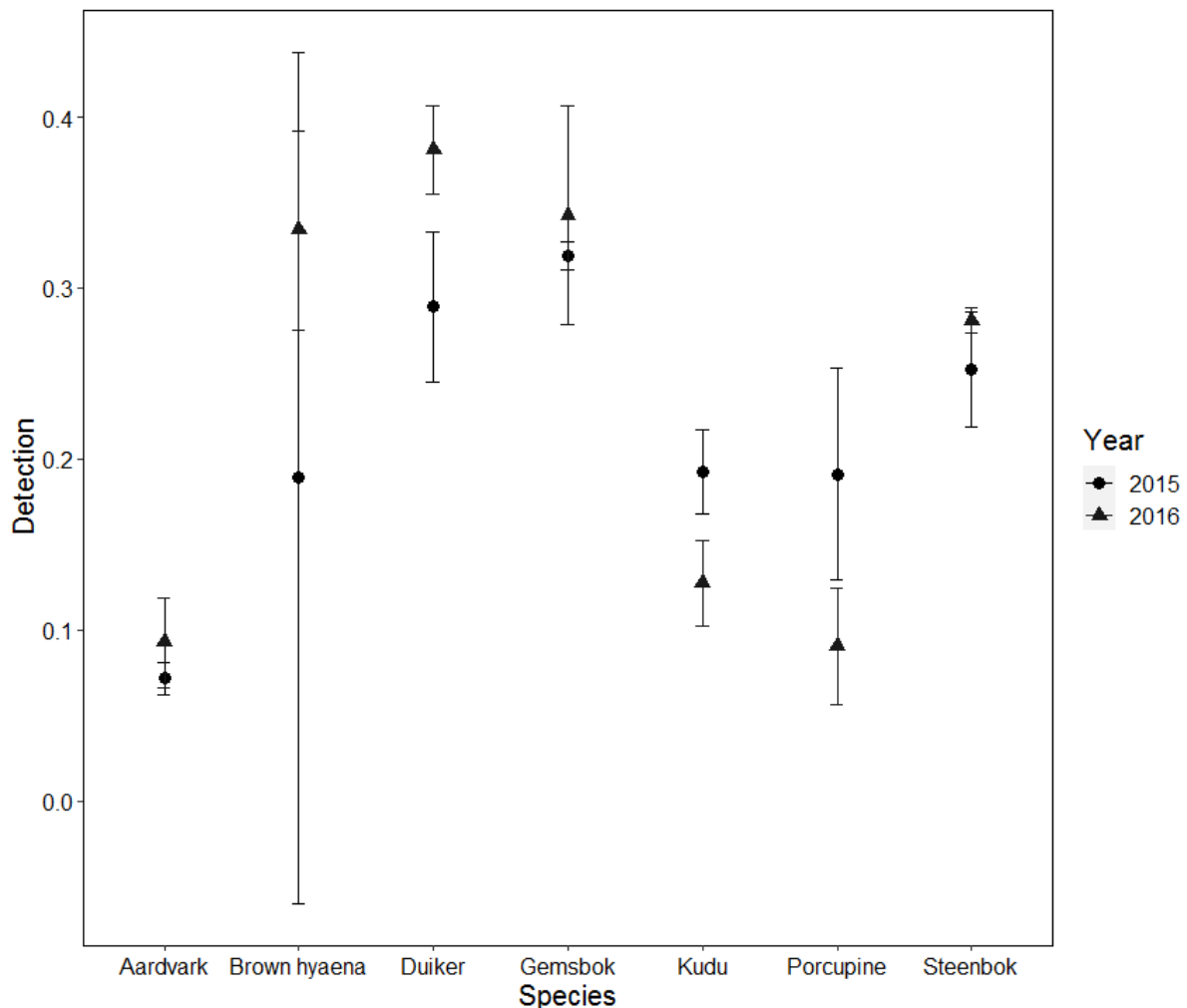

Source: The Author

Gemsbok

In 2015, the most parsimonious model for occupancy did not include any effect of habitat (AIC = 176.18, AICω = 0.69; Supplementary Table 1); neither did the most parsimonious model for detection (AIC = 176.13, AICω = 0.69; Supplementary Table 1). Summed model weights for occupancy and detection models containing the effect of habitat were <0.5 (Table 4), indicating that habitat did not have an effect on occupancy or detection.

In 2016, the most parsimonious model for occupancy did not contain any effect of habitat or distance to water (AIC = 182.94, AICω = 0.39; Supplementary Table 1) and summed model weights for models including the effect of habitat and distance to water were <0.5 (Table 4), suggesting that neither habitat nor distance to water had an effect on occupancy. The most parsimonious model for detection included the effects of habitat and distance to water (AIC = 184.54, AICω = 0.33; Supplementary Table 1) and summed model weights for models containing the effect of habitat and distance to water were 0.63 and 0.56, respectively (Table 4), implying that habitat had some effect on detection, but distance to water did not. Gemsbok were more detectable in scrub than grassland (Fig 4). Overall, gemsbok occupancy and detection rates increased from 2015 to 2016 (Figure 2 and Figure 3).

Figure 3: Detection of seven mammal species in NG3, north-western Botswana, in the 2015 and 2016 dry seasons
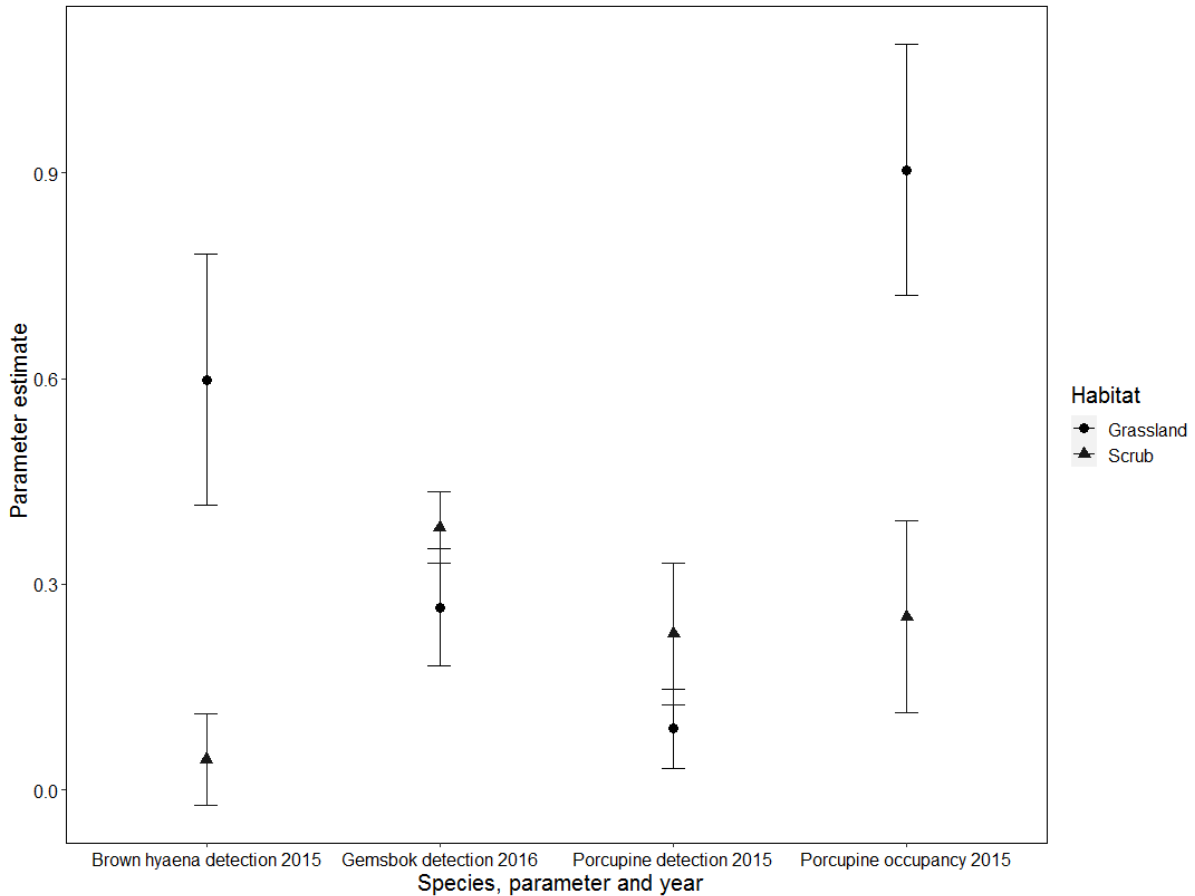

Source: The Author

Kudu

In 2015, the most parsimonious model for occupancy did not contain any effect of habitat (AIC = 57.72, AICω = 0.73; Supplementary Table 1); neither did the most parsimonious model for detection (AIC = 57.73, AICω = 0.73; Supplementary Table 1). Summed model weights for occupancy and detection models including the effect of habitat were <0.5 (Table 4), indicating that habitat did not affect occupancy or detection.

In 2016, neither the most parsimonious model for occupancy (AIC = 77.88, AICω = 0.49; Supplementary Table 1) nor the most parsimonious model for detection (AIC = 77.07, AICω = 0.48; Supplementary Table 1) included any effect of habitat or distance to water. Summed occupancy and detection model weights for models containing the effect of habitat and distance to water were <0.5 (Table 4), showing that neither habitat nor distance to water had an effect on occupancy or detection. Overall, kudu occupancy rates increased from 2015 to 2016 (Figure 2), while detection rates decreased (Figure 3)

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Figure 4: Habitat effects on occupancy and detection of mammal species in NG3, north-western Botswana, in the 2015 and 2016 dry seasons

Source: The Author

Porcupine

In 2015, the most parsimonious model for occupancy contained the effect of habitat (AIC = 73.62, AICω = 0.74; Supplementary Table 1) and summed model weights for models including the effect of habitat were 0.74 (Table 4), indicating that habitat had an effect on occupancy: porcupine occupancy was higher in grassland than scrub (Fig 4). The most parsimonious model for detection included the effect of habitat

(AIC = 73.62, AICω = 0.70; Supplementary Table 1) and summed model weights for models containing the effect of habitat were 0.70 (Table 4), suggesting that habitat had an effect on detection: porcupine were more detectable in scrub than in grassland (Figure 4).

In 2016, the most parsimonious model for occupancy did not include any effect of habitat or distance to water (AIC = 40.99, AICω = 0.44; Supplementary Table 1); neither did the most parsimonious model for detection (AIC = 39.55, AICω = 0.64; Supplementary Table 1). Summed occupancy and detection model weights for models containing the effect of habitat and distance to water were <0.5 (Table 4), implying that neither habitat nor distance to water affected occupancy or detection. Overall, porcupine occupancy rates increased slightly from 2015 to 2016 (Figure 2), while detection rates decreased (Figure 3).

Steenbok
In 2015, neither the most parsimonious model for occupancy (AIC = 110.21, AICω = 0.73; Supplementary Table 1) nor the most parsimonious model for detection (AIC = 111.70, AICω = 0.56; Supplementary Table 1) contained the effect of habitat. Summed model weights for occupancy and detection models including the effect of habitat were <0.5 (Table 4), indicating that habitat did not affect occupancy or detection.

In 2016, the most parsimonious model for occupancy did not contain any effect of habitat or distance to water (AIC = 140.84, AICω = 0.50; Supplementary Table 1); neither did the most parsimonious model for detection (AIC = 140.67, AICω = 0.52; Supplementary Table 1). Summed occupancy and detection model weights for models including the effect of habitat and distance to water were <0.5 (Table 4), suggesting that neither habitat nor distance to water had an effect on occupancy or detection. Overall, steenbok occupancy and detection rates increased from 2015 to 2016 (Figure 2 and Figure 3).

Discussion
Artificial water provision in semi-arid landscapes has the potential to affect many desert-adapted resident species through indirect effects of changes to the mammalian community that could lead to impacts on vegetation and changes in competition and predation levels (James *et al.* 1999). In this study, contrary to our hypothesis, there was no increase in species richness from 2015 to 2016 in the north-western Botswana study site. Between the 2015 and 2016 dry seasons in our study area, there was a decrease in occupancy for aardvark, brown hyaena and common duiker, and an increase in occupancy for gemsbok, kudu, porcupine and steenbok (Table 5).

With the exception of duikers, all herbivores showed increasing occupancy, refuting our hypotheses that grazers and browsers would respond differently to artificial water provision. Gemsbok, the only resident wild grazing species detected, did not show increasing occupancy with proximity to water, refuting our hypothesis to that effect, and there was no evidence to support the hypothesis that browsers would not respond to artificial water provision. Overall, our results were inconclusive, but the study does provide a framework for future research into the effects of artificial water provision in semi-arid environments, a topic that is likely to become increasingly important in light of predicted hotter temperatures (Midgley and Thuiller 2011) and more extreme rainfall events in Botswana (Byakatonda *et al.* 2018).

Table 5: Summary of changes in occupancy and detection rates between 2015 and 2016 for seven mammalian species in NG3, north-western Botswana

Species	Diet	Occupancy trend	Detection trend
Aardvark	Insectivore	Decrease	Increase
Brown hyaena	Carnivore	Decrease	Increase
Common duiker	Mixed feeder	Decrease	Increase
Gemsbok	Grazer	Increase	Increase
Kudu	Browser	Increase	Decrease
Porcupine	Granivore	Increase	Decrease
Steenbok	Mixed feeder	Increase	Increase

Source: The Author

Species richness did not vary between 2015 and 2016, although the particular species found in the study area may have differed. Water-dependent species such as livestock and medium-bodied grazers (Muposhi *et al.* 2016) could have entered the study area following artificial water provision. In Kenya, livestock and human settlements around water have been linked to negative effects on the distribution and abundance of arid-adapted wildlife (de Leeuw *et al.* 2001). Invasive species populations can be boosted by artificial water provision (Letnic *et al.* 2015), as can water-dependent species (Owen-Smith 1996). The latter occupy areas close to water sources, potentially leading to over-utilisation of vegetation resources and increased predator presence, which can be detrimental to water-independent species in the surrounding area (Hall *et al.* 2013). This study did not have the resources to collect data on vegetation, but future studies should monitor potential changes in vegetation characteristics with proximity to artificial water points.

Overall, three species showed declining occupancy between 2015 and 2016: an insectivore, a carnivore and a mixed feeder. Four species showed increasing occupancy over the same period: a grazer, a browser, a granivore and a mixed feeder. Detection increased for most species, except kudu and porcupine, possibly because lower rainfall in 2016 would have resulted in lower vegetation growth and hence increased visibility and probability of detection. Rainfall in north-western Botswana in 2015 and 2016 totalled 422.2 and 349.2mm, respectively[3], which could have affected vegetation growth and mammalian population trends, although lower rainfall would more likely have resulted in occupancy decreases in 2016 rather than the observed increases for most herbivores. Changes to the environment predicted by climate change could mean delayed rainfall, hotter temperatures and more extreme rainfall events for north-western Botswana (Byakatonda *et al.* 2018 and 2019), so occupancy of resident Kalahari species may well decline over time, highlighting the need for sustained monitoring programs in the region.

Initial occupancy in 2015 did not appear to be a factor determining subsequent trends, since initial occupancy for decreasing and increasing species ranged from 0.37-0.69 and 0.32-0.79, respectively. Aardvark and brown hyena, which both displayed decreasing occupancy, are nocturnal, but so are porcupines, which showed increasing occupancy. There were, therefore, no distinct patterns that could be used to predict occupancy trends, but these results highlight the variety of responses by mammals to artificial water provision, which affects every species in different ways (Harris *et al.* 2015). Several species should, therefore, be considered by any future studies on the effects of artificial water provision in semi-arid environments.

Competing carnivores entering the study area may have caused declines in brown hyena occupancy, but other species were recorded only occasionally by the camera traps so no sudden increase in their

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presence was evident. Additionally, the increasing occupancy for most herbivores was not indicative of higher predation pressure. Grazers, such as gemsbok, are more water-dependent than browsers (Redfern *et al.* 2003), so we expected differences in their occupancy responses to artificial water provision, but no species showed changes in occupancy with proximity to water, perhaps because insufficient time had elapsed since artificial water was first provided in November 2015.

Detection of brown hyena in 2015 was higher in grassland than scrub, presumably because of greater visibility, which could also have led to higher levels of porcupine occupancy in 2015. Detection of gemsbok in 2016 and porcupine in 2015 was higher in scrub than grassland, where they could have been moving more slowly, allowing the camera traps to detect them more readily. There is no clear reason why these habitat-specific differences in occupancy and detection were not replicated in both years. Occupancy increased for most herbivore species, apart from common duiker, in keeping with previous findings that herbivore populations initially respond positively to artificial water provision (Sutherland *et al.* 2018). Higher herbivore populations lead to increased grazing pressure and potential over-utilisation of resources and homogenisation of resource availability (Chamaille-Jammes *et al.* 2016), particularly during periods of drought (Smit and Grant 2009). Hotter conditions with more extreme rainfall patterns are predicted for Botswana within the next few decades (Byakatonda *et al.* 2018), so inflated herbivore populations in north-western Botswana linked to artificial water provision could have substantial effects on vegetation in the study site.

Most desert dwellers are highly adapted to semi-arid conditions, with efficient muscles and high water retention that means they do not need to drink (Curtin *et al.* 2018) and can obtain moisture from their food, but many species will drink if water is provided (Selebatso *et al.* 2018). Drinking could reduce the need to hunt for carnivores that secure moisture from prey and reduce physiological stress for all species (Hall *et al.* 2013), but water does not benefit all species equally, so could potentially lead to biodiversity loss (Harris *et al.* 2015). Animals living in semi-arid environments are naturally drought-resistant, but over-stocking linked to artificial water provision can deplete resource buffers, reducing the availability of palatable forage and ultimately leading to herbivore population declines during drought periods (Parker and Witkowski 1999). Artificial water provision can reduce home range size (McKee *et al.* 2015) and promote residency in otherwise migratory species, which increases vulnerability to stochastic environmental effects, including water pump failure (Selebatso *et al.* 2018). Long-term studies of semi-arid landscapes where artificial water provision is initiated and sustained are needed to understand how the mammalian community will respond over time.

Conclusion

In semi-arid landscapes, water is a naturally scarce resource, so providing artificial water year-round represents a substantial environmental change that is likely to affect many aspects of the ecosystem (James *et al.* 1999). Artificial water provision in environments such as those in north-western Botswana should, therefore, be conducted with caution, and further studies on the long-term effects of continuous artificial water provision on the surrounding ecosystem should be carried out. Results from this study were inconclusive, but our findings allowed us to identify several limitations that could be used to inform future study design. The study was conducted over a relatively short time scale, with artificial water having been provided by The Lodge for less than a year by the start of the 2016 late dry season. We also only had access to the area surrounding one source of artificial water, whereas additional study sites may have provided greater insight. Prolonged research over a lengthier time scale and covering more study sites may have produced different results, but resources did not allow such a study. Future studies should allow for several years of study following initial artificial water provision in multiple locations.

Camera traps were used to record a variety of species, rather than placed in an optimal position for

single species. The camera traps may, therefore, have failed to record some images of species because they did not trigger the cameras (Burton *et al.* 2015). However, we did not attempt to estimate density from these data, and the occupancy analysis approach that we used took detection, which varied with species, into consideration (MacKenzie *et al.* 2006). Several species not included in the occupancy analyses were captured by the camera traps, but their detection rates were too low to allow occupancy estimation. Some rare or specialised species may have been affected by artificial water provision, but we could not reliably incorporate them into models. Ideally, camera traps would have been positioned at the artificial water source to identify species drinking, but we did not have sufficient camera traps to monitor grid sites as well as waterhole activity. Several parameters other than artificial water availability could have varied between 2015 and 2016, so we cannot definitively state that observed changes in occupancy were only linked to artificial water provision, but it probably contributed substantially to changes in environmental conditions.

Semi-arid landscapes, including those of north-western Botswana, are often sparsely populated by wildlife, which can reduce their value for eco-tourism and their conservation importance could be overlooked, so artificial water provision can increase the attraction for tourism enterprises with vested interests in conservation of natural resources sought after by tourists. Ecosystems in semi-arid landscapes are highly valuable because they provide insight into future conditions under climate change scenarios for many parts of the world (Duncan *et al.* 2012 and Razgour *et al.* 2018), including Botswana. Rising temperatures and declining rainfall patterns could push the physiological capabilities of arid-adapted mammals to the point that they can no longer extract water from their forage and must drink to balance water losses.

However, the long-term effects of water provision are not well understood, and there is a risk of losing species that are behaviourally and physiologically adapted to drought conditions. There is, therefore, a fine balance between attracting sufficient animals to marginal areas for viable photographic tourism and maintaining a functional ecosystem, which warrants further research. Results from this study can be used to guide future research endeavours, which should be carried out over a longer time period, encompass multiple sites with artificial water provision, consider a variety of species, record data on vegetation characteristics and account for relevant environmental variables.

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